Living Murray icon site wetlands within the Mallee CMA region: monitoring program designs and 2004-05 monitoring results.

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Report citation

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Cover: Carp (*Cyprinus carpio*) and cumbungi (*Typha domingensis*) in Webster's Lagoon (photos: I. Ellis and O. Scholz)

Summary

The ecological condition of many Murray River floodplain ecosystems has suffered as a consequence of changes to their flow regime. In 2002 the Murray Darling Basin Ministerial Council established the Living Murray Initiative to return the Murray River system to a healthy working river through the delivery of 500 GL of environmental flows to targeted wetlands within six 'Icon Sites' (Significant Ecological Assets). The ability to deliver these flows and to demonstrate environmental benefits during the life time of the Living Murray Initiative (2002-2012) will involve the installation and/or modification of regulating structures and the implementation of long-term wetland monitoring programs, respectively.

The ambit of this study was to develop scientifically credible monitoring programs capable of detecting environmental responses to managed changes in flows for five targeted Icon Site wetlands located within the Mallee CMA region: Webster's Lagoon and Lake Wallawalla (Lindsay Island), Potterwalkagee Creek (Mulcra Island), Horseshoe Lagoon (Wallpolla Island), and Hattah Lakes (target wetland to be determined).

To date, structural options for achieving site-specific flow and ecological objectives have been developed for Webster's Lagoon, Lake Wallawalla, Horseshoe Lagoon and Potterwalkagee Creek. The installation of the control structures developed for each wetland is expected to commence in the near future.

Monitoring programs could be fully developed for three wetlands (Webster's Lagoon, Lake Wallawalla and Horseshoe Lagoon). However, further hydraulic modelling is required to complete this process for Potterwalkagee Creek. The development of a monitoring program for the Hattah Lakes Icon Site will become possible once flow and ecological objectives have been finalised.

The development of monitoring programs for each wetland followed a six step process involving: i) the definition of key ecological objectives and management priorities, ii) the development of conceptual ecological response models, iii) the definition of specific testable ecological response hypotheses, iv) the identification of key ecological response variables, v) the determination of a suitable experimental design, and vi) the optimisation of sampling effort and statistical power.

Several experimental design options are available to test hypotheses based on the identification of changes in response variables before and after an environmental disturbance such as wetland drying/flooding. These design options vary both in their complexity and in their ability to detect change. Generally, the incorporation of experimental controls within an experimental design is favoured as it permits the elimination of potentially confounding artefacts introduced by the experimental procedure and provides the greatest statistical power to test hypotheses and to demonstrate cause and effect.

Monitoring to demonstrate cause and effect is expensive in both time and money and cannot reasonably be undertaken for all Icon Site wetlands. Nor is it possible or desirable. Webster's Lagoon was the only wetland for which a suitable control site could be located, permitting the use of a Before, After, Control and Impact (BACI) experimental design. Alternative monitoring program designs were used for the other sites. These included Intervention Analysis (IA), which whilst less scientifically rigorous offers the capacity to monitor ecological responses (rather than demonstrate causality), and Instantaneous Adaptive Management, in which ecological outcomes are monitored directly and real-time data is used to inform structure operational rules.

Effort was made to maximise consistency between wetland monitoring programs in terms of the hypotheses that were generated and the sampling protocols that were developed, thereby increasing the potential for assessing ecosystem responses to changes in hydrology at the landscape scale. Whilst emphasis within each of the monitoring programs was placed on the identification of single wet/dry cycle ecosystem responses, this does not preclude the identification of longer term multi-cycle ecosystem responses as data becomes available.

Field monitoring of Webster's Lagoon commenced during December 2004. Data collected to date validated initial assumptions of the conceptual model developed for the Lagoon and validated statistical requirements of the sampling protocols used. Carp caught within Webster's Lagoon were of two broad size classes, with most (74.4 %) within the 70-150 mm FL size range. This suggests that the mesh size of grates incorporated into the proposed regulator will need to be sufficiently small to exclude these for there to be a measurable treatment effect.

With Before-After type experimental designs (BACI and IA), the ability to detect the effects of management intervention is greatly influenced by the duration of the monitoried pre-intervention period. As such, we strongly recommend that at least 12 months (or preferably 24 months) of pre-intervention (*i.e.* 'Before') data be collected prior to the operation of regulating structures.

1 Project Overview

In 2002, the Murray Darling Basin Ministerial Council established the Living Murray Initiative – a long term program of collective actions aimed at returning the Murray River system to a healthy working river. The vision for the Living Murray Initiative is '*… a healthy River Murray system, sustaining communities and preserving unique values'*. The first step of the Living Murray Initiative aims to recover 500 GL of water to improve environmental flows and achieve ecological objectives at six Significant Ecological Assets (SEAs) along the Murray River: Barmah-Millewa Forests, Gunbower-Koondrook-Pericoota Forests, Hattah Lakes, Chowilla Floodplain including Lindsay, Mulcra and Wallpolla islands, the Murray Mouth including the Coorong and the Lower Lakes, and the River Murray Channel (Figure 1.1).

Figure 1.1: Locations of the 6 Living Murray Icon Sites. (source Mallee CMA 2005).

The restoration of each of these 'Icon Sites' through the provision of environmental flows and the management of key identified threats is to be achieved through the development of Icon Site Water Management Plans, part of which will involve the installation and/or modification of regulating structures and the development and implementation of long-term monitoring programs capable of demonstrating environmental benefits.

Two Icon Sites occur within the Mallee Catchment Management Authority (CMA) region; Hattah Lakes and the Lindsay-Mulcra-Wallpolla islands floodplain. Lindsay, Mulcra and Wallpolla islands are the Victorian component of the Chowilla-Lindsay-Wallpolla Icon Site, which extends from 40 km west of Mildura to the South Australian border. Lindsay Island lies within the Murray-Sunset National Park and consists of 15000 ha of land bounded by the

Murray River and the Lindsay River anabranch. Mulcra Island consists of 2156 ha of State Forest between Lindsay and Wallpolla islands bounded by the Murray and the Potterwalkagee Creek anabranch. Wallpolla Island, also an anabranch of the Murray River, consists of 9000 ha of State Forest bounded by the Murray River and by Wallpolla Creek. The Hattah Lakes Icon Site is situated 80 km upstream of Mildura and lies within the Hattah Kulkyne National Park.

Wetlands within the Lindsay-Mulcra-Wallpolla islands Icon Sites that have been identified as priority sites for the delivery of environmental flows include Webster's Lagoon and Lake Wallawalla (Lindsay Island), Potterwalkagee Creek (Mulcra Island) and Horseshoe Lagoon (Wallpolla Island). No target site has yet been identified for the Hattah Lakes Icon Site.

To date, assessments of the ecological water requirements have been made and management options for delivering site-specific environmental flows through the installation of regulating structures have been developed for Webster's Lagoon, Lake Wallawalla and Horseshoe Lagoon, and the installation of the control structures developed for each wetland is expected to commence in the near future (refer to review by Reed 2004 and Zukowski and Meredith 2005). However, this process has not yet been completed for Potterwalkagee Creek and Hattah Lakes.

Monitoring to demonstrate cause and effect is expensive in both time and money and cannot reasonably be undertaken for all Icon Site wetlands. Nor is it possible. Webster's Lagoon is currently permanently inundated, making it possible to locate suitable control site/s and thus to implement a more rigorous experimental design that permits the demonstration of cause and effect (*i.e.* BACI-based; Underwood 1996). In contrast, Horseshoe Lagoon, Lake Wallawalla, Potterwalkagee Creek and Lake Hattah have a wet/dry regime that is unique to their floodplains, discounting the possibility of locating suitable control sites. Because of this, less intensive Before-After monitoring program designs need be employed that are capable of informing structural operational rules, and that permit assessments of whether ecological objectives are being met without demonstrating causality.

The objectives of this investigation were, firstly, to develop long-term monitoring programs capable of assessing the efficacy of management initiatives in meeting pre-defined flow and ecological objectives for each of the targeted wetlands and, secondly, to present initial monitoring data collected from these wetlands as part of this process. Data collected prior to the installation of the proposed regulating structures in each of the targeted wetlands will provide a benchmark against which future changes in wetland condition can be assessed.

This report includes only monitoring data for Webster's Lagoon up to mid-April 2005. No data was collected from the other targeted wetlands as they were either dry (Horseshoe Lagoon, Lake Wallawalla) or their management objectives had not yet been finalised (Potterwalkagee Creek and Lake Hattah).

2 Monitoring Program Design Framework

The success of any management program, and thus the ability to protect ecosystem values, depends on the implementation of an effective monitoring program. The adaptive management approach to monitoring we describe in this study is based on the premise that managed ecosystems are complex and inherently unpredictable. The adaptive approach embraces the uncertainties in system responses and attempts to structure management actions as experiments from which learning is a critical product. Central to this process is the structuring of monitoring efforts within a scientifically rigorous experimental framework in which collected data can be used iteratively to re-assess the conceptual understanding of ecosystem processes and to inform the management decision making framework (Lee and Lawrence 1986, Lee 1989, Nyberg 1999a).

The shortcomings of monitoring programs instituted in the past and the issue of how to improve program design have been the subject of much discussion (*e.g.* Underwood 1991, Stewart-Oaten *et al.* 1986, 1992, Green 1993, Finlayson 1996). In this section we provide a synopsis of the key considerations involved in developing a generic adaptive management based monitoring program. As management issues are likely to vary both between systems and through time, this framework will form the basis for developing site-specific programs for each of the identified Icon Site wetlands.

A number of publications have outlined a sequence of steps based on the falsification experimental protocol described by Popper (1968) to achieve the best possible design for, and outcomes from, monitoring programs to detect human impacts (*i.e.* impacts of altered hydrological regimes) (*e.g.* Underwood 1996, Green 1979, Chapman and Underwood 2000, Thoms *et al.* 2001, Downes *et al* 2002, King *et al.* 2003) (Figure 2.1). The suggested sequential steps for designing an effective monitoring program described below provided the framework used in the current study:

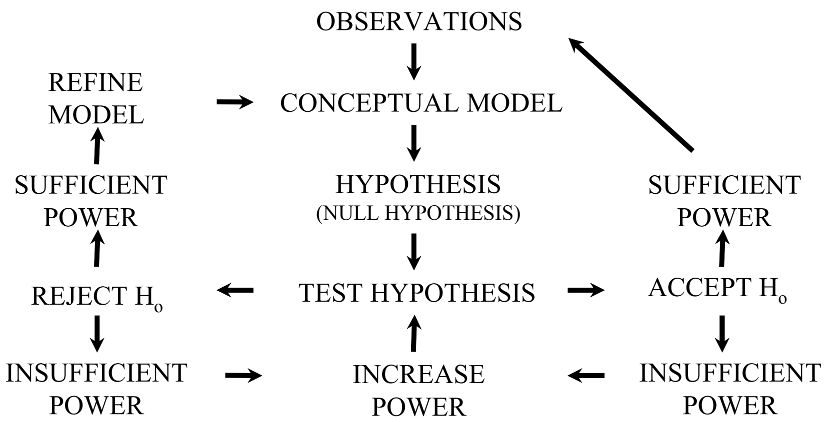


Figure 2.1: Components of a falsification experimental protocol (based on Popper 1968).

1: Define monitoring scale, key ecological objectives and management priorities

Define the spatial scale of management/monitoring activity, identify and prioritise ecological values that are of management concern and identify ecological objectives for each.

2: Develop a conceptual model for the system in question

The main aim of this step is to synthesise available knowledge of the relationships between the various ecosystem components and to identify management actions likely to achieve desired outcomes. This process will help identify both flow and management objectives.

a. Flow objectives: the water regime most likely to achieve desired outcome for each ecological value. Roberts and Marston (2000), SKM (2002a), Jaensch (2002), and Tucker *et al.* (2003) all provide information on water regime requirements for many species of wetland flora and fauna.
b. Management objectives: the combination of flow and other management activities that will enable the ecological objectives to be met.

3: Define specific testable hypotheses

Define specific testable hypotheses regarding expected changes in ecological values arising from the imposition of flow and management objectives. Not specifying testable questions reduces the ability to feed collected data back into the adaptive management process. As many of the potential responses of ecological values are likely to be driven by changes in hydrology, it is critical that an accurate assessment of hydrology (*e.g.* water levels) be included within the monitoring design. This will allow for the testing of two key hypotheses:

H_1: that water management actions have an effect on system hydrology, and
H_2: that water management actions have an effect on ecological values.

The ability to test H_1 will depend on the accuracy of water level gauging and the magnitude of the hydrological change. The ability to test H_2 will depend on the precision and statistical power with which key natural assets are monitored and the magnitude of the ecological change.

4: Select variables and indicators for measuring

The generation of hypotheses based on the conceptual model will assist in the identification of key response variables. The variables chosen should be relevant to the question asked, be strongly associated with the putative impact, and be ecologically significant. Considerations in choosing variables include the availability of potential indicator groups, the potential for redundancy between variables, the mobility of individual organisms, the response times of variable or indicator groups, and the reliability of data collection (sampling and analytical methodology) (Downes *et al.* 2002). Indicators should either directly assess progress toward an ecological, hydrological or management objective or they should be linked to the achievement of an objective through the conceptual model. Refer to Reid and Brooks (1998) for more information on the selection of indicators and their relation to hydrological regime.

5: Develop a program design

Collecting data to detect the presence or measure the size of an environmental impact requires careful design of the sampling program. Several experimental design options are available to test hypotheses based on the identification of changes in a response variable before and after an environmental disturbance, such as wetland drying (*e.g.* Underwood 1996, Downes *et al.* 2002). These design options vary both in their complexity and in their ability to detect change.

The incorporation of experimental controls or reference sites is favoured as it permits the elimination of potentially confounding artefacts introduced by the experimental procedure and provides the greatest statistical power to test hypotheses. Where control or reference sites can be identified, statistical designs incorporating Before, After, Control, and Impact (BACI) data should be used (Underwood 1996). Where control or reference sites are not available it will be necessary to rely on a more simple program design that compares data collected before and after implementation of the proposed water management strategy (Intervention Analysis). Unfortunately, such a design is limited by the fact that temporal variation in the selected indicators may occur independently of any changes related to the proposed water management action. Evidence for the impact of management actions will thus depend on the length of the period over which before and after data are collected and the frequency of interventions.

A 'levels of evidence' approach may be adopted to increase the confidence of conclusions generated by experimentally based monitoring. This refers to a method of analysis that requires an assessment based on a variety of different types of evidence. Typically, the relationship between management action (impact) and ecosystem response is a complicated one. There are often long time lags and usually many other a/biotic factors which act on the response variables being measured, making it difficult to separate the effects due to different causes. Downes *et al.* (2002) describe a number of causal criteria, such as strength of association, consistency of association (is a similar response observed elsewhere?), biological gradient (impact-response relationship) and biological plausibility, which can be examined concurrently to increase the strength of conclusions.

<u>6: Optimise sampling effort and statistical power</u>

The objective of this step is to design a sampling regime that provides the most efficient (in terms of resources) and precise estimates of selected response parameters. Most importantly, consideration needs be given to the type of spatial and temporal sampling (random, stratified, systematic) and to power analysis based on considerations of spatial and temporal variation. Power analyses provide a measure of the confidence of conclusions reached for a given sampling effort. For many systems there is either no data or the data that is available does not provide insight into the spatial and/or temporal variation of the response variables within the system. An initial pilot study offers the opportunity to provide such information. Key references for power analysis and the optimisation of sampling effort include Elliott (1979), Andrew and Mapstone (1987), Green (1989), Underwood (1993, 1996), Miller *et al.* (1997), Thoms *et al*. (2001), and Keough and Quinn (2002).

<u>7: Feed-back of monitoring data into the adaptive management process</u>

Integral to the adaptive management process is the periodic review of monitoring data, and its interpretation in terms of the conceptual model and management targets. From this, potential changes in management responses may be identified. Where appropriate, this review process may also involve modification of the conceptual model and subsequent hypotheses. Augmentation of the monitoring process with targeted small-scale experimentation may provide an effective means of examining, over much shorter time-frames, process responses predicted by the conceptual model. It will also allow for testing of hypotheses requiring field manipulations (*i.e.* changes in water level) that cannot otherwise be implemented.

3 Webster’s Lagoon

3.1 Background

Webster’s Lagoon is a permanently inundated wetland located on Lindsay Island in northwest Victoria (Figure 3.1). As a well-defined cut-off meander of the Murray River, Webster’s Lagoon is flat-bottomed and asymmetric in cross-section, with the southern bank being higher and steeper than the northern bank. The southern bank is generally steeper as a result of the former Murray River cutting into the high alluvial floodplain, whereas the northern bank is depositional in origin and comprises younger alluvium (SKM 2004a).

The hydrological characteristics of Webster’s Lagoon have changed substantially during the past 100 years as a consequence of the regulation of the Murray River and changes in the intake and flow capacity of the various associated anabranches and creek channels. Because it lies within the weir pool created by Lock 6, Webster’s Lagoon has become an impounded wetland with a permanent connection at its western end to the Murray River through Toupnein Creek. Regulation of the Murray River has also greatly decreased the frequency and duration of inflows at the eastern end of Webster’s Lagoon. Under these conditions, the lagoon does not experience the long summer and autumn complete dry periods thought to have occurred under natural conditions (SKM 2004a), although small changes in flow/water level result in the periodic exposure of extensive areas of the lagoon (wet-dry littoral zone) (Egis 2001) (Figure 3.2). At the Lock 6 pool level (19.2 m AHD), approximately 40 ha of the lagoon is inundated. During medium to large floods this area may increase up to a maximum of 80 ha (>21 m AHD) (Beovich 1994).

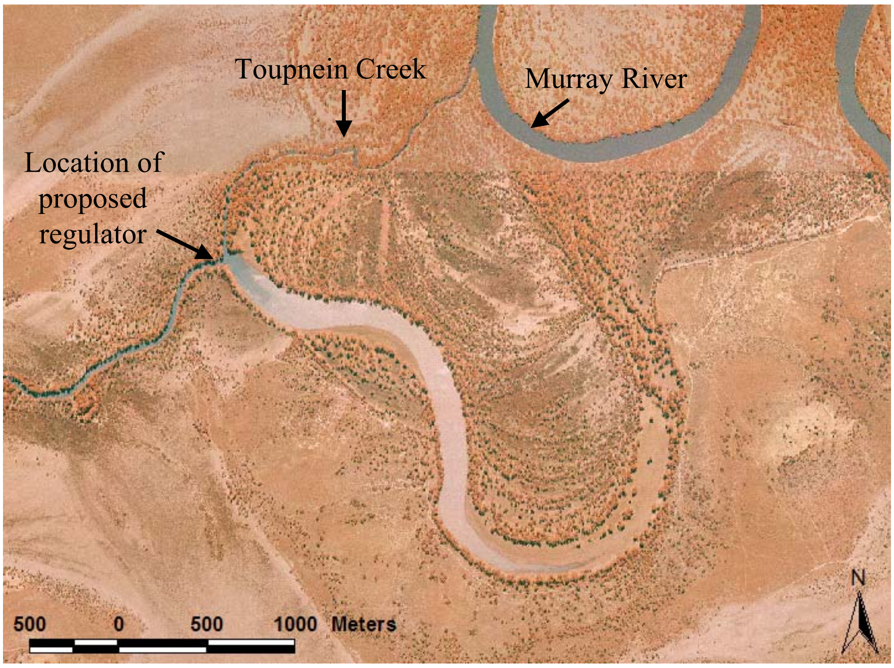


Figure 3.1: Webster’s Lagoon showing location of proposed structure (source MDBC River Murray Mapping 2nd edn. May 1996).

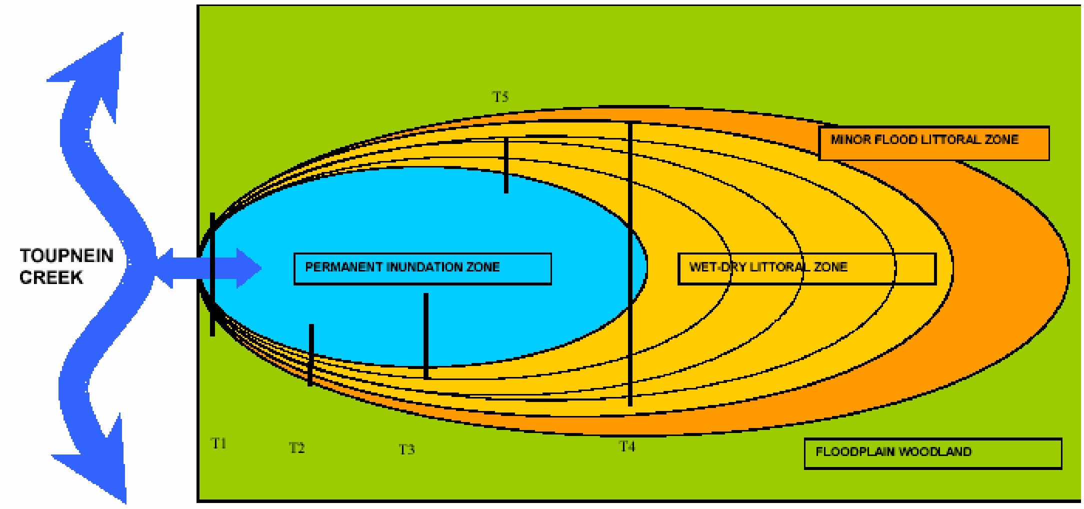


Figure 3.2: Schematic diagram of the hydraulic zones within Webster's Lagoon (source J. Roberts in SKM 2004a).

Webster's Lagoon is considered to be of high ecological value on both regional and state levels. Much of this stems from the broad range of habitat it provides and its ability to support a variety of significant flora and fauna (SKM and Roberts 2003). Key ecosystem values and processes supported by Webster's Lagoon have been compromised either directly by changes in flow regime or indirectly through associated threats, such as the presence of carp (*Cyprinus carpio*) and the undesired expansion of native species such as cumbungi (*Typha* spp.) and red gum (*Eucalyptus camaldulensis*) in the lagoon bed. The management of these threats and other identified ecological values and processes is to be achieved through the installation and operation of a regulator at the junction of Toupnein Creek and Webster's Lagoon (refer to Zukowski and Meredith 2005 for design details).

Ecological and flow objectives for each identified ecological value, process and threat developed by SKM (2004a) are shown in Table 3.1. The regulating structure proposed for Webster's Lagoon will:

- Permit the partial and tempory disconnection of the lagoon from Toupnein Creek.
- Allow for the restoration of an intermittent water regime within the wetland.
- Permit the retention of water within the Lagoon at higher elevations (19.8 m AHD) following medium to large flood events or if water is pumped into the lagoon.
- Permit the inundation of the wet-dry littoral zone for longer than is currently occurring.
- Incorporate the capacity to exclude larger fish (*e.g.* carp) from inflows.

Operating guidelines for the regulating structure are to:

- Dry the wet-dry littoral zone for a minimum of 6 months in summer and autumn.
- Completely dry the existing permanently inundated zone for 6 - 10 months.
- Wet the wet-dry littoral zone for between 3 - 6 months during winter and spring.

Ecological value/process	Ecological objectives	Flow objectives
Lateral connectivity	Maintain connections between Webster's Lagoon and Toupnein Creek	Insufficient data currently available to indicate the most appropriate timing and duration of periods of dis/connection.
Carbon and nutrient cycling	Rehabilitate	Dry out wet-dry zone annually over summer/autumn for at least 2 weeks. Inundate wet-dry zone nearly every year during spring for 1-3 months.
Vegetation	Rehabilitate extent and diversity	Dry out wet-dry zone annually between December-June for at least 6 months. Inundate wet-dry zone nearly every year between June-December for 3-6 months.
Small native fish	Rehabilitate habitat and feeding opportunities	Dry out wet-dry zone annually during summer-autumn for at least 6 months. Inundate wet-dry zone nearly every year during winter-spring for at least 3 months.
Waterbirds	Rehabilitate habitat and feeding opportunities	Dry out wet-dry zone annually during summer-autumn for at least 6 months. Inundate wet-dry zone nearly every year during winter-spring for at least 3 months.
Turtles (Broad-shelled)	Maintain	Insufficient data currently available to indicate whether structure will exclude re-colonisation after re-flooding.
Ecological threat		
Saline groundwater discharges	Minimise saline discharges to the wetland and river	Minimise period of wet-dry zone drying. Insufficient data currently available to indicate maximum period of drying permissible.
Cumbungi	Reduce vigour and extent in persistent pool	Complete drying between December-June (ideally September-June) for 7-10 months.
Red gum	Reduce abundance/extent of regeneration in wet-dry zone	Inundation to a depth of 50-100 cm for 4-6 weeks between June-October, or shallow flooding for 12-26 weeks during autumn-winter. Complete drying between October-June for 6 months.
Carp	Reduce abundance	Complete drying anytime for more than 1 day.

Table 3.1: Summary of ecological objectives and flow requirements for Webster's Lagoon (source SKM 2004a).

3.2 Conceptual framework and hypotheses

Floodplain wetlands have intrinsic local values, and represent an important component of the larger floodplain ecosystem (Ward 1989, Ward and Stanford 1995). They are naturally diverse and productive habitats, and whose management through the alteration of natural flow regimes has resulted in either real or perceived declines in wetland condition (*e.g.* Kingsford 2000a,b). It is generally agreed that both wet and dry periods are important in maintaining ecosystem integrity in ephemeral wetlands (Boulton and Lloyd 1992, Bunn *et al*. 1997, Boulton and Jenkins 1998). Disturbances, such as flooding and drying, drive aquatic and terrestrial successional processes and facilitate biotic and abiotic exchanges between elements of the floodplain and the riverine environment (*cf.* Flood Pulse Concept; Junk *et al.* 1989). Because of this, ephemeral wetlands are potentially sites of high productivity and diversity within floodplain ecosystems. Consequently, the management of these wetlands has implications for ecosystem productivity and diversity at the landscape scale. Thus, any changes in wetland function initiated through hydraulic modification need to be viewed from both the perspective of the individual wetland and from a landscape perspective.

Available information, conceptual models (*e.g*. Flood-Pulse Concept - Junk *et al.* 1989, Alternative-States Model - Scheffer *et al.* 1993, Geomorphic-Trophic Model - Hershey *et al.* 1999, Trophic-Cascade Model - Carpenter *et al.* 2001, Ephemeral Deflation Basin Lake Model - Scholz and Gawne 2004a,b) and expert consultation (refer to Acknowledgements) were used to develop a conceptual model of the links between the various ecosystem components and their hypothesised responses under existing and proposed management scenarios (Figures 3.3 and 3.4). From these we generated testable hypotheses which were used to guide the development of the monitoring program.

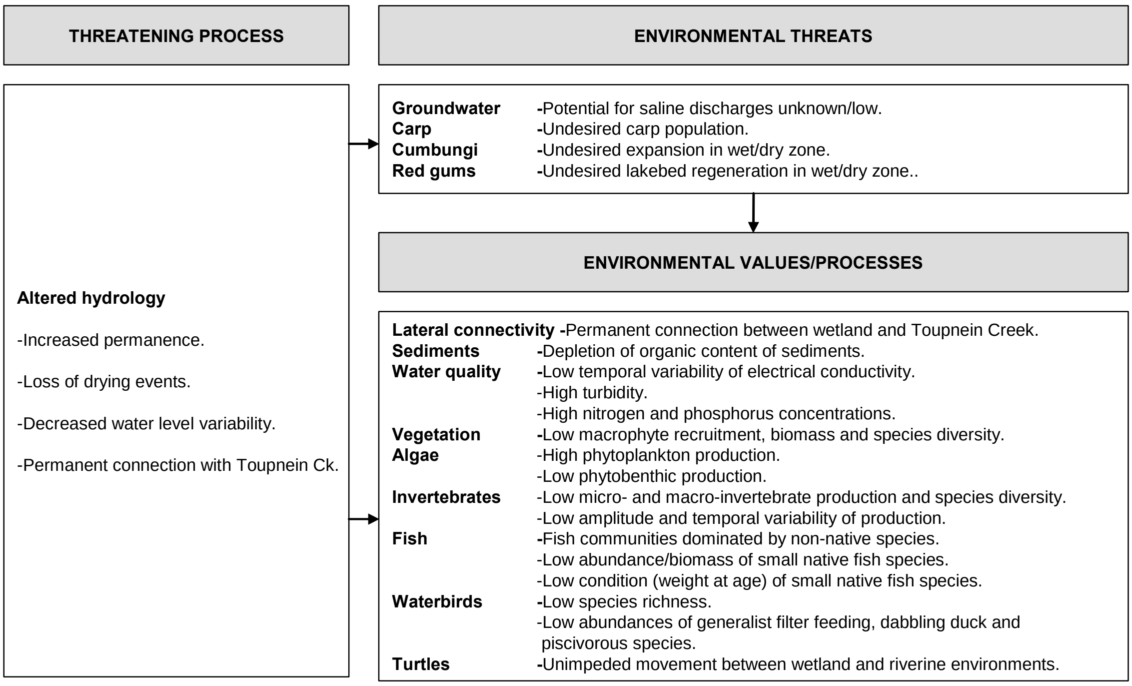


Figure 3.3: Conceptualisation of ecosystem function under existing flow conditions at Webster's Lagoon.

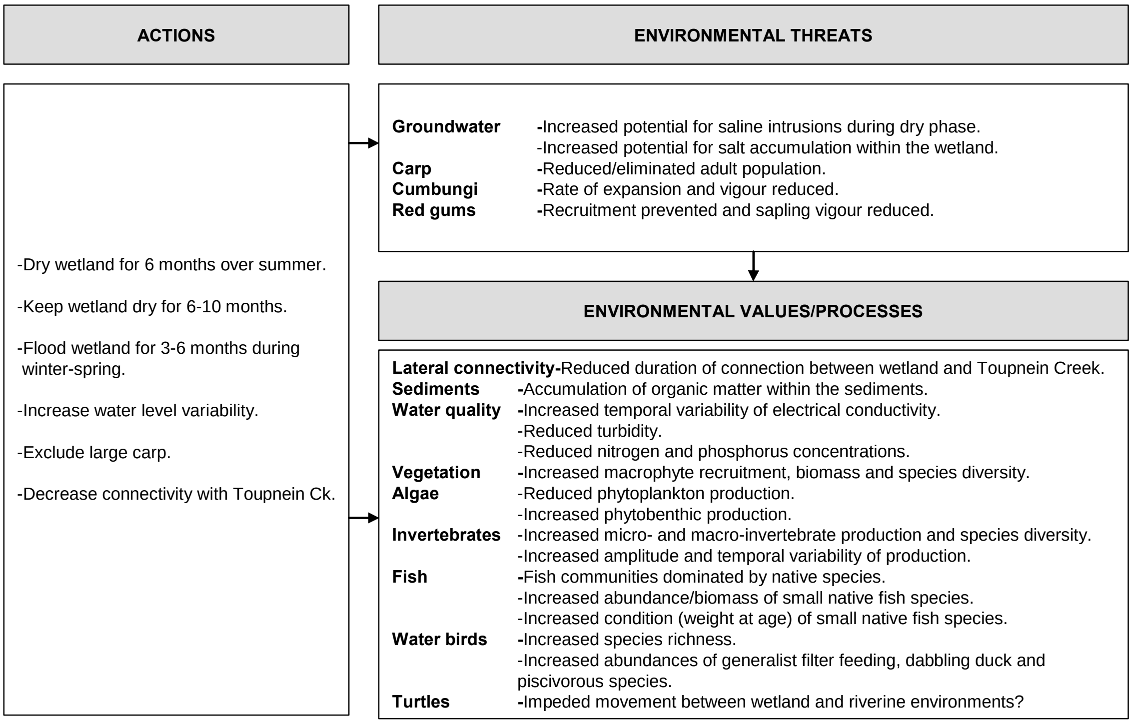


Figure 3.4: Conceptualisation of ecosystem function under the flow regime proposed for Webster's Lagoon.

Trophic cascade or food web models suggest that aquatic ecosystem productivity and community composition are determined by a combination of top-down and bottom-up forces (Carpenter and Kitchell 1988, Carpenter *et al.* 2001). The strengths of these interactions (*e.g.* competition, predation) tend to become weaker as food web complexity increases (Shapiro 1990, Lazzaro 1997, Borer *et al.* 2005), and for ephemeral systems biotic interactions are likely to increase during the drying phase (Schneider and Frost 1996, Golladay *et al.* 1997, Tockner *et al.* 1999). This latter response has been linked to declining habitat or refuge availability (Golladay *et al.* 1997). For example, higher water levels offer fish refuge from avian predation, but eventually this refuge is lost and avian predation becomes more effective (Kahl 1964, Kushlan 1976). Consequently, not only do fish numbers decrease as wetlands dry, but the fish community becomes increasingly dominated by smaller fish species (Scholz and Gawne 2004b). These changes in the fish community may thus stimulate a downward cascading change in trophic structure by offering invertebrate predators (*e.g.* notonectids, dytiscid beetles) an opportunity to proliferate (*e.g.* Boulton and Lloyd 1991, Jeffries 1994).

Viewed independently, hypothesized responses of each trophic level to the re-instatement of a wet/dry phase and the exclusion of carp from Webster's Lagoon are suggestive of an overall increase in productivity and diversity. Whilst we recognise that trophic interactions are likely to play a key role in determining measured responses at each level, the strength of such interactions is difficult to either measure or predict. It will thus be important to consider likely strengths of trophic interactions when interpreting the observations derived for each of the outlined hypotheses.

Flow/hydrology

Hypothesis A1: The proposed control structure in Webster's Lagoon will have an effect on water levels within the wetland in compliance with the management strategy.

Many ecological responses are likely to be driven by changes in hydrology, making it critical that an accurate assessment of water levels be included within the monitoring design. The flow objectives for Webster's Lagoon are to maintain water stage within the lagoon at 19.8 mAHD for 6 months, and to allow for complete drying of the lagoon (AHD < 19.4m) for 6 months. Achieving this will allow for the testing of impacts on ecological assets. Water level data will feed directly into an adaptive management framework, allowing for real time management responses during periods of non-compliance.

Lateral connectivity

Hypothesis B1: The proposed control structure in Webster's Lagoon will introduce periods of hydraulic isolation to the Lagoon.

Many wetland processes and biota are dependent on exchanges between wetland and riverine environments. Under the current flow regime Webster's Lagoon is permanently connected to Toupnein Creek. The construction and operation of a regulating structure at the mouth of the Lagoon will introduce periods of hydraulic isolation and thus potentially inhibit exchanges between aquatic environments. This may adversely impact, for example, on the ability of turtle populations to re-colonise the Lagoon after an imposed drying event.

Whilst the regime of wetland dis/connection will be monitored directly, the identification of appropriate duration and timing of periods of dis/connection will be addressed within the monitoring program for each trophic component as desribed below.

Groundwater/salinity

Hypothesis C1: Re-instating dry phases in Webster's Lagoon will raise groundwater levels beneath the wetland during the dry periods. *Hypothesis C2: Re-instating dry phases in Webster's Lagoon will cause an increase in groundwater salinity beneath the wetland during the dry periods.* *Hypothesis C3: Re instating dry phases in Webster's Lagoon will result in an increase in the salinity of wetland sediments during the dry periods.*

The geology of Webster's Lagoon comprises three distinct layers commencing at the surface with Fine Alluvium (0-5 m thick), underlain by Channel Sands (aquifer 5-10 m thick), and at a depth of 20-25 m by Parilla Sand (aquitard >10 m thick) (SKM 2004a). The water table is thought to reside within the Fine Alluvium at a depth between 2-5 m below the ground surface, with little vertical flow occurring between the Parilla Sand and Channel Sand layers. Groundwater salinity under Webster's Lagoon is thought to increase with depth, ranging from 1000-20000 $\mu S\ cm^{-1}$ within the upper layers and between 20000-40000 $\mu S\ cm^{-1}$ within the Channel Sands (SKM 2004a).

SKM (2004a) indicated that surface flooding has the potential to influence both the direction of groundwater flows and the formation of groundwater mounds. However, no information is currently available to suggest what impact on groundwater the drying of Webster's Lagoon is likely to have. Data from other systems suggests that drying is likely to reduce downward hydraulic pressure on saline groundwater, allowing it to rise. This may increase the potential for soil salinisation and saline groundwater intrusion into the wetland, and thus also the potential for the movement of salt downstream into the river. The issue related to minimising salinity going back to the Murray River has been examined as part of the Initial Salinity Impact Assessment for Webster's Lagoon (Evans *et al.* 2004). Although no environmental objective was developed for groundwater/salinity, monitoring groundwater responses to changes in surface hydrology will help fill an important knowledge gap and assist with minimising salinisation risks.

<u>Sediments</u>

Hypothesis D1: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the mass of benthic coarse particulate organic matter (CPOM) in/on the sediments.

Re-instating a wet/dry regime in Webster's Lagoon is likely to impact on the amount of organic matter present within the wetland sediments in several ways. Firstly, wetland drying will allow the establishment of terrestrial vegetation within the wet/dry zone, and greater water level variability will likely stimulate aquatic macrophyte production (*e.g.* Walker *et al.* 1994, Blanch *et al.* 1999, 2000), especially in the absence of carp (refer to carp section below). These are likely to represent a potentially important source of organic matter within the wetland as they decompose. Secondly, the proposed changes in hydrology are likely to increase access to CPOM deposited at higher elevations. And thirdly, physical barriers such as natural sills and regulating structures have the potential to reduce lateral exchanges of organic matter between wetland and riverine environments, thereby increasing the potential for the net accumulation of organic matter originating within the wetland.

The increases in sediment organic matter content predicted to occur as a consequence of altering the flow regime of Webster's Lagoon will likely play an important role in carbon and nutrient cycling within the wetland and in structuring trophic development by increasing food and habitat resources for detritivorous biota (*e.g.* invertebrates). Although not specifically identified as a key ecological value, measuring sediment organic matter content will increase our ability to interpret the responses of key biota to changes in flow.

<u>Water quality</u>

Electrical conductivity

Hypothesis E1: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will increase the temporal variability of water column electrical conductivity.

Hypothesis E2: Re-instating a wet/dry regime and reducing connectivity between Webster's Lagoon and Toupnein Creek will lead to the loading of salt within the wetland.

In its current permanently inundated and connected state, electrical conductivity (EC) within Webster's Lagoon increases with distance from Toupnein Creek (MDFRC unpublished data). This gradient demonstrates the incomplete mixing of less saline Toupnein Creek inflows within the lagoon. Increases in EC with distance from connection are presumed to be driven by evaporative concentration. Although no environmental objective was developed for surface water salinity, monitoring responses to changes in hydrology is necessary to assess salinisation risks.

It is anticipated that the installation of a regulating structure will have several consequences on wetland EC. Firstly, the structure itself is likely to impede diluting exchanges between Toupnein Creek and the Lagoon, allowing EC to increase at sites closer to the regulator.

Secondly, as the wetland dries evaporative concentration is likely to elevate salt concentrations throughout the wetland to levels exceeding those currently encountered. In the final stages of drying the concentration of salts can be extreme with hyper-saline conditions often developing within remnant pools (*e.g.* Scholz and Gawne 2004a). Such changes are likely to impact on both physical processes (Grace *et al.* 1997, Baldwin *et al.* in review) and the structure of aquatic communities (Hart *et al.* 1991, Bailey and James 2000, Nielsen *et al.* 2003, Baldwin *et al.* in review). Thirdly, post-inundation EC may be elevated above that of the floodwaters by the intrusion and surface encrustation of saline groundwater during the drying phase. And fourthly, the hydraulic isolation of Webster's Lagoon between inflow events is likely to reduce the potential for salt to be exported from the wetland, possibly resulting in the accumulation of salt within the wetland over successive wet/dry cycles.

Turbidity

Hypothesis E3: Re-instating a wet/dry regime to Webster's Lagoon will result in turbidity increasing during the initial drying phase in the presence of large carp.

Hypothesis E4: Re-instating a wet/dry regime and excluding of large carp from Webster's Lagoon will result in a decrease in post flood pulse turbidity.

Turbidity is a measure of light attenuation in water and primarily reflects the amount of suspended particulate matter in the water column. It is a principal determinant of photic depth and, especially at high levels, has the potential to limit primary production and thus ecosystem structure and function. Turbidity is influenced by a number of factors such as the quality of source waters, the capacity for entrained particles to settle, and the susceptibility of the sediments to re-suspend. The susceptibility of sediments to re-suspension is influenced by sediment structure, water depth, and bioturbation, by carp for example. Numerous studies have attributed increases in turbidity to the benthic feeding behaviour of carp (Lamarra 1975, Meijer *et al*. 1990, King *et al.* 1997, Robertson *et al.* 1997), and this is likely also true for Webster's Lagoon. No environmental objective was developed for surface water turbidity (SKM 2004a). However, monitoring responses to management actions will assist with assessing changes in primary production and wetland trophic structure.

In its current permanently inundated and connected state, turbidity in Webster's Lagoon increases with distance from Toupnein Creek (MDFRC unpublished data). This gradient is suggestive of incomplete mixing of less turbid Toupnein Creek inflows within the lagoon. Increases in turbidity with distance into the wetland are likely functions of deceasing depth and hence the increased effects of wind mixing, and increased carp impact. It is anticipated that the imposition of a wet/dry cycle and the exclusion of large carp will have several consequences on wetland turbidity.

Firstly, the structure itself is likely to impede diluting exchanges between Toupnein Creek and the Lagoon, allowing turbidity to increase at sites closer to the regulator. Secondly, during the initial drying episode, carp densities within the receding pool (in the absence of significant avian predation) are likely to increase, increasing their impact on turbidity. Thirdly, the exposure of sediments as the wetland dries promotes their consolidation (Van der Wielen in press) and the establishment of terrestrial vegetation (Scholz and Gawne 2004a). These processes reduce the susceptibility of the sediments to re-suspension when inundated. As the efficacy of both processes in reducing post-inundation turbidity is related to the duration of exposure, it is likely that the influence of these will increase with distance from the regulator. Finally, the exclusion of large carp from Webster's Lagoon on its re-filling is likely to reduce a potentially significant source of disturbance to wetland sediments.

Nutrients (nitrogen and phosphorus)

Hypothesis E5: Re-instating a wet/dry regime to Webster's Lagoon will result in water column N and P concentrations increasing during the initial drying phase in the presence of large carp.

Hypothesis E6: Re-flooding of Webster's Lagoon after a drying event will stimulate the release of a short-lived large pulse of N and P.

Hypothesis E7: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in a decrease in post-flood pulse water column N and P concentrations.

The rehabilitation of nutrient cycling has been identified as a key ecological objective for Webster's Lagoon. This objective is to be met by re-instating a wet/dry regime and by excluding large adult carp from the wetland. However, no data is available to indicate the extent to which nutrient cycling within Webster's Lagoon has been impacted as a consequence of increased permanency of water. It is anticipated that the imposition of a wet/dry cycle and the exclusion of large carp will have several consequences on suspended nutrient concentrations.

Firstly, the installation of a regulating structure is likely to impede the nutrient exchanges between the wetland and Toupnein Creek and to eliminate such exchanges altogether when this structure is closed. No information is currently available to suggest either the magnitude or direction of these exchanges under the current operating scenario.

Secondly, as the wetland dries initially, nutrient concentrations are likely to increase to levels exceeding those currently encountered due to both evaporative concentration (*e.g.* Scholz *et al.* 2002, Scholz and Gawne 2004a) and turbation of the sediments. For example, as the wetland dries initially, carp densities within the receding pool are likely to increase (in the absence of significant avian predation), increasing the level of disturbance of the sediments, and thus the flux of nutrient-rich interstitial water and particle associated phosphorus from the sediments. Increases in fish densities will also increase the relative significance of excretion to the suspended nutrient pools (Lamarra 1975, Meijer *et al.* 1990, King *et al.* 1997, Robertson *et al.* 1997).

Thirdly, the exposure of wetland sediments during drying episodes is likley to stimulate the mineralisation of nutrients in the sediments (McComb and Qiu 1998, Baldwin and Mitchell 2000). Depending on sediment organic matter content and the duration of sediment exposure, a potentially significant pool of mineralised (or bio-available) nutrients may be flushed from the sediments on its subsequent inundation (Baldwin 1996, McComb and Qiu 1998, Baldwin and Mitchell 2000). Flooding is commonly associated with increased system productivity fuelled by an initial pulse of nutrients, derived from both the inflowing water and sediments releases (Junk *et al.* 1989, Scholz *et al.* 2002). This nutrient pulse is, however, short-lived (weeks) with water column nutrient concentrations declining steadily as nutrients become more tightly coupled with biotic and abiotic uptake and release processes (*e.g.* Baldwin and Mitchell 2000, Scholz *et al*. 2002). We therefore anticipate that flooding of Webster's Lagoon after a complete drying event will stimulate an initial pulse of increased water column N and P concentrations.

And fourthly, the exclusion of large carp following the re-flooding of the Lagoon will likely reduce the potential for disturbance of the sediments, reducing both turbidity and suspended nutrient concentrations.

Wetland vegetation

Aquatic vegetation

Hypothesis F1: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the aereal extent of submerged aquatic vegetation.

Hypothesis F2: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the distribution of submerged aquatic vegetation.

Macrophytes play an important role in aquatic ecosystems (refer to review by Carpenter and Lodge 1986). They may contribute to reducing turbidity, provide an important conduit for the transfer of oxygen to the sediments during plant growth, and nutrients to the water column during decay (Boon and Sorrell 1991), and provide substrata for the development of biofilms, which provide an important food resource for a wide range of invertebrates and fish (Cattaneo 1983, Bunn and Boon 1993). Macrophytes also provide important refuges for invertebrates and fish (Lillie and Budd 1992). The annual cycles of macrophyte growth and die-back may also have important consequences for water quality (dissolved oxygen, pH) and the accrual of organic matter on the sediments (Carpenter and Lodge 1986).

The distribution and abundance of aquatic vegetation is strongly influenced by hydrology (Welcomme 1979, Brock 1986). For example, longer periods of inundation and decreases in the variability of water levels, such as has occurred within Webster’s Lagoon, have been linked with reductions of both diversity and the width of the vegetated littoral zone (Brock and Casanova 1991, Poiani and Johnson 1993, Walker *et al*. 1994, Nielsen and Chick 1997, Blanch *et al.* 1999, 2000). Prolonging the period of inundation has also been shown to disadvantage the development of ephemeral or terrestrial plant taxa (Maher 1984, Briggs and Maher 1985). The establishment and persistence of aquatic vegetation may also be influenced by turbidity, which influences light availability, by the intensity of grazing pressure, and by the presence of carp through their destructive feeding habits (Hume *et al*. 1983, Roberts *et al*. 1995).

We anticipate that the re-instatement of the wet/dry cycle and the exclusion of large carp from Webster’s Lagoon will increase the capacity for submerged aquatic vegetation to establish, increasing their contribution to wetland function. However, we recognise that observed responses of the macrophyte community after implementing the proposed actions may be confounded by the diversity and availability of plant propagules (either already present within the wetland or colonisers arriving via water or aerial pathways), the timing of the proposed disturbance (seasonality of germination and establishment), and grazing interactions. These issues will not be examined directly as part of the current program.

Cumbungi

Hypothesis F3: Drying (exposing) cumbungi in Webster’s Lagoon between December and June will reduce stand vigour.

Cumbungi (*Typha domingensis*) is one of the most productive emergent native macrophytes, forming tall (2-4 m) dense mono-specific stands in water up to 2 metres deep. Cumbungi has the ability to expand by rhizome extension throughout the year, although most rapid rhizome extension typically occurs after mid-summer once the canopy has reached maximum biomass and flowering has occurred (Roberts and Ganf 1986). Cumbungi grows vigorously in hydraulically stable environments and as such has the potential to be used as an indicator of environmental (hydraulic) change (*e.g*. Roberts and Wylks 1992, Roberts and Marston 2000).

Cumbungi can tolerate exposure (dry conditions) only for short periods (3-4 months in summer-autumn) once the growing season is over without loss of vigour. However, its rhizomes can survive longer periods of exposure (possibly a few years) where soil moisture is sufficient to prevent desiccation. After flowering in spring-summer, germination and seedling establishment is dependent on the availability of water (depth < 5 cm) (Froend and McComb 1994, Roberts and Marston 2000). Because of these attributes, water management initiatives aimed at controlling/eliminating cumbungi are likely to be most effective where they induce stress during the most active period of growth, thereby reducing rhizome development and the establishment of shoots and flowers. This may be achieved by drying the wetland over the winter – mid-summer period. Whilst cumbungi may also be stressed by prolonged exposure to water depth >2 m, no opportunity exists within Webster's Lagoon to achieve such conditions given the maximum stage capacity of the proposed regulator (19.8 m AHD) (SKM 2004a). We therefore anticipate that the drying of Webster's Lagoon during the suggested period will reduce stand vigour, measured as the rate of stand expansion, shoot height and density and reproductive stem density. However, we note that using water management to control cumbungi is likely to require repeated drying episodes.

Red gums

Hypothesis F4: Drying Webster's Lagoon during spring-autumn will reduce mean health of red gum saplings present within the current wet/dry zone.

Hypothesis F5: Inundating the wet-dry zone of Webster's Lagoon during autumn-winter will reduce mean health of red gum saplings present within the current wet/dry zone.

Webster's Lagoon is fringed on higher areas by black box (*Eucalyptus largiflorens*) and on lower areas by red gum (*Eucalyptus camaldulensis*). Under the current hydrologic regime, soil moisture in the lower parts of the wet-dry littoral zone favours the recruitment of red gums into this zone. Whilst red gums play an important functional role within floodplain and wetland systems through their provision of carbon (leaf litter) and habitat for fauna (*e.g.* Briggs and Maher 1983, Briggs *et al.* 1997, Baldwin 1999), regeneration especially in the wet-dry littoral zone of Webster's Lagoon is considered a threat to the biodiversity and character of the wetland (SKM 2004a). Accordingly, reducing the abundance/extent of red gum regeneration within the wet-dry zone has been identified as a key environmental objective (SKM 2004a).

Conceptually, the recruitment of red gum seedlings may be controlled through hydraulic manipulation by subjecting seedlings to periods of flooding/immersion and/or desiccation. Seedling mortality may be achieved by complete submergence (carbon dioxide deprivation) or by being shallow flooded (oxygen deprivation of root zone). For seedlings that are 6-9 months old mortality may be achieved either by flooding to a depth of 50–100 cm for 4–6 weeks between June and October, or by shallow flooding for 12–26 weeks during autumn and winter (but not summer) (SKM 2004a). Management of lakebed red gums by desiccation may be achieved by drying the wetland for 6 months between October and June.

These recommendations are more likely to be effective in the control of recent recruits (< 1 year old) than of deeper rooted older red gums that are present within the wet/dry zone. Other methods of tree removal will need to be identified if no response in wet/dry zone red gums to hydraulic manipulation can be detected over successive wet/dry cycles.

Algae

Hypothesis G1: The re-flooding of Webster's Lagoon after a drying event will stimulate an initial short-lived pulse of increased algal biomass.

Hypothesis G2: Re-instating a wet/dry regime and excluding large carp from Webster's lagoon will result in an increase in the ratio of phytobenthic to phytoplanktonic biomass after the initial flood pulse.

Algae represent a key component of total wetland primary production (*e.g.* Wetzel 1964, Hargrave 1969, Wetzel *et al.* 1972). This is especially true in systems where macrophytes are absent or their distribution is restricted to a relatively small area, such as occurs in Webster's Lagoon. Algae occur either suspended within the water column (phytoplankton) or attached to substrata (phytobenthos). Numerous studies have shown production by either fraction to be influenced by light availability, nutrients and by substratum quality (*e.g.* Turner *et al.* 1983, Cox 1988, 1990a,b,c, Reynolds and Descy 1996, Havens *et al.* 1998), although taxon specific responses do vary greatly (Reynolds 1997, Huszar and Caraco 1998).

In its current permanently inundated state, the presence of carp has likley contributed to increasing turbidity and suspended nutrient concentrations (*cf.* King *et al.* 1997, Robertson *et al.* 1997). This is likely to directly influence the relative abundances of planktonic and benthic algae in several ways. Firstly, benthivory of carp increases the potential for benthic algae to become light limited by physically disturbing the sediments and reducing the penetration of light. And secondly, physical disturbance of the water column and sediments by carp maintains phytoplankton in suspension, increasing their retention within the photic zone, and increasing their access to light and nutrients. Accordingly, phytoplankton production is likely to be elevated and phytobenthic production inhibited (*cf.* Ogilvie and Mitchell 1998).

The reintroduction of wetland drying events and the exclusion of carp from Webster's Lagoon are likely to influence both phytoplankton and phytobenthic production in different ways throughout the wet/dry cycle. During the drying phase, increases in turbidity and nutrient concentrations (discussed earlier) are likely to favour phytoplankton production. This in conjunction with likely evaporation driven increases in phytoplankton density increases the potential for algal blooms to establish, especially during the warmer months when rates of production tend to be greatest (*e.g.* Scholz and Gawne 2004a). Sediment consolidation during the dry phase (Van der Wielen in press) in conjunction with the exclusion of carp on filling is likely to result in greater water column transparency and to increase sediment stability. Both of these are likely to favour the establishment and production of benthic biofilms. With the exception of an initial post-inundation pulse of nutrients from the sediments and concomitant pulse of phytoplankton production, much of the post-inundation release of nutrients from the sediments is likely to be assimilated by the benthic biofilms before it reaches the water column (*cf.* microbial loop; Boulton and Brock 1999).

Several studies have indicated that rates of benthic production can easily exceed that of the phytoplankton in shallow wetlands (Wetzel 1964, Wetzel *et al.* 1972, Stanley 1976, Loeb *et al.* 1983) and provide a major source of energy driving food webs (Bunn and Davies 1999). The anticipated shift in algal production from planktonic to benthic arising from the proposed management actions is also likely to influence the accessibility of food resources available to aquatic grazer populations, and thereby influence wetland trophic structure.

Invertebrates

Zooplankton

Hypothesis H1: The re-flooding of Webster's Lagoon will stimulate an initial short-lived pulse of increased zooplankton density lasting approximately 4 weeks.

Hypothesis H2: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the density of zooplankton after the initial flood pulse.

Hypothesis H3: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in a shift in zooplankton community composition after the initial flood pulse.

Zooplankton provide major links in aquatic food chains by facilitating the transfer of nutrients, carbon and energy between bacteria, algae and higher consumers, such as fish and water fowl (*e.g.* Boon and Shiel 1990, Boulton and Jenkins 1998, Humphries *et al*. 1999). Because of this, zooplankton plays a key role in structuring ecosystem function (*e.g.* Shapiro *et al*. 1975, Lazzaro 1997).

Zooplankton community structure is influenced by many factors, including life history characteristics, food availability, predation pressure, water quality, habitat diversity and complexity, and exchanges between wetland and riverine environments (Wiggins *et al.* 1980, Shiel and Walker 1984, Williams 1985, Shiel 1985, 1986, 1995). Many of these factors are influenced by season and in ephemeral systems by drying and flooding (*e.g*. Scholz and Gawne 2004a). Despite potentially complex interactions between these factors, generalised responses of zooplankton to the re-instatement of a wet/dry regime in Webster's Lagoon are to be expected.

We anticipate that the inundation of Webster's Lagoon after it has dried will stimulate an initial pulse of zooplankton density driven initially by floodwater borne immigrants which are subsequently replaced by emergents from the sediments during the first month after flooding (Maher and Carpenter 1984, Boulton and Lloyd 1992, Jenkins and Boulton 2003). Such pulses in post-inundation zooplankton density are thought to increase feeding opportunities and the recruitment success of native fish species and waterbirds (*e.g*. Cushing 1975, Maher 1984, Maher and Carpenter 1984, Crome 1986, Cushing 1990).

The duration of the initial post-flood pulse of zooplankton production has been shown to vary between wetlands, but generally decline over the first month of inundation as the intensity of trophic interactions increase (*e.g.* Carpenter *et al.* 2001, Scholz and Gawne 2004a,c). The magnitude of initial increases in zooplankton density after flooding has been linked to both the frequency of inundation and to the duration of the preceding dry period. For example, Boulton and Lloyd (1992) reported that wetlands that experience frequent (annual) episodes of flooding tend to be more productive than those that flood only infrequently (once in 22 years). Whilst longer dry phases tend to reduce the viability of aestivating individuals and of eggs and cysts deposited in the sediments, studies in dryland systems indicate that resting eggs are extremely long lived and eggs can survive dry periods of 20, 50 and even 100 years and still emerge once the sediments are flooded (Hairston *et al*. 1995, Jenkins and Briggs 1997, Jenkins and Boulton 1998).

Invertebrate productivity (Brinson *et al.* 1981, Maher and Carpenter 1984, Briggs and Maher 1985) and diversity (Boulton and Jenkins 1998) have been shown to suffer in response to reductions in the duration of wetland drying and to increases in the permanency of water, such as has occurred at Webster's Lagoon. This may be due, in part, to 'bottom-up' effects through reductions in nutrient availability and primary production, and to the loss of lake drying associated environmental cues needed by invertebrates to stimulate developmental shifts.

Based on these considerations, we hypothesized that re-imposing a wet/dry regime on Webster's Lagoon would stimulate an initial post-inundation pulse of increased zooplankton density, and greater post-flood-pulse zooplankton density relative to that of the current 'permanent system'. We also hypothesised that the re-imposition of a wet/dry cycle would mediate a shift in zooplankton communities towards taxa more typical of ephemeral water bodies. Such taxa tend to have more flexible life history strategies, increasing their resilience to disturbance events, such as drying (*e.g.* Baird *et al.* 1987).

Macro-invertebrates

Hypothesis H4: The re-flooding of Webster's Lagoon will stimulate an initial short-lived pulse of increased macro-invertebrate density lasting approximately 8 weeks.

Hypothesis H5: Re-instating a wet/dry regime and excluding adult carp from Webster's Lagoon will result in a longer term increase in the density of macro-invertebrates after the initial flood pulse.

Hypothesis H6: Re-instating a wet/dry regime and excluding adult carp from Webster's Lagoon will result in a shift in macro-invertebrate community composition after the initial flood pulse.

Macro-invertebrates are important consumers within wetland ecosystems, occupying a range of functional groups (*e.g.* grazers, detritivores, filter feeders, predators) and provide the principal food source for many vertebrates such as fish and birds (Bunn and Boon 1993). Macro-invertebrates have proven effective indicators of ecosystem health in stream environments (Cranston *et al.* 1996) and their use as indicators in wetlands has been advocated (*e.g.* Davis *et al.* 1993).

Macro-invertebrate community structure is influenced by many factors, including life history characteristics, water quality, habitat quality, and exchanges between wetland and riverine environments (Wiggins *et al.* 1980, Shiel and Walker 1984, Williams 1985) and by trophic interactions (*e.g.* Carpenter and Kitchell 1988, Carpenter *et al.* 2001). Numerous studies have shown hydrology to be a major determinant of macro-invertebrate community structure and productivity (*e.g.* Wiggins *et al.* 1980, Bataille and Baldassarre 1993, Jeffries 1994, Leslie *et al.* 1997). Although differences in the structure of macro-invertebrate communities of ephemeral and more permanent wetlands have been documented (*e.g.* Schalles and Shure 1989, Batzer and Resh 1992, Davis *et al.* 1993, Jeffries 1994), little consensus exists as to whether ephemeral systems are intrinsically more productive habitats for macro-invertebrates than more permanent systems (Batzer and Wissinger 1996). This lack of consensus is likely due to un-controlled-for differences in the strength of trophic interactions between the systems studied. Despite potentially complex interactions between determinants of macro-invertebrate community structure, generalised responses of macro-invertebrates to the re-instatement of a wet/dry regime in Webster's Lagoon and the exclusion of carp are to be expected.

We anticipate that the inundation of Webster's Lagoon after a drying event will stimulate an initial pulse of macro-invertebrate abundance driven by two processes; the rapid recolonisation of newly created habitat and favourable trophic interactions. Firstly, recolonisation may occur via several pathways, such as emergence from the sediments of desiccation resistant eggs and larvae or desiccation resistant adults, passive movement with the incoming waters, active migration, chance introduction by other animals, and aerial dispersal (Talling 1951, Wiggins *et al.* 1980, Batzer and Wissinger 1996, Hillman and Quinn 2002). And secondly, trophic cascade or food web models suggest that increases in the availability of food resources (*e.g.* zooplankton; Maher and Carpenter 1984, Boulton and Lloyd 1992) combined with reductions in predation pressure that are commonly encountered during the initial post-inundation period (*e.g.* Lake *et al.* 1989, Batzer and Wissinger 1996, Battle and Golladay 2001) will favour the establishment of abundant macro-invertebrate communities. The little data that is available suggests that initial post-flood increases in macro-invertebrate density may persist for as little as 1-2 months (*e.g.* Scholz and Gawne 2004c) or for as long as 2 years (*e.g.* Maher and Carpenter 1984). Subsequent declines in macro-invertebrate density are anticipated once the availability of food resources decreases and the intensity of biotic interactions (*e.g.* competition, predation) increases.

The presence of carp has been shown to exert a significant negative pressure on invertebrate abundances, either directly through predation or indirectly through habitat modification (*e.g.* Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al*. 1994, Tatrai *et al*. 1994). We anticipate that the removal of carp from the system will result in a longer term (*i.e.* after an initial post-inundation pulse) increase in macro-invertebrate abundances within the wetland and that this may also be reflected in community composition.

<u>Fish</u>

Hypothesis I1: The initial drying of Webster's Lagoon will result in an increase in the relative abundance of large carp.

Hypothesis I2: The initial drying of Webster's Lagoon will result in an increase in the relative biomass of large carp.

Hypothesis I3: The initial drying of Webster's Lagoon will result in a progressive reduction in the relative abundance of small fish populations.

Hypothesis I4: The initial drying of Webster's Lagoon will result in a progressive reduction in the relative biomass of small fish populations.

Hypothesis I5: The initial drying of Webster's Lagoon will result in a progressive reduction in the diversity of small fish populations.

Hypothesis I6: The operation of carp screens during re-filling will result in a decrease in the relative abundance of large carp in Webster's Lagoon.

Hypothesis I7: The operation of carp screens during re-filling will result in a decrease in the relative biomass of large carp in Webster's Lagoon.

Hypothesis I8: Inundating Webster's Lagoon after a drying event will stimulate spawning in small-bodied native fish species.

Hypothesis I9: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the relative abundance of small fish populations.

Hypothesis I10: Re-instating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the relative biomass of small fish populations.

Fish provide an important link in wetland food webs through their consumption of invertebrate and fish prey and their consumption by birds. Wetland hydrology plays an important role in structuring fish assemblages through its influence on ecosystem productivity, habitat availability and connectivity, and cues for fish spawning. Native fish throughout the Murray-Darling Basin have been severely impacted by altered flow regimes, the loss of habitat, and barriers to passage (MDBMC 2002). Altered flow regimes, in particular, are thought to have impacted on the distribution of native fish species and to have favoured the expansion of alien species, such as carp (*Cyprinus carpio*) and mosquitofish (*Gambusia holbrooki*) (Harris and Gehrke 1997). Whilst fish distributions within the Basin have been reported (Llewellyn 1983, Harris and Gerhke 1997), most fish work in the past has focussed on riverine populations. Very little information is currently available for wetland fish communities and their responses to wetting and drying (*e.g.* Scholz and Gawne 2004a,c).

The fish community of Webster's Lagoon is dominated by the exotic carp, goldfish (*Carassius auratus*) and mosquitofish (*Gambusia holbrooki*) and by natives species including bony herring (*Nematalosa erebi*) and flathead gudgeon (*Philypnodon grandiceps)* (Ho *et al*. 2004). Carp are widely considered responsible for the degradation of aquatic ecosystems. Although the evidence is fragmented and sometimes contradictory, carp have been implicated in the demise of native fish stocks by reducing spawning site availability (*i.e*. destruction of macrophytes; Hume *et al*. 1983, Brown 1996), by reducing the efficiency of visual feeding (*i.e*. increasing turbidity; Lamarra 1975, Roberts *et al.* 1995), and by competing for food resources (Hume *et al.* 1983, Fletcher 1986, Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al*. 1994, Tatrai *et al*. 1994).

The ecological objectives to 'rehabilitate habit and feeding opportunities' for the small native fish community at Webster's Lagoon and to 'reduce carp abundance' will be met by re-imposing a wet/dry cycle and by excluding large carp on re-filling (SKM 2004a). During the initial drying phase we anticipate that carp densities will increase (in the absence of avian predation) and that their adverse impacts on small fish will increase. In addition to establishing the efficacy of the regulating structure in excluding large carp, we will monitor fish populations directly as relative abundance rather than measuring habitat and feeding opportunities as stated in the objective. We anticipate the exclusion of adult carp from the wetland as it re-fills to have beneficial impacts on the abundance of smaller native fish populations for the reasons stated above. As large-bodied native fish do not use the Lagoon under current conditions (Ho *et al.* 2004) the benefits of excluding carp are likley to outweigh negative impacts of excluding large native fish. We also anticipate that flooding after a dry event will stimulate increases in the relative abundances of small fish populations due to factors including initial increases in post-flood food abundances, increases in habitat availability, and by cueing spawning (*e.g.* Meredith and McCasker in prep).

<u>Waterbirds</u>

Hypothesis J1: Reinstating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in the cumulative species richness of waterbirds.

Hypothesis J2: Reinstating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in post inundation abundances of generalist filter feeding and dabbling duck birds (all birds of the genus Anas, Pink eared-duck, Hardhead).

Hypothesis J3: Reinstating a wet/dry regime and excluding large carp from Webster's Lagoon will result in an increase in post inundation abundances of piscivorous birds that prefer smaller fish (Caspian Tern, White-faced heron, Great Egret, Little Egret).

Hypothesis J4: Bird assemblage composition of Webster's Lagoon will change over time, becoming increasingly dominated by piscivorous species after re-flooding.

Waterbird abundance is determined by the reproductive success of adults and the survival of post-juvenile birds. The breeding and survival of most waterbirds in the Murray-Darling Basin have been linked to cycles of flooding and drying (Briggs 1990); with most species being capable of moving large distances between wetlands as local breeding and feeding opportunities fluctuate. Almost all common waterbirds of the Murray-Darling Basin breed following wetland flooding, with some also breeding seasonally, usually in spring. In only 2 species (both ducks) is breeding always seasonal and apparently unaffected by wetland hydrology (Briggs and Lawler 1991).

Flow regulation within the Murray-Darling Basin has altered the temporal and spatial distribution of inundated floodplain wetlands available to waterbirds. Regulation has permanently inundated many formerly intermittent wetlands (*e.g.* Webster's Lagoon), and reduced the average duration of inundation for many others. Whilst this has increased the availability of survival habitat for waterbirds at the landscape scale, it has effectively reduced the availability of breeding habitat. Thus, the overall effect of river regulation on waterbirds is likely to be reduced recruitment of young, but enhanced survival of adults (Briggs and Lawler 1991). Because of this, it is important that the number of breeding opportunities for waterbirds within the Murray-Darling Basin be increased. As is discussed below, re-instating a wet/dry cycle to Webster's Lagoon is likely to achieve this objective.

The nesting requirements of Australian waterbirds vary (Serventy 1985), but all require adequate food to satisfy the energetic and other costs of nest finding or building, the laying and incubating of eggs, and the rearing of young. Whilst these costs may be partly provided for by the accumulation of body fat and/or protein reserves prior to laying (*e.g.* Thomas 1988, Norman and Hurley 1984, Briggs 1988), most waterbirds of the Murray-Darling Basin require readily available food of high quality at nesting time for successful breeding (Briggs and Lawler 1991). Many studies have shown the inundation of previously dry wetlands to initiate a highly productive succession of potential food resources (*e.g.* invertebrates and fish), which attracts many waterbird species and stimulates breeding (*e.g.* Maher 1984, Maher and Carpenter 1984, Crome 1986, 1988). Recently flooded wetlands thus provide a potentially important window of opportunity for breeding by many waterbird species. Over time after inundation and as wetlands begin to dry; changes in the composition and availability of food resources tend also to stimulate changes in the structure of waterbird assemblages (Scott 1997), increasing the cumulative diversity of species utilising the wetland.

In addition to food availability, post-inundation waterbird breeding success is reliant on the persistence of adequate food resources and of nesting habitat throughout the breeding (rearing) cycle. For example, many colonial water birds require inundation under mature red gum trees for between 3-6 months and up to 10 months for successful breeding (Briggs and Thornton 1999). This does not currently occur within Webster's Lagoon. Whilst re-introducing a wet/dry cycle may encourage other species of waterbirds to breed in the Lagoon, water will not be held high enough within the Lagoon to inundate the red gums, making local nesting by colonial waterbirds unlikely. To achieve this would require the construction of a much larger regulating structure capable of holding larger floods at higher elevations for longer (Zukowski and Meredith 2004).

For purposes of developing testable hypotheses relating to the responses of waterbirds to the re-imposition of a wet/dry regime in Webster's Lagoon, bird species were divided into guilds (groups of species that exploit similar resources in a similar manner, but that are not necessarily closely related taxonomically). Two guilds were identified; the generalist filter-feeders and dabbling ducks and the piscivores that prefer smaller fish.

We anticipate that the creation of additional nichès in response to the re-instatement of wet/dry cycles in Webster's Lagoon will stimulate an increase in the cumulative species richness of waterbirds utilising the wetland. Further, successional shifts in the availability and distribution of major food types and habitat throughout the wet/dry cycle will be reflected by shifts in the structure of waterbird assemblages. We anticipate that on re-flooding of the wetland, an initial burst of wetland productivity (0.5-2 months) will stimulate increases in the abundance of waterbirds, particularly the generalist filter-feeding and dabbling duck guild. This guild comprises all native species of Anas and pink-eared duck (*Malacorhynchus membranaceus*) and the hardhead (*Athya australis*) (*cf.* groupings of Roshier *et al*. 2002). We also anticipate a directional shift in waterbird assemblage composition over the 6-8 month of inundation, with the establishment of species that eat small fish as a major part of their diet, particularly Caspian tern (*Sterna caspia*), white-faced heron (*Egretta novaehollandiae*), great egret (*Egretta alba*), and perhaps little egret (*Egretta garzetta*). It is possible that abundances of these taxa will increase initially as wetland drying increases the density of small fish within the receding pool, and subsequently fall as fish numbers decline in response to elevated predation pressure and or changes in water quality. Wetland drying is also likely to increase feeding opportunities for scavenging land and waterbirds as fish become stranded. We also anticipate the development of a longer term shift in assemblage composition if the ‘structural biophysical’ nature of Webster’s Lagoon steadily changes, *e.g.* cover-dependent wetland-dependent species may colonise if dense reed beds and sedge beds develop. Species likely to establish as a consequence of such changes include; the clamorous reed warbler (*Acrocephalus stentoreus*), little grassbird (*Megalurus gramineus*), crakes and rails, and grazing waterfowl.

As wetland waterbird populations are highly mobile and are influenced by processes occurring at landscape scales, they are inherently very variable. Whilst we anticipate that we will be able to detect short-term responses in waterbird numbers between dry and flooded phases, the ability to reliably detect such responses in waterbird assemblages to changes in hydrology will likely require the collection of long term data extending over multiple wet/dry cycles.

Turtles

Hypothesis K1: Re-instating a wet/dry phase and the use of carp screens upon refilling will impede the recovery of aquatic turtles in Webster's Lagoon.

Three native species of freshwater turtle are known to occur within the Mallee region; the Broad-shelled river turtle (*Chelodina expansa*), the Eastern long-neck turtle (*Chelodina longicollis*), and the Murray turtle (*Emydura macquarii*) (NRE 2001).

Broad-shelled river turtles are considered threatened under the FFG Act and endangered under the ALTVFV (DSE 2003, 2004a), although this status may partly reflect the species’ secretive nature (Spencer 2001). Broad-shelled turtles are known to inhabit floodplain wetlands, but are principally inhabitants of river environments. This species is a selective predator which conceals itself in mud and uses its long neck to catch fish, yabbies and other prey (Scott 2001). Because of their threatened status there is a statutory requirement that the impact of changes in flow regime on the populations within Webster’s Lagoon be monitored.

The Eastern long-neck turtle is common and widespread both within the Mallee region (Ho *et al.* 2004) and throughout eastern Australia (Scott 2001). This species is thought to be better suited to wetland habitats than to river channel environments because it generally can not catch large prey such as fish and predominantly feeds on small invertebrates that are often more abundant in floodplain wetlands (Scott 2001).

The Murray turtle is thought to be an inhabitant primarily of river environments and is only sometimes present in deep permanent wetlands (Scott 2001). Of the three species, the Murray turtle has not been recorded in the vicinity of Webster’s Lagoon (Ho *et al.* 2004).

Freshwater turtles spend most of their time in the water, emerging only to bask, to lay their eggs in a chamber dug in dry ground away from water, or to move between wetlands (Boulton and Brock 1999). Movement between wetlands may be stimulated by competition with fish for food resources (Chessman 1984), or the loss of aquatic habitat as wetlands dry. The pulse of productivity generally associated with wetland re-flooding is thought to stimulate the dispersal of turtles back to the wetland (Boulton and Brock 1999). The re-establishment of turtle populations after a wetland drying event may occur via both overland and aquatic pathways. The relative importance of either pathway for either species has not been determined. We, therefore, decided to test whether the use of carp screens would inhibit the re-colonisation of Webster's Lagoon by Eastern long-neck and Broad-shelled turtles.

3.3 Response variables

Response variables to be measured as part of the monitoring program were identified for each of the hypotheses stated in the previous section. Standard sampling and analytical protocols for each response variable are listed in Table 3.2.

Response variable	Field and laboratory method
Surface water	
Height (mAHD)	Automated logging using Odyssey pressure sensors (Dataflow Systems Pty Ltd, Christchurch New Zealand).
Area and volume	Capacity tables constructed from profile surveys.
Groundwater	
Depth (m)	Measured from piezometers located along four transects within the lagoon: immediately downstream of the proposed regulator, immediately upstream of the proposed regulator, midway along the currently wetted area, and at the top of the currently wetted area. Piezometers to be spaced approximately 50 m apart, with transects extending from the centre of the lagoon into the large fringing red gums. Surveying of piezometers to mAHD will enable comparisons of water tables.
Electrical conductivity (EC; $\mu S\ cm^{-1}$)	EC (standardized to 25 ^oC) of water samples extracted from piezometers measured in the field using a U-10 multi-probe (HORIBA Ltd., Australia).
Sediments	
Electrical conductivity (EC; $\mu S\ cm^{-1}$)	EC (standardized to 25 ^oC) using a U-10 multi-probe (HORIBA Ltd., Australia) of soil samples collected at depth intervals of 20 cm (down to 1.0 m) and at intervals of 50 cm down to the water table. 10 replicate sediment cores to be taken within 50 m of the three transects within the lagoon.
Coarse particulate organic matter (CPOM; g CPOM l^{-1})	After processing and the removal of organisms from macro-invertebrate samples (20m sweep) the remaining organic matter sieved and the >1 mm size fraction dried at 80^oC for 48 hours and weighed.
Water quality	
Electrical conductivity (EC; $\mu S\ cm^{-1}$)	EC standardized to 25 ^oC determined *in situ* using a U-10 multi-probe (HORIBA Ltd., Australia).
Turbidity (NTU)	Turbidity of 500 ml sub-surface water samples determined using a U-10 multi-probe (HORIBA Ltd., Australia). Samples diluted 1:3 with distilled water where turbidity exceeds the max. level of detection of 1000 NTU.
Total nitrogen (TN; mg N l^{-1})	Unfiltered sub-surface 200 ml samples stored frozen. TN determined colorimetrically after pre-digestion in $NaOH\text{-}K_2S_2O_8$ and oxidation to nitrate (APHA 1995). Detection limit ± 0.019 mg N l^{-1}.
Total phosphorus (TP; mg P l^{-1})	Unfiltered sub-surface 200 ml samples stored frozen. TP determined colorimetrically using the phosphomolybdate-blue method after pre-digestion in $NaOH\text{-}K_2S_2O_8$ and oxidation to orthophosphate (APHA 1995). Detection limit ± 0.0025 mg P l^{-1}.
Oxides of nitrogen (NOx; mg N l^{-1}),	10 ml of 0.2 µm filtered water samples stored frozen. NOx determined colorimetrically after its reduction to nitrite using a cadmium column (APHA 1995). Detection limit ± 0.003 mg N l^{-1}.
Filterable reactive phosphorus (FRP; mg P l^{-1})	10 ml of 0.2 µm filtered water samples stored frozen. FRP determined colorimetrically using the phosphomolybdate-blue method (APHA 1995). Detection limit ± 0.001 mg P l^{-1}.
Wetland vegetation	
Aereal cover (ha) and distribution	Aereal extent and distribution of aquatic vegetation (*Valisneria* sp.), cumbungi (*Typha domingensis*) and red gum (*Eucalyptus camaldulensis*) determined from aerial photographs (0.2m pixel resolution) taken during the peak plant biomass period (November-January).
Vigour (cumbungi)	Vigour determined annually between November-January as rate of aereal expansion of stands, above-ground shoot height, shoot and reproductive stem density and the ratio of dead and growing shoots sampled from 4 x 1 m^{-2} quadtrats placed within the mature growth zone of each of three of the largest stands present. Quadrats placed *ca*. 3-4 m into the stand so as to avoid both the outer colonising edge and the inner-most core of the stand.

Table 3.2: Response variables, measurement units and analytical methods used to test hypotheses developed for Webster's Lagoon.

Response variable	Field and laboratory method
Wetland vegetation	
Recruitment and vigour (red gum)	Recruitment and vigour determined annually between November-January by recording the elevation (mAHD), leaf condition (colour) and tree height of each individual within 25 m wide quadrats extending vertically from the centre of the wetland up to the mature red gums fringing the wetland. Quadrats positioned horizontally in the centre of each stand of red gum saplings and quadrat positions fixed between years. This data will permit the identification of relationships between age cohorts and condition to the prevailing hydrological regime.
Algae	
Phytoplankton biomass (CHLp; μg CHL l^{-1}, CHL m^{-2})	Phytoplankton biomass measured indirectly as chlorophyll *a*. 500 ml of sub-surface water passed through glass fibre filters (Whatman Pty. Ltd. GF/C). 10 ml 100 % ethanol added to filters and chlorophyll extracted for 24 hours in the dark at 4^oC then at 80^oC for 10 minutes. Supernatant centrifuged and chlorophyll *a* measured spectro-photometrically at 665 nm and 750 nm without acidification (APHA 1995). Chlorophyll concentrations expressed per unit volume and, by recording water depth at time of sampling, per unit area.
Phytobenthos biomass (CHLb; μg CHL m^{-2})	Phytobenthic biomass measured indirectly as chlorophyll *a*. 100 ml 100 % ethanol added to 18 cm^{-2} sediment surface cores (1 cm depth) chlorophyll extracted for 24 hours in the dark at 4^oC then at 80^oC for 10 minutes. Supernatant centrifuged, and chlorophyll *a* measured spectro-photometrically at 665 nm and 750 nm without acidification (APHA 1995). Chlorophyll concentrations expressed per unit area.
Invertebrates	
Zooplankton density (individuals l^{-1}) and species richness	75 l water passed through 50µm mesh size net and concentrates preserved in 70 % ethanol. A minimum of 100 individuals counted from at least 4x1 ml sub-samples using a Sedgewick-Rafter Cell (APHA 1995). Identifications to at least genus according to Shiel (1995) and Ingram *et al*. (1997).
Macro-invertebrate density (individuals l^{-1}) and species richness	20 m benthic sweep samples collected using a 250 µm mesh size net and concentrates preserved in 70 % ethanol. A minimum of 100 individuals or whole sample counted and identified to genus level according to Williams (1980), Hawking and Smith (1997) and Anderson and Weir (2004). This is the rapid bio-assessment protocol of Tiller and Metzeling (1998) (see also Coysh *et al*. 2001). Compared with other commonly used active and passive sampling methods, sweep netting provides similar diversity though lower abundances (Humphries *et al*. 1998).
Fish	
Relative abundance (fish $CPUE^{-1}$) and biomass (kg $ha^{-1}CPUE^{-1}$)	For each sampling event 6 large fyke nets (LFN) (set tangential to shore with cod end away from shore), 6 small fyke nets (SFN) (set parallel to shore with pairs set in opposite directions) and 6 larval light traps (LT) set overnight. Set and pull times recorded and total catch adjusted to 24 hours (= 1 CPUE). Standard lengths (±0.05 mm for LFN and ±0.005 mm for SFN and LT) and weights (±0.5 g for LFN and ±0.0005 g for SFN and LT) recorded for the first 30 individuals (including the largest and smallest individuals caught) per species. Fish identifications follow McDowall (1996). Carp gudgeons identified to genus level only (*i.e*. *Hypseleotris* spp.) owing to the current taxonomic uncertainty at the species level (Bertozzi *et al*. 2000). All large native fish returned. Ethics approval obtained prior to sampling.
Waterbirds	
Abundance (birds $wetland^{-1}$)	Survey protocol to be determined. This will be consistent across Icon Site wetlands to permit comparisons at the regional scale.
Turtles	
Relative abundance, individual movements	Individuals caught using large fyke nets (deployed for fish) marked (using nail polish) before being released. Re-capture used to estimate population size identify individual returns to wetland after drying.

Table 3.2 cont.: Response variables, measurement units and analytical methods used to test hypotheses developed for Webster's Lagoon.

3.4 Experimental Design

3.4.1 Design options

Several experimental design options are available to test hypotheses based on the identification of changes in a response variable before and after an environmental disturbance such as wetland drying/flooding. These design options vary both in their complexity and in their ability to detect change. Generally, the incorporation of experimental controls or reference sites within an experimental design is favoured as it permits the elimination of potentially confounding artefacts introduced by the experimental procedure and provides the greatest statistical power to test hypotheses and to demonstrate cause and effect. Where control or reference sites can be identified, experimental designs incorporating Before, After, Control and Impact (BACI) data can and should be used (Underwood 1996). However, this is not always possible, requiring the implementation of alternative approaches that offer either a less scientifically rigorous means of demonstrating an ecological response, or that are capable of assessing the momentary status of key ecological response variables (*i.e.* management trigger levels). Alternatively, where the required level of scientific rigour can not be met it may prove more appropriate to do nothing.

Each of the ecological response hypotheses that were developed for Webster's Lagoon in Section 3.2 were assigned to one of four monitoring design categories based on considerations of the availability and/or requirement of a suitable control and the type of output required. These were:

i) Before-After Control-Impact (BACI) design. This approach represents the most scientifically credible monitoring option, and will enable the unambiguous determination of changes to the system that have occurred as a direct result of management intervention by comparing environmental variables in the 'Impact' system with that in an unmanaged 'Control' system 'Before' and 'After' the intervention. Note that because this approach requires a large number of measurements paired in time (also referred to as a BACIP design; Green 1997, Stewart-Oaten *et al.* 1986, Underwood 1991) across 2 (or more) locations, it represents the most expensive of the four options. Note also that if employed across multiple ecological response hypotheses within one system, this option has the advantage of being able to explain failures to detect expected ecological responses by considering conceptual models that link different trophic levels (*i.e.* trophic interactions).

ii) Intervention Analysis (IA) or Before-After (BA) design (*cf.* Box and Tiao 1975). This design option is used in cases where no control site is possible due to the uniqueness of the water regime, but where it is still necessary to monitor/demonstrate the ecological response within a system. Because no control is available, it will not be possible to demonstrate whether or not structural operation has 'caused' any observed 'effect' due to the fact that the data are 'pseudoreplicated', *i.e.* any change in the response variable may be due to any number of confounding effects (Hurlbert 1984, Underwood 1996).

iii) Instantaneous adaptive management (IAM) feedback. In cases where the ecological response hypothesis defines a key ecological outcome that is expected from the operation of structures on the floodplain, these ecological responses will be monitored and their real-time analysis will determine structure operation rules. In such cases, no control site is required (or in some cases possible), only the post-impact (After) period is monitored, and no establishment of scientifically credible cause and effect is necessary (or in some cases possible). However, in these cases a hierarchy of ecological outcomes is necessary to determine the sequence of management events.

iv) Do nothing. In some cases, the ecological response hypotheses for systems where no control is possible will be examined in other systems (*i.e.* other Living Murray Icon Sites). Rather than invest considerable resources into monitoring outcomes without being able to demonstrate cause and effect, it may be pertinent to transfer that effort to systems where causality can be determined.

Of the 40 ecological response hypotheses developed for Webster's Lagoon, 30 hypotheses require that cause and effect be demonstrated. These will be tested using a BACI experimental design. For 9 hypotheses (*i.e.* those relating to, lateral connectivity, groundwater/ salinity and wetland vegetation) no experimental control will be possible due to the uniqueness of responses to the wetland. These will be tested using Intervention Analysis (Table 3.3). Wetland flow/hydrology data will permit instantaneous adaptive management (*i.e.* compliance monitoring).

Our conceptual model (Section 3.2) indicates that the ecological response variables identified for each hypothesis (Section 3.3) are likely to vary considerably throughout the wet/dry cycle (Figure 3.5). In most cases, testing each hypothesis using either BACI or IA experimental designs will involve the comparison of variable responses between specific phases of the wet/dry cycle. The hydrological phases relevant to each hypothesis are presented in Table 3.3. Other design considerations such as the number of suitable control wetlands that can be located for BACI designs and the optimisation of temporal and spatial sampling regimes employed (for both BACI and AI designs) have the capacity to greatly effect the ability to detect treatment effects (*i.e.* statistical power). These issues are considered in the following sections.

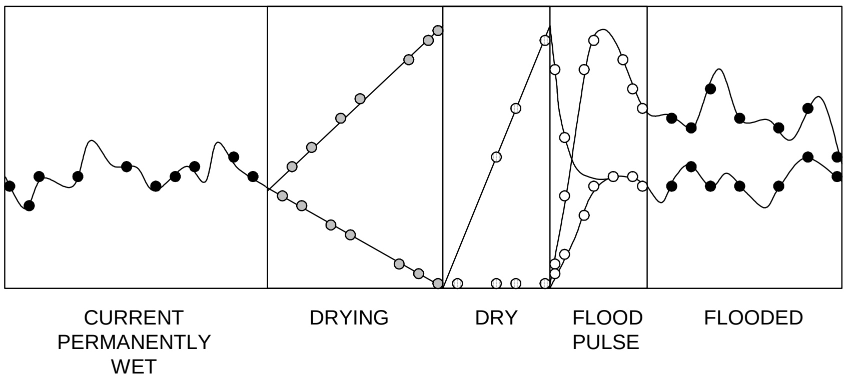


Figure 3.5: Phases of the wet/dry cycle showing conceptualised responses of ecological response variables.

Environmental value	Hypotheses	Establish cause and effect (BACI)	Intervention Analysis (Before-After)	Instantaneous adaptive management	Do nothing	Hydrological phases examined	Statistical analysis
Flow/hydrology	A1	×	×	✓	×	All	No statistics
Lateral connectivity	B1	×	✓	×	×	All	No statistics
Groundwater/salinity	C1	×	✓	×	×	All	Repeated measures ANOVA
	C2	×	✓	×	×	All	Repeated measures ANOVA
	C3	×	✓	×	×	Dry	Repeated measures ANOVA
Sediments	D1	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
Water quality (electrical conductivity)	E1	✓	×	×	×	All	2-factor fixed-model ANOVA
	E2	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
Water quality (turbidity)	E3	✓	×	×	×	Drying	Repeated measures ANOVA
	E4	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
Water quality (nutrients)	E5	✓	×	×	×	Drying	Repeated measures ANOVA
	E6	✓	×	×	×	Flood pulse	Repeated measures ANOVA
	E7	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
Wetland vegetation (aquatic vegetation)	F1	×	✓	×	×	Annual assessment when wet	No statistics (no measure of variance possible)
	F2	×	✓	×	×	Annual assessment when wet	No statistics (no measure of variance possible)
Wetland vegetation (cumbungi)	F3	×	✓	×	×	Annual assessment	1-factor fixed-model ANOVA
Wetland vegetation (red gum)	F4	×	✓	×	×	Annual assessment	2-factor non-parametric ANOVA (rank data) ?
	F5	×	✓	×	×	Annual assessment	2-factor non-parametric ANOVA (rank data) ?
Algae	G1	✓	×	×	×	Flood pulse	Repeated measures ANOVA
	G2	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
Invertebrates (zooplankton)	H1	✓	×	×	×	Flood pulse	Repeated measures ANOVA
	H2	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
	H3	✓	×	×	×	Permanently flooded v flooded	Multivariate analysis (*e.g.* MDS) ?
Invertebrates (macro-invertebrates)	H4	✓	×	×	×	Flood pulse	Repeated measures ANOVA
	H5	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
	H6	✓	×	×	×	Permanently flooded v flooded	Multivariate analysis (*e.g.* MDS) ?
Fish	I1	✓	×	×	×	Drying	Repeated measures ANOVA
	I2	✓	×	×	×	Drying	Repeated measures ANOVA
	I3	✓	×	×	×	Drying	Repeated measures ANOVA
	I4	✓	×	×	×	Drying	Repeated measures ANOVA
	I5	✓	×	×	×	Drying	Repeated measures ANOVA
	I6	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
	I7	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
	I8	✓	×	×	×	Flood pulse	Repeated measures ANOVA
	I9	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
	I10	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA
Waterbirds	J1	✓	×	×	×	All	No statistics (no measure of variance possible)
	J2	✓	×	×	×	Permanently flooded v flood pulse and flooded	2-factor fixed-model ANOVA
	J3	✓	×	×	×	Permanently flooded v flood pulse and flooded	2-factor fixed-model ANOVA
	J4	✓	×	×	×	Flood pulse	Multivariate analysis (*e.g.* MDS) ?
Turtles	K1	✓	×	×	×	Permanently flooded v flooded	2-factor fixed-model ANOVA

Table 3.3: Summary of monitoring approach to be used to test each of the ecological response hypotheses developed for Webster's Lagoon, the hydrological phases for which data is required, and the statistical treatment of data.

3.4.2 Control selection

3.4.2.1 Introduction

Several variants of the BACI experimental design are available depending on the number of available Impact and Control locations (*e.g.* Underwood 1996, Downes *et al.* 2002). These design variants differ in their complexity and in their ability to detect change. Generally, maintaining an equal number of Control and Impact sites (a balanced or symmetrical BACI design) is desirable for analytical and inferential convenience (Underwood 1993). As only one Impact site (Webster's Lagoon) was available, maintaining a balanced design would eliminate replication at the experimental treatment level. Whilst temporal replication of sampling within such a design makes it possible to control for some random elements of differences between the two locations, there is no reason to expect a response variable to have the same time-course changes at both sites. Increasing the number of control sites used overcomes such short comings, resulting in an unbalanced or asymmetrical BACI design. The detection of an impact using such designs is, however, more complex because it may show up in very different ways depending on the spatial and temporal consistency of the univariate response variables being measured (Underwood 1993, 1996, Quinn and Keough 2002).

Because average values of ecological response variables likely differ between Control and Impact locations, the BACIP design focuses on any changes at the Impact location relative to the Control, and the variable that is analysed is the difference between Control and Impact values. Sampling through time is used to estimate the variation in these differences, and this variation is used to assess the average difference Before and After the impact event commences (Downes *et al.* 2002). Treament effects are thus shown by a significant statistical interaction between the two sources of variation (Control *vs* Impact and Before *vs* After) determined using two-factor fixed model (sites and times fixed) ANOVAs and repeated measures ANOVAs (RMAs) (refer to Table 3.3). However, for a treatment effect to be detected it is important that Impact and Control locations (wetlands) 'track' each other well through time in the absence of impact. This second criteria cannot be established *a priori*.

Based on these considerations, we were faced with either selecting a single Control wetland that was most likely to 'track' Webster's Lagoon in the absence of impact or selecting multiple Control wetlands that whilst increasing scientific rigour also increased substantially monitoring costs and complexity of data interpretation. Ultimately this decision would be influenced by the number wetlands that could be located that satisfied key hydrological, physical and ecological criteria.

3.4.2.2 Methods

A desk-top survey of wetlands within the Mallee Catchment Management region was undertaken to identify potential candidates to serve as controls for Webster's Lagoon. Wetlands were screened using the selection criteria shown in Table 3.4. Only two wetlands satisfied all physical and ecological requirements; the Lock 7 wetland and Wetland 491 (Figure 3.6). Pilot surveys of these two wetlands and Webster's Lagoon were undertaken between the 10^{th} and 19^{th} January 2005 to ground truth desk top survey data and to provide a 'snap-shot' of wetland ecological condition.

For each wetland, standard field and laboratory methods were used to measure and collect samples from 3 sites for each of the following response variables: CPOM, EC, turbidity, TN, NOx, TP, FRP, CHLp, CHLb, invertebrate abundances, and fish and turtle community structure (Table 3.2). Data are shown as means ± SE for all response variables except for fish and turtles, which are shown as CPUE (individuals 24 net hours^{-1}). No statistical comparisons between wetlands were made.

Control selection criteria	
Location	• Wetland associated with the Murray River within the Mallee Region. • Distance from Webster's Lagoon (ideally < 20 km).
Geomorphology	• Wetland formed as a cut-off meander of the main channel of the Murray River.
Connectivity	• Wetland has a single primary connection with the main channel of the Murray River (*i.e.* does not flow).
Hydrology	• Wetland naturally ephemeral, but currently maintains a permanent connection with the main channel of the Murray River.
Dimensions	• Wetland similar to Webster's Lagoon in area, length, width and depth.
Biota	• Carp (*Cyprinus carpio*) present. • Cumbungi (*Typha domingensis*) present within the wet/dry zone. • Red gums (*Eucalyptus camaldulensis*) present within the wet/dry zone. • Submerged macrophytes present.

Table 3.4: Criteria used to select suitable control wetlands for Webster's Lagoon.

3.4.2.3 Results

Data for each of the key selection criteria for each of the wetlands examined are shown in Table 3.5. Of the two candidate wetlands, Wetland 491 was much further from Webster's Lagoon than was the Lock 7 wetland. Both candidate wetlands were much smaller than Webster's Lagoon. Although shorter in length, the Lock 7 wetland was more similar morphologically to Webster's Lagoon, having a similar mean width and littoral profile, than was Wetland 491. Wetland 491 was slightly narrower and had much steeper littoral gradients than the other 2 wetlands examined, and hence had a much narrower wet/dry zone. Each wetland was fringed by mature red gums at higher elevations. Because of its steeper bank profile, these trees were much closer to the water's edge at Wetland 491 and for much of its perimeter the canopy of these extended over the water. These trees contribute significant amounts of shade and leaf litter/woody debris directly into the water at current pool levels (O. Scholz pers. obs.). However, this was not supported by the CPOM data collected during the survey (Table 3.6). At Webster's Lagoon and the Lock 7 wetland, much of this leaf litter is depostited at elevations above the current pool level and thus not directly available for aquatic processing. Differences in the littoral profile between wetlands also likely accounted for the absence of red gum regeneration within the wet/dry zone of Wetland 491.

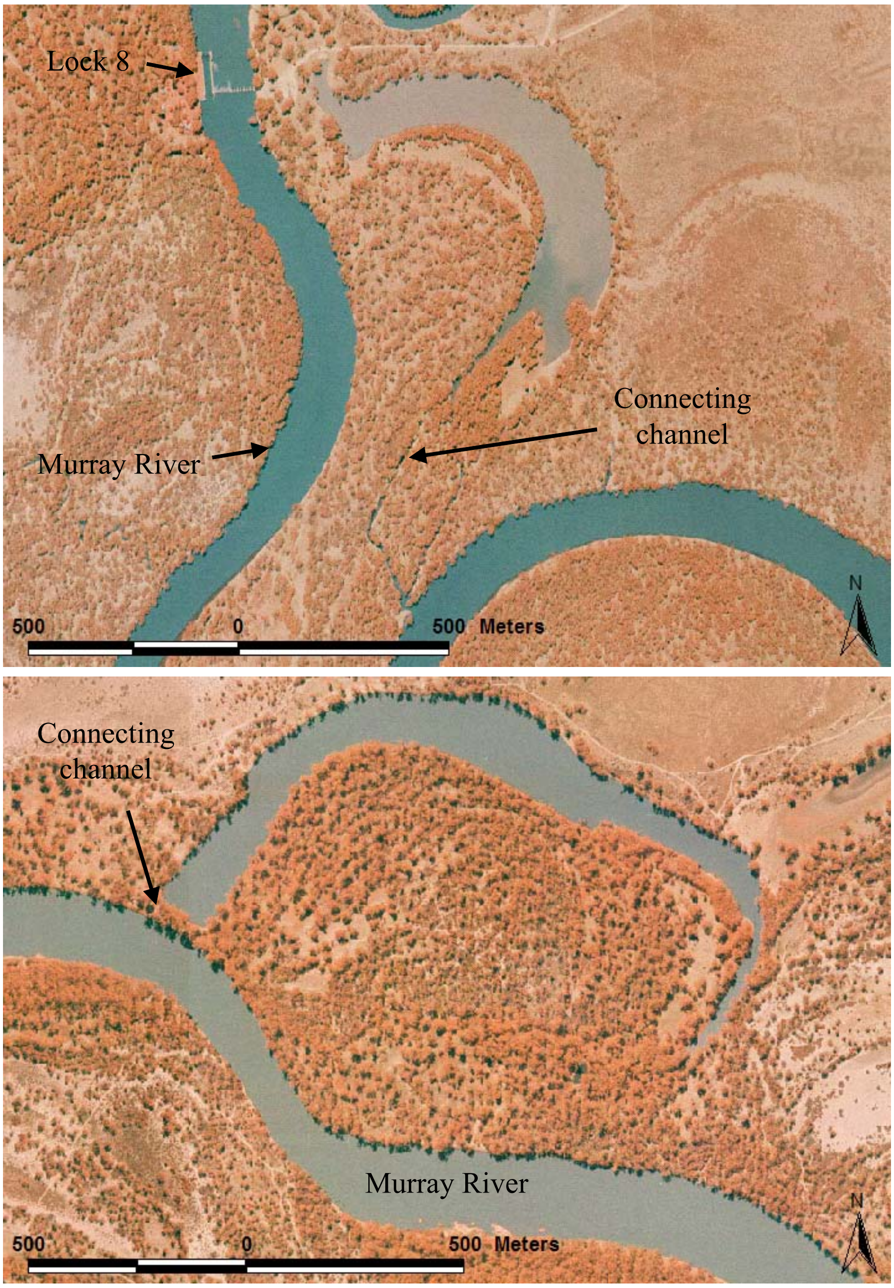


Figure 3.6: Lock 7 wetland (top) and Wetland 491 (bottom) surveyed for their potential to function as experimental controls for the Webster's Lagoon monitoring program (source MDBC River Murray Mapping 2nd edn. May 1996).

Selection criteria		Wetlands		
		Webster's Lagoon	Lock 7 wetland	Wetland 491
Location	Within the Mallee region	✓	✓	✓
	Distance from Webster's Lagoon (km)	0	19	60
Geomorphology	Cut-off meander	✓	✓	✓
Connectivity	Connection with Murray River	2.1 km via Toupnein Ck, upstream of Lock 6	200 m via feeder creek upstream of Lock 7	50 m through levee upstream of Lock 9
Hydrology	Naturally ephemeral	✓	✓	✓
	Currently permanent	✓	✓	✓
Dimensions	Area (ha)	40.0	13.8	19.1
	Length (km)	3.25	1.06	2.03
	Width (mean±sd) (m)	139.8 ± 10.6	132.6 ± 19.9	108.2 ± 32.3
	Depth (max.) (m)	2.2	1.5	1.3
Fauna	Carp	✓	✓	✓
Flora	Cumbungi	✓	✓	✓
	Red gums (regeneration)	✓	✓	×
	Submerged macrophytes	✓	✓	✓

Table 3.5: Comparison of selection criteria for Webster's Lagoon and the two candidate control wetlands; Lock 7 wetland and Wetland 491.

Summary statistics for the ecological response variables measured at each wetland are shown in Table 3.6. Although these data were obtained from the single survey and provide no indication of temporal variability, several important differences between the wetlands were apparent. Wetland 491 was less saline, was much less turbid, and had much lower suspended nutrient concentrations than either Webster's Lagoon or the Lock 7 wetland. As a likely consequence of being less turbid, submerged macrophyte beds within the littoral zone of Wetland 491 were much more developed (pers. obs. O. Scholz). Although these differences were not reflected in the algal and invertebrate trophic levels, they were in the structure of fish assemblages. Carp and bony herring were present in large fyke net (LFN) catches from each wetland. However, golden perch (a top predator) and rainbowfish were recorded only in Wetland 491. Although each wetland supported a similar suite of small bodied fish, much greater small fyke net (SFN) catches were recorded in Wetland 491 than were in either of the other wetlands. The two species of turtle previously recorded in Webster's Lagoon (Ho *et al.* 2004) were again recorded in this survey and were also present in both candidate wetlands.

3.4.2.4 Discussion/Conclusion

Our objective was to identify whether we would use one or two wetlands as controls in the experimental design for monitoring ecological response hypotheses at Webster's Lagoon. By conducting a desk top survey we were able to select two wetlands that satisfied most, if not all, key selection criteria; the Lock 7 wetland and Wetland 491. The analysis of field survey data collected from each wetland indicated that we should discount using Wetland 491 based on what we regarded as significant differences in wetland character and function between it and Webster's Lagoon. By comparison, the risks associated with the need for Impact and Control locations to 'track' each other well through time in the absence of impact were considered far less by selecting the Lock 7 wetland to serve as our experimental control. The availability of only a single experimental control (Lock 7 wetland) thus dictated the use of a balanced unreplicated BACI experimental design, the cheaper and more easily interpretable, though less rigorous of the available design options. Whilst not ideal, the single control BACI approach was our only option and has been widely used with some success in the past, for example, in whole-lake biomanipulations (*e.g.* Carpenter and Kitchell 1988, Carpenter *et al.* 2001).

Response variable		Wetlands		
		Webster's Lagoon	Lock 7 wetland	Wetland 491
Sediments				
CPOM (g CPOM m^{-2})		0.331 ± 0.089	2.129 ± 0.541	0.476 ± 0.158
Water quality				
EC (µS cm^{-1})		351 ± 50.8	545 ± 44.6	241 ± 5.15
Turbidity (NTU)		628 ± 222	200 ± 14.6	34 ± 1.6
TN (µg N l^{-1})		2069 ± 506	2974 ± 166	1183 ± 26
TP (µg P l^{-1})		335 ± 76.2	307 ±15.8	73.2 ± 2.10
NOx (µg N l^{-1})		25.6 ± 2.75	9.56 ± 2.04	4.44 ± 0.18
FRP (µg P l^{-1})		18.0 ± 2.68	26.3 ± 5.26	7.99 ± 0.17
Algae				
CHLp (µg CHL l^{-1})		46.3 ± 8.8	63.7 ± 2.9	34.7 ± 2.2
CHLb (µg CHL m^{-2})		12.0 ± 2.8	8.0 ± 0.5	10.1 ± 1.3
Invertebrates				
Zooplankton (individuals l^{-1})		954 ± 191	1170 ± 308	955 ± 124
Macro-invertebrates (individuals l^{-1})		7.52 ± 1.11	23.94 ± 4.04	15.99 ± 2.93
Fish				
Fish LFN (CPUE)		7.67	0.67	0.68
% of catch	Carp	67.4	75.0	25.9
% of catch	Bony herring	32.6	25.0	49.2
% of catch	Golden perch	0.0	0.0	24.9
Fish SFN (CPUE)		49.33	40.82	205.78
% of catch	Flatheaded gudgeon	62.2	0.8	5.2
% of catch	Bony bream	30.1	0.0	6.1
% of catch	Australian smelt	2.7	0.4	1.0
% of catch	Carp	1.7	0.0	0.0
% of catch	Carp gudgeon	1.4	7.8	73.6
% of catch	Gambusia	1.0	89.4	0.4
% of catch	Hardyhead	1.0	1.6	13.2
% of catch	Rainbowfish	0.0	0.0	0.4
Turtles				
Turtle (CPUE)		1.67	8.31	3.05
% of catch	Eastern long-neck	90.0	96.1	66.1
% of catch	Broad-shelled	10.0	3.9	33.9

Table 3.6: Ranges of water quality, algal and invertebrate response variables and catches of fish and turtles recorded in Webster's Lagoon, Lock 7 wetland and Wetland 491.

3.4.3 Temporal sampling

With Before-After type experimental designs (*e.g.* BACI), statistical power is influenced by both temporal and spatial variation. Power may be improved by increasing the number of samples taken per sampling event, by increasing the sampling period, and by increasing the number of sampling events.

For hypotheses in which Before and After time periods are to be compared, each sampling event provides only one value. As sub-samples collected from the wetland within sampling events do not add replicates to test the hypothesis about treatment effects, their impact on statistical power is likely to be small, especially where spatial variability is less than temporal variability. For example, if there is considerable spatial variation within each location, and only a few samples are taken at each time the variation in Control-Impact differences may reflect this small-scale spatial variation, rather than the temporal variation that is of interest. Thus, increasing the number of sub-samples taken within sampling events reduces the spatial variance (*i.e.* increases sampling precision) of estimated response variable means, and the data begins to reflect primarily temporal variation.

Increasing the sampling period is generally considered more important than is increasing the frequency of sampling (*e.g.* Downes *et al.* 2002). As no estimates of either temporal or spatial variation of any of the response variables were available for Lake Wallawalla, it was not possible to estimate the minimum sampling period or minimum number of sampling events required to reliably detect meaningful treatment effects (*i.e.* minimum detectable treatment effect size between Before and After time periods; *cf* Zar 1984). We anticipate that a full annual cycle (preferably 2 annual cycles) of Before data will be required to detect intervention responses with sufficient statistical power.

The choice of when to sample is often arbitrary, and is commonly set by previous practise (*e.g.* monthly intervals), logistic constraints (minimum time between sampling events to allow for sample processing) or some sort of conceptual model ('seasonal variation is important'). However many times the univariate response variable is to be sampled and at whatever interval, the times chosen should be random, not regular, and all locations (impact and controls) should be sampled at the same time (Green 1997, Stewart-Oaten *et al.* 1986, Underwood 1991). Random (temporal) sampling reduces the potential for cyclic processes (*e.g.* lunar, season) to influence the magnitude of differences observed before and after a disturbance. Underwood (1991) suggests that the best design incorporates sampling at several temporal scales. This is especially important where the temporal scale of response is unknown. For example, where there is no replication of sampling within seasons, comparisons among seasons are confounded with shorter-term changes. This may be overcome by having several independent (non serially-correlated) randomly chosen sampling events in each season (Underwood 1993).

The most appropriate timing and frequency of sampling should be determined by the scales over which response processes and individual organisms operate (Underwood 1996). These are likely to vary considerably between response variables. Identification of the most appropriate sampling frequency for each response variables is complicated by the requirement that data be independent (*i.e.* not serially-correlated). The independence of data is a major assumption of most statistical procedures and is often overlooked (Underwood 1996). Data will not be independent where samples are taken at such short intervals that the same individuals or cohort of individuals are repeatedly counted. The use of repeated-measures ANOVA is favoured where lack of independence between sample events is likely to be an issue, such as with the identification of post-flood responses in the current program (Underwood 1993).

For the current monitoring program, consideration of the time required for sample processing limited the number of sampling events possible per year to eight. Whilst the identification of seasonal variation was not to be tested explicitly, equal sampling effort (n=2) will be allocated to each season. Within seasons, the timing of sampling is to be determined on a 'random' basis, although true 'randomness' will need to be balanced against the requirement of independence. Each wetland (Control and Impact) is to be sampled at the same time over a 4 day period. Whilst sampling at this frequency will be appropriate for most ecological response variables, others will require more frequent monitoring (*e.g.* wetland water levels – daily) or less frequent monitoring (*e.g.* groundwater – 3 monthly, vegetation – annually, soil salinity – coinciding with the end of the dry phase). For hypotheses relating to post-flood responses, sampling frequency will be increased over the first 8 weeks so as to increase temporal resolution and power for repeated measures analyses. Sampling during these events will occur weekly for the first month, then every two weeks for the second month. Sampling effort at these times will be in addition to the longer-term monitoring framework.

3.4.4 Spatial sampling

3.4.4.1 Introduction

As described above, statistical power in Before-After experimental designs is influenced by both temporal and spatial replication. The ability to detect temporal variation (Before *vs* After) is reduced where spatial variation within the wetland is large relative to temporal variation. Observed spatial variation is a function of sampling precision, which is influenced by both the size and number of samples taken at any one time to obtain an estimate of the variable mean. Methods used to assess sampling precision vary widely. One of the more commonly used statistics for determining precision is the ratio SE/mean (Andrew and Mapstone 1987). Desired precision is arbitrarily set *a priori*, with target values generally ≤ 0.25 (Elliott 1979, Andrew and Mapstone 1987).

Our objective was to test whether the standard sampling program (5 replicates x 3 sites) was capable of achieving such levels of precision for each of the response variables at site and wetland scales. Consideration of the time required for invertebrate sample processing necessarily limited the number of replicates collected per site and sampling event to 5. We anticipate that temporal variation will be greater than ±25 % of the mean. Achieving a sampling precision ≤ 0.25 this will increase our ability to optimise the allocation of sampling effort within sampling events and to detect changes in temporal variation at both site and wetland scales. However, these will need to be validated once the monitoring program commences.

3.4.4.2 Methods

Two pilot surveys of Webster's Lagoon were undertaken to assess sampling precision for each of the response variables. On the 14th December 2004 we collected samples and measured electrical conductivity (EC), turbidity (NTU), total nitrogen (TN) and oxides of nitrogen (NOx) concentrations, total phosphorus (TP) and filterable reactive phosphorus (FRP) concentrations. On the 10th January 2005 we collected samples for benthic coarse particulate organic matter (CPOM) dry weight, phytoplankton chlorophyll *a* (CHLp) and phytobenthic chlorophyll *a* (CHLb) concentrations, and zooplankton and macro-invertebrate densities and species richness.

Standard field and laboratory methods used for each response variable are described in Table 3.2. For each response variable 5 replicate samples were collected from 3 open water sites (Figure 3.7). Additional zooplankton and macro-invertebrates samples were collected from amongst macrophyte beds present within the wetland to examine whether stratification of sampling at this level was necessary. Macrophyte beds comprised *Vallisneria* sp. and *Ludwegia peploides*, whose distribution was restricted to Site A. Other species of aquatic vegetation present within the wetland were not sampled as they were either exposed at the time of sampling (*Eleocharis* sp., *Juncus* sp.; present at each site), or were too dense (*Typha domingensis*; Site C only).

Sampling precision at site and wetland scales was calculated as SE/mean for two levels of sampling effort; 3 replicates and 5 replicates, in order to assess the most appropriate level of replication for each response variable. Calculations based on 3 replicates used the minimum, median and maximum values obtained for the 5 replicates. Relationships between cumulative species richness and number of replicates were examined for zooplankton and macro-invertebrates to assess the likelihood of missing taxa at the proposed levels of sampling. Differences between sample means were examined using *t*-tests and fixed factor one-way ANOVAs with Tukey's *post hoc* testing at $\alpha = 0.05$. Data did not require transformation to satisfy assumptions of normality. All analyses were done using SYSTAT® V10.2 (SPSS Inc., Chicago USA).

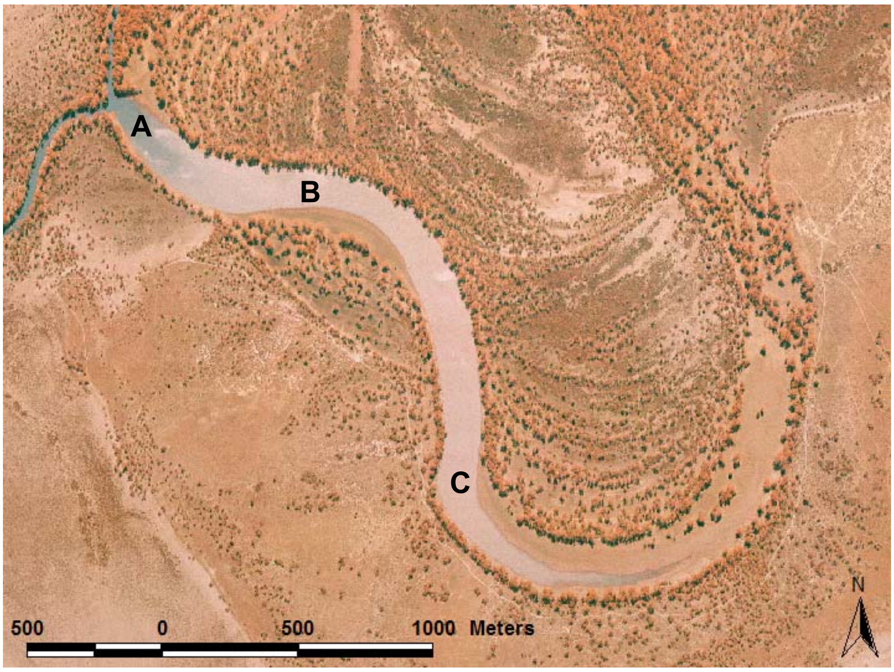


Figure 3.7. Webster's Lagoon showing the location of the three sampling sites (source MDBC River Murray Mapping 2nd edn. May 1996).

3.4.4.3 Results

Summary plots of mean ± SE (for n=5) at site and wetland scales for each response variable examined are shown in Figure 3.8.

Benthic coarse particulate organic matter (CPOM) ranged from 0.020-1.067 g CPOM m^{-2} and did not differ significantly between sites (One-way ANOVA $p = 0.149$).

Electrical conductivity (EC) ranged from 201-450 µS cm^{-1} and increased significantly with distance from Toupnein Creek (One-way ANOVA $p < 0.001$, Tukey *post-hoc* test A<B<C). Turbidity (NTU) ranged from 315-2128 NTU and increased significantly with distance from Toupnein Creek (One-way ANOVA $p < 0.001$, Tukey *post-hoc* test A<B<C).

Total nitrogen (TN) and total phosphorus (TP) concentrations ranged from 355-4640 µg N l^{-1} and 56-640 µg P l^{-1}, respectively. Oxides of nitrogen (NOx) and filterable reactive phosphorus (FRP) concentrations ranged from 14-39 µg N l^{-1} and 9-29 µg P l^{-1}, respectively. Concentrations of TN, TP and FRP increased significantly with distance from Toupnein Creek (all One-way ANOVAs $p < 0.001$, Tukey *post-hoc* test A<B<C). NOx concentrations did not differ significantly between sites (One-way ANOVA $p = 0.209$).

Phytoplankton chlorophyll *a* (CHLp) concentrations ranged from 6.52-59.86 µg CHL l^{-1} and increased significantly with distance from Toupnein Creek (One-way ANOVA $p < 0.001$, Tukey *post-hoc* test A<B<C). Phytobenthic chlorophyll *a* (CHLb) concentrations ranged from 0.60-22.67 µg CHL cm^{-2} and were significantly greater at Site A than those measured at sites B and C (One-way ANOVA $p < 0.001$, Tukey *post-hoc* test A>B=C).

Total open water zooplankton densities ranged from 30.0-2320.0 individuals l^{-1} and increased significantly with distance from the Toupnein Creek (One-way ANOVA $p < 0.001$, Tukey *post-hoc* test A<B<C) (Figure 3.8). Although both rotifer and micro-invertebrate abundances increased with distance from Toupnein Creek (One-way $ANOVA_{rotifers}$ $p < 0.001$, Tukey *post-hoc* test A<B=C, One-way $ANOVA_{micro-invertebrates}$ $p < 0.001$, Tukey *post-hoc* test A<B<C), rotifers became numerically less dominant (Site A 92.3 %, Site B 62.7 %, Site C 60.1 %).

Forty six zooplankton taxa were encountered in samples taken from Webster's Lagoon. These included 31 rotifer and 15 micro-invertebrate taxa. Rotifer taxon richness decreased and micro-invertebrate taxon richness increased with distance from Toupnein Creek (One-way $ANOVA_{rotifers}$ $p = 0.004$, Tukey *post-hoc* test A=B, B=C, A>C, One-way $ANOVA_{micro-invertebrates}$ $p < 0.001$, Tukey *post-hoc* test A<B=C). Plots of cumulative species richness *vs.* sampling effort for each zooplankton group indicated that few additional taxa were likely to be encountered beyond the 5^{th} replicate within each site (Figure 3.9).

Total densities of zooplankton sampled from amongst *Vallisneria* sp./*Ludwegia peploides* beds present at Site A ranged from 45.0-233.3 individuals l^{-1} and did not differ significantly from that of samples collected from the adjacent open water (*t*-test; $t = -0.574$, $p = 0.597$). Whilst rotifer densities did not differ significantly between adjacent habitats at site A (*t*-test; $t = 0.723$, $p = 0.510$), significantly more micro-invertebrates were present within the macrophyte beds (*t*-test; $t = -2.919$, $p = 0.043$). Only 2 rotifer taxa encountered throughout the wetland were unique to the macrophyte beds. These were rare, constituting < 0.5% of total individuals examined.

Total open water macro-invertebrate densities ranged from 0.48-13.21 individuals m^{-2} and did not differ significantly between sites (One-way ANOVA $p = 0.649$) (Figure 3.8). Thirteen macro-invertebrate taxa (11 arthropods, 2 molluscs) were encountered in samples taken from Webster's Lagoon. Ten taxa were recorded from Site A. No difference in composition was recorded between sites B and C, for which 5 taxa were identified. Only 2 taxa (*Chironominae* sp., *Micronecta* sp.) were present at all three sites. Plots of cumulative species richness *vs.* sampling effort indicated that few additional taxa were likely to be encountered beyond the 5th replicate within each site (Figure 3.9).

Total densities of macro-invertebrates sampled from amongst *Vallisneria* sp./*Ludwegia peploides* beds present at Site A ranged from 10.00-17.02 individuals m^{-2} and were more abundant than those present in the adjacent open water (*t*-test; $t = -6.009$, $p = 0.004$). Only 2 taxa encountered throughout the wetland were unique to the macrophyte beds. These were rare, constituting < 1% of total individuals examined.

The precision (SE/mean) achieved at site and wetland scales for each response variable at the two levels of sampling effort (3 and 5 replicates) are shown in Table 3.7. These data indicate that 5 replicates were sufficient to achieve the required precision (≤ 0.25) at both wetland and site scales for each of the response variables except for CPOM. By reducing sampling effort to 3 replicates it was still possible to achieve adequate precision for most water quality variables.

Sampling precision was greater (*i.e.* lower precision value) at the site-scale than it was at the wetland-scale for water quality and algal response variables, indicating that these varied more between sites than they did within sites. This is reflected in the plots shown in Figure 3.8.

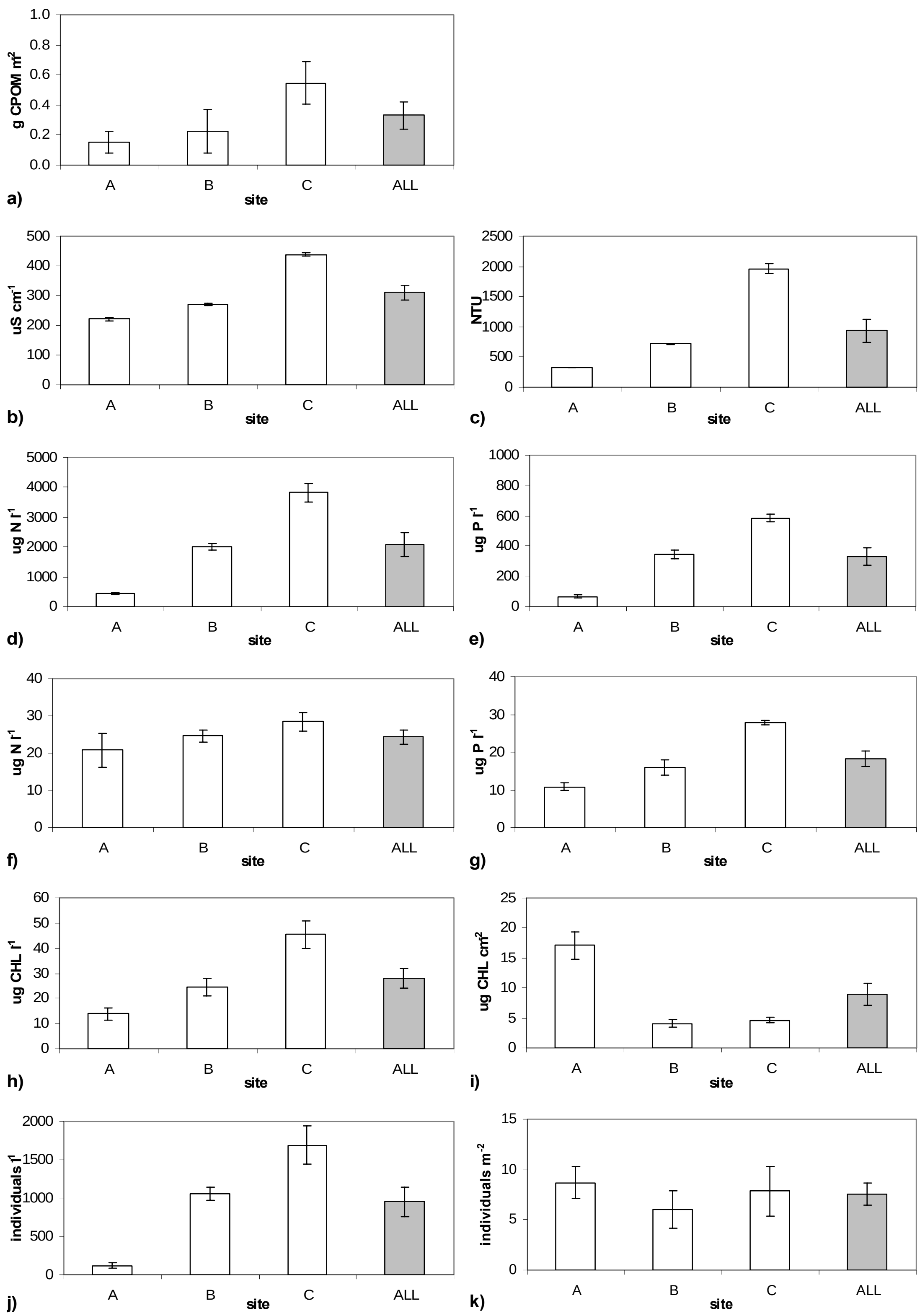


Figure 3.8: Mean ± SE values (n=5) recorded for a) CPOM (g CPOM m^{-2}), b) EC (µS cm^{-1}), c) turbidity (NTU), d) TN (µg N l^{-1}), e) TP (µg P l^{-1}), f) NOx (µg N l^{-1}), g) FRP (µg P l^{-1}), h) CHLp (µg CHL l^{-1}), i) CHLb (µg CHL cm^{-2}) j) zooplankton (individuals l^{-1}) and k) macro-invertebrates (individuals m^{-2}) at each site (A, B, C) and for all sites pooled (ALL) from Webster's Lagoon.

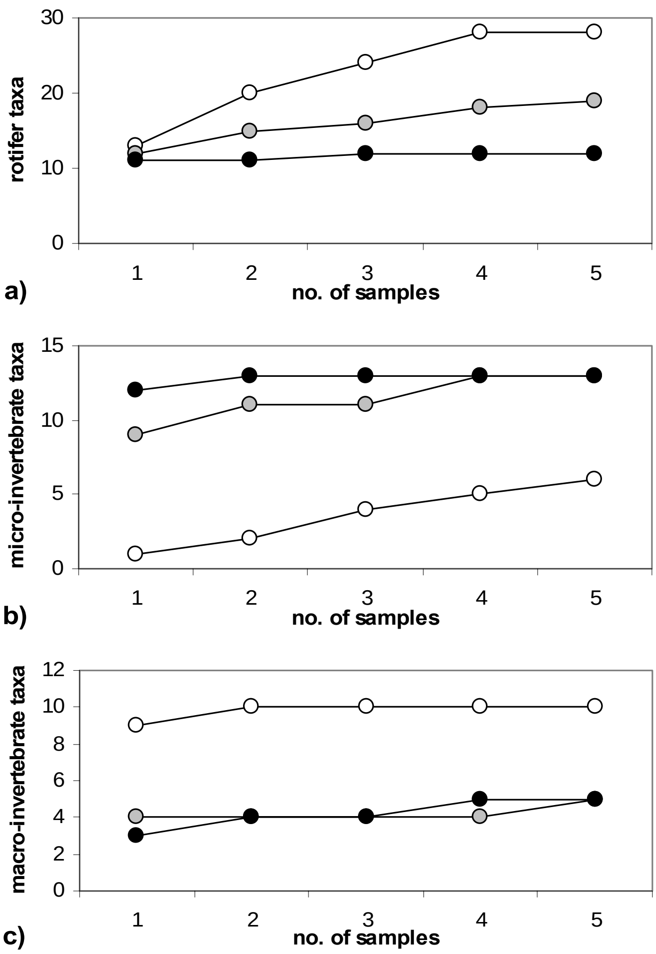


Figure 3.9: Species-area relationships for different levels of sampling effort at site A (○), Site B (●) and Site C (●) in Webster's Lagoon for a) rotifers, and b) micro-invertebrates, and c) macro-invertebrates.

Response variable	Precision (n=3) Sites min	max	Wetland	Precision (n=5) Sites min	max	Wetland
CPOM	**0.38**	**0.88**	**0.35**	0.25	**0.65**	**0.27**
EC	0.02	0.05	0.11	0.01	0.03	0.08
Turbidity	0.02	0.05	0.25	0.01	0.04	0.20
TN	0.02	0.05	0.21	0.01	0.03	0.16
TP	0.02	0.06	0.19	<0.01	0.04	0.15
NOx	0.08	0.24	0.18	0.05	0.13	0.13
FRP	0.04	0.07	**0.31**	0.03	0.04	0.23
CHLp	0.17	0.24	**0.27**	0.11	0.14	0.19
CHLb	0.11	0.25	**0.31**	0.06	0.17	0.23
Zooplankton	0.14	**0.44**	**0.28**	0.08	**0.30**	0.20
Macro-invertebrates	**0.26**	**0.52**	0.24	0.18	**0.31**	0.15

Table 3.7: Summary of precision (SE/mean) estimates calculated for sampling efforts of n=3 and n=5 replicates for each response variable at the site and wetland scales of sampling. Site and wetland scale levels of precision ≥ 0.25 are highlighted.

3.4.4.4 Discussion

For each response variable the standard sampling regime (5 replicates x 3 sites) was sufficient to achieve levels of precision $\leq$ 0.25 at the wetland scale, the primary scale for which treatment effects are to be tested. Measurements of CPOM were the least precise of all variables measured (precision = 0.267). Estimates of precision indicated that routine sampling effort for water quality and algal response variables could be reduced to 3 replicates per site without seriously compromising precision at the wetland scale. However, sampling effort will only be reduced to this level for nutrients, whose anlysis represents a significant financial cost. At approximately $20 per combined TN and TP sample and $20 per combined NOx and FRP sample, reducing sampling effort from 5 to 3 replicates will generate a savings of approximately $240 per wetland per sampling event.

Within site variance was generally lower for water quality and algal response variable than was the variance encountered between sites. Whilst CPOM and macro-invertebrates were uniformly distributed amongst open water sites throughout the wetland, significant gradients through the wetland were encountered for EC, turbidity, TN, TP, FRP, phytoplankton and phytobenthic chlorophyll *a* concentrations and zooplankton densities. These data indicate that water within the wetland was pooly mixed. The influence of Toupnein Creek on wetland water quality extended mid-way between sites A and B, where the strongest quality gradients were encountered. Whilst whole-wetland estimates of response variables will be used to test the ecological response hypotheses, for some variables (EC, turbidity), where within site variance was much lower than was between site variance, experimental treatment effects are likely to vary between sites and should therefore be examined at both site and wetland scales.

Both the zooplankton- and macro-invertebrate-specific hypotheses involve the estimation of community density and species richness. As numerous studies have demonstrated habitat preferences by these organisms (Beck 2000, Tanaguchi *et al.* 2003), consideration was given to identifying potential habitats and the need to stratify sampling at the habitat level. Two primary habitat types were identified; open water and macrophyte beds (*Vallisneria* sp. and *Ludwegia peploides*). The occurrence of submerged macrophytes in Webster's Lagoon was restricted to the area closest to and within the mixing zone of Toupnein Creek (Site A) where turbidity and EC were lowest and water depth greatest (*ca.* 0.2-0.5 m). Whilst there was no difference in zooplankton density between open water and macrophyte habitats, significantly more macro-invertebrates were present in samples collected from the macrophyte beds. Of the 48 zooplankton and 15 macro-invertebrate taxa encountered only 2 rare taxa from each group were unique to macrophyte habitats.

Based on these considerations and the limited ability to increase sampling effort above 5 replicates per site for invertebrates we decided not to introduce habitat as a sampling strata. Whilst this is unlikely to influence our estimations of zooplankton community structure, it may lead to our underestimating macro-invertebrate densities especially if macrophyte habitats expand as a consequence of hydrological manipulations. To counter this we will apportion sampling effort (*i.e.* 5 replicates) within sites according to the relative areas occupied by each habitat type.

3.5 Monitoring Results for 2004-05

3.5.1 Introduction

Forty hypotheses were developed to test wetland ecological responses to the re-instatment of a wet/dry regime and the exclusion of large adult carp from Webster's Lagoon (Section 3.2). For each hypothesis ecological response variables were identified (Section 3.3), and appropriate experimental designs and sampling protocols developed, and statistical models selected (Section 3.4). In this section we present the first of the long-term monitoring data from Webster's Lagoon and the Lock 7 experimental control wetland collected between mid December 2004 and mid April 2005.

For experimental designs dealing with the examination of changes in variance between Before and After time periods, each sampling event provides only one value for Control and Impact locations. The data presented here represent the start of the Before experimental period against which all future impacts will be assessed. At present, too few sampling events ($n \leq 4$ over 5 months) preclude accurate estimation of the variation in differences between Control and Impact response variable means or of minimum detectable treatment effect sizes between time periods (*cf.* Zar 1984). This will become possible once data covering a full annual cycle becomes available. Similarly, the current data set does not lend itself to examining the extent to which response variables in both wetlands 'track' each other, a key assumption underlying the rigour of unreplicated BACI experimental designs. Because of these considerations, meaningful analyses of data with respect to hypothesis testing are at present premature.

The objectives of this initial report of monitoring results were to:

- i) report on the status of flow/hydrology and groundwater/salinity monitoring,
- ii) present available data relating to the distribution of aquatic macrophytes, cumbungi (*Typha domingensis*) and red gum (*Eucalyptus camaldulensis*) within Webster's Lagoon,
- iii) demonstrate the presence of large adult carp (*Cyprinus carpio*) in Webster's Lagoon and the Lock 7 wetland and present available fish community data,
- iv) validate the ability of the standard monitoring protocols to achieve the required level of sampling precision (SE/mean ≤ 0.25), and
- v) present a summary of wetland scale data.

3.5.2 Methods

Webster's Lagoon was sampled 4 times and the Lock 7 wetland 3 times prior to the end of April 2005 (Table 3.8). Standard field and laboratory methods used for each response variable are described in Table 3.2. Three replicate samples were collected from 3 open water sites (Figure 3.7) for the determination of electrical conductivity (EC) and nutrient concentrations. Five replicate samples per site were collected for sediment organic matter (CPOM) turbidity, algae and invertebrates. Fish communities were sampled by setting 6 large and 6 small fyke nets in each wetland and sampling event. Aerial photographs flown on the 25th January 2005 by Aerometrix Pty. Ltd. (20 cm resolution) were used to quantify the distribution of submerged macrophytes, cumbungi (*Typha domingensis*) and red gum (*Eucalyptus camaldulensis*). Vegetation types identified from aerial photographs were validated in the field.

Webster’s Lagoon	Lock 7 wetland
14.12.2004	not sampled
10.1.2005	17.1.2005
14.2.2005	17.2.2005
26.4.2005	28.4.2005

Table 3.8: Dates of field sampling events for Webster’s Lagoon and the Lock 7 wetland.

3.5.3 Results

Flow/hydrology and lateral connectivity

Water levels as determined from fixed marker posts during the sampled period fluctuated by 7.5 cm in Webster’s Lagoon and by 11.0 cm in the Lock 7 wetland. Automated water level loggers were installed in both wetlands at the end of April 2005 and are currently recording water levels on a daily basis. This data when linked to AHD and wetland sill heights will allow for the testing of hypotheses A1 and B1.

Groundwater/salinity

Groundwater bores were installed along each of four transects at Webster’s Lagoon in early 2005 by REM Pty Ltd.. Bore construction and location details are provided by Jeuken (2005). Transect A is situated 10m downstream and transect B is situated 10 m upstream of the proposed control structure. Transects C and D are located midway along and toward the terminal end of the Lagoon, respectively. Monitoring of these bores will commence shortly and will facilitate the testing of hypotheses C1 and C2. The monitoring of sediment salinity (Hypothesis C3) will not commence until the regulating structure has been installed and the wetland has dried completely.

Wetland vegetation

Aquatic macrophytes

Macrophyte beds present within Webster’s Lagoon during January 2005 consisted principally of ribbonweed (*Vallisneria* sp.) and water primrose (*Ludwegia peploides)* (Figure 3.10). Figure 3.11 shows the distribution of these species within the Lagoon. *Vallisneria* sp. beds were more extensive (3775 m^2) than were beds of *Ludwegia peploides* (1085 m^2), which were restricted to a narrow littoral band at the highest inundated elevations. The distribution of both macrophyte species in the Lagoon was restricted to the area closest to and within the mixing zone of Toupnein Creek (Site A) where turbidity and EC were lowest and water depth greatest (*ca.* 0.2-0.5 m). This mixing zone is evident in the aerial imagery and extends approximately 200 m into the wetland from Toupnein Creek (340 m along the western edge).

Treatment effects on the distribution of ribbonweed (*Vallisneria* sp.) and water primrose (*Ludwegia peploides)* will be monitored on an annual basis using aerial photography. Other emergent species, including *Eleocharis* sp., *Juncus* sp., were present during January 2005. These formed a narrow littoral band at the highest inundated elevations and extended discontinuously around the entire wetland. Elevational shifts in this community will also be recorded annually. At present, however, digital imagery has not been linked with AHD.

a) b)

Figure 3.10: a) Ribbonweed and b) water primrose beds close to the mouth of Websters Lagoon at Site A (photos O. Scholz).

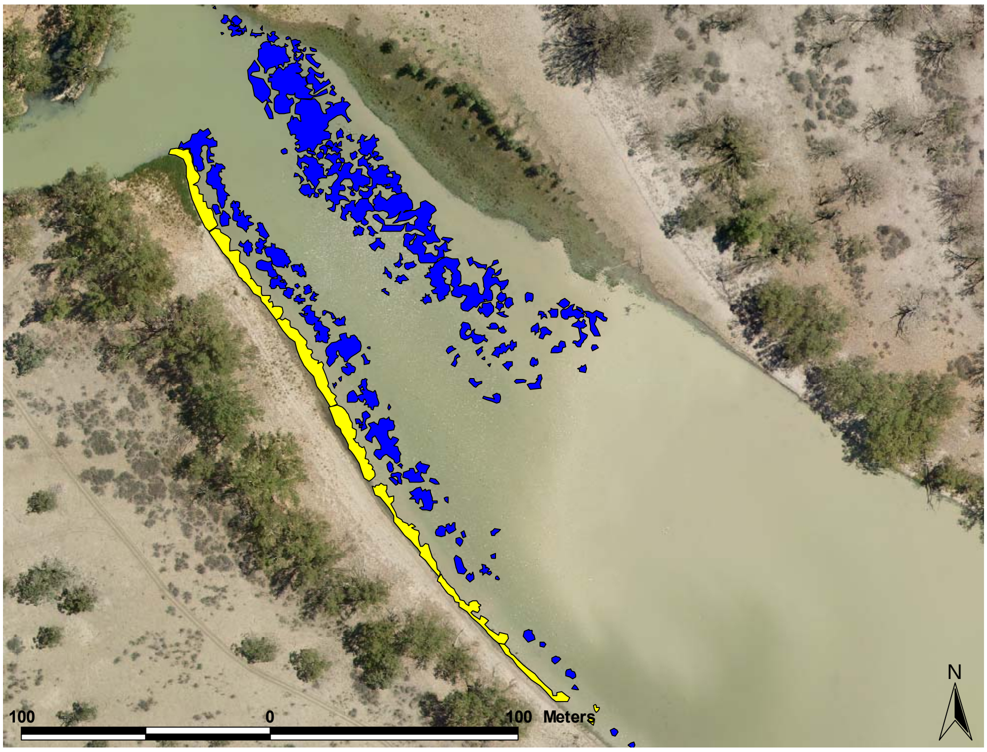


Figure 3.11: Distribution of ribbonweed (blue) and water primrose (yellow) within mixing zone of Toupnein Creek and Webster's Lagoon at Site A (photo January 2005 Aerometrix Pty. Ltd.).

Cumbungi

The distribution of cumbungi (*Typha domingensis*) was restricted to the distal end of Webster's Lagoon either side of Site C (Figure 3.12). Fourteen individual stands of cumbungi covering a combined area of 2830 m^2 were present during January 2005. Mean stand diameters ranged from 1.50-22.75 m, indicating a wide range in stand ages (stand age could not be determined).

Mature cumbungi stands comprised three distinct concentric zones; an actively expanding outer edge characterised by the lateral extension of colonising shoots, an inner denser mature zone characterised by reproductive shoots and taller leaf shoots, and a central core of senescent and senescing shoots (Figure 3.13). No assessment of stand vigour was possible within the current reporting time-frame.

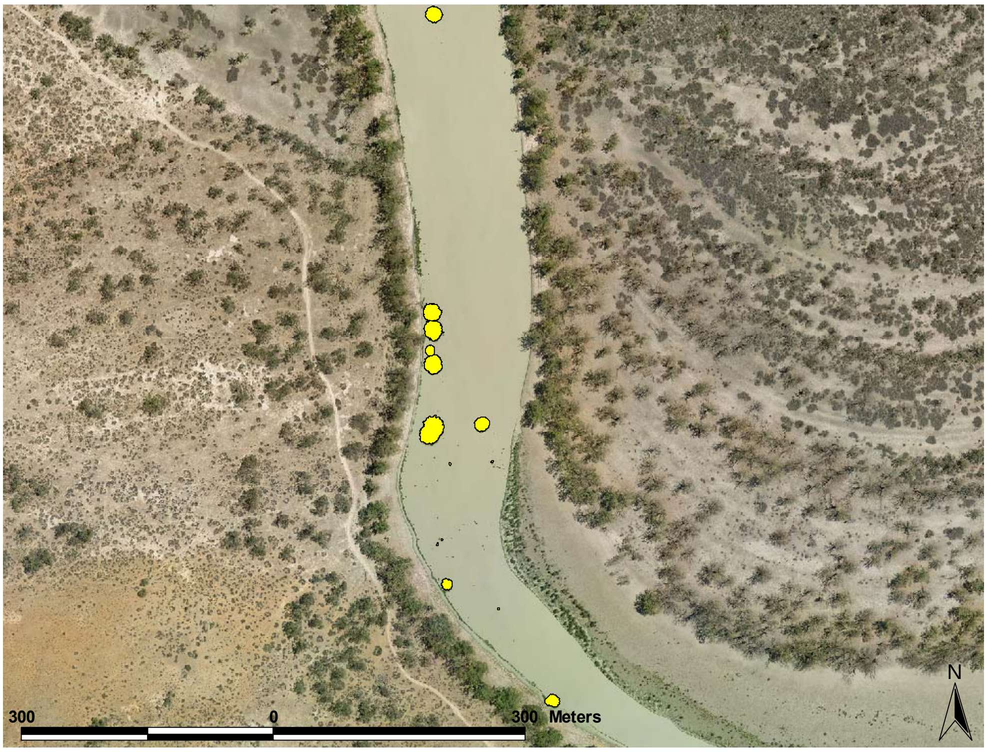
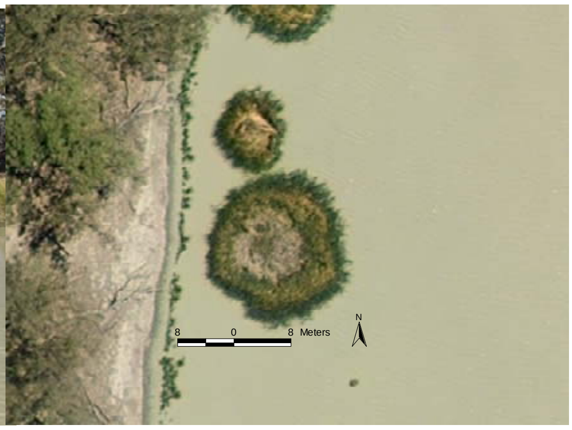


Figure 3.12: Distribution of cumbungi (yellow) at Site C within Webster's Lagoon (photo January 2005 Aerometrix Pty. Ltd.).



a) **b)**

c) **d)**

Figure 3.13: a) Mature stand of cumbungi, b) aerial view of cumbungi showing the three growth zones, c) reproductive shoots of cumbungi, and d) colonising outer edge of cumbungi stand (photos January 2005; O. Scholz and Aerometrix Pty. Ltd.).

Red gums

The distribution of red gum (*Eucalyptus camaldulensis*) saplings within the wet/dry zone of Webster's Lagoon was concentrated in three distinct areas within the Lagoon (Figure 3.14). The first field inspection of these stands during April 2005 indicated that each stand consisted of at least 3 size (*cf.* age) cohorts, each located within distinct elevation bands (Figure 3.15a). No assessment of individual sapling or cohort condition, height or elevation was possible within the current reporting time-frame. However, a plant condition index method based on average leaf colour was trialled (Figure 3.15b).

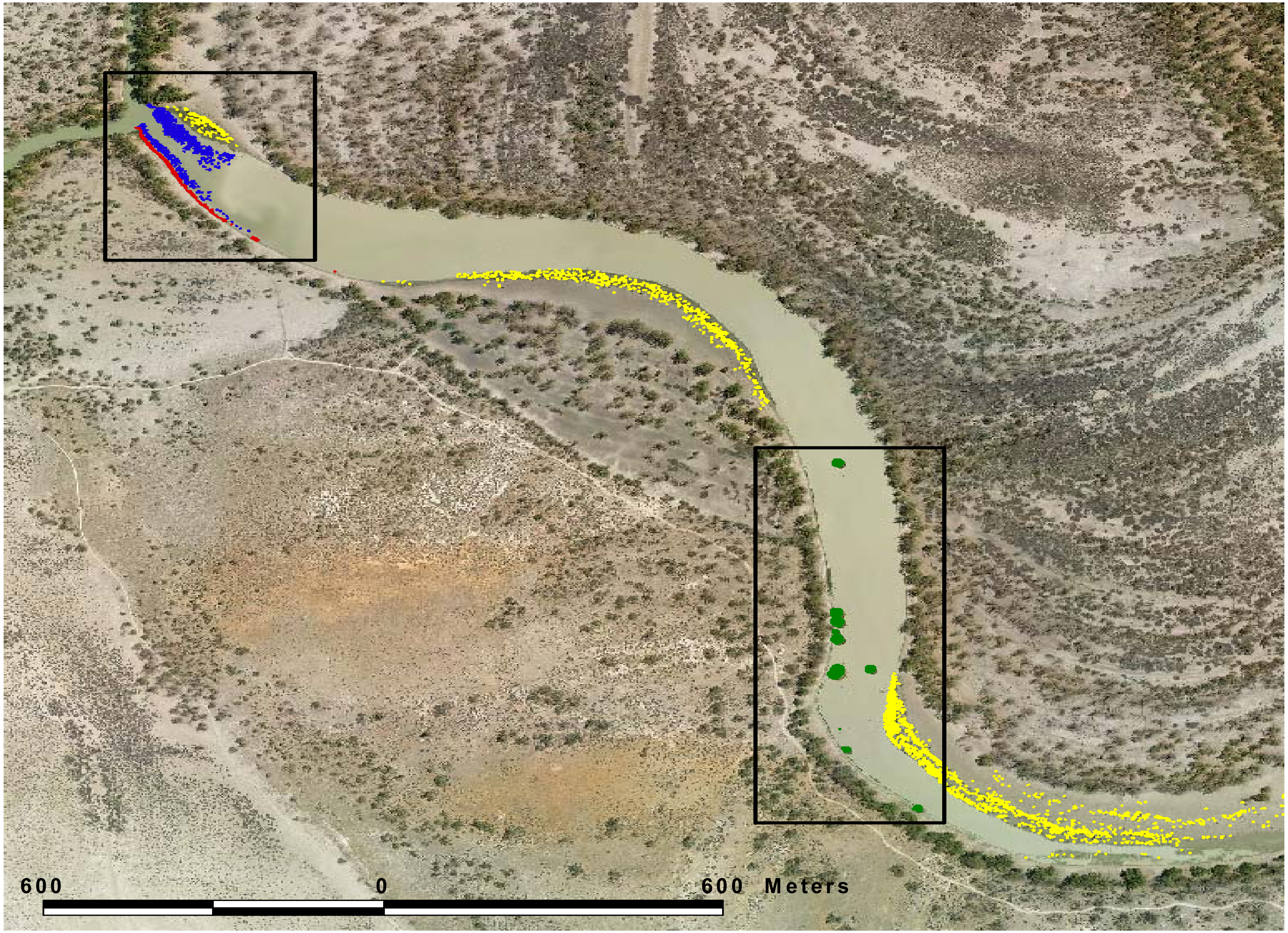


Figure 3.14: Distribution of red gums within Webster's Lagoon (yellow). Enlargements of outlined areas are shown in figures 3.11 and 3.12 (photo January 2005 Aerometrix Pty. Ltd.).

a) **b)**

Figure 3.15: a) Stand of red gums at Site A comprising at least 3 distinct height (age) cohorts, and b) leaf condition index trialled during 2005 showing from top to bottom a healthy, a stressed and a scenescent leaf (photos O. Scholz).

<u>Fish</u>

A total of 7 fish species were recorded from Webster's Lagoon and 8 fish species were recorded from the Lock 7 wetland between January and April 2005 (Table 3.9). Dwarf flatheaded gudgeon (n=1) was only recorded in the Lock 7 wetland. Alien species (carp and mosquitofish) accounted for 23.2 % of fish caught in Webster's Lagoon and 93.2 % of fish caught in the Lock 7 wetland.

Common name	Scientific name	Large fyke nets		Small fyke nets	
		Webster's Lagoon	Lock 7 wetland	Webster's Lagoon	Lock 7 wetland
Carp*	*Cyprinus carpio* (Linnaeus)	79	20	5	1
Bony herring	*Nematalosa erebi* (Gunther)	38	2	213	0
Flatheaded gudgeon	*Phylypnodon grandiceps* (Krefft)	0	0	297	2
Mosquitofish*	*Gambusia holbrooki* (Girard)	0	0	99	459
Flyspecked hardyhead	*Ceratocephalus stercusmuscarum* (Thorn)	0	0	27	4
Australian smelt	*Retropinna semoni* (Weber)	0	0	24	1
Carp gudgeon	*Hypseleotris* spp.	0	0	6	25
Dwarf flatheaded gudgeon	*Phylypnodon* sp. (H.K. Larson)	0	0	0	1
	TOTAL	117	22	671	493

Table 3.9: Total numbers of each fish species collected form Webster's Lagoon and the Lock 7 wetland between January – April 2005 using both large and small fyke nets. *-alien species

Carp and bony herring were the only fish species caught using large fyke nets. Carp were generally the most abundant large fish species in both wetlands, and could generally be seen from the waters edge (Figure 3.16). CPUE for carp caught in Webster's Lagoon during the study period was 4.30 ± 2.17 fish net hr^{-1} and in the Lock 7 wetland was 1.14 ± 0.35 fish net hr^{-1}. Carp constitututed ≥ 75.0 % of catches with one exception (Webster's Lagoon Site C in April 2005) (Figure 3.17). The relatively low abundance of carp caught at Site C in Websters Lagoon during April 2005 likely reflected reduced net efficiency due to the shallowness of water in which nets were set (*ca.* 5-10 cm deep) rather than to real reductions in fish abundance, as numerous large carp were observed at Site C at that time. Wetland water levels at that time were the lowest recorded during the study period.

Two distinct size classes of carp were identified from total large fyke net catches in Webster's Lagoon (sampling events combined) (Figure 3.18). The largest size class (500 – 615 mm FL) accounted for 20.5 % of the total catch (n=78). Most carp caught (74.4 %) were in the 70 - 150 mm FL size range and likely comprised several age cohorts.

Total small fyke net CPUE for Webster's Lagoon during the study period was 38.62 ± 5.49 fish net hr^{-1} and for the Lock 7 wetland was 27.69 ± 9.10 fish net hr^{-1}. Small fyke net catches from Webster's Lagoon were numerically dominated by flatheaded gudgeons and bony herring and in April by mosquitofish, which together constituted 85 – 93 % of catches. In contrast, small fyke net catches from the Lock 7 wetland were numerically dominated by mosquitofish throughout the study period. These constituted 90 – 98 % of catches (Figure 3.19).

Figure 3.16: Large adult carp in Webster's Lagoon observed at Site C during the April 2005 survey (photo I. Ellis MDFRC).

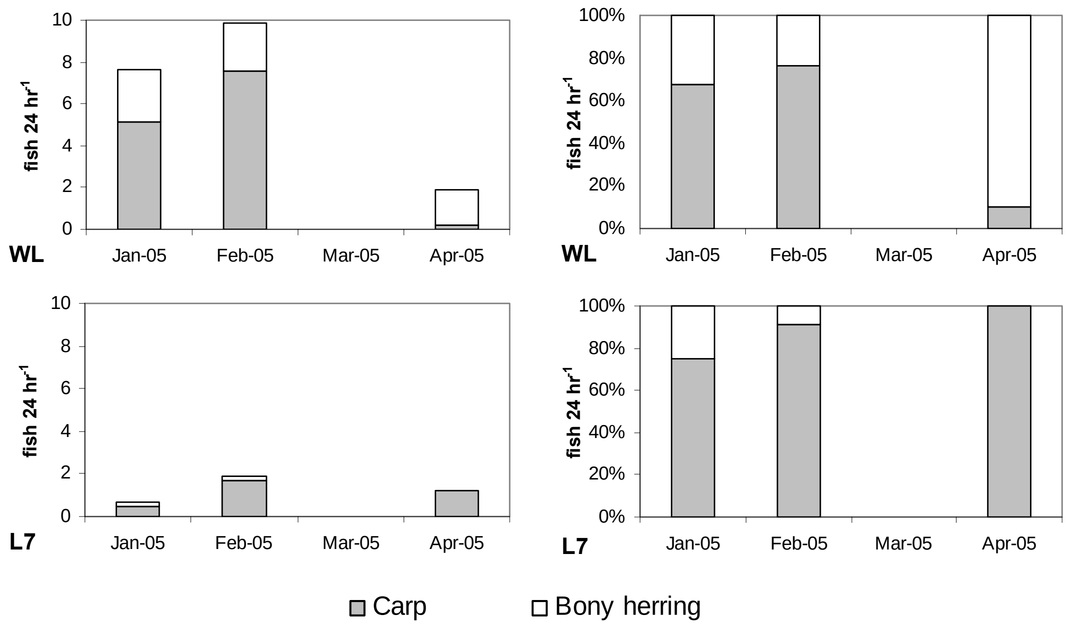


Figure 3.17: CPUE (fish 24 net hours^{-1}) and relative abundance of fish caught using large fyke nets set in Webster's Lagoon (WL) and the Lock 7 wetland (L7).

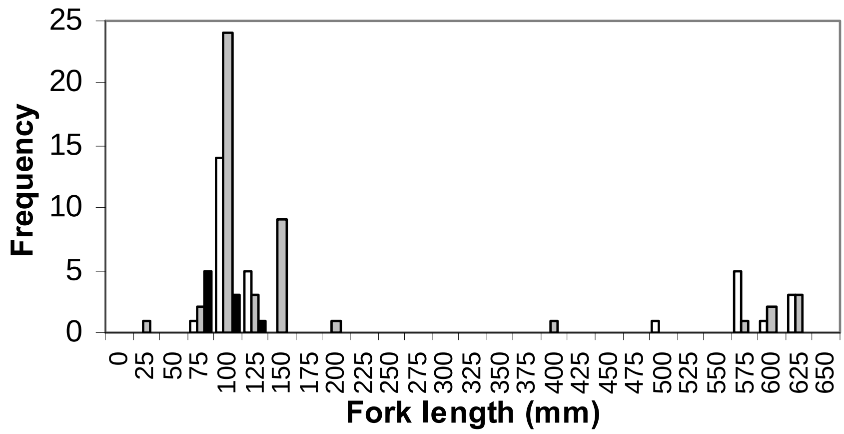


Figure 3.18: Frequency distribution of carp size classes (mm) caught in Webster's Lagoon during January 2005 (□, n=30), February 2005 (■, n=47) and April 2005 (■, n=9).

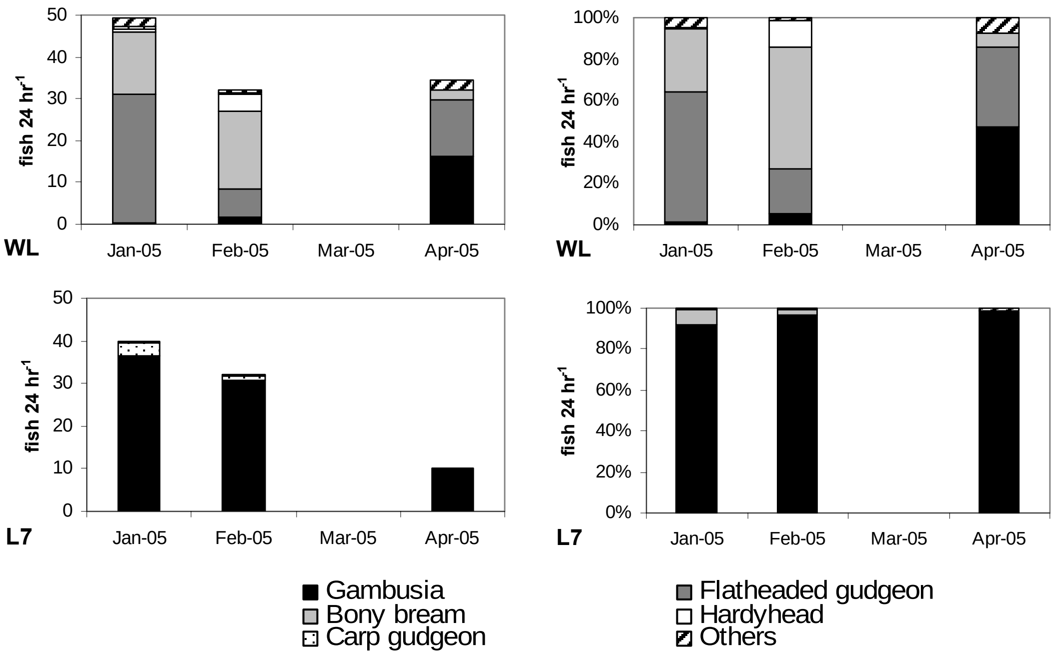


Figure 3.19: CPUE (fish 24 net hours^{-1}) and relative abundance of fish caught using small fyke nets set in Webster's Lagoon (WL) and the Lock 7 wetland (L7).

Sampling precision

The precision (SE/mean) achieved at site and wetland scales using the sampling regime developed during the pilot survey for sediment organic matter (CPOM), water quality, algae and invertebrates are shown in Table 3.10. 90th percentile values were used to provide a more representative indication of achievable precision. For most ecological response variables a precision of ≤ 0.25 was achieved. Lowest precision was achieved for estimates of macro-invertebrate density, suggesting that greater sampling effort may be warranted.

Wetland scale data

Temporal trends in wetland scale data (mean ± SE) for sediment organic content (CPOM), water quality, algal and invertebrate response variables are shown in Figure 3.20. For reasons discussed in the introduction of this section, no further analysis of the data at this stage was warranted.

Response variable	Wetland	Number of replicates per site	Precision (SE/mean) 90th percentile	
			Sites	Wetland
CPOM	Webster's	5	**0.52**	**0.31**
	Lock 7	5	**0.43**	0.25
EC	Webster's	3	0.05	0.13
	Lock 7	3	0.10	0.07
Turbidity	Webster's	5	0.15	**0.32**
	Lock 7	5	0.18	0.15
TN	Webster's	3	**0.35**	**0.26**
	Lock 7	3	0.10	0.05
TP	Webster's	3	**0.36**	**0.27**
	Lock 7	3	0.13	0.12
NOx	Webster's	3	0.22	0.20
	Lock 7	3	0.19	0.20
FRP	Webster's	3	0.17	**0.28**
	Lock 7	3	0.08	0.17
CHLp	Webster's	5	0.14	0.21
	Lock 7	5	0.17	0.15
CHLb	Webster's	5	0.24	0.23
	Lock 7	5	0.22	0.09
Zooplankton	Webster's	5	**0.49**	**0.28**
	Lock 7	5	**0.28**	0.25
Macro-invertebrates	Webster's	5	**0.45**	**0.28**
	Lock 7	5	**0.78**	**0.42**

Table 3.10: 90th percentiles of site and wetland scale sampling precision (SE/mean) achieved for each response variable between December 2004 – April 2005 in Webster's Lagoon and the Lock 7 wetland. Precision values ≥ 0.25 are highlighted.

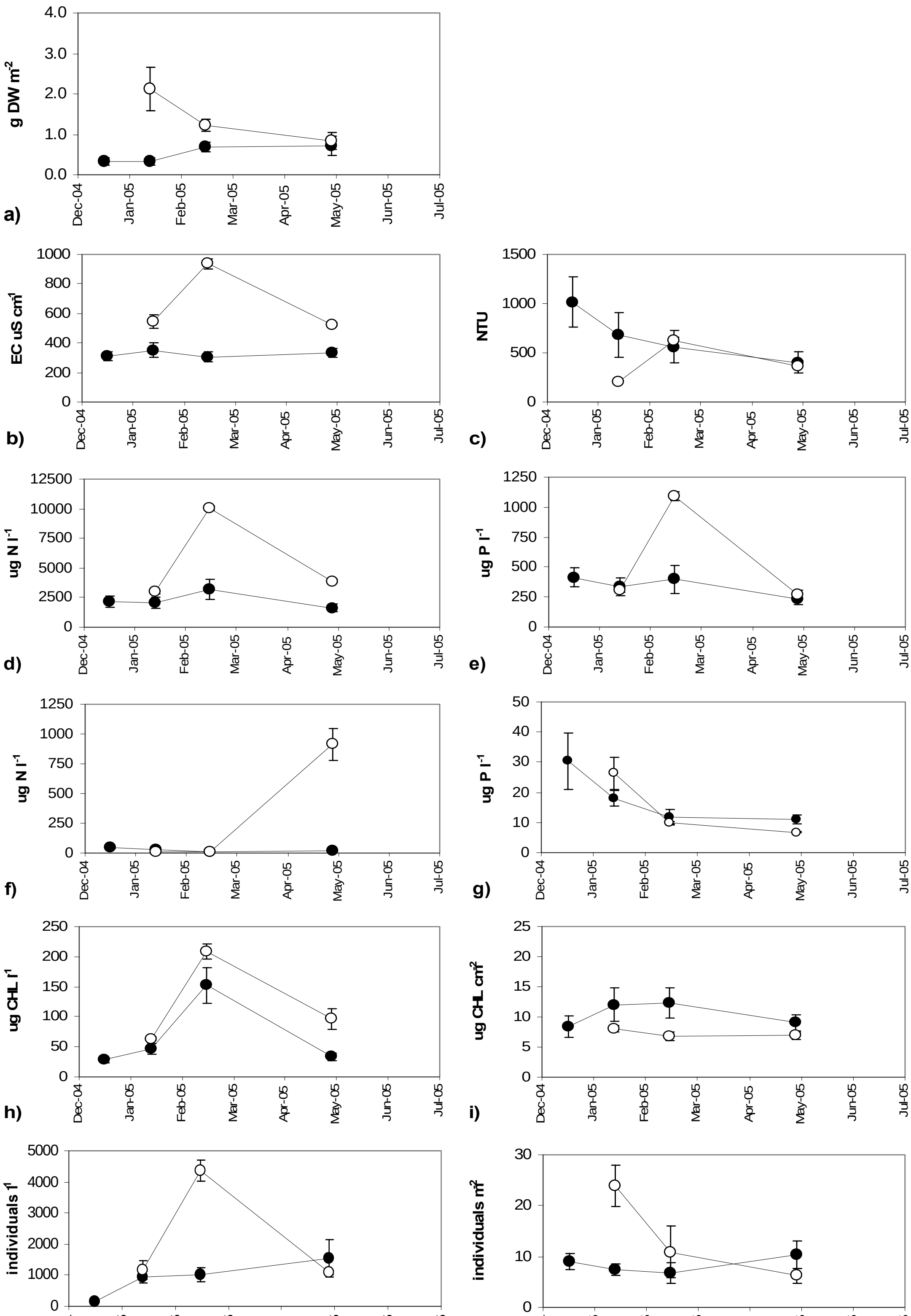


Figure 3.20: Mean ± SE wetland scale values recorded for a) CPOM (g CPOM m^{-2}), b) EC (µS cm^{-1}), c) turbidity (NTU), d) TN (µg N l^{-1}), e) TP (µg P l^{-1}), f) NOx (µg N l^{-1}), g) FRP (µg P l^{-1}), h) CHLp (µg CHL l^{-1}), i) CHLb (µg CHL cm^{-2}) j) zooplankton (individuals l^{-1}) and k) macro-invertebrates (individuals m^{-2}) at Webster's Lagoon (●) and the Lock 7 wetland (○).

3.5.4 Discussion

At the time of writing 4 field sampling events have taken place extending over a 5 month period. Whilst monitoring protocols have been developed for most ecological response variables, those for some are still being developed (*i.e.* groundwater, vegetation, waterbirds) and will be incorporated into the routine monitoring programme as they become available.

The removal of carp from Webster's Lagoon has been identified as one of the key ecological objectives for the wetland. It was therefore important that the presence of these during the experimental Before period be demonstrated and that they were being caught using the standard sampling gear (large and small fyke nets). Carp were present within both Webster's Lagoon and the Lock 7 experimental control wetland. Large numbers could usually be seen from the water's edge and appeared in catches at both wetlands. Carp caught from within Webster's Lagoon were of two broad size classes consisting of an unknown number of age cohorts. As most (74.4 %) of the carp caught were within the 70 - 150 mm FL size range, it will be necessary that the mesh size of grates incorporated into the proposed regulator be sufficiently small to exclude these for there to be a measurable treatment effect.

As stated in sections 3.4.3 and 3.4.4, reducing sampling variance increases our ability to detect temporal variation (Before vs After). Based on the available monitoring data for sediment organic matter (CPOM), water quality, algae and invertebrates we were able to demonstrate that the standard sampling effort employed for each of these response variables (either 3 or 5 replicates at each of 3 sites within the wetland) had the capacity to routinely achieve a precision of ≤ 0.25. The lower precision achieved for macro-invertebrate samples suggests that greater sampling effort of these may be warranted. At this stage we are not in a position to determine the level of statistical power we will be able to achieve in testing each of the ecological response hypotheses (*i.e.* minimum detectable treatment effect size between experimental Before and After time periods). This will become possible once at least a full annual cycle of monitoring has been completed.

4 Horseshoe Lagoon

4.1 Background

Horseshoe Lagoon is a wetland connected to the Wallpolla Island anabranch system of the Murray River in northwest Victoria (Figure 4.1). As a well defined cut-off meander, Horseshoe Lagoon is flat-bottomed and asymmetric in cross-section, with the southern bank being higher and steeper than the northern bank. The southern bank is generally steeper as a result of the former Murray River cutting into the high alluvial floodplain, whereas the northern bank is depositional in origin and comprises younger alluvium (SKM 2004b).

The hydrology of Horseshoe Lagoon has changed substantially as a consequence of the regulation of the Murray River, the weir pool created by Lock 9, and deepening of the sill separating the wetland from Finnigans Creek. The current commence to fill level for Horseshoe Lagoon is just above the normal pool level for Lock 9 (27.5 ± 0.2 mAHD). At this level approximately 19.9 ha of the lagoon is inundated. Currently there are only short periods (January–March) where there are no inflows to the wetland. Under these conditions, the lagoon experiences less frequently the long summer and autumn complete dry periods that are thought to have occurred under natural conditions (SKM 2004b), although small changes in flow/water level result in the periodic exposure of extensive areas of the lagoon (wet-dry littoral zone). Regulation of the Murray River has also eliminated inflows at the eastern end of Horseshoe Lagoon, which under natural conditions would have occurred once Murray River flows exceed the current equivalent of 60-66000 ML day^{-1} at Lock 10 (SKM 2004b).

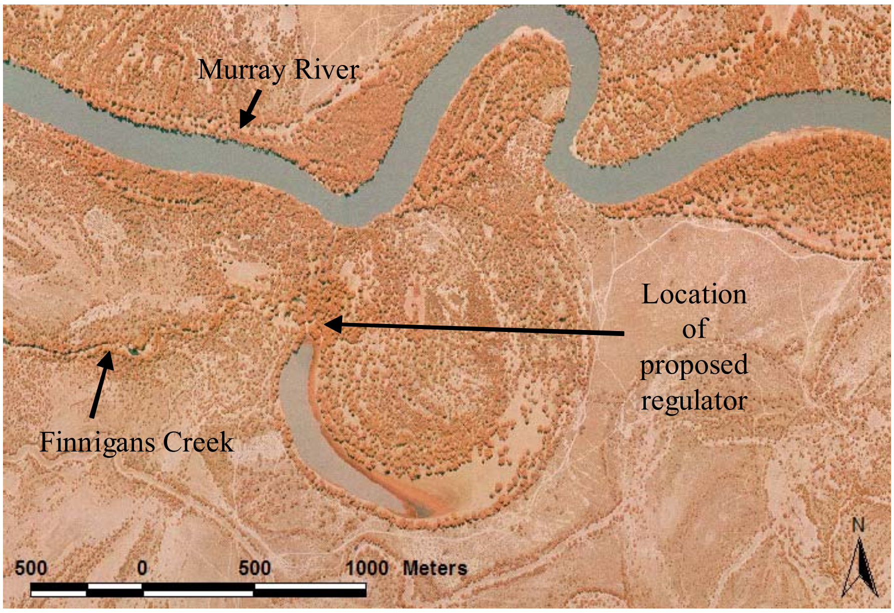


Figure 4.1: Horseshoe Lagoon showing location of proposed structure.

Horseshoe Lagoon is considered to be of high ecological value on both regional and state levels. Much of this stems from the broad range of habitat it provides and its ability to support a variety of significant flora and fauna (SKM and Roberts 2003). Key ecosystem values and processes supported by Horseshoe Lagoon have been compromised either directly by changes in flow regime or indirectly through associated threats, such as the presence of carp (*Cyprinus carpio*) and the undesired expansion of native species such as cumbungi (*Typha* spp.) and red gum (*Eucalyptus camaldulensis*) in to the lagoon bed. The management of these threats and other identified ecological values and processes is to be achieved through the installation and operation of a regulator at the junction of Finnigans Creek and Horseshoe Lagoon (refer to Zukowski and Meredith 2005 for design details).

Ecological and flow objectives for each identified ecological value, process and threat developed by SKM (2004b) are shown in Table 4.1. The regulating structure proposed for Horseshoe Lagoon will:

- Permit the partial and tempory disconnection of the lagoon from Finnigans Creek.
- Allow for the restoration of a more frequent wet/dry cycle within the wetland.
- Permit the retention of water within the Lagoon at higher elevations (28.8 m AHD) following medium to large flood events or if water is pumped into the lagoon.
- Permit the inundation of the wet-dry littoral zone for longer than is currently occurring.
- Incorporate the capacity to exclude larger fish (*e.g.* carp) from inflows.

Operating guidelines for the regulating structure are to:

- Dry the wet-dry littoral zone for a minimum of 6 months in summer and autumn.
- Completely dry the existing permanently inundated zone for 6 - 10 months.
- Wet the wet-dry littoral zone for between 3 - 6 months during winter and spring.

Ecological value/process	Ecological objectives	Flow objectives
Lateral connectivity	Maintain connections between Horseshoe Lagoon and Finnigans Creek	Insufficient data currently available to indicate the most appropriate timing and duration of periods of dis/connection.
Carbon and nutrient cycling	Rehabilitate	Dry out wet-dry zone annually over summer/autumn for at least 2 weeks. Inundate wet-dry zone nearly every year during spring for 1-3 months.
Vegetation	Rehabilitate extent and diversity	Dry out and wet-dry zone annually between December-June for at least 6 months. Inundate wet-dry zone nearly every year between June-December for 3-6 months.
Small native fish	Rehabilitate habitat and feeding opportunities	Dry out and wet-dry zone annually during summar-autumn for at least 6 months. Inundate wet-dry zone nearly every year during winter-spring for at least 3 months.
Waterbirds	Rehabilitate habitat and feeding opportunities	Dry out and wet-dry zone annually during summar-autumn for at least 6 months. Inundate wet-dry zone nearly every year during winter-spring for at least 3 months.
Turtles (Broad-shelled)	Maintain	Insufficient data currently available to indicate whether structure will exclude re-colonisation after re-flooding.
Ecological threat		
Saline groundwater discharges	Minimise	Minimise period of wet-dry zone drying. Insufficient data currently available to indicate maximum period of drying permissible.
Cumbungi	Reduce vigour and extent in persistent pool	Complete drying between December-June (ideally September-June) for 7-10 months.
Red gum	Reduce abundance/extent of regeneration in wet-dry zone	Inundation to a depth of 50-100 cm for 4-6 weeks between June-October, or shallow flooding for 12-26 weeks during autumn-winter. Complete drying between October-June for 6 months.
Carp	Reduce abundance	Complete drying anytime for more than 1 day.

Table 4.1: Summary of ecological objectives and flow requirements for Horseshoe Lagoon (source SKM 2004b).

4.2 Conceptual framework and hypotheses

Floodplain wetlands have intrinsic local values, and represent an important component of the larger floodplain ecosystem (Ward 1989, Ward and Stanford 1995). They are naturally diverse and productive habitats, whose management through the alteration of natural flow regimes has resulted in either real or perceived declines in wetland condition (*e.g.* Kingsford 2000a,b). It is generally agreed that both wet and dry periods are important in maintaining ecosystem integrity in ephemeral wetlands (Boulton and Lloyd 1992, Bunn *et al.* 1997, Boulton and Jenkins 1998). Disturbances, such as flooding and drying, drive aquatic and terrestrial successional processes and facilitate biotic and abiotic exchanges between elements of the floodplain and the riverine environment (*cf.* Flood Pulse Concept; Junk *et al.* 1989). Because of this, ephemeral wetlands are potentially sites of high productivity and diversity within floodplain ecosystems. Consequently, the management of these wetlands has implications for ecosystem productivity and diversity at the landscape scale. Thus, any changes in wetland function initiated through hydraulic modification needs to be viewed from both the perspective of the individual wetland and from a landscape perspective.

Available information, conceptual models (*e.g.* Flood-Pulse Concept - Junk *et al.* 1989, Alternative-States Model - Scheffer *et al.* 1993, Geomorphic-Trophic Model - Hershey *et al.* 1999, Trophic-Cascade Model - Carpenter *et al.* 2001, Ephemeral Deflation Basin Lake Model - Scholz and Gawne 2004a,b) and expert consultation (refer to Acknowledgements) were used to develop a conceptual model of the links between the various ecosystem components and their hypothesised responses under existing and proposed management scenarios (Figures 3.3 and 3.4). From these we generated testable hypotheses which were used to guide the development of the monitoring program.

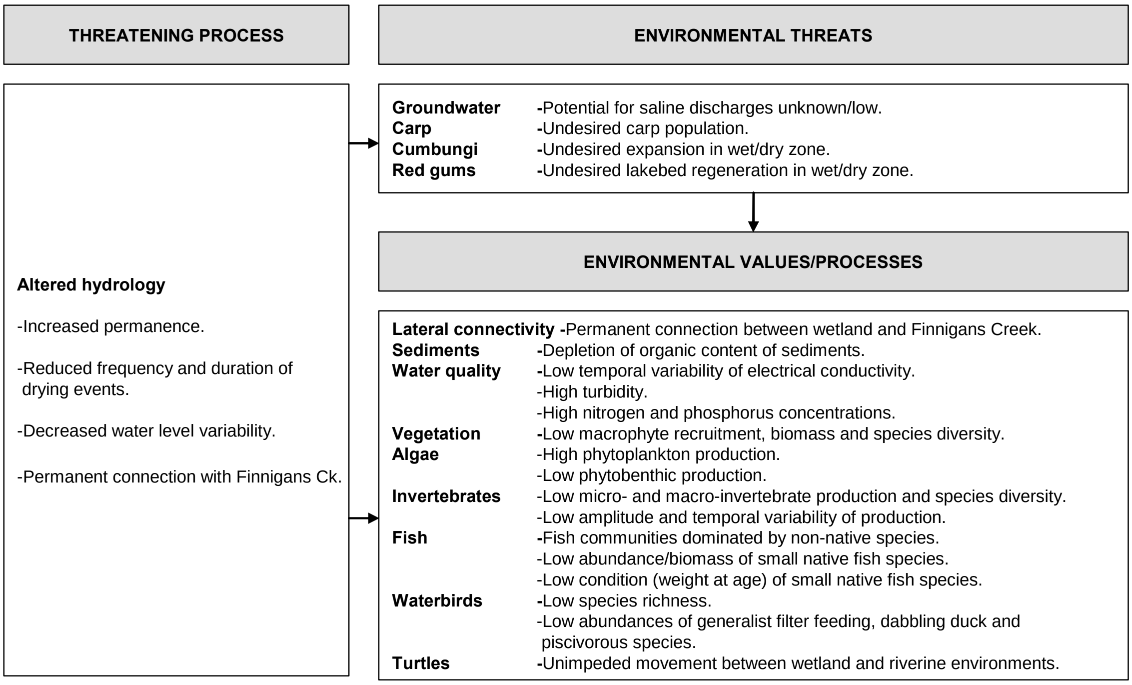


Figure 4.2: Conceptualisation of ecosystem function under existing flow conditions at Horseshoe Lagoon.

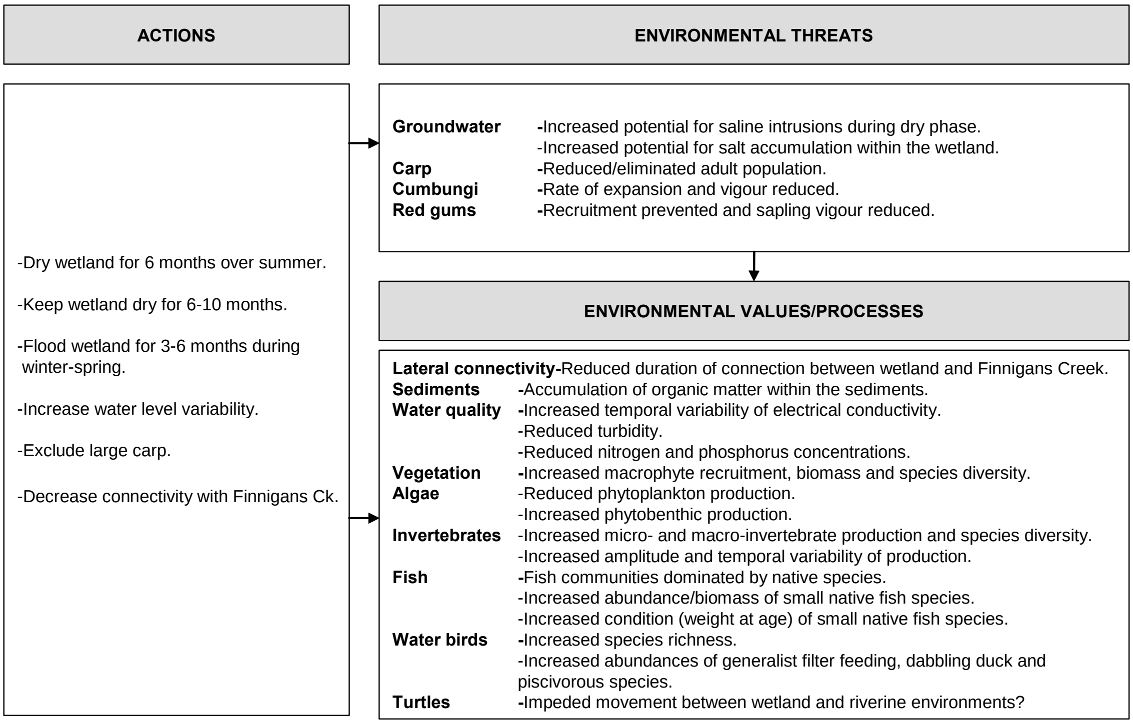


Figure 4.3: Conceptualisation of ecosystem function under the flow regime proposed for Horseshoe Lagoon.

Trophic cascade or food web models suggest that aquatic ecosystem productivity and community composition are determined by a combination of top-down and bottom-up forces (Carpenter and Kitchell 1988, Carpenter *et al.* 2001). The strengths of these interactions (*e.g.* competition, predation) tend to become weaker as food web complexity increases (Shapiro 1990, Lazzaro 1997, Borer *et al.* 2005), and for ephemeral systems biotic interactions are likely to increase during the drying phase (Schneider and Frost 1996, Golladay *et al.* 1997, Tockner *et al.* 1999). This latter response has been linked to declining habitat or refuge availability (Golladay *et al.* 1997). For example, higher water levels offer fish refuge from avian predation, but eventually this refuge is lost and avian predation becomes more effective (Kahl 1964, Kushlan 1976). Consequently, not only do fish numbers decrease as wetlands dry, but the fish community becomes increasingly dominated by smaller fish species (Scholz and Gawne 2004b). These changes in the fish community may thus stimulate a downward cascading change in trophic structure by offering invertebrate predators (*e.g.* notonectids, dytiscid beetles) an opportunity to proliferate (*e.g.* Boulton and Lloyd 1991, Jeffries 1994).

Viewed independently, hypothesized responses of each trophic level to the re-instatement of a wet/dry phase and the exclusion of carp from Horseshoe Lagoon are suggestive of an overall increase in productivity and diversity. Whilst we recognise that trophic interactions are likely to play a key role in determining measured responses at each level, the strength of such interactions is difficult to either measure or predict. It will thus be important to consider likely strengths of trophic interactions when interpreting the observations derived for each of the outlined hypotheses.

Flow/hydrology

Hypothesis A1: The proposed control structure in Horseshoe Lagoon will have an effect on water levels within the wetland in compliance with the management strategy.

Many ecological responses are likely to be driven by changes in hydrology, making it critical that an accurate assessment of water levels be included within the monitoring design. The flow objectives for Horseshoe Lagoon are to maintain water stage within the lagoon at 28.3 mAHD for 6 months, and to allow for complete drying of the lagoon (AHD < 19.4m) for 6 months. This will allow for the testing of impacts on ecological assets. Water level data will feed directly into an adaptive management framework, allowing for real time management responses during periods of non-compliance.

Lateral connectivity

Hypothesis B1: The proposed control structure in Horseshoe Lagoon will introduce periods of hydraulic isolation to the Lagoon.

Many wetland processes and biota are dependent on exchanges between wetland and riverine environments. Under the current flow regime Horseshoe Lagoon is nearly permanently connected to Finnigans Creek. The construction and operation of a regulating structure at the mouth of Horseshoe Lagoon will introduce periods of hydraulic isolation and thus potentially inhibit exchanges between aquatic environments. This may adversely impact, for example, on the ability of turtle populations to re-colonise the Lagoon after an imposed drying event.

Whilst the regime of wetland dis/connection will be monitored directly, the identification of appropriate duration and timing of periods of dis/connection will be addressed within the monitoring program for each trophic component as desribed below.

Groundwater/salinity

Hypothesis C1: Re-instating dry phases in Horseshoe Lagoon will raise groundwater levels beneath the wetland during the dry periods. *Hypothesis C2: Re-instating dry phases in Horseshoe Lagoon will cause an increase in groundwater salinity beneath the wetland during the dry periods.* *Hypothesis C3: Re-instating dry phases in Horseshoe Lagoon will result in an increase in the salinity of wetland sediments during the dry periods.*

The geology of Horseshoe Lagoon comprises four distinct layers commencing at the surface with Fine Alluvium (0-5 m thick), underlain by Channel Sands (aquifer 5-10 m thick), Blanchetown Clay (5-10 m thick), and at a depth of 20-25 m by Parilla Sand (aquitard >10 m thick) (SKM 2004b). The water table is thought to reside within the Fine Alluvium at a depth between 2-5 m below the ground surface, with little vertical flow occurring between the Parilla Sand and Channel Sand layers. Groundwater salinity under Horseshoe Lagoon is thought to increase with depth, ranging from 1000-40000 $\mu S\ cm^{-1}$ depending on the direction of groundwater flows, which are influenced by the Murray River and flow conditions in nearby streams (SKM 2004b).

Little detailed information is currently available regarding interactions between ground and surface water in Horseshoe Lagoon (SKM 2004b), and no information is currently available to suggest what impact on groundwater the drying of Horseshoe Lagoon is likely to have. Data from other systems suggests that drying is likely to reduce downward hydraulic pressure on saline groundwater, allowing it to rise. This may increase the potential for soil salinisation and saline groundwater intrusion into the wetland, and thus also the potential for the movement of salt downstream into the river. Although no environmental objective was developed for groundwater/salinity, monitoring groundwater responses to changes in surface hydrology will help fill an important knowledge gap and assist with assessing salinisation risks.

Sediments

Hypothesis D1: Re-instating a more frequent wet/dry cycle and excluding large carp from Horseshoe Lagoon will result in an increase in the mass of benthic coarse particulate organic matter (CPOM) in/on the sediments.

Re-instating a more frequent wet/dry regime in Horseshoe Lagoon is likely to impact on the amount of organic matter present within the wetland sediments in several ways. Firstly, wetland drying will allow the establishment of terrestrial vegetation within the wet/dry zone, and greater water level variability will likely stimulate aquatic macrophyte production (*e.g.* Walker *et al.* 1994, Blanch *et al.* 1999, 2000), especially in the absence of carp (refer to carp section below). These are likely to represent a potentially important source of organic matter within the wetland as they decompose. Secondly, the proposed changes in hydrology are likely to increase access to CPOM deposited at higher elevations. And thirdly, physical barriers such as natural sills and regulating structures have the potential to reduce lateral exchanges of organic matter between wetland and riverine environments, thereby increasing the potential for the net accumulation of organic matter originating within the wetland.

The increases in sediment organic matter content predicted to occur as a consequence of altering the flow regime of Horseshoe Lagoon will likely play an important role in carbon and nutrient cycling within the wetland and in structuring trophic development by increasing food and habitat resources for detritivorous biota (*e.g.* invertebrates). Although not specifically identified as a key ecological value, measuring sediment organic matter content will increase our ability to interpret the responses of key biota to changes in flow.

Water quality

Electrical conductivity

Hypothesis E1: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will increase the temporal variability of water column electrical conductivity.

Hypothesis E2: Re-instating a more frequent wet/dry regime and reducing connectivity between Horseshoe Lagoon and Finnigans Creek will lead to the loading of salt within the wetland.

Limited electrical conductivity (EC) data is available for Horseshoe Lagoon. That which is available (May – November 2003; range 172 – 424 $\mu S\ cm^{-1}$) indicates that EC was uniformly distributed within the wetland and that wetland EC was consistently greater than that present within the Murray River upstream of Lock 9 (difference range 91–273 $\mu S\ cm^{-1}$) (SKM 4004b, MDFRC unpublished data). These data do not reflect the extent of variability in EC that is likely to occur throughout the wet/dry cycle. Although no environmental objective was developed for surface water salinity, monitoring responses will assist with assessing salinisation risks.

It is anticipated that the installation of a regulating structure will have several consequences on wetland EC. Firstly, the structure itself is likely to impede diluting exchanges between Finnigans Creek and the Lagoon, allowing EC to increase at sites closer to the regulator. Secondly, as the wetland dries evaporative concentration is likely to elevate salt concentrations throughout the wetland to levels exceeding those currently encountered. In the final stages of drying the concentration of salts can be extreme with hyper-saline conditions often developing within remnant pools (*e.g.* Scholz and Gawne 2004a). Such changes are likely to impact on both physical processes (Grace *et al.* 1997, Baldwin *et al.* in review) and the structure of aquatic communities (Hart *et al.* 1991, Bailey and James 2000, Nielsen *et al.* 2003, Baldwin *et al.* in review). Thirdly, post-inundation EC may be elevated above that of the floodwaters by the intrusion and surface encrustation of saline groundwater during the drying phase. And fourthly, the hydraulic isolation of Horseshoe Lagoon between inflow events is likely to reduce the potential for salt to be exported from the wetland, possibly resulting in the accumulation of salt within the wetland over successive wet/dry cycles.

Turbidity

Hypothesis E3: Re-instating a more frequent wet/dry regime to Horseshoe Lagoon will result in turbidity increasing during the initial drying phase in the presence of large carp.

Hypothesis E4: Re-instating a more frequent wet/dry regime and excluding of large carp from Horseshoe Lagoon will result in a decrease in turbidity.

Turbidity is a measure of light attenuation in water and primarily reflects the amount of suspended particulate matter in the water column. It is the main determinant of photic depth and, especially at high levels, has the potential to limit primary production and thus ecosystem structure and function. Turbidity is influenced by a number of factors such as the quality of source waters, the capacity for entrained particles to settle, and the susceptibility of the sediments to re-suspend. The susceptibility of sediments to re-suspension is influenced by sediment structure, water depth, and bioturbation, by carp for example. Numerous studies have attributed increases in turbidity to the benthic feeding behaviour of carp (Lamarra 1975, Meijer *et al*. 1990, King *et al.* 1997, Robertson *et al.* 1997), and this is likely also true for Horseshoe Lagoon. No environmental objective was developed for surface water turbidity (SKM 2004b). However, monitoring responses to management actions will assist with assessing changes in primary production and wetland trophic structure.

Little turbidity data is available for Horseshoe Lagoon. That which is available (May–November 2003; range 170–>999 NTU) indicates that turbidity was consistently greater than that present within the Murray River upstream of Lock 9 (difference range 132–>966 NTU) (SKM 4004b, MDFRC unpublished data). It is anticipated that the imposition of a more frequent wet/dry cycle and the exclusion of large carp will have several consequences on wetland turbidity.

Firstly, the structure itself is likely to impede diluting exchanges between Finnigans Creek and the Lagoon, allowing turbidity to increase at sites closer to the regulator. Secondly, during the initial drying episode carp densities within the receding pool (in the absence of significant avian predation) are likely to increase, increasing their impact on turbidity. Thirdly, the exposure of sediments as the wetland dries promotes their consolidation (Van der Wielen *et al*. in press) and the establishment of terrestrial vegetation (Scholz and Gawne 2004a). Both processes reduce the susceptibility of the sediments to re-suspension when inundated. As the efficacy of both processes in reducing post-inundation turbidity is related to the duration of exposure, it is likely that the influence of these will increase with distance from the regulator. Finally, the exclusion of large carp from Horseshoe Lagoon on its re-filling is likely to eliminate a potentially significant source of disturbance to wetland sediments.

Nutrients (nitrogen and phosphorus)

Hypothesis E5: Re-instating a more frequent wet/dry regime to Horseshoe Lagoon will result in water column N and P concentrations increasing during the initial drying phase in the presence of large carp.

Hypothesis E6: Re-flooding of Horseshoe Lagoon after a drying event will stimulate the release of a short-lived large pulse of N and P from the sediments.

Hypothesis E7: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in a decrease in post-flood pulse water column N and P concentrations.

The rehabilitation of nutrient cycling has been identified as a key ecological objective for Horseshoe Lagoon. This objective is to be met by re-instating a wet/dry regime and by excluding large adult carp from the wetland. However, no data is available to indicate the extent to which nutrient cycling within Horseshoe Lagoon has been impacted as a consequence of increased permanency of water. It is anticipated that the imposition of a more frequent wet/dry cycle and the exclusion of large carp will have several consequences on suspended nutrient concentrations.

Firstly, the installation of a regulating structure is likely to impede the nutrient exchanges between the wetland and Finnigans Creek and to eliminate such exchanges altogether when this structure is closed. No information is available to suggest either the magnitude or direction of these exchanges under the current operating scenario.

Secondly, as the wetland dries, nutrient concentrations are likely to increase to levels exceeding those currently encountered due to evaporative concentration (*e.g.* Scholz *et al.* 2002, Scholz and Gawne 2004a) and turbation of the sediments. For example, as the wetland dries initially, carp densities within the receeding pool are likley to increase (in the absence of significant avian predation), increasing the level of disturbance of the sediments, and thus the flux of nutrient-rich interstitial water and particle associated bio-available phosphorus from the sediments. Increases in fish densities will also increase the relative significance of excretion to the suspended nutrient pools (Lamarra 1975, Meijer *et al*. 1990, King *et al.* 1997, Robertson *et al.* 1997).

Thirdly, the exposure of wetland sediments during drying episodes is likley to stimulate the mineralisation of nutrients in the sediments (McComb and Qiu 1998, Baldwin and Mitchell 2000). Depending on sediment organic matter content and the duration of sediment exposure, a potentially significant pool of mineralised (or bio-available) nutrients may be flushed from the sediments on its subsequent inundation (Baldwin 1996, McComb and Qiu 1998, Baldwin and Mitchell 2000). Flooding is commonly associated with increased system productivity fuelled by an initial pulse of nutrients, derived from both the inflowing water and sediments releases (Junk *et al.* 1989, Scholz *et al.* 2002). This nutrient pulse is, however, short-lived (weeks) with water column nutrient concentrations declining steadily as nutrients become more tightly coupled with biotic and abiotic uptake and release processes (*e.g*. Baldwin and Mitchell 2000, Scholz *et al*. 2002). We therefore anticipate that flooding of Horseshoe Lagoon after a complete drying event will stimulate an initial pulse of increased water column N and P concentrations.

And fourthly, the exclusion of large carp following the re-flooding of the Lagoon will likely reduce the potential for disturbance of the sediments, reducing both turbidity and suspended nutrient concentrations.

Wetland vegetation

Aquatic vegetation

Hypothesis F1: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in the aereal extent of aquatic vegetation.

Hypothesis F2: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in the distribution of aquatic vegetation.

Macrophytes play an important role in aquatic ecosystems (refer to review by Carpenter and Lodge 1986). They may contribute to reducing turbidity, provide an important conduit for the transfer of oxygen to the sediments during plant growth, and nutrients to the water column during decay (Boon and Sorrell 1991), and provide substrata for the development of biofilms, which provide an important food resource for a wide range of invertebrates and fish (Cattaneo 1983, Bunn and Boon 1993). Macrophytes also provide important refuges for invertebrates and fish (Lillie and Budd 1992). The annual cycles of macrophyte growth and die-back may also have important consequences on water quality (dissolved oxygen, pH) and the accrual of organic matter on the sediments (Carpenter and Lodge 1986).

The distribution of aquatic vegetation is strongly influenced by the prevailing hydrology (Welcomme 1979, Brock 1986). For example, longer periods of inundation and decreases in the variability of water levels, such as has occurred within Horseshoe Lagoon, have been linked with reductions of both diversity and the width of the vegetated littoral zone (Brock and Casanova 1991, Poiani and Johnson 1993, Walker *et al.* 1994, Nielsen and Chick 1997, Blanch *et al.* 1999, 2000). Prolonging the period of inundation has also been shown to disadvantage the development of ephemeral or terrestrial plant taxa (Maher 1984, Briggs and Maher 1985). The establishment and persistence of aquatic vegetation may also be influenced by turbidity, which influences light availability, by the intensity of grazing pressure, and by the presence of carp through their destructive feeding habits (Hume *et al.* 1983, Roberts *et al.* 1995).

We anticipate that the re-instatement of a more frequent wet/dry cycle and the exclusion of large carp from Horseshoe Lagoon will increase the capacity for aquatic vegetation to establish, increasing their contribution to wetland function. Drying of the wetland may potentially change the composition of the macrophyte community. SKM (2004b) have indicated that elimination of the persistent pool will likely reduce the abundance of emergent species such as sedge (*Cyperus* sp.), river club-rush (*Schoenoplectus* sp.) and cumbungi (*Typha domingensis*), that water couch (*Paspalum distichum*) will become less abundant and be replaced by couch (*Cynodon dactylon*), that spikerush (*Eleocharis* sp.) are likely to persist, and that aquatic herbs such as water primrose (*Ludwigia peploides*) and water milfoil (*Myriophyllum* sp.) will be present only during late floods and flood recession. Despite such predictions, we recognise that observed responses of the macrophyte community after imposing changes in hydrology and removing carp may be confounded by the diversity and availability of plant propagules (either already present within the wetland or colonisers arriving via water or aerial pathways), the timing of the proposed disturbance (seasonality of germination and establishment), and grazing interactions. These issues will not be examined directly as part of the current program.

Cumbungi

Hypothesis F3: Drying (exposing) cumbungi in Horseshoe Lagoon between December and June will reduce stand vigour.

Cumbungi (*Typha domingensis*) is one of the most productive emergent native macrophytes, forming tall (2-4 m) dense mono-specific stands in water up to 2 metres deep. Cumbungi has the ability to expand by rhizome extension throughout the year, although most rapid rhizome extension typically occurs after mid-summer once the canopy has reached maximum biomass and flowering has occurred (Roberts and Ganf 1986). Cumbungi grows vigorously in hydraulically stable environments and as such has the potential to be used as an indicator of environmental (hydraulic) change (*e.g.* Roberts and Wylks 1992, Roberts and Marston 2000).

Cumbungi can tolerate exposure (dry conditions) only for short periods (3-4 months in summer-autumn) once the growing season is over without loss of vigour. However, its rhizomes can survive longer periods of exposure (possibly a few years) where soil moisture is sufficient to prevent desiccation. After flowering in spring-summer, germination and seedling establishment is dependent on the availability of water (depth < 5 cm) (Froend and McComb 1994, Roberts and Marston 2000). Because of these attributes, water management initiatives aimed at controlling/eliminating cumbungi are likely to be most effective where they induce stress during the most active period of growth, thereby reducing rhizome development and the establishment of shoots and flowers. This may be achieved by drying the wetland over the winter – mid-summer period. Whilst cumbungi may also be stressed by prolonged exposure to water depth >2 m, no opportunity exists within Horseshoe Lagoon to achieve such conditions given the maximum stage capacity of the proposed regulator (28.3 m AHD) (SKM 2004b). We therefore anticipate that the drying of Horseshoe Lagoon during the suggested period will reduce stand vigour, measured as the rate of stand expansion, shoot height and density and reproductive stem density. However, we note that using water management to control cumbungi is likely to require repeated drying episodes.

Red gums

Hypothesis F4: Drying Horseshoe Lagoon during spring-autumn will reduce mean health of red gum saplings present within the current wet/dry zone.
Hypothesis F5: Inundating the wet-dry zone of Horseshoe Lagoon during autumn-winter will reduce mean health of red gum saplings present within the current wet/dry zone.

Horseshoe Lagoon is fringed on higher areas by black box (*Eucalyptus largiflorens*) and on lower areas by red gum (*Eucalyptus camaldulensis*). Under the current hydraulic regime, soil moisture in the lower parts of the wet-dry littoral zone favours the expansion of red gums into this zone. Whilst red gums play an important functional role within floodplain and wetland systems through their provision of carbon (leaf litter) and habitat for fauna (*e.g.* Briggs and Maher 1983, Briggs *et al.* 1997, Baldwin 1999), regeneration especially in the wet-dry littoral zone of Horseshoe Lagoon is considered a threat to the biodiversity and character of the wetland (SKM 2004b). Accordingly, reducing the abundance/extent of red gum regeneration within the wet-dry zone has been identified as a key environmental objective (SKM 2004b).

Conceptually, the recruitment of red gum seedlings may be controlled through hydraulic manipulation by subjecting seedlings to periods of flooding/immersion and/or desiccation. Seedling mortality may be achieved by complete submergence (carbon dioxide deprivation) or by being shallow flooded (oxygen deprivation of root zone). For seedlings that are 6-9 months old mortality may be achieved either by flooding to a depth of 50–100 cm for 4–6 weeks between June and October, or by shallow flooding for 12–26 weeks during autumn and winter (but not summer) (SKM 2004b). Management of lakebed red gums by desiccation may be achieved by drying the wetland for 6 months between October and June.

These recommendations are more likely to be effective in the control of recent recruits (< 1 year old) than of deeper rooted older red gums that are present within the wet/dry zone. Other methods of tree removal will need to be identified if no response in wet/dry zone red gums to hydraulic manipulation can be detected over successive wet/dry cycles.

<u>Algae</u>

Hypothesis G1: The re-flooding of Horseshoe Lagoon after a drying event will stimulate an initial short-lived pulse of increased algal biomass.

Hypothesis G2: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe lagoon will result in an increase in the ratio of phytobenthic to phytoplanktonic biomass after the initial flood pulse.

Algae represent a key component of total wetland primary production (*e.g.* Wetzel 1964, Hargrave 1969, Wetzel *et al.* 1972). This is especially true in systems where macrophytes are absent or restricted to a relatively small area. Algae occur either suspended within the water column (phytoplankton) or attached to substrata (epiphyton and phytobenthos). Numerous studies have shown production by either fraction to be influenced by light availability, nutrients and by substratum quality (*e.g.* Turner *et al.* 1983, Cox 1988, 1990a,b,c, Reynolds and Descy 1996, Havens *et al.* 1998), although taxon specific responses vary greatly (Reynolds 1997, Huszar and Caraco 1998).

In its current permanently inundated state, the presence of carp has likley contributed to increasing turbidity and suspended nutrient concentrations (King *et al.* 1997, Robertson *et al.* 1997). This is likely to directly influence the relative abundances of planktonic and benthic algae in several ways. Firstly, benthivory of carp increases the potential for benthic algae to become light limited by physically disturbing the sediments and reducing the penetration of light. And secondly, physical disturbance of the water column and sediments by carp maintains phytoplankton in suspension, increasing their retention within the photic zone, and increasing their access to light and nutrients. Accordingly, phytoplankton production is likely to be elevated and phytobenthic production inhibited (*cf.* Ogilvie and Mitchell 1998).

The reintroduction of more frequent wetland drying events and the exclusion of carp from Horseshoe Lagoon are likely to influence both phytoplankton and phytobenthic production in different ways throughout the wet/dry cycle. During the drying phase, increases in turbidity and nutrient concentrations (discussed earlier) are likely to favour phytoplankton production. This in conjunction with evaporation driven increases in phytoplankton density increases the potential for algal blooms to establish, especially during the warmer months when rates of production tend to be greatest (*e.g.* Scholz and Gawne 2004a). Sediment consolidation during the dry phase (Van der Wielen in press) in conjunction with the exclusion of carp on filling is likely to result in greater water column transparency and an increase in sediment stability. Both of these are likely to favour the establishment and production of benthic biofilms. With the exception of an initial post-inundation flush of nutrients from the sediments and concomitant flush of phytoplankton production, much of the post-inundation release of nutrients from the sediments is likely to be assimilated by the benthic biofilms before it reaches the water column (*cf.* microbial loop; Boulton and Brock 1999).

Several studies have indicated that rates of benthic production can easily exceed that of the phytoplankton in shallow wetlands (Wetzel 1964, Wetzel *et al*. 1972, Stanley 1976, Loeb *et al.* 1983) and provide a major source of energy driving food webs (Bunn and Davies 1999).

The anticipated shift in algal production from planktonic to benthic arising from the proposed management actions is also likely to influence the accessibility of food resources available to aquatic grazer populations, and thereby influence wetland trophic structure.

Invertebrates

Zooplankton

Hypothesis H1: The re-flooding of Horseshoe Lagoon will stimulate an initial short-lived pulse of increased zooplankton density lasting approximately 4 weeks.

Hypothesis H2: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in the density of zooplankton after the initial flood pulse.

Hypothesis H3: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in a shift in zooplankton community composition after the initial flood pulse.

Zooplankton provide major links in aquatic food chains by facilitating the transfer of nutrients, carbon and energy between bacteria, algae and higher consumers, such as fish and water fowl (*e.g.* Boon and Shiel 1990, Boulton and Jenkins 1998, Humphries *et al*. 1999). Because of this, zooplankton plays a key role in structuring ecosystem function (*e.g.* Shapiro *et al*. 1975, Lazzaro 1997).

Zooplankton community structure is influenced by many factors, including life history characteristics, food availability, predation pressure, water quality, habitat diversity and complexity, and exchanges between wetland and riverine environments (Wiggins *et al.* 1980, Shiel and Walker 1984, Williams 1985, Shiel 1985, 1986, 1995). Many of these factors are influenced by season and in ephemeral systems by drying and flooding (*e.g.* Scholz and Gawne 2004a). Despite potentially complex interactions between these factors, generalised responses of zooplankton to the re-instatement of a wet/dry regime in Horseshoe Lagoon are to be expected.

We anticipate that the inundation of Horseshoe Lagoon after it has dried will stimulate an initial pulse of zooplankton density driven initially by floodwater borne immigrants which are subsequently replaced by emergents from the sediments during the first month after flooding (Maher and Carpenter 1984, Boulton and Lloyd 1992, Jenkins and Boulton 2003). Such pulses in post-inundation zooplankton density are thought to increase feeding opportunities and the recruitment success of native fish species and waterbirds (*e.g*. Cushing 1975, Maher 1984, Maher and Carpenter 1984, Crome 1986, Cushing 1990).

The duration of the initial post-flood pulse of zooplankton production has been shown to vary between wetlands, but generally decline over the first month of inundation as the intensity of trophic interactions increase (*e.g.* Carpenter *et al.* 2001, Scholz and Gawne 2004a,c). The magnitude of initial increases in zooplankton density after flooding has been linked to both the frequency of inundation and to the duration of the preceding dry period. For example, Boulton and Lloyd (1992) reported that wetlands that experience frequent (annual) episodes of flooding tend to be more productive than those that flood only infrequently (once in 22 years). Whilst longer dry phases tend to reduce the viability of aestivating individuals and of eggs and cysts deposited in the sediments, studies in dryland systems indicate that resting eggs are extremely long lived and eggs can survive dry periods of 20, 50 and even 100 years and still emerge once the sediments are flooded (Hairston *et al*. 1995, Jenkins and Briggs 1997, Jenkins and Boulton 1998).

Invertebrate productivity (Brinson *et al.* 1981, Maher and Carpenter 1984, Briggs and Maher 1985) and diversity (Boulton and Jenkins 1998) have been shown to suffer in response to reductions in the duration of wetland drying and to increases in the permanency of water, such as has occurred at Horseshoe Lagoon. This may be due, in part, to 'bottom-up' effects through reductions in nutrient availability and primary production, and to the loss of lake drying associated environmental cues needed by invertebrates to stimulate developmental shifts.

Based on these considerations, we hypothesized that re-imposing a more frequent wet/dry regime on Horseshoe Lagoon would stimulate an initial post-inundation pulse of increased zooplankton density, and greater post-flood-pulse zooplankton density relative to that of the current 'permanent system'. We also hypothesised that the re-imposition of a wet/dry cycle would mediate a shift in zooplankton communities towards taxa more typical of ephemeral water bodies. Such taxa tend to have more flexible life history strategies, increasing their resilience to disturbance events, such as drying (*e.g.* Baird *et al.* 1987).

Macro-invertebrates

Hypothesis H4: The re-flooding of Horseshoe Lagoon will stimulate an initial short-lived pulse of increased macro-invertebrate density lasting approximately 8 weeks.

Hypothesis H5: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in a longer term increase in the density of macro-invertebrates after the initial flood pulse.

Hypothesis H6: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in a shift in macro-invertebrate community composition after the initial flood pulse.

Macro-invertebrates are important consumers within wetland ecosystems, occupying a range of functional groups (*e.g.* grazers, detritivores, filter feeders, predators) and provide the principal food source for many vertebrates such as fish and birds (Bunn and Boon 1993). Macro-invertebrates have proven effective indicators of ecosystem health in stream environments (Cranston *et al.* 1986) and their use as indicators in wetlands has been advocated (*e.g.* Davis *et al.* 1993).

Macro-invertebrate community structure is influenced by many factors, including life history characteristics, water quality, habitat quality, and exchanges between wetland and riverine environments (Wiggins *et al.* 1980, Shiel and Walker 1984, Williams 1985) and by trophic interactions (*e.g.* Carpenter and Kitchell 1988, Carpenter *et al.* 2001). Numerous studies have shown hydrology to be a major determinant of macro-invertebrate community structure and productivity (*e.g.* Wiggins *et al.* 1980, Bataille and Baldassarre 1993, Jeffries 1994, Leslie *et al.* 1997). Although differences in the structure of macro-invertebrate communities of ephemeral and more permanent wetlands have been documented (*e.g.* Schalles and Shure 1989, Batzer and Resh 1992, Davis *et al.* 1993, Jeffries 1994), little consensus exists as to whether ephemeral systems are intrinsically more productive habitats for macro-invertebrates than more permanent systems (Batzer and Wissinger 1996). This lack of consensus is likely due to un-controlled-for differences in the strength of trophic interactions between the systems studied. Despite potentially complex interactions between determinants of macro-invertebrate community structure, generalised responses of macro-invertebrates to the re-instatement of a wet/dry regime in Horseshoe Lagoon and the exclusion of carp are to be expected.

We anticipate that the inundation of Horseshoe Lagoon after a drying event will stimulate an initial pulse of macro-invertebrate abundance driven by two processes; the rapid recolonisation of newly created habitat and favourable trophic interactions. Firstly, recolonisation may occur via several pathways, such as emergence from the sediments of desiccation resistant eggs and larvae or desiccation resistant adults, passive movement with the incoming waters, active migration, chance introduction by other animals, and aerial dispersal (Talling 1951, Wiggins *et al.* 1980, Batzer and Wissinger 1996, Hillman and Quinn 2002). And secondly, trophic cascade or food web models suggest that increases in the availability of food resources (*e.g.* zooplankton; Maher and Carpenter 1984, Boulton and Lloyd 1992) combined with reductions in predation pressure that are commonly encountered during the initial post-inundation period (*e.g.* Lake *et al.* 1989, Batzer and Wissinger 1996, Battle and Golladay 2001) will favour the establishment of abundant macro-invertebrate communities. The little data that is available suggests that initial post-flood increases in macro-invertebrate density may persist for as little as 1-2 months (*e.g.* Scholz and Gawne 2004c) or for as long as 2 years (*e.g.* Maher and Carpenter 1984). Subsequent declines in macro-invertebrate density are anticipated once the availability of food resources decreases and the intensity of biotic interactions (*e.g.* competition, predation) increases.

The presence of carp has been shown to exert a significant negative pressure on invertebrate abundances, either directly through predation or indirectly through habitat modification (*e.g.* Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al*. 1994, Tatrai *et al*. 1994). We anticipate that the removal of carp from the system will result in a longer term (*i.e.* after an initial post-inundation pulse) increase in macro-invertebrate abundances within the wetland and that this may also be reflected in community composition.

Fish

Hypothesis I1: The initial drying of Horseshoe Lagoon will result in an increase in the relative abundance of large carp.

Hypothesis I2: The initial drying of Horseshoe Lagoon will result in an increase in the relative biomass of large carp.

Hypothesis I3: The initial drying of Horseshoe Lagoon will result in a progressive reduction in the relative abundance of small fish populations.

Hypothesis I4: The initial drying of Horseshoe Lagoon will result in a progressive reduction in the relative biomass of small fish populations.

Hypothesis I5: The initial drying of Horseshoe Lagoon will result in a progressive reduction in the diversity of small fish populations.

Hypothesis I6: The operation of carp screens during re-filling will result in a decrease in the relative abundance of large carp in Horseshoe Lagoon.

Hypothesis I7: The operation of carp screens during re-filling will result in a decrease in the relative biomass of large carp in Horseshoe Lagoon.

Hypothesis I8: Inundating Horseshoe Lagoon after a drying event will stimulate spawning in small-bodied native fish species.

Hypothesis I9: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in the relative abundance of small fish.

Hypothesis I10: Re-instating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in the relative biomass of small fish.

Fish provide an important link in wetland food webs through their consumption of invertebrate and fish prey and their consumption by birds. Wetland hydrology plays an important role in structuring fish assemblages through its influence on ecosystem productivity, habitat availability and connectivity, and cues for fish spawning. Native fish throughout the Murray-Darling Basin have been severely impacted by altered flow regimes, the loss of habitat, and barriers to passage (MDBMC 2002). Altered flow regimes, in particular, are thought to have impacted on the distribution of native fish species and to have favoured the expansion of alien species, such as carp (*Cyprinus carpio*) and mosquitofish (*Gambusia holbrooki*) (Harris and Gehrke 1997). Whilst fish distributions within the Basin have been reported (Llewellyn 1983, Harris and Gerhke 1997), most fish work in the past has focused on riverine populations. Very little information is currently available for wetland fish communities and their responses to wetting and drying (*e.g.* Scholz and Gawne 2004a,b,c).

The fish community of Horseshoe Lagoon is dominated by the exotic carp, goldfish (*Carassius auratus*) and mosquitofish (*Gambusia holbrooki*) and by natives species including bony herring (*Nematalosa erebi*) and carp gudgeon (*Hypseleotris* spp.) (Ho *et al*. 2004). Carp are widely considered responsible for the degradation of aquatic ecosystems. Although the evidence is fragmented and sometimes contradictory, carp have been implicated in the demise of native fish stocks by reducing spawning site availability (*i.e*. destruction of macrophytes; Hume *et al*. 1983, Brown 1996), by reducing the efficiency of visual feeding (*i.e*. increasing turbidity; Lamarra 1975, Roberts *et al.* 1995), and by competing for food resources (Hume *et al.* 1983, Fletcher 1986, Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al*. 1994, Tatrai *et al*. 1994).

The ecological objectives to ‘rehabilitate habit and feeding opportunities’ for the small native fish community at Horseshoe Lagoon and to ‘reduce carp abundance’ will be met by re-imposing a more frequent wet/dry cycle and by excluding large carp on re-filling (SKM 2004b). During the initial drying phase we anticipate that carp densities will increase (in the absence of avian predation) and that their adverse impacts on small fish will increase. In addition to establishing the efficacy of the regulating structure in excluding large carp, we will monitor fish populations directly as relative abundance rather than measuring habitat and feeding opportunities as stated in the objective. We anticipate the exclusion of large carp from the wetland as it re-fills to have beneficial impacts on the abundance of smaller native fish populations for the reasons stated above. We also anticipate that flooding after a dry event will stimulate increases in the relative abundances of small fish populations due to factors including initial increases in post-flood food abundances, increases in habitat availability, and by cueing spawning (*e.g.* Meredith and McCasker in prep).

Waterbirds

Hypothesis J1: Reinstating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in the cumulative species richness of waterbirds.

Hypothesis J2: Reinstating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in post inundation abundances of generalist filter feeding and dabbling duck birds (all birds of the genus Anas, Pink eared-duck, Hardhead).

Hypothesis J3: Reinstating a more frequent wet/dry regime and excluding large carp from Horseshoe Lagoon will result in an increase in post inundation abundances of piscivorous birds that prefer smaller fish (Caspian Tern, White-faced heron, Great Egret, Little Egret).

Hypothesis J4: Bird assemblage composition of Horseshoe Lagoon will change over time, becoming increasingly dominated by piscivorous species after re-flooding.

Waterbird abundance is determined by the reproductive success of adults and the survival of post-juvenile birds. The breeding and survival of most waterbirds in the Murray-Darling Basin have been linked to cycles of flooding and drying (Briggs 1990); with most species being capable of moving large distances between wetlands as local breeding and feeding opportunities fluctuate. Almost all common waterbirds of the Murray-Darling Basin breed following wetland flooding, with some also breeding seasonally, usually in spring. In only 2 species (both ducks) is breeding always seasonal and apparently unaffected by wetland hydrology (Briggs and Lawler 1991).

Flow regulation within the Murray-Darling Basin has altered the temporal and spatial distribution of inundated floodplain wetlands available to waterbirds. Regulation has permanently inundated many formerly intermittent wetlands, and reduced the average duration of inundation for many others. Whilst this has increased the availability of survival habitat for waterbirds at the landscape scale, it has effectively reduced the availability of breeding habitat. Thus, the overall effect of river regulation on waterbirds is likely to be reduced recruitment of young, but enhanced survival of adults (Briggs and Lawler 1991). Because of this, it is important that the number of breeding opportunities for waterbirds within the Murray-Darling Basin be increased. As is discussed below, re-instating a more frequent wet/dry cycle to Webster's Lagoon is likely to achieve this objective.

The nesting requirements of Australian waterbirds vary (Serventy 1985), but all require adequate food to satisfy the energetic and other costs of nest finding or building, the laying and incubating of eggs, and the rearing of young. Whilst these costs may be partly provided for by the accumulation of body fat and/or protein reserves prior to laying (*e.g.* Thomas 1988, Norman and Hurley 1984, Briggs 1988), most waterbirds of the Murray-Darling Basin require readily available food of high quality at nesting time for successful breeding (Briggs and Lawler 1991). Many studies have shown the inundation of previously dry wetlands to initiate a highly productive succession of potential food resources (*e.g.* invertebrates and fish), which attracts many waterbird species and stimulates breeding (*e.g.* Maher 1984, Maher and Carpenter 1984, Crome 1986, 1988). Recently flooded wetlands thus provide a potentially important window of opportunity for breeding by many waterbird species. Over time after inundation and as wetlands begin to dry; changes in the composition and availability of food resources tend also to stimulate changes in the structure of waterbird assemblages (Scott 1997), increasing the cumulative diversity of species utilising the wetland.

In addition to food availability, post-inundation waterbird breeding success is reliant on the persistence of adequate food resources and of nesting habitat throughout the breeding (rearing) cycle. One of the objectives identified by SKM and Roberts (2003) is to provide occaisional breeding and roosting habitat for colonial waterbirds on Wallpolla Island. There are no records of colonial waterbirds using Horseshoe Lagoon. The current water regime of Horseshoe Lagoon is not tailored for colonial waterbird breeding. Many colonial water birds require inundation under mature red gum trees for between 3-6 months and up to 10 months for successful breeding (Briggs and Thornton 1999). This does not currently occur within Horseshoe Lagoon. Creating suitable nesting sites in Horseshoe Lagoon would require the establishment of large areas of Lignum or mature trees (*e.g.* red gums, *cf.* Briggs and Thornton 1999) within the wet/dry zone of the wetland and/or increasing the capacity of the regulating structure to retain higher flood levels. Consequently no objective for providing for the breeding requirements of colonial waterbirds has been included.

For purposes of developing testable hypotheses relating to the responses of waterbirds to the re-imposition of a more frequent wet/dry regime in Horseshoe Lagoon, bird species were divided into guilds (groups of species that exploit similar resources in a similar manner, but that are not necessarily closely related taxonomically). Two guilds were identified; the generalist filter-feeders and dabbling ducks and the piscivores that prefer smaller fish.

We anticipate that the creation of additional nichès in response to the re-instatement of more frequent wet/dry cycles in Horseshoe Lagoon will stimulate an increase in the cumulative species richness of waterbirds utilising the wetland. Further, successional shifts in the availability and distribution of major food types and habitat throughout the wet/dry cycle will be reflected by shifts in the structure of waterbird assemblages. We anticipate that on re-flooding of the wetland, an initial burst of wetland productivity (0.5-2 months) will stimulate increases in the abundance of waterbirds, particularly the generalist filter-feeding and dabbling duck guild. This guild comprises all native species of Anas and pink-eared duck (*Malacorhynchus membranaceus*) and the hardhead (*Athya australis*) (*cf.* groupings of Roshier *et al*. 2002). We also anticipate a directional shift in waterbird assemblage composition over the 6-8 month of inundation, with the establishment of species that eat small fish as a major part of their diet, particularly Caspian tern (*Sterna caspia*), white-faced heron (*Egretta novaehollandiae*), great egret (*Egretta alba*), and perhaps little egret (*Egretta garzetta*). It is possible that abundances of these taxa will increase initially as wetland drying increases the density of small fish within the receding pool, and subsequently fall as fish numbers decline in response to elevated predation pressure and or changes in water quality. Wetland drying is also likely to increase feeding opportunities for scavenging land and waterbirds as fish become stranded. We also anticipate the development of a longer term shift in assemblage composition if the 'structural biophysical' nature of Webster's Lagoon steadily changes, *e.g.* cover-dependent wetland-dependent species may colonise if dense reed beds and sedge beds develop. Species likely to establish as a consequence of such changes include; the clamorous reed warbler (*Acrocephalus stentoreus*), little grassbird (*Megalurus gramineus*), crakes and rails, and grazing waterfowl.

As wetland waterbird populations are highly mobile and are influenced by processes occurring at landscape scales, they are inherently very variable. Whilst we anticipate that we will be able to detect short-term responses in waterbird numbers between dry and flooded phases, the ability to reliably detect such responses in waterbird assemblages to changes in hydrology will likely require the collection of long term data extending over multiple wet/dry cycles.

Waterbird abundance is particularly expensive in both time and resources to monitor adequately. As no control site exists for Horseshoe Lagoon, we recommend that the monitoring of waterbirds at Horseshoe Lagoon be foregone in favour of concentrating effort at Webster's Lagoon for which a suitable experimental control exists permitting the establishment of cause and effect relationships, and for which we would expect similar post-inundation responses of waterbird populations.

Turtles

Hypothesis K1: Re-instating a wet/dry phase and the use of carp screens upon refilling will impede the recovery of aquatic turtles in Horseshoe Lagoon.

Three native species of freshwater turtle are known to occur within the Mallee region; the Broad-shelled river turtle (*Chelodina expansa*), the Eastern long-neck turtle (*Chelodina longicollis*), and the Murray turtle (*Emydura macquarii*) (NRE 2001). However, none were recorded during a recent survey of Horseshoe Lagoon (Ho *et al.* 2004).

Broad-shelled river turtles are considered threatened under the FFG Act and endangered under the ALTVFV (DSE 2003, 2004a), although this status may partly reflect the species' secretive nature (Spencer 2001). Broad-shelled turtles are known to inhabit floodplain wetlands, but are principally inhabitants of river environments. This species is a selective predator which conceals itself in mud and uses its long neck to catch fish, yabbies and other prey (Scott 2001). Because of their threatened status there is a statutory requirement that the impact of changes in flow regime on the populations within Horseshoe Lagoon (if any) be monitored.

The Eastern long-neck turtle is common and widespread both within the Mallee region (Ho *et al.* 2004) and throughout eastern Australia (Scott 2001). This species is thought to be better suited to wetland habitats than to river channel environments because it generally can not catch large prey such as fish and predominantly feeds on small invertebrates that are often more abundant in floodplain wetlands (Scott 2001).

The Murray turtle is thought to be an inhabitant primarily of river environments and is only sometimes present in deep permanent wetlands (Scott 2001).

Freshwater turtles spend most of their time in the water, emerging only to bask, to lay their eggs in a chamber dug in dry ground away from water, or to move between wetlands (Boulton and Brock 1999). Movement between wetlands may be stimulated by competition with fish for food resources (Chessman 1984), or the loss of aquatic habitat as wetlands dry. The pulse of productivity generally associated with wetland re-flooding is thought to stimulate the dispersal of turtles back to the wetland (Boulton and Brock 1999). The re-establishment of turtle populations after a wetland drying event may occur via both overland and aquatic pathways. The relative importance of either pathway for either species has not been determined. We, therefore, decided to test whether the use of carp screens would inhibit the re-colonisation of Horseshoe Lagoon by Eastern long-neck and Broad-shelled turtles.

4.3 Response variables

Response variables to be measured as part of the monitoring program were identified for each of the hypotheses stated in the previous section. Standard sampling and analytical protocols for each response variable are listed in Table 4.2.

Response variable	Field and laboratory method
Surface water	
Height (mAHD)	Automated logging using Odyssey pressure sensors (Dataflow Systems Pty Ltd, Christchurch New Zealand).
Area and volume	Capacity tables constructed from profile surveys.
Groundwater	
Depth (m)	Measured from piezometers located along four transects within the lagoon: immediately downstream of the proposed regulator, immediately upstream of the proposed regulator, midway along the currently wetted area, and at the top of the currently wetted area. Piezometers to be spaced approximately 50 m apart, with transects extending from the centre of the lagoon into the large fringing red gums. Surveying of piezometers to mAHD will enable comparisons of water tables.
Electrical conductivity (EC; $\mu S\ cm^{-1}$)	EC (standardized to 25 ^oC) of water samples extracted from piezometers measured in the field using a U-10 multi-probe (HORIBA Ltd., Australia).
Sediments	
Electrical conductivity (EC; $\mu S\ cm^{-1}$)	EC (standardized to 25 ^oC) using a U-10 multi-probe (HORIBA Ltd., Australia) of soil samples collected at depth intervals of 20 cm (down to 1.0 m) and at intervals of 50 cm down to the water table. 10 replicate sediment cores to be taken within 50 m of the three transects within the lagoon.
Coarse particulate organic matter (CPOM; g CPOM l^{-1})	After processing and the removal of organisms from macro-invertebrate samples (20m sweep) the remaining organic matter sieved and the >1 mm size fraction dried at 80^{o}C for 48 hours and weighed.
Water quality	
Electrical conductivity (EC; $\mu S\ cm^{-1}$)	EC standardized to 25 ^oC determined *in situ* using a U-10 multi-probe (HORIBA Ltd., Australia).
Turbidity (NTU)	Turbidity of 500 ml sub-surface water samples determined using a U-10 multi-probe (HORIBA Ltd., Australia). Samples diluted 1:3 with distilled water where turbidity exceeds the max. level of detection of 1000 NTU.
Total nitrogen (TN; mg N l^{-1})	Unfiltered sub-surface 200 ml samples stored frozen. TN determined colorimetrically after pre-digestion in $NaOH\text{-}K_2S_2O_8$ and oxidation to nitrate (APHA 1995). Detection limit ± 0.019 mg N l^{-1}.
Total phosphorus (TP; mg P l^{-1})	Unfiltered sub-surface 200 ml samples stored frozen. TP determined colorimetrically using the phosphomolybdate-blue method after pre-digestion in $NaOH\text{-}K_2S_2O_8$ and oxidation to orthophosphate (APHA 1995). Detection limit ± 0.0025 mg P l^{-1}.
Oxides of nitrogen (NOx; mg N l^{-1}),	10 ml of 0.2 µm filtered water samples stored frozen. NOx determined colorimetrically after its reduction to nitrite using a cadmium column (APHA 1995). Detection limit ± 0.003 mg N l^{-1}.
Filterable reactive phosphorus (FRP; mg P l^{-1})	10 ml of 0.2 µm filtered water samples stored frozen. FRP determined colorimetrically using the phosphomolybdate-blue method (APHA 1995). Detection limit ± 0.001 mg P l^{-1}.
Wetland vegetation	
Aereal cover (ha) and distribution	Aereal extent and distribution of aquatic vegetation (*Valisneria* sp.), cumbungi (*Typha domingensis*) and red gum (*Eucalyptus camaldulensis*) determined from aerial photographs (0.2 m pixel resolution) taken during the peak plant biomass period (November-January).
Vigour (cumbungi)	Vigour determined annually between November-January as rate of aereal expansion of stands, above-ground shoot height, shoot and reproductive stem density and the ratio of dead and growing shoots sampled from 4 x 1 m^{-2} quadtrats placed within the mature growth zone of each of three of the largest stands present. Quadrats placed *ca*. 3-4 m into the stand so as to avoid both the outer colonising edge and the inner-most core of the stand.

Table 4.2: Response variables, measurement units and analytical methods used to test hypotheses developed for Horseshoe Lagoon.

Response variable	Field and laboratory method
Wetland vegetation	
Recruitment and vigour (red gum)	Recruitment and vigour determined annually between November-January by recording the elevation (mAHD), leaf condition (colour) and tree height of each individual within 25 m wide quadrats extending vertically from the centre of the wetland up to the mature red gums fringing the wetland. Quadrats positioned horizontally in the centre of each stand of red gum saplings and quadrat positions fixed between years. This data will permit the identification of relationships between age cohorts and condition to the prevailing hydrological regime.
Algae	
Phytoplankton biomass (CHLp; µg CHL l^{-1}, CHL m^{-2})	Phytoplankton biomass measured indirectly as chlorophyll *a*. 500 ml of sub-surface water passed through glass fibre filters (Whatman Pty. Ltd. GF/C). 10 ml 100 % ethanol added to filters and chlorophyll extracted for 24 hours in the dark at 4^oC then at 80^oC for 10 minutes. Supernatant centrifuged and chlorophyll *a* measured spectro-photometrically at 665 nm and 750 nm without acidification (APHA 1995). Chlorophyll concentrations expressed per unit volume and, by recording water depth at time of sampling, per unit area.
Phytobenthos biomass (CHLb; µg CHL m^{-2})	Phytobenthic biomass measured indirectly as chlorophyll *a*. 100 ml 100 % ethanol added to 18 cm^{-2} sediment surface cores (1 cm depth) chlorophyll extracted for 24 hours in the dark at 4^oC then at 80^oC for 10 minutes. Supernatant centrifuged, and chlorophyll *a* measured spectro-photometrically at 665 nm and 750 nm without acidification (APHA 1995). Chlorophyll concentrations expressed per unit area.
Invertebrates	
Zooplankton density (individuals l^{-1}) and species richness	75 l water passed through 50µm mesh size net and concentrates preserved in 70 % ethanol. A minimum of 100 individuals counted from at least 4x1 ml sub-samples using a Sedgewick-Rafter Cell (APHA 1995). Identifications to at least genus according to Shiel (1995) and Ingram *et al*. (1997).
Macro-invertebrate density (individuals l^{-1}) and species richness	20 m benthic sweep samples collected using a 250 µm mesh size net and concentrates preserved in 70 % ethanol. A minimum of 100 individuals or whole sample counted and identified to genus level according to Williams (1980), Hawking and Smith (1997) and Anderson and Weir (2004). This is the rapid bio-assessment protocol of Tiller and Metzeling (1998) (see also Coysh *et al*. 2001). Compared with other commonly used active and passive sampling methods, sweep netting provides similar diversity though lower abundances (Humphries *et al*. 1998).
Fish	
Relative abundance (fish $CPUE^{-1}$) and biomass (kg $ha^{-1}CPUE^{-1}$)	For each sampling event 6 large fyke nets (LFN) (set tangential to shore with cod end away from shore), 6 small fyke nets (SFN) (set parallel to shore with pairs set in opposite directions) and 6 larval light traps (LT) set overnight. Set and pull times recorded and total catch adjusted to 24 hours (= 1 CPUE). Standard lengths (±0.05 mm for LFN and ±0.005 mm for SFN and LT) and weights (±0.5 g for LFN and ±0.0005 g for SFN and LT) recorded for the first 30 individuals (including the largest and smallest individuals caught) per species. Fish identifications follow McDowall (1996). Carp gudgeons identified to genus level only (*i.e*. *Hypseleotris* spp.) owing to the current taxonomic uncertainty at the species level (Bertozzi *et al*. 2000). All large native fish returned. Ethics approval obtained prior to sampling.
Waterbirds	
Abundance (birds $wetland^{-1}$)	Survey protocol to be determined. This will be consistent across Icon Site wetlands to permit comparisons at the regional scale.
Turtles	
Relative abundance, individual movements	Individuals caught using large fyke nets (deployed for fish) marked (using nail polish) before being released. Re-capture used to estimate population size identify individual returns to wetland after drying.

Table 4.2 cont.: Response variables, measurement units and analytical methods used to test hypotheses developed for Horseshoe Lagoon.

4.4 Experimental Design

4.4.1 Design options

Several experimental design options are available to test hypotheses based on the identification of changes in a response variable before and after an environmental disturbance such as wetland drying/flooding. These design options vary both in their complexity and in their ability to detect change. Generally, the incorporation of experimental controls or reference sites within an experimental design is favoured as it permits the elimination of potentially confounding artefacts introduced by the experimental procedure and provides the greatest statistical power to test hypotheses and to demonstrate cause and effect. Where control or reference sites can be identified, experimental designs incorporating Before, After, Control and Impact (BACI) data can and should be used (Underwood 1996). However, this is not always possible, requiring the implementation of alternative approaches that offer either a less scientifically rigorous means of demonstrating an ecological response, or that are capable of assessing the momentary status of key ecological response variables (*i.e.* management trigger levels). Alternatively, where the required level of scientific rigour can not be met it may prove more appropriate to do nothing.

Each of the ecological response hypotheses that were developed for Horseshoe Lagoon in Section 4.2 were assigned to one of four monitoring design categories based on considerations of the availability and/or requirement of a suitable control and the type of output required. These were:

i) Before-After Control-Impact (BACI) design. This approach represents the most scientifically credible monitoring option, and will enable the unambiguous determination of changes to the system that have occurred as a direct result of management intervention by comparing environmental variables in the 'Impact' system with that in an unmanaged 'Control' system 'Before' and 'After' the intervention. Note that because this approach requires a large number of measurements paired in time (also referred to as a BACIP design; Green 1997, Stewart-Oaten *et al.* 1986, Underwood 1991) across 2 (or more) locations, it represents the most expensive of the four options. Note also that if employed across multiple ecological response hypotheses within one system, this option has the advantage of being able to explain failures to detect expected ecological responses by considering conceptual models that link different trophic levels (*i.e.* trophic interactions).

ii) Intervention Analysis (IA) or Before-After (BA) design (*cf.* Box and Tiao 1975). This design option is used in cases where no control site is possible due to the uniqueness of the water regime, but where it is still necessary to monitor/demonstrate the ecological response within a system. Because no control is available, it will not be possible to demonstrate whether or not structural operation has 'caused' any observed 'effect'.due to fact the data are 'pseudoreplicated', *i.e.* any change in the response variable may be due to any number of confounding effects (Hurlbert 1984, Underwood 1996).

iii) Instantaneous adaptive management (IAM) feedback. In cases where the ecological response hypothesis defines a key ecological outcome that is expected from the operation of structures on the floodplain, these ecological responses will be monitored and their real-time analysis will determine structure operation rules. In such cases, no control site is required (or in some cases possible), only the post-impact (After) period is monitored, and no establishment of scientifically credible cause and effect is necessary (or in some cases possible). However, in these cases a hierarchy of ecological outcomes is necessary to determine the sequence of management events.

iv) Do nothing. In some cases, the ecological response hypotheses for systems where no control is possible will be examined in other systems (*i.e.* other Living Murray Icon Sites). Rather than invest considerable resources into monitoring outcomes without being able to demonstrate cause and effect, it may be pertinent to transfer that effort to systems where causality can be determined.

Monitoring to demonstrate cause and effect is expensive in both time and money and cannot reasonably be undertaken for all Icon site wetlands. Nor is it possible. For Webster's Lagoon, which is currently permananently inundated, it was possible to locate a suitable control site and thus to implement a BACI experimental design. However, this was not possible for Horseshoe Lagoon. As we expect ecological responses following the re-instatement of a more frequent wet/dry cycle to the Lagoon to be similar to those that will be tested at Webster's Lagoon using a BACI design, it was decided to limit monitoring effort at Horseshoe Lagoon to the compliance monitoring of wetland water levels using an IAM design and the demonstration of ecological resposes for key management issues (*i.e.* wetland vegetation, carp populations, groundwater) and surface water electrical conductivity and turbidity using an IA design (Table 4.3).

Our conceptual model (Section 4.2) indicates that the ecological response variables identified for each hypothesis (Section 4.3) are likely to vary considerably throughout the wet/dry cycle (Figure 4.4). The testing of hypotheses using an IA experimental design will involve the comparison of variable responses between specific phases of the wet/dry cycle. The IA design identifies treatment effects as a significant interaction between the sources of variation (Before *vs* After) determined using one-factor ANOVAs and repeated measures ANOVAs (RMAs). The hydrological phases relevant to each hypothesis and the statistical treatment of data are presented in Table 4.3. Other design considerations such as the optimisation of temporal and spatial sampling regimes employed have the capacity to greatly affect the ability to detect treatment effects (*i.e.* statistical power). These issues are considered in the following sections.

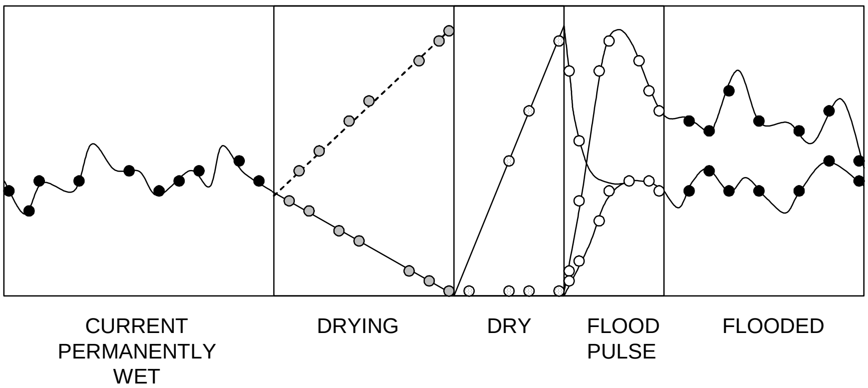


Figure 4.4: Phases of the wet/dry cycle showing conceptualised responses of response variables.

Environmental value	Hypotheses	Establish cause and effect (BACI)	Intervention Analysis (Before-After)	Instantaneous adaptive management	Do nothing	Hydrological phases examined	Statistical analysis
Flow/hydrology	A1	×	×	✓	×	All	No statistics
Lateral connectivity	B1	×	✓	×	×	All	No statistics
Groundwater/salinity	C1	×	✓	×	×	All	Repeated measures ANOVA
	C2	×	✓	×	×	All	Repeated measures ANOVA
	C3	×	✓	×	×	Dry	Repeated measures ANOVA
Sediments	D1	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
Water quality (electrical conductivity)	E1	×	✓	×	×	All	1-factor fixed-model ANOVA
	E2	×	✓	×	×	Permanently flooded v flooded	1-factor fixed-model ANOVA
Water quality (turbidity)	E3	×	✓	×	×	Drying	Repeated measures ANOVA
	E4	×	✓	×	×	Permanently flooded v flooded	1-factor fixed-model ANOVA
Water quality (nutrients)	E5	×	×	×	✓	Drying	Repeated measures ANOVA
	E6	×	×	×	✓	Flood pulse	Repeated measures ANOVA
	E7	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
Wetland vegetation (aquatic vegetation)	F1	×	✓	×	×	Annual assessment when wet	No statistics (no measure of variance possible)
	F2	×	✓	×	×	Annual assessment when wet	No statistics (no measure of variance possible)
Wetland vegetation (cumbungi)	F3	×	✓	×	×	Annual assessment	1-factor fixed-model ANOVA
Wetland vegetation (red gum)	F4	×	✓	×	×	Annual assessment	1-factor non-parametric ANOVA (rank data)
	F5	×	✓	×	×	Annual assessment	1-factor non-parametric ANOVA (rank data)
Algae	G1	×	×	×	✓	Flood pulse	Repeated measures ANOVA
	G2	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
Invertebrates (zooplankton)	H1	×	×	×	✓	Flood pulse	Repeated measures ANOVA
	H2	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
	H3	×	×	×	✓	Permanently flooded v flooded	Multivariate analysis (*e.g.* MDS) ?
Invertebrates (macro-invertebrates)	H4	×	×	×	✓	Flood pulse	Repeated measures ANOVA
	H5	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
	H6	×	×	×	✓	Permanently flooded v flooded	Multivariate analysis (*e.g.* MDS) ?
Fish	I1	×	✓	×	×	Drying	Repeated measures ANOVA
	I2	×	✓	×	×	Drying	Repeated measures ANOVA
	I3	×	×	×	✓	Drying	Repeated measures ANOVA
	I4	×	×	×	✓	Drying	Repeated measures ANOVA
	I5	×	×	×	✓	Drying	Repeated measures ANOVA
	I6	×	✓	×	×	Permanently flooded v flooded	1-factor fixed-model ANOVA
	I7	×	✓	×	×	Permanently flooded v flooded	1-factor fixed-model ANOVA
	I8	×	✓	×	×	Flood pulse	Repeated measures ANOVA
	I9	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
	I10	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA
Waterbirds	J1	×	×	×	✓	All	No statistics (no measure of variance possible)
	J2	×	×	×	✓	Permanently flooded v flood pulse and flooded	1-factor fixed-model ANOVA
	J3	×	×	×	✓	Permanently flooded v flood pulse and flooded	1-factor fixed-model ANOVA
	J4	×	×	×	✓	Flood pulse	Multivariate analysis (*e.g.* MDS) ?
Turtles	K1	×	×	×	✓	Permanently flooded v flooded	1-factor fixed-model ANOVA

Table 4.3: Summary of monitoring approach to be used to test each of the ecological response hypotheses developed for Horseshoe Lagoon, the hydrological phases for which data is required, and the statistical treatment of data.

4.4.2 Temporal sampling

With Before-After type experimental designs (*e.g.* BACI), statistical power is influenced by both temporal and spatial variation. Power may be increased by increasing the number of samples taken per sampling event, by increasing the sampling period, and by increasing the number sampling events.

For hypotheses in which Before and After time periods are to be compared, each sampling event provides only one value. As sub-samples collected from the wetland within sampling events do not add replicates to test the hypothesis about treatment effects, their impact on statistical power is likely to be small, especially where spatial variability is less than temporal variability. Increasing the number of sub-samples taken within sampling events reduces the spatial variance (*i.e.* increases sampling precision) of estimated response variable means, and the data begins to reflect primarily temporal variation.

Increasing the sampling period is generally considered more important than is increasing the frequency of sampling (*e.g.* Downes *et al.* 2002). As no estimates of either temporal or spatial variation of any of the response variables were available for Lake Wallawalla, it was not possible to estimate the minimum sampling period or minimum number of sampling events required to reliably detect meaningful treatment effects (*i.e.* minimum detectable treatment effect size between Before and After time periods; *cf* Zar 1984). We anticipate that a full annual cycle (preferably 2 annual cycles) of Before data will be required to detect intervention responses with sufficient statistical power.

The choice of when to sample is often arbitrary, and is commonly set by previous practise (*e.g.* monthly intervals), logistic constraints (minimum time between sampling events to allow for sample processing) or some sort of conceptual model ('seasonal variation is important'). However many times the univariate response variable is to be sampled and at whatever interval, the times chosen should be random, not regular (Green 1997, Stewart-Oaten *et al.* 1986, Underwood 1991). Random (temporal) sampling reduces the potential for cyclic processes (*e.g.* lunar, season) to influence the magnitude of differences observed before and after a disturbance. Underwood (1991) suggests that the best design incorporates sampling at several temporal scales. This is especially important where the temporal scale of response is unknown. For example, where there is no replication of sampling within seasons, comparisons among seasons are confounded with shorter-term changes. This may be overcome by having several independent (non serially-correlated) randomly chosen sampling events in each season (Underwood 1993).

The most appropriate timing and frequency of sampling should be determined by the scales over which response processes and individual organisms operate (Underwood 1996). These are likely to vary considerably between response variables. For example, generation and population turnover times of zooplankton are much shorter (weeks) than they are for fish (months/years). Identification of the most appropriate sampling frequency for each is complicated by the requirement that data be independent. The independence of data is a major assumption of most statistical procedures and is often overlooked (Underwood 1996). Data will not be independent where samples are taken at such short intervals that the same individuals or cohort of individuals are repeatedly counted. The use of repeated-measures ANOVA is favoured where lack of independence between sample events is likely to be an issue, such as with the identification of post-flood responses in the current program (Underwood 1993).

For consistency between Icon Site wetland monitoring programs, sampling events at Horseshoe Lagoon will occur at the same frequency and time as those developed for

Webster's Lagoon, *i.e.* 2 'randomly' timed sampling events within each season. Note that the requirement for 'randomness' will need to be balanced against the requirement for independence. Whilst sampling at this frequency will be appropriate for most ecological response variables, others will require more frequent monitoring (*e.g.* wetland water levels – daily) or less frequent monitoring (*e.g.* groundwater – 3 monthly, vegetation – annually, soil salinity – coinciding with the end of the dry phase). For hypotheses relating to post-flood responses, sampling frequency will be increased over the first 8 weeks so as to increase temporal resolution and power for repeated measures analyses. Sampling during these events will occur weekly for the first month, then every two weeks for the second month. Sampling effort at these times will be in addition to the longer-term monitoring framework.

4.4.3 Spatial sampling

As described above, the ability to detect temporal variation is reduced where spatial variation within the wetland is relatively large. Observed spatial variation is a function of sampling precision, which is influenced by both the size and number of samples taken at any one time to obtain an estimate of the variable mean. Methods used to assess sampling precision vary widely. One of the more commonly used statistics for determining precision is the ratio SE/mean (Andrew and Mapstone 1987). Desired precision is arbitrarily set *a priori*, with target values generally ≤0.25 (Elliott 1979, Andrew and Mapstone 1987).

At the time of writing, Horseshoe Lagoon was dry precluding the examination of spatial variation of key response variables within the wetland and thus of the optimum allocation of sampling effort within sampling events. This will be possible once the wetland is flooded and will follow the procedure as described for Webster's Lagoon (Section 3.4.4).

5 Lake Wallawalla

5.1 Background

Lake Wallawalla is a large riverine lake and lunette system situated south of Lindsay Island in the Murray-Sunset National Park (Figure 5.1). Lunette lakes, such as Lake Wallawalla, are widely distributed throughout the Murray-Darling Basin, but tend to be concentrated in the Riverine Plains and the Mallee (Bowler and Magee 1978, Seddon *et al.* 1997). Typical of lunette lakes of the Murray-Darling Basin, Lake Wallawalla is an elliptical lake enclosed by the lunette on the eastern side with the long axis oriented north-south (Bowler and Magee 1978). The lake covers an area of approximately 815 ha and is connected by two channels (approximately 600 m long) to the Lindsay River (an anabranch of the Murray River). The inlet channels pass under the Mail Route Road via one large pipe culvert (eastern channel, 1.5 m diameter) and two smaller pipe culverts (western channel, 0.75 m diameter) (SKM 2003a).

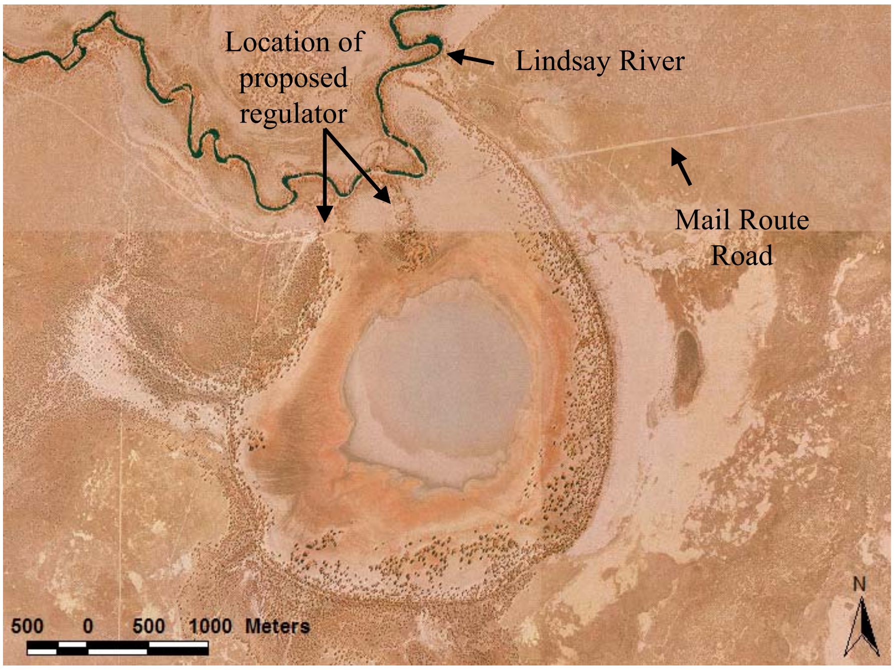


Figure 5.1: Lake Wallawalla showing location of proposed structures (source MDBC River Murray Mapping 2nd edn. May 1996).

The current flooding threshold for flows entering Lake Wallawalla from the Lindsay River is approximately 50000 ML day^{-1} at Lock 8 (SKM 2003a). Weir pool levels do not affect inflows to Lake Wallawalla as the lake becomes inundated only during large floods which overtop existing regulating structures (Pressey 1986, Beovich 1994). The levee created by the Mail Route Road and pipe culverts has restricted the rate of flow between Lake Wallawalla and Lindsay River, increasing the duration of inundation at low stages (< 21.60 mAHD) from 14.1 months to 27.2 months. However, inundation durations are consistently shorter under existing conditions for higher stages where the road has only a small influence (2.94 months *vs* 3.77 months) (SKM 2003a). Decreases in average flood elevations in Lake Wallawalla are attributable to reductions in Murray River flow peaks (SKM 2003a). Regulation of the Murray River has also delayed the timing of floods by 2 months (inflows now generally occur around September), although Beovich (1994) suggests there has been no delay with floods generally occurring between July and October, and increased the duration of dry periods

(SKM 2003a). However, the flooding frequency has not changed, with the lake still receiving floods approximately once in every 3–5 years (SKM 2003a).

Lake Wallawalla is considered to be of high ecological value at both regional and state levels due to the broad range of habitat it provides and its ability to support a variety of significant flora and fauna (EA 2001, SKM and Roberts 2003). The ecosystem supported by Lake Wallawalla is under threat as a result of flow regulation (SKM and Roberts 2003). For example, reductions in the duration of red gum flooding are believed to be detrimental to the breeding success of some waterbird species, and the existing culverts are believed to be restricting fish passage.

Ecological and flow objectives for each identified ecological value, process and threat were developed by SKM (2003a) and are shown in Table 5.1. These objectives are to be met by modifying the existing regulating features. The proposed upgrade of the Mail Route Road and associated culverts will:

- Change the water regime to allow the lake bed to drain to a lower level by replacing perched culverts with regulators recessed below the channel bed.
- Allow for higher stage water levels to be held for longer periods. This will be achieved by raising the Mail Route Road to an elevation of 22.35 mAHD and upgrading culverts so they can be closed. At this elevation, water in the lake would inundate the majority of the red gum community. Holding water under these trees for longer than is currently possible will facilitate the management of bird breeding events.
- Prevent the movement of larger fish (*e.g.* carp) into the wetland by incorporating carp screens into the structure.
- Improve passage for small native fish at all times and for large native fish at times when the carp screens are not installed. In particular, the structures will aim to provide downstream fish passage to allow native fish to enter or leave the lake with filling or receding flows.

Operating guidelines for the regulating structure are to:

- Maintain inundation frequency.
- Increase flows through the culverts into the lake during small/medium floods.
- Increase elevation and duration of water held in the Lake following a flood to enhance water bird breeding opportunities.
- Reinstate a more natural rate of flow back to the river from the lake after flooding.
- Increase fish passage into and out of lake.

It is important to note here that the proposed structural modifications will not allow for the management of ecological objectives that require flows/water levels exceeding 22.35 mAHD (*i.e.* Black Box/Chenopod Woodland and Alluvial Plains Shrubland community). Further, the management of fish passage (ecological threat) will be addressed as a structural objective (*i.e.* increasing culvert sizes) rather than as a flow objective.

Ecological value/process	Ecological objectives	Flow objectives
Lateral connectivity	Maintain connections between Lake Wallawalla-Lindsay River-Murray River-floodplain	Promote flooding (> 2 months), maintain current frequency (<1 in 3 yrs) and timing (Aug Nov).
Carbon and nutrient cycling	Maintain	Promote flooding (> 2 months), maintain current frequency (<1 in 3 yrs) and timing (Aug-Nov).
Lakebed Herbland community	Maintain	Maintain winter-spring flood duration (1-6 months) and frequency (1 in 2-5 years) of LH zone.
Riverine Grassy Forest community (red gums)y	Maintain	Maintain winter-spring flood duration (1-6 months) and frequency (1 in 2-5 years) of RGF zone.
Black Box/Chenopod Woodland community	Maintain	Flood BCW zone for 2-4 months once every 3-5 years.
Alluvial Plains Shrubland community	Maintain	Flood APS zone for 2-4 months once every 10 years.
Zooplankton	Maintain	Flood LH zone for > 1 month.
Early breeding fish	Maintain populations of Australian smelt	Flood LH zone for > 2 months at least once in 3 years between Aug. and Oct.
Mid-breeding fish	Maintain populations of flatheaded gudgeon, rainbow-fish, cod, carp gudgeons	Flood LH zone for > 2 months at least once in 3 years between Nov. and Jan.
Late breeding fish	Maintain populations of bony herring, hardyheads, carp gudgeons	Flood LH zone for > 2 months at least once in 3 years between Feb. and Apr.
Flood dependent breeding fish	Restore populations of golden and silver perch	Flood LH zone for > 2 months at least once in 3 years between Aug. and Nov.
Waterbirds	Maintain colonial nesting bird community and promote breeding	Flood RGF zone for 5-8 months (winter/spring) and/or 7-10 months (autumn) at least once every 5 years.
	Maintain duck community and promote breeding	Flood RGF zone for 5-6 months (at any time) at least once every 5 years.
Ecological threat		
Fish passage	Reduce restriction	Structural objective- maintain/ increase duration of connection with Lindsay River.
Saline groundwater discharges	Minimise saline discharges to Lindsay River	Maintain mean flood duration less than 163 days
Soil salinisation	Minimise soil salinisation	Maintain flood return period less than 14 months
Exotic fish	Minimise any incease in abundance of carp, redfin, mosquitofish	Reduce flood durations and frequency in LH zone. Increase drawdown rate of wetland for egg desiccation.

Table 5.1: Summary of ecological objectives and flow requirements for Lake Wallawalla (source SKM 2003a).

5.2 Conceptual framework and hypotheses

Floodplain wetlands have intrinsic local values, and represent an important component of the larger floodplain ecosystem (Ward 1989, Ward and Stanford 1995). They are naturally diverse and productive habitats, whose management through the alteration of natural flow regimes has resulted in either real or perceived declines in wetland condition (*e.g.* Kingsford 2000a,b). It is generally agreed that both wet and dry periods are important in maintaining ecosystem integrity in ephemeral wetlands (Boulton and Lloyd 1992, Bunn *et al.* 1997, Boulton and Jenkins 1998). Disturbances, such as flooding and drying, drive aquatic and terrestrial successional processes and facilitate biotic and abiotic exchanges between elements of the floodplain and the riverine environment (*cf.* Flood Pulse Concept; Junk *et al.* 1989). Because of this, ephemeral wetlands are potentially sites of high productivity and diversity within floodplain ecosystems. Consequently, the management of these wetlands has implications for ecosystem productivity and diversity at the landscape scale. Thus, any changes in wetland function initiated through hydraulic modification need to be viewed from both the perspective of the individual wetland and from a landscape perspective.

Available information, conceptual models (*e.g*. Flood-Pulse Concept - Junk *et al.* 1989, Alternative-States Model - Scheffer *et al.* 1993, Geomorphic-Trophic Model - Hershey *et al.* 1999, Trophic-Cascade Model - Carpenter *et al.* 2001, Ephemeral Deflation Basin Lake Model - Scholz and Gawne 2004a,b) and expert consultation (refer to Acknowledgements) were used to develop a conceptual model of the links between the ecosystem components described in Table 5.1 and their hypothesised responses under existing and proposed management scenarios (Figures 5.2 and 5.3). From these we generated testable hypotheses which were used to guide the development of the monitoring program.

THREATENING PROCESS

Altered hydrology

-Restricted flow between the lake and Lindsay River.

-Increased duration of inundation of low stages (<21.6 m AHD).

-Decreased duration of inundation of higher stages (>21.6 m AHD).

-Delayed timing of flooding (Jul-Oct).

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ENVIRONMENTAL VALUES/PROCESSES/THREATS

Lateral connectivity -Restriction of fish passage between lake and river.

Groundwater -Interactions between surface and groundwater poorly understood.
-Current freshwater lens creates lateral pressure on the movement of saline groundwater to Lindsay River.

Water quality -Low potential for the movement of saline groundwater in to the lake.
-High turbidity.

Vegetation -Lakebed Herbland, Riverine Grassy Forest (red gums) and Alluvial Plains Shrubland in good condition.
-Decreased condition of Black Box/Chenopod Woodland.
-Duration of red rum inundation too short.

Fish -Fish communities dominated by non-native species (*e.g.* carp).
-Low abundance/biomass of small native fish species.

Water birds- -Reduced breeding opportunity.

Figure 5.2: Conceptualisation of ecosystem function under existing flow conditions at Lake Wallawalla.

ACTIONS	ENVIRONMENTAL RESPONSE
-Raise Mail Route Road to 22.35 mAHD. -Increase culvert diameter. -Increase inflows during small/medium floods (<22.35 mAHD). -Exclude large carp during low/medium floods (<22.35 mAHD) (not possible for floods >22.35 mAHD). -Maintain inundation frequency. -Increase elevation of water held in the lake following a flood. -Increase duration of water held in the lake following a flood. -Reinstate a more natural slow flow back to the river from the lake after flooding.	**Lateral connectivity-** Increased capacity for fish passage between lake and river. **Groundwater** -Increased potential for movement of saline groundwater to Lindsay River. **Water quality** -Increased potential for movement of saline groundwater in to the lake. -Turbidity reduced during low/medium floods (<22.35 mAHD). **Vegetation** -Lakebed Herbland, Riverine Grassy Forest (red gums), Black Box / Chenopod Woodland and Alluvial Plains Shrubland in good condition. -Vegetation community boundaries shifted to higher elevations. -Duration of Red Gum inundation increased. **Fish** -Adult carp population reduced during low/medium floods (<22.35 mAHD). -Fish communities dominated by native species during low/ medium floods (<22.35 mAHD). -Increased abundance/biomass of small native fish species during low/ medium floods (<22.35 mAHD). **Water birds** -Increased breeding oppurtunity.

Figure 5.3: Conceptualisation of ecosystem function under the flow regime proposed for Lake Wallawalla.

Flow/hydrology

Hypothesis A1: Operation of the proposed control structure in Lake Wallawalla will have an effect on water levels within the wetland in compliance with the management strategy.

Many ecological responses are likely to be driven by changes in hydrology, making it critical that an accurate assessment of water levels be included within the monitoring design. The flow objectives for Lake Wallawalla are to maintain water stage within the lake at 22.35 mAHD for as long as required to achieve key ecological objectives (*e.g.* waterbird breeding success), and to reduce the rate of stage recession following a flood event. There is currently limited understanding of how water moves into and out of the lake and the relative contributions of each channel to filling and draining flows. This information will be required following the modification of in/outlet structures to develop an accurate stage-discharge relationship and determine fill and empty rates. Water level data will feed directly into an adaptive management framework, allowing for real time management responses during periods of non-compliance.

Lateral connectivity

Many wetland processes and biota are dependent on exchanges between wetland and riverine environments. With the current structural configuration of the regulating Mail Route Road structure such exchanges (*i.e* fish passage) are restricted by the width of the culverts. Whilst the proposed structural modifications will increase the width of this connection, the capacity for lateral exchanges will also be influenced by the management of water levels within the Lake.

As will be discussed later, the operating rules for opening and closing the structure will be largely influenced by the adaptive management of waterbird populations. For example, following a flood event water will be held within the lake at higher elevations and for longer than is currently possible with the intent of promoting waterbird breeding. Only once this has occurred will the structure be opened and water released. As lateral connectivity will be directly influenced by this sequence of operating events it will not be monitored directly and no hypothesis will be tested.

Groundwater/salinity

Hypothesis B1: Inundating Lake Wallawalla will raise groundwater levels adjacent to the lake.

The geology of Lake Wallawalla comprises five distinct layers; Fine Alluvium (aquifer 2-7 m in depth), Channel Sand (aquifer 7-11 m in depth), Parilla Sand (aquifer), Bookpurnong Beds (aquitard), and Murray Group Limestone (SKM 2003a). Regional groundwater within the Channel Sands aquifer in the vicinity of Lake Wallawalla is saline (22000–40800 mg l^{-1} TDS or 13200-24500 $\mu S\ cm^{-1}$) and flows in a north-east direction.

Regional groundwater levels are strongly influenced by locks on the Murray River, which permanently elevate surface water levels and influence seepage, and by seasonal flooding. Locally, there appears to be a strong relationship between lake level and groundwater SKM (2003a), with the flooding of Lake Wallawalla resulting in the formation of a groundwater mound beneath the lake. This exerts a strong influence on the direction and rate of local groundwater flow, increasing sub-surface discharges of saline groundwater to Lindsay River upstream of the lake. SKM (2002b) estimated this process to contribute approximately 2.8 T day^{-1} of groundwater derived salt to Lindsay River, equating to a rise of 0.5 $\mu S\ cm^{-1}$ at Morgan. Thus, any increase in the flooding duration or frequency of Lake Wallawalla is likely to elevate the groundwater mound beneath the lake and so increase inputs to the Lindsay River. Note that the proposed scheme will have no impact on flood frequency.

Groundwater data will inform our understanding of the effect of Lake Wallawalla hydrology both on regional groundwater pattern and saline groundwater discharge to Lindsay River. Monitoring of changes in flow-adjusted salinity (electrical conductivity) of the Lindsay and Murray rivers before and during lake flooding will be used to identify the magnitude of potential downstream impacts.

As for all groundwater related questions (refer to other wetland sections of this report) the lack of suitable controls precludes the establishment of cause and effect. Testing of the hypothesis will thus be restricted to examining intervention outcomes.

Water quality

Electrical conductivity

Hypothesis C1: Salt loads within Lake Wallawalla will increase over time after inundation.

The downstream movement of salt from floodplain wetlands is recognised as a key threat at the regional landscape scale. Salt may be mobilised from the floodplain via both ground- and surface-water pathways. Whilst changes in hydrology (*i.e.* increased elevation and duration of water held in the lake) will likely increase the lateral movement of saline groundwater directly into Lindsay River, it may also increase the potential for the movement of saline groundwater directly into the lake water column. At present, no surface water salinity data is available for Lake Wallawalla, nor is there any information relating to the potential for saline intrusions within the lakebed. The monitoring of surface water salinity (as electrical conductivity) in conjunction with lake volumes will facilitate assessment of the potential for surface waters to contribute to downstream salt loads.

Terrestrial vegetation

Hypothesis D1: Inundating Lake Wallawalla will not reduce the condition of the Lakebed Grassy Herbland (LGH) community.

Hypothesis D2: inundating Lake Wallawalla will not reduce the condition of the Riverine Grassy Forest (RGF; red gum) community.

One hundred and seventy three plant taxa have been recorded from within the normally flooded area of Lake Wallawalla (NRE 2002). Of these, no species are currently listed as rare or threatened nationally or listed under the Commonwealth's Environment and Biodiversity Conservation Act 1999. However, 12 species are considered vulnerable and 10 species rare in Victoria (SKM 2003a).

Flood frequency is a major determinant of vegetation distribution patterms within Lake Wallawalla. The current water regime supports four distinct vegetation zones. The Lakebed Grassy Herbland (LGH) occupies the open central areas of the lake that are inundated when water stage height exceeds 20.70 mAHD. This community is surrounded by concentric rings of red gum dominated Riverine Grassy Forest (RGF), which is inundated at stages >21.50 mAHD, Black Box/Chenopod Woodland (BCW), which is inundated at stages >22.35 mAHD, and at the highest elevations by Alluvial Plains Shrubland (APS), which is inundated at stages >23.20 mAHD (SKM 2003a).

For all except the lowest lying LGH zone, the change to the current regulated water regime has reduced the period of inundation (on average by 91 %) and reduced their frequency (on average by 67 %). A recent survey (July 2002) indicated that all vegetation types excluding BCW were in good health and showed signs of recruitment. SKM and Roberts (2003) reported that patches of black box along the western margin of the lake appeared to be in poor condition and were not showing any signs of recruitment. Factors contributing to this loss of condition remain to be identified.

The key ecological objective identified for each of the identified vegetation communities is that they be maintained. Whilst condition assessments have been recommended for each community (SKM 2003a), the proposed structural modifications and concomitant changes in hydrology will not allow for the management of ecological objectives that require flows/water levels exceeding 22.35 mAHD (*i.e.* BCW and APS communities). The provision of water to these communities will remain reliant on larger flood events. Monitoring will thus focus on identifying changes in the condition of LGH and RGF communities. Key condition response variables and survey protocols have yet to be developed.

Fish

Hypothesis E1: The use of carp screens during low-medium flood events (<22.35 mAHD) will eliminate large carp in Lake Wallawalla.

Hypothesis E2: Inundating Lake Wallawalla will stimulate spawning in small-bodied native fish species.

Fish provide an important link in wetland food webs through their consumption of prey and their consumption by birds. Wetland hydrology plays an important role in structuring fish assemblages though its influence on ecosystem productivity, habitat availability and flow related cues for fish spawning. Native fish throughout the Murray-Darling Basin have been severely impacted by altered flow regimes, the loss of habitat, and barriers to passage

(MDBMC 2002). Altered flow regimes, in particular, are thought to have impacted on the distribution of native fish species and to have favoured the expansion of alien species, such as carp (*Cyprinus carpio*) and mosquitofish (*Gambusia holbrooki*) (Harris and Gehrke 1997). Whilst fish distributions within the Basin have been reported (Llewellyn 1983, Harris and Gerhke 1997), and responses of fish communities to flooding in Australian floodplain systems have received some attention (*e.g.* Gerhke 1991, 1994, Gehrke *et al.* 1997, McKinnon 1997, Humphries *et al.* 1999, King *et al.* 2003, Scholz and Gawne 2004a,c), there is a general lack of information on the fish passage requirements of most species, particularly smaller fish species (*e.g.* Mallen-Cooper 2001).

A key assumption behind the high ecological value attributed to Lake Wallawalla is that large floodplain depressions are important for the overall lowland river native fish community, by providing important spawning, nursery and adult habitat. However, the importance of a large lake that holds water for a long period following flooding is poorly understood and difficult to test.

Fish passage requirements are likely to vary greatly between fish species and fish life stages, and to vary over different temporal scales during the flood period. Whilst separate flow objectives were developed for early- (Australian smelt), mid- (flatheaded gudgeon, crimson-spotted rainbow-fish, western carp gudgeons), late- (bony herring, flyspecked hardyheads, freshwater catfish) and flood dependent -breeding native fish (golden perch, silver perch, Murray cod) to accommodate likely seasonal passage requirements, recent work has shown that several species such as golden perch, bony herring, flyspecked hardyhead and gudgeons also move between lake and river habitats diurnally (Tucker and Nichols 2001).

Fish passage requirements will be addressed principally within the design of the proposed regulating structure by increasing the size of culverts under the Mail Route Road. As the operating rules for the regulating structure are to be guided primarily by the breeding responses of resident waterbird populations, the scope for proactive management with respect to the temporal patterns of fish passage requirements will be limited. We anticipate that increasing the duration of floods >20.70 mAHD (upper limit of LGH vegetation community) and re-instating a more natural slow flow back to the river following a flood (*i.e.* increasing the duration of connection) will provide a wider window of opportunity for native fish species to utilise the lake, and so increase the potential to achieve the ecological objective of maintaining native fish populations.

Exotic fish such as carp, redfin and mosquitofish are a recognised threat to the ecological values of Lake Wallawalla. Although the evidence is fragmented and sometimes contradictory, carp have been implicated in increasing turbidity and suspended nutrient loads (Lamarra 1975 Roberts *et al.* 1995), uprooting of macrophytes arising from their benthivory around the plant base (Hume *et al.* 1983, Brown 1996), and reducing benthic invertebrate biomass through predation (Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al.* 1994, Tatrai *et al.* 1994). Carp have also been shown to adversely impact on native fish stocks by impairing the efficiency of visual feeding (increased turbidity), by reducing spawning site availability (reduction of macrophytes) and by reducing food resource availability (reduction of invertebrate biomass) (Hume *et al.* 1983, Fletcher 1986).

Whilst minimising any increases in abundance of exotic species is a key ecological objective, species specific control measures are difficult to implement. Available management options include their elimination by complete lake drying, and the exclusion of larger individuals during small/medium inflow events (<22.35 mAHD) using screens over the inlet culverts. However, as these measures also impact on native fish species their efficacy needs to be tested. Whilst, the complete and continuous exclusion of carp is not feasible due to the embankment formed by Mail Route Road being lower than many annual peak water levels, the exclusion of larger carp during smaller flood events may have some beneficial impacts on the abundance of smaller native and threatened fish populations. Although the exclusion of carp from the Lake during low/medium flood events can be tested, the likely absence of an experimental (flooded) Before time period precludes the formulation of a directional hypothesis (*i.e.* Hypothesis E1). It will thus not be possible to establish whether differences in carp abundances recorded after re-filling are due to the carp screens or to differences in carp abundances in the flood water. Note also that this hypothesis will be rejected by the capture of only a single carp.

The overall ecological objective for the fish community in Lake Wallawalla is to maintain a diverse community of native fish. Whilst this is to be addressed by increasing opportunities for fish passage as described above and by excluding carp during low/medium flood events, we anticipate that the flooding of Lake Wallawalla will trigger spawning in most small-bodied native (and exotic) fish species. Although most small-bodied native fish exhibit seasonal preferences for spawning, recent work has shown that most (if not all) have the capacity to spawn opportunistically during an over-bank flood in response to increases in the availability of food (post-flood pulse of invertebrate production) and critical spawning and nursery habitat (Meredith and McCasker in prep.). This is also true for exotic species such as carp. Similar post-inundation responses are likley to occur at the other Icon Site wetlands.

Waterbirds

Hypothesis F1: Flooding red gums in Lake Wallawalla for more than 5 months will result in waterbird breeding success.

Lake Wallawalla when flooded is an important area for waterbirds. Thirty four waterbird species have been recorded at the lake, including 6 threatened species (LCC 1987, NRE 1995, DSE 2004b).

Waterbird abundance is determined by the reproductive success of adults and the survival of post-juvenile birds. The breeding and survival of most waterbirds in the Murray-Darling Basin have been linked to cycles of flooding and drying (Briggs 1990); with most species being capable of moving large distances between wetlands as local breeding and feeding opportunities fluctuate. Almost all common waterbirds of the Murray-Darling Basin breed following wetland flooding, with some also breeding seasonally, usually in spring. In only 2 species (both ducks) is breeding always seasonal and apparently unaffected by wetland hydrology (Briggs and Lawler 1991).

Flow regulation within the Murray-Darling Basin has altered the temporal and spatial distribution of inundated floodplain wetlands available to waterbirds. Regulation has permanently inundated many formerly intermittent wetlands, and reduced the average duration of inundation for many others (e.g. Lake Wallawalla - reduced duration of high stage inundation). Whilst this has increased the availability of survival habitat for waterbirds at the landscape scale, it has effectively reduced the availability of breeding habitat. Thus, the overall effect of river regulation on waterbirds is likely to be reduced recruitment of young, but enhanced survival of adults (Briggs and Lawler 1991). Because of this, it is important that the number of breeding opportunities for waterbirds within the Murray-Darling Basin be increased. As is discussed below, maintaining higher water levels for longer within Lake Wallawalla is likely to achieve this objective.

The nesting requirements of Australian waterbirds vary (Serventy 1985), but all require adequate food to satisfy the energetic and other costs of nest finding or building, the laying and incubating of eggs, and the rearing of young. Whilst these costs may be partly provided for by the accumulation of body fat and/or protein reserves prior to laying (*e.g.* Thomas 1988, Norman and Hurley 1984, Briggs 1988), most waterbirds of the Murray-Darling Basin require readily available food of high quality at nesting time for successful breeding (Briggs and Lawler 1991). Many studies have shown the inundation of previously dry wetlands to initiate a highly productive succession of potential food resources (*e.g.* invertebrates and fish), which attracts many waterbird species and stimulates breeding (*e.g.* Maher 1984, Maher and Carpenter 1984, Crome 1986, 1988). Recently flooded wetlands thus provide a potentially important window of opportunity for breeding by many waterbird species. Over time after inundation and as wetlands begin to dry; changes in the composition and availability of food resources tend also to stimulate changes in the structure of waterbird assemblages (Scott 1997), increasing the cumulative diversity of species utilising the wetland.

In addition to food availability, post-inundation waterbird breeding success is reliant on the persistence of adequate food resources and of nesting habitat throughout the breeding (rearing) cycle. Within Lake Wallawalla important nesting habitat is provided by the fringing red gums. Under current operating conditions, water is not being held under these trees long enough for birds to sussessfully breed, especially those that require relatively longer inundation periods (*e.g.* darters, cormorants, herons, egrets, spoonbills and Australian white ibis).

Red gum inundation requirements vary between waterbird species and have been shown to range from 4–10 months (Briggs *et al.* 1997). There is generally a 2–3 month time lag between a flood event and when birds start to breed (Maher 1991, Briggs *et al.* 1994), and many waterbirds take about 3.5 months to build their nests, lay and incubate their eggs and to fledge their young (Marchant and Higgins 1990, Briggs *et al.* 1994). Peak breeding for many waterbirds occurs approximately 3–6 months after wetland flooding (Maher 1991, Briggs *et al.* 1994). Assuming that some birds at the peak breeding stage are still laying (and hence need a further 3.5 months) (Briggs *et al.* 1994), then a total of 7–10 months flooding may be desirable, during which water would need to be kept under the red gums at Lake Wallawalla.

The ecological objective of maintaining the community of colonial nesting waterbirds and ducks and promoting breeding will thus be addressed by inundating red gums (> 21.50 mAHD) for at least 5 months. The retention of water under red gums to promote breeding by waterbirds is the primary management objective for the Lake. Monitoring of waterbird breeding success during the post-inundation period will to a large extent guide the operation of the regulating structure. At this stage, key response variables and survey protocols for quantifying breeding success have yet to be developed.

5.3 Response variables

Response variables to be measured as part of the monitoring program were identified for each of the hypotheses stated in the previous section. Standard sampling and analytical protocols for each response variable are listed in Table 5.2.

Response variable	Field and laboratory method
Surface water	
Height (mAHD)	Automated logging using Odyssey pressure sensors (Dataflow Systems Pty Ltd, Christchurch New Zealand).
Area and volume	Capacity tables constructed from profile surveys (*e.g.* digital elevation mapping).
Groundwater	
Depth (m)	Measured from piezometers located along five transects extending radially out from the Lake (refer to Jeuken 2005). Surveying of piezometers to mAHD will enable comparisons of water tables.
Electrical conductivity (EC; µS cm^{-1})	EC (standardized to 25 ^{o}C) of water samples extracted from piezometers measured in the field using a U-10 multi-probe (HORIBA Ltd., Australia).
Water quality	
Electrical conductivity (EC; µS cm^{-1})	EC standardized to 25 ^{o}C determined *in situ* using a U-10 multi-probe (HORIBA Ltd., Australia).
Turbidity (NTU)	Turbidity of 500 ml sub-surface water samples determined using a U-10 multi-probe (HORIBA Ltd., Australia). Samples diluted 1:3 with distilled water where turbidity exceeds the max. level of detection of 1000 NTU.
Wetland vegetation	
Community condition (response variables to be determined)	Survey protocol to be determined. This will be consistent across Icon Site wetlands to permit comparisons at the regional scale.
Fish	
Relative abundance (fish $CPUE^{-1}$) and biomass (kg $ha^{-1}CPUE^{-1}$)	For each sampling event 6 large fyke nets (LFN) (set tangential to shore with cod end away from shore), 6 small fyke nets (SFN) (set parallel to shore with pairs set in opposite directions) and 6 larval light traps (LT) set overnight. Set and pull times recorded and total catch adjusted to 24 hours (= 1 CPUE). Standard lengths (±0.05 mm for LFN and ±0.005 mm for SFN and LT) and weights (±0.5 g for LFN and ±0.0005 g for SFN and LT) recorded for the first 30 individuals (including the largest and smallest individuals caught) per species. Fish identifications follow McDowall (1996). Carp gudgeons identified to genus level only (*i.e. Hypseleotris* spp.) owing to the current taxonomic uncertainty at the species level (Bertozzi *et al.* 2000). All large native fish returned. Ethics approval obtained prior to sampling.
Waterirds	
Breeding opportunities (response variables to be determined)	Survey protocol to be determined. This will be consistent across Icon Site wetlands to permit comparisons at the regional scale.

Table 5.2: Response variables, measurement units and analytical methods used to test hypotheses developed for Lake Wallawalla.

5.4 Experimental Design

5.3.1 Design considerations

Several experimental design options are available to test hypotheses based on the identification of changes in a response variable before and after an environmental impact such as increasing the elevation and duration of lake water levels. These design options vary both in their complexity and in their ability to detect change. Generally, the incorporation of experimental controls or reference sites within an experimental design is favoured as it permits the elimination of potentially confounding artefacts introduced by the experimental procedure and provides the greatest statistical power to test hypotheses and to demonstrate cause and effect. Where control or reference sites can be identified, experimental designs incorporating Before, After, Control and Impact (BACI) data can and should be used (Underwood 1996). However, this is not always possible, requiring the implementation of alternative approaches that offer either a less scientifically rigorous means of demonstrating an ecological response, or that are capable of assessing the momentary status of key ecological response variables (*i.e.* management trigger levels). Alternatively, where the required level of scientific rigour can not be met it may prove more appropriate to do nothing.

Each of the ecological response hypotheses that were developed for Lake Wallawalla in Section 5.2 were assigned to one of four monitoring design categories based on considerations of the availability and/or requirement of a suitable control and the type of output required. These were:

i) Before-After Control-Impact (BACI) design. This approach represents the most scientifically credible monitoring option, and will enable the unambiguous determination of changes to the system that have occurred as a direct result of management intervention by comparing environmental variables in the 'Impact' system with that in an unmanaged 'Control' system 'Before' and 'After' the intervention. Note that because this approach requires a large number of measurements paired in time (also referred to as a BACIP design; Green 1997, Stewart-Oaten *et al.* 1986, Underwood 1991) across 2 (or more) locations, it represents the most expensive of the four options. Note also that if employed across multiple ecological response hypotheses within one system, this option has the advantage of being able to explain failures to detect expected ecological responses by considering conceptual models that link different trophic levels (*i.e.* trophic interactions).

ii) Intervention Analysis (IA) or Before-After (BA) design (*cf.* Box and Tiao 1975). This design option is used in cases where no control site is possible due to the uniqueness of the water regime, but where it is still necessary to monitor/demonstrate the ecological response within a system. Because no control is available, it will not be possible to demonstrate whether or not structural operation has 'caused' any observed 'effect'.due to fact the data are 'pseudoreplicated', *i.e.* any change in the response variable may be due to any number of confounding effects (Hurlbert 1984, Underwood 1996).

iii) Instantaneous adaptive management (IAM) feedback. In cases where the ecological response hypothesis defines a key ecological outcome that is expected from the operation of structures on the floodplain, these ecological responses will be monitored and their real-time analysis will determine structure operation rules. In such cases, no control site is required (or in some cases possible), only the post-impact (After) period is monitored, and no establishment of scientifically credible cause and effect is necessary (or in some cases possible). However, in these cases a hierarchy of ecological outcomes is necessary to determine the sequence of management events.

iv) Do nothing. In some cases, the ecological response hypotheses for systems where no control is possible will be examined in other systems (*i.e.* other Living Murray Icon Sites). Rather than invest considerable resources into monitoring outcomes without being able to demonstrate cause and effect, it may be pertinent to transfer that effort to systems where causality can be determined.

Monitoring to demonstrate cause and effect is expensive in both time and money and cannot reasonably be undertaken for all Icon site wetlands. Nor is it possible. For Webster's Lagoon which is currently permananently inundated, it was possible to locate a suitable control site and thus to implement a BACI experimental design to test a wide range of ecosystem responses to altered hydrology. Whilst the demonstration of cause and effect was considered most appropriate for Webster's and Horseshoe lagoons, the emphasis at Lake Wallawalla was to monitor and adaptively manage for key ecological outcomes (*e.g.* flow/hydrology and waterbird breeding success). For these an IAM feedback experimental design was most appropriate. For other response variables, such as groundwater/salinity and vegetation condition, it will be possible to obtain both Before (dry phase) and After (flooded phase) data, allowing an IA experimental design to be used. The data obtained from these may also be incorporated into the overall adaptive management framework. At the time of writing we do not anticipate that Lake Wallawalla will experience a flooding event prior to the modification of the regulating structure. This precludes the collection of a Before wet-phase data set or the development of testable directional hypotheses for water quality (electrical conductivity) and fish communities. The monitoring of these will thus be limited to demonstrating responses following lake inundation. Because of this, it will not be possible to establish whether differences in carp abundances recorded after re-filling are due to the carp screens or to differences in carp abundances in the flood water, and Hypothesis E1 will be rejected by the capture of only a single carp.

Of the suite of ecological objectives developed for Lake Wallawalla (Table 5.1), three are outside of the influence of the regulating structure (Black Box/Chenopod Woodland community, Alluvial Plains Shrubland communities and soil salinisation), one will be addressed as a structural objective (culvert widening to improve fish passage) and two will not be monitored as they are not considered critical to the management of the Lake (carbon and nutrient cycling and zooplankton) (*i.e.* refer to 'do nothing' above).

As stated above, monitoring for ecological outcomes using IAM requires that a hierarchy of ecological outcomes and response variable 'trigger' levels be established *a priori* to define the priority of management actions/responses in circumstances where there is a conflict in appropriate actions between ecological values. For example, whilst prolonging the inundation of red gums may stimulate the desired breeding responses in bird communities, it may have an adverse impact on red gum tree condition. Whilst differences in lag times between impact initiation and detectable response need to be factored in to the identification of management 'trigger' levels, doing so is not straight foreward. After consultation with key stakeholders and experts (refer to Acknowledgements) a heirarchy of ecological issues was developed (Table 5.3). Underlying all management responses is the statutory requirement that the contribution of saline discharges to Lindsay River be maintained below 1 uS cm^{-1} at Morgan. It is likley that any groundwater impacts arising from the holding of water for longer within the lake will be evident after the first filling cycle, at which time management responses and priorities may be revised.

Priority rank	Ecological issue	Management responses
1	Waterbirds	Maintain water under red gums until bird breeding has occurred. Response variable 'trigger' levels to be determined.
2	Red gums	Maintain water under red gums until condition deteriorates. Response variable/ 'trigger' levels and response lag times to be determined.
3	Fish	Maintain fish passage for as long as possible.

Table 5.3: Heirarchy of instantaneous adaptive management (IAM) actions/responses developed for Lake Wallawalla.

Based on our conceptual understanding of ecosystem function developed in Section 5.2, each of the ecological response variable/s identified for each hypothesis are likely to vary considerably throughout the wet/dry cycle (Figure 5.4). Table 5.4 provides a summary for each hypothesis of the experimental design to be used, the hydrological phases that will be examined and the statistical treatment of data. We note that the stated statistical models to be used in identifying responses are provisional based on the eventuality of Lake Wallawalla experiencing a flooding event prior to the completion of the proposed structural modifications. Other design consideration, such as the optimisation of temporal and spatial sampling regimes employed, have the capacity to greatly affect the ability to detect treatment effects (*i.e.* statistical power). These issues are considered in the following sections.

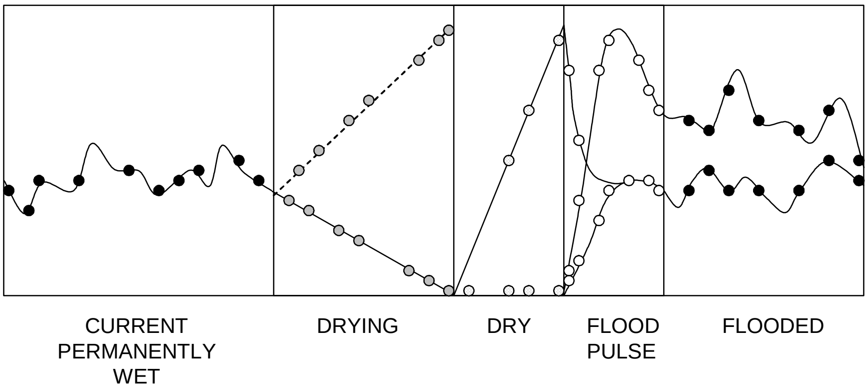

Figure 5.4: Phases of the wet/dry cycle showing conceptualised responses of response variables.

Environmental value	Hypotheses	Establish cause and effect (BACI)	Intervention Analysis (Before-After)	Instantaneous adaptive management	Do nothing	Hydrological phases examined	Statistical analysis
Flow/hydrology	A1	×	×	✓	×	All	No statistics
Groundwater/salinity	B1	×	✓	×	×	All	1-factor fixed-model ANOVA
Water quality (electrical conductivity)	C1	×	×	✓	×	Flooded	1-factor fixed-model ANOVA
Wetland vegetation (LGH)	D1	×	✓	×	×	Dry	1-factor fixed-model ANOVA
Wetland vegetation (RGF)	D2	×	✓	✓	×	All	1-factor fixed-model ANOVA
Fish	E1	×	×	✓	×	Flooded	No statistics, capture of one carp sufficient to reject hypothesis
	E2	×	×	✓	×	Flood pulse	Repeated measures ANOVA
Waterbirds	F1	×	×	✓	×	Flooded	No statistics, management response based on *a priori* defined trigger level

Table 5.4: Summary of monitoring approach to be used to test each of the ecological response hypotheses developed for Lake Wallawalla, the hydrological phases for which data is required, and the statistical treatment of data. Note that instantaneous adaptive management responses will be possible only for flow/hydrology and waterbirds.

5.3.2 Temporal sampling

With Before-After type experimental designs (*e.g.* Intervention Analysis), statistical power is influenced by both temporal and spatial variation. Power may be improved by increasing the number of samples taken per sampling event, by increasing the sampling period, and by increasing the number of sampling events.

For hypotheses in which Before and After time periods are to be compared, each sampling event provides only one value. As sub-samples collected from the wetland within sampling events do not add replicates to test the hypothesis about treatment effects, their impact on statistical power is likely to be small, especially where spatial variability is less than temporal variability. Increasing the number of sub-samples taken within sampling events reduces the spatial variance (*i.e.* increases sampling precision) of estimated response variable means, and the data begins to reflect primarily temporal variation.

Increasing the sampling period is generally considered more important than is increasing the frequency of sampling (*e.g.* Downes *et al.* 2002). As no estimates of either temporal or spatial variation of any of the response variables were available for Lake Wallawalla, it was not possible to estimate the minimum sampling period or minimum number of sampling events required to reliably detect meaningful treatment effects (*i.e.* minimum detectable treatment effect size between Before and After time periods; *cf* Zar 1984). We anticipate that a full annual cycle (preferably 2 annual cycles) of Before data will be required to detect intervention responses with sufficient statistical power.

The choice of when to sample is often arbitrary, and is commonly set by previous practise (*e.g.* monthly intervals), logistic constraints (minimum time between sampling events to allow for sample processing) or some sort of conceptual model ('seasonal variation is important'). However many times the univariate response variable is to be sampled and at whatever interval, the times chosen should be random, not regular (Green 1997, Stewart-Oaten *et al.* 1986, Underwood 1991). Random (temporal) sampling reduces the potential for cyclic processes (*e.g.* lunar, season) to influence the magnitude of differences observed before and after a disturbance. Underwood (1991) suggests that the best design incorporates sampling at several temporal scales. This is especially important where the temporal scale of response is unknown. For example, where there is no replication of sampling within seasons, comparisons among seasons are confounded with shorter-term changes. This may be overcome by having several independent (non serially-correlated) randomly chosen sampling events in each season (Underwood 1993).

The most appropriate timing and frequency of sampling should be determined by the scales over which response processes and individual organisms operate (Underwood 1996). These are likely to vary considerably between response variables. Identification of the most appropriate sampling frequency for each response variables is complicated by the requirement that data be independent (*i.e.* not serially-correlated). The independence of data is a major assumption of most statistical procedures and is often overlooked (Underwood 1996). Data will not be independent where samples are taken at such short intervals that the same individuals or cohort of individuals are repeatedly counted. The use of repeated-measures ANOVA is favoured where lack of independence between sample events is likely to be an issue, such as with the identification of post-flood responses in the current program (Underwood 1993).

For consistency between Icon Site wetland monitoring programs, sampling events at Lake Wallawalla will occur at the same frequency and time as those developed for Webster's Lagoon, *i.e.* 2 'randomly' timed sampling events within each season. Note that the requirement for 'randomness' will need to be balanced against the requirement for independence. Whilst sampling at this frequency will be appropriate for water quality (electrical conductivity) and fish, others response variables will require either more frequent monitoring (*e.g.* wetland water levels – daily) or less frequent monitoring (*e.g.* groundwater – 3 monthly). As response variables for vegetation condition and waterbird breeding success have yet to be determined, comment on sampling frequency for these is premature. For hypotheses relating to post-flood responses, sampling frequency will be increased over the first 8 weeks so as to increase temporal resolution and power for repeated measures analyses. Sampling during these events will occur weekly for the first month, then every two weeks for the second month. Sampling effort at these times will be in addition to the longer-term monitoring framework.

5.3.3 Spatial sampling

As described above, the ability to detect temporal variation is reduced where spatial variation within the wetland is relatively large. Observed spatial variation is a function of sampling precision, which is influenced by both the size and number of samples taken at any one time to obtain an estimate of the variable mean. Methods used to assess sampling precision vary widely. One of the more commonly used statistics for determining precision is the ratio SE/mean (Andrew and Mapstone 1987). Desired precision is arbitrarily set *a priori*, with target values generally ≤ 0.25 (Elliott 1979, Andrew and Mapstone 1987).

At this stage key response variables have not been established for vegetation and waterbirds, and no data is available for other response variables within the Lake from which an *a priori* assessment of spatial variance could be obtained. The collection of such data was also precluded by the current dry state of the Lake. Estimating spatial variance and the sampling effort required to achieve an adequate level of precision will be possible once the wetland is flooded and will follow the procedure as described for Webster's Lagoon (Section 3.4.4).

6 Potterwalkagee Creek

6.1 Background

Mulcra Island (2400 ha) is loacated on the Murray River approximately 80 km west of Mildura between Lindsay and Wallpolla islands. It is bounded by the Murray River and by Potterwalkagee Creek, an anabranch of the Murray River (Figure 6.1).

There are three main inflows to Potterwalkagee Creek from the Murray River upstream of Lock 8. One of these, Stony Crossing (sill height 24.1 mAHD), provides permanent flows in the downstream section of Potterwalkagee Creek at normal Lock 8 operating pool level (24.65±0.10 mAHD for flows at Lock 9 <35000 ML day^{-1}) (SKM 2004c). Flows through the two tributaries upstream of Stony Crossing are intermittent, commencing once discharge at Lock 9 exceeds 40700 ML day^{-1} (Beovich 1994) or > 60000 Ml day^{-1} (SKM 2004c). Under current conditions, flows through these creeks occur approximately once every two to three years for an average duration of 60 days between May and November, although no flows have occurred over the past ten years. Under natural unregulated Murray River conditions, flows through these tributaries would have been twice as frequent and twice as long, and Stony Crossing would not have flowed (SKM 2004c).

Two instream structures were originally constructed for irrigation supply in Potterwalkagee Creek; a dam near the western end of Potterwalkagee Creek which raised water levels in the creek to approximately that of Lock 8, and a culvert further downstream where the creek meets a backwater created by the Lock 7 weir. These two structures are no longer required as extractions for irrigation no longer take place. Whilst the dam has been breached and allows water to pass through unhindered, the culvert restricts flow (SKM 2004c).

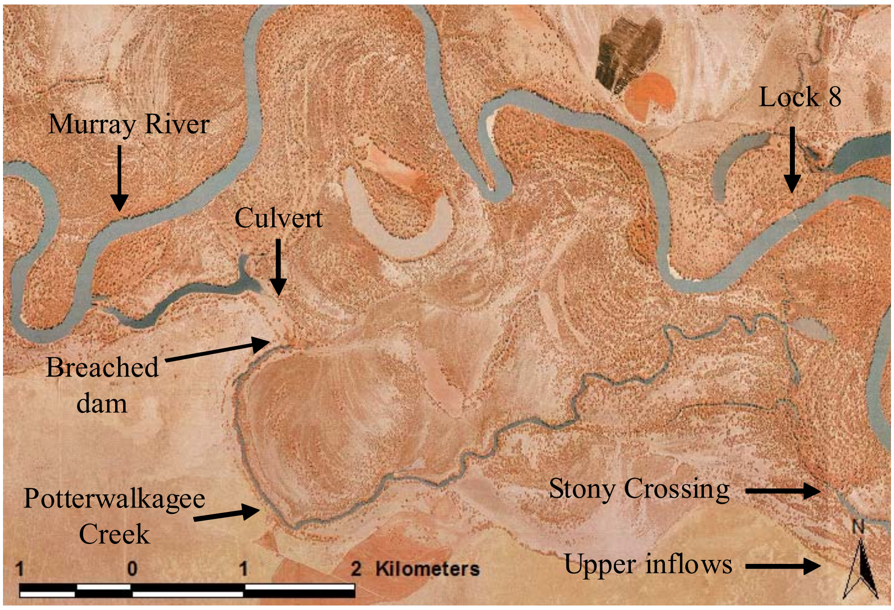

Figure 6.1: Mulcra Island showing locations of proposed structures on Potterwalkagee Creek (source MDBC River Murray Mapping 2nd edn. May 1996).

Regulation of the Murray River has changed the flow regime of waterways on the Island and has impacted on the Island's ecological values. In the upper reaches of Potterwallkagee Creek (upstream of Stony Crossing) where inundation frequency and duration have been reduced, vegetation is exhibiting signs of moderate to high stress as a result of lack of water. The mid and lower reaches of Potterwallkagee Creek experience permanent flow and contain diverse instream habitat that supports a relatively diverse small native fish community. Whilst suitable habitat also exists in this section for large species such as golden perch and Murray cod, their access to these habitats is currently restricted. Further, most wetlands across the island are dry and have been dry for several years. Under natural conditions, most of these wetlands would have experienced inundation once every one to two years. Vegetation in and around these wetlands is showing moderate signs of stress due to lack of water (SKM 2004c).

Ecological and flow objectives were developed for four flow management options by SKM (2004c) (Table 6.1). Following initial expert consultations it was decided that Option 3, the 'Breached Dam' option, was the most likley to proceed and that this should be the first option for which a monitoring program was designed.

Ecological and flow objectives for each identified ecological value developed by SKM (2004c) for Option 3 are shown in Table 6.2. These objectives are to be met by installing a regulator at the downstream end of Potterwalkergee Creek. This will enable water levels in Potterwalkergee Creek to be increased and backed onto the floodplain to create a wetland. Preliminary Digital Elevation Modelling indicated that this could be achieved with a regulator 1.5 m above the current road level at the lower Potterwallkergee Creek crossing. However, further hydraulic modelling is required to identify the stage height-inundated area relationship.

Preliminary operating guidelines for the "Breached Dam' option regulating structure are:

- to increase the frequency and duration of inundation of floodplain wetlands that are currently too dry.

Option	Management area	Key flow objective
1	Potterwalkagee Ck. upstream of Stony Crossing.	-Increase frequency of low and high flow events.
2 Stony Crossing	Potterwalkagee Ck. downstream of Stony Crossing.	-Maintain permanent flow but increase variability by establishing a low flow period during summer and initiating higher flow events during winter that increase water velocity. -Ensure adequate passage for large bodied fish at upper and lower ends of reach.
3 Breached Dam	Wetlands connected to Potterwalkagee Ck.	-Increase frequency and duration of inundation of wetlands that are currently too dry.
4	Wetlands connected to Murray River.	-Increase frequency and duration of inundation of wetlands that are currently too dry.

Table 6.1: Preliminary flow options using regulators developed for Mulcra Island (source SKM 2004c).

Ecological value/process	Ecological objectives	Flow objectives
Lateral connectivity	Rehabilitate (connection between flood runners and low floodplain	Inundate floodplain for 2-3 months during winter-spring every 1-2 years.
Carbon and nutrient cycling	Rehabilitate	Inundate floodplain for 6-12 months every year. Timing not important.
Biofilms	Rehabilitate (sustain autotrophic processes)	Inundate floodplain for 6 months during summer-autumn every year.
Macro-invertebrates	Maintain	Inundate floodplain for 2-3 months every year during winter/spring.
Wetland vegetation	Rehabilitate (subistence)	Inundate floodplain for 3-5 months every 1-2 years during winter-spring.
Wetland vegetation	Rehabilitate (recruitment)	Inundate floodplain for 3-5 months every 1-2 years during late spring-early summer.
Waterbirds (colonial nesting)	Rehabilitate foraging opportunities	Inundate floodplain for 4-12 months during spring-early summer. Frequency not important.
Waterbirds (waterfowl and grebes)	Rehabilitate foraging opportunities	Inundate floodplain for 2-8 months during spring-early summer. Frequency not important.

Table 6.2: Summary of ecological objectives and flow requirements for Potterwalkergee Creek under Option 3 ('Breached Dam' option) (source SKM 2004c).

6.2 Conceptual framework and hypotheses

The floodplain wetlands have intrinsic local values, and represent an important component of the larger floodplain ecosystem (Ward 1989, Ward and Stanford 1995). They are naturally diverse and productive habitats, whose management through the alteration of natural flow regimes has resulted in either real or perceived declines in wetland condition (*e.g.* Kingsford 2000a,b). It is generally agreed that both wet and dry periods are important in maintaining ecosystem integrity in ephemeral wetlands (Boulton and Lloyd 1992, Bunn *et al.* 1997, Boulton and Jenkins 1998). Disturbances, such as flooding and drying, drive aquatic and terrestrial successional processes and facilitate biotic and abiotic exchanges between elements of the floodplain and the riverine environment (*cf.* Flood Pulse Concept; Junk *et al.* 1989). Because of this, ephemeral wetlands are potentially sites of high productivity and diversity within floodplain ecosystems. Consequently, the management of these wetlands has implications for ecosystem productivity and diversity at the landscape scale. Thus, any changes in wetland function initiated through hydraulic modification need to be viewed from both the perspective of the individual wetland and from a landscape perspective.

Available information, conceptual models (*e.g.* Flood-Pulse Concept - Junk *et al.* 1989, Alternative-States Model - Scheffer *et al.* 1993, Geomorphic-Trophic Model - Hershey *et al.* 1999, Trophic-Cascade Model - Carpenter *et al.* 2001, Ephemeral Deflation Basin Lake Model - Scholz and Gawne 2004a,b) and expert consultation (refer to Acknowledgements) were used to develop a conceptual model of the links between the various ecosystem components and their hypothesised responses under Option 3 ('Breached Dam' option) (Figures 3.3 and 3.4). From these we generated a preliminary set of testable hypotheses which will be used to guide the development of the monitoring program. These hypotheses will be subject to some revision once hydraulic modelling data showing the extent of floodplain inundation under Option 3 becomes available.

THREATENING PROCESS	ENVIRONMENTAL VALUES/PROCESSES
-Reduced frequency and duration of inundation of floodplain wetlands. (no overbank flooding during past 10 years) →	**Lateral connectivity**-Reduced connection between Potterwalkergee Creek and floodplain wetlands. **Groundwater** -Groundwater levels historically low. **Water quality** -None. **Vegetation** -No macrophytes. -Fringing red gum and black box stressed by lack of water. **Algae** -None. **Invertebrates** -None. **Fish** -None. -Fish community of adjacent channel dominated by carp, goldfish and mosquitofish. **Water birds** -Low species richness. -Low abundances of generalist filter feeding, dabbling duck and piscivorous species.

Figure 6.2: Conceptualisation of ecosystem function of floodplain wetlands under existing Potterwalkagee Creek flows.

THREATENING PROCESS	ENVIRONMENTAL VALUES/PROCESSES
-Increased frequency and duration of inundation of floodplain wetlands. (aerial extent and duration of inundation to be determined)	**Lateral connectivity**-Increased connection between Potterwalkergee Creek and floodplain wetlands. **Groundwater** -Increased groundwater levels. **Water quality** -High turbidity. **Vegetation** -Increased macrophyte recruitment, biomass and species diversity. -Increased condition of fringing red gum and black box. **Algae** -Increased algal production. **Invertebrates** -Increased invertebrate production and diversity. **Fish** -Fish community dominated by exotic species (carp, goldfish, mosquitofish). -Increased recruitment potential for riverine fish community. **Water birds** -Low species richness. -Low abundances of generalist filter feeding, dabbling duck and piscivorous species.

Figure 6.3: Conceptualisation of ecosystem function of floodplain wetlands after implementation of Option 3 ('Breached Dam' option) for Potterwalkagee Creek.

In its present state, the floodplain wetlands associated with the lower reaches of Potterwalkagee Creek below Stony Crossing have not been inundated for at least 10 years. The flooding of theses areas as proposed under Option 3 ('Breached Dam' option) will likely initiate a wide ranging sequence of ecosystem processes similar to those described in previous sections for Webster's Lagoon and Horseshoe Lagoon. The primary differences between the managed inundation of the Potterwalkagee Creek floodplain and that of Webster's and Horseshoe lagoons are the periods of lakebed exposure and that it will not be possible to exclude the movement of exotic fish (*i.e.* carp, goldfish, mosquitofish) onto the Potterwalkagee Creek floodplain. Because of these differences it is anticipated that the post-inundation ecosystem responses and trajectories will differ. This is reflected in our conceptualization of ecosystem responses developed here. Note also that we have at this stage excluded from our conceptual model and the monitoring program responses likely to occur during the flood recession, or 'drying', phase. This will be possible once hydraulic modelling has been completed and the relationship between stage height and connectivity between wetland and Potterwalkagee Creek has been identified. For details of possible ecosystem responses to wetland drying we refer the reader to the models developed for Webster's and Horseshoe lagoons.

Trophic interactions

Trophic cascade or food web models suggest that aquatic ecosystem productivity and community composition are determined by a combination of top-down and bottom-up forces (Carpenter and Kitchell 1988, Carpenter *et al.* 2001). The strengths of these interactions (*e.g.* competition, predation) tend to become weaker as food web complexity increases (Shapiro 1990, Lazzaro 1997, Borer *et al.* 2005). Viewed independently, hypothesized responses of each trophic level to wetland inundation are suggestive of an overall increase in productivity and diversity. Whilst we recognise that trophic interactions are likely to play a key role in determining measured responses at each level, the strength of such interactions is difficult to either measure or predict. It will thus be important to consider likely strengths of trophic interactions when interpreting the observations derived for each of the outlined hypotheses.

Flow/hydrology

Hypothesis A1: The flow regulating structure proposed for Potterwalkagee Creek under Option 3 will have an effect on water levels in compliance with the management strategy.

The installation of a regulator at the breached dam on Potterwalkagee Creek (Option 3) will enable water levels to be increased and backed onto the floodplain to create a wetland. Preliminary Digital Elevation Modelling indicates that this could be achieved with a regulator 1.5 m above the current road level at the lower Potterwallkergee Creek crossing. However, further hydraulic modelling is required to identify the stage height-inundated area relationship. Establishing this relationship and monitoring stage heights will be critical in assessing ecosystem responses.

Lateral connectivity

Under Option 3, the backing up of Potterwalkergee Creek will increase both the availability of and connectivity between inundated areas. Hydraulic connectivity between components of the floodplain (*i.e.* channel and wetland) facilitates lateral exchanges of materials and biota. Such exchanges are widely considered as being of prime importance to maintaining ecosystem integrity (*e.g.* Junk *et al.* 1989, Bunn *et al.* 1997). For example, inundation tends to stimulate increases in system productivity and the relative availability of food resources for fish at critical times, such as spawning and breeding (*e.g.* Cushing 1990, Meredith and McCasker in prep). Subsequent recruitment success of many native riverine fish species may thus be largely dependent on the temporal regime of lateral connections between the different components of the floodplain.

The ecological objective of rehabilitating lateral connections between the channel and floodplain wetland environments will not be monitored directly, but will be inferred from assessments of stage height and hydraulic modelling.

Groundwater/salinity

Hypothesis B1: Inundating the Potterwalkagee Creek floodplain (Option 3) will raise groundwater levels.

The geology of Mulcra Island comprises four distinct layers commencing at the surface with Fine Alluvium (aquitard 0-5 m thick), underlain by Channel Sand (aquifer 5-10 m thick) and Blanchetown Clay (aquitard 5-10 m thick), and at a depth of 20 – 25 m by Parilla Sand (aquifer >10 m thick) (SKM 2004c). Salinity within the Fine Alluvium and Channel Sand ranges from 1000 – 40000 $\mu S\ cm^{-1}$.

Regional groundwater flow within Mulcra Island is generally from east to west along a hydraulic gradient which increases under the influence of the Murray River. Local groundwater levels are influenced by the pattern of overbank flooding. As no overbank flooding has occurred during the past 10 years, groundwater levels are currently at their lowest point since records commenced in 1983 (SKM 2004c).

Altering discharge or water level in Potterwalkagee Creek and associated floodplain wetlands is likely to influence the direction of groundwater flow. This in turn will impact on the volume of saline groundwater discharged in to Potterwalkergee Creek. For example, increasing channel flows during non-flood periods reduces the potential for saline groundwater discharges in to Potterwalkergee Creek. However, any decrease in groundwater discharge is likely to cause groundwater levels adjacent to the channel to rise, which may impact on vegetation if the salinity is high (SKM 2004c). The risk of this is thought considerably lower in the normally dry upper Potterwalkagee Creek area.

At present there is insufficient groundwater and hydrological data to quantitively assess potential risks to vegetation and the movement of salt within Mulcra Island. Because of the unavailability of suitable controls it will not be possible to establish causality. Testing of the hypothesis will thus be restricted to examining monitoring outcomes.

Water quality

Turbidity

Turbidity is a measure of light attenuation in water and primarily reflects the amount of suspended particulate matter in the water column. It is a principal determinant of photic depth and, especially at high levels, has the potential to limit primary production and thus ecosystem structure and function. Turbidity is influenced by a number of factors such as the quality of source waters, the capacity for entrained particles to settle, and the susceptibility of the sediments to re-suspension. The susceptibility of sediments to re-suspension is influenced by sediment structure, water depth, and bioturbation, by carp for example. Numerous studies have attributed increases in turbidity to the benthic feeding behaviour of carp (Lamarra 1975, Meijer *et al*. 1990, King *et al.* 1997, Robertson *et al.* 1997), and this is likely also to be true for the Potterwalkagee Creek floodplain following its inundation. Although no environmental objective was developed for surface water turbidity, monitoring turbitity will assist with assessing changes in trophic structure and function.

Nutrients (nitrogen and phosphorus)

Hypothesis C1: Inundating the Potterwalkergee Creek floodplain (Option 3) will stimulate the release of a short-lived large pulse of N and P.

Although no data is available to indicate the extent to which nutrient cycling within the Potterwalkagee Creek floodplain has been impacted as a consequence of prolonged drying, the rehabilitation of nutrient cycling has been identified as a key ecological objective under Option 3 (‘Breached Dam’ option). This objective is to be met by inundating the floodplain wetland for 6 – 12 months every year.

The exposure of wetland sediments stimulates the mineralisation of nutrients in the sediments (McComb and Qiu 1998, Baldwin and Mitchell 2000). Depending on sediment organic matter content and the duration of exposure, a potentially significant pool of mineralised (or bio-available) nutrients may be flushed from the sediments on inundation (Baldwin 1996, McComb and Qiu 1998, Baldwin and Mitchell 2000). Flooding is commonly associated with increased system productivity fuelled by an initial pulse of nutrients, derived from both the inflowing water and sediments releases (*e.g.* Scholz *et al*. 2002, Scholz and Gawne 2004a). This nutrient pulse is, however, short-lived (weeks) with water column nutrient concentrations declining steadily as nutrients become more tightly coupled with biotic and abiotic uptake and release processes (*e.g.* Baldwin and Mitchell 2000, Scholz *et al*. 2002). We therefore anticipate that flooding of the previously dry Potterwalkagee Creek floodplain will stimulate an initial pulse of increased water column N and P concentrations.

Wetland vegetation

Hypothesis D1: Inundating the Potterwalkergee Creek floodplain (Option 3) will result in an increase in the aereal extent of submerged aquatic vegetation.

Hypothesis D2: Inundating the Potterwalkergee Creek floodplain (Option 3) will result in an increase in the diversity of submerged aquatic vegetation.

The distribution of native vegetation (ecological vegetation classes or EVCs) is to a large extent influenced by flood patterns and soil type. Eight of the ten EVCs identified on Mulcra Island are considered directly dependent on flooding and the creek system (White *et al.* 2003 in SKM 2004c). In a recent survey (May 2004), SKM (2004c) found that all water dependent vegetation communities on the island had experienced a decline in health, either as a result of too much water or too little water. Whilst flow objectives have been developed for 5 of the 8 EVCs present on Mulcra Island, only one EVC (wetland vegetation) is likely to be directly impacted on under Option 3 ('Breached Dam' option). Although we have restricted our discussion here to the wetland vegetation community, the assessment of potential impacts on the condition of other EVCs will become possible once more detailed EVC mapping and hydraulic modelling data becomes available.

Wetland vegetation (*i.e.* aquatic macrophytes) play an important role in aquatic ecosystems (refer to review by Carpenter and Lodge 1986). They may contribute to reducing turbidity, provide an important conduit for the transfer of oxygen to the sediments during plant growth, and nutrients to the water column during decay (Boon and Sorrell 1991), and provide substrata for the development of biofilms, which in turn provide an important food resource for a wide range of invertebrates and fish (Cattaneo 1983, Bunn and Boon 1993). Macrophytes also provide important refuges for invertebrates and fish (Lillie and Budd 1992). The annual cycles of macrophyte growth and die-back may also have important consequences for water quality (dissolved oxygen, pH) and the accrual of organic matter on the sediments (Carpenter and Lodge 1986).

Flow objectives for in-channel and wetland vegetation are aimed at rehabilitating aquatic vegetation communities by increasing the variability of instream flows and overbank flooding. We anticipate that inundating the Potterwalkagee Creek floodplain wetland under Option 3 will favour the establishment of a more abundant and diverse macrophyte community within the wetland than is present within the currently inundated areas of Potterwalkagee Creek. However, we recognise that observed responses of the macrophyte community after inundation may be confounded by the diversity and availability of plant propagules (either already present within the wetland or colonisers arriving via water or aerial pathways), the timing of the proposed disturbance (seasonality of germination and establishment), grazing interactions and physical disturbances by carp for example.

Algae

Hypothesis E1: Inundating the Potterwalkergee Creek floodplain (Option 3) will stimulate an initial short-lived pulse of increased algal biomass.

Algae represent a key component of total wetland primary production (*e.g.* Wetzel 1964, Hargrave 1969, Wetzel *et al.* 1972). This is especially true in systems where macrophytes are absent or their distribution is restricted to a relatively small area, such as occurs on the Potterwalkagee Creek floodplain. Algae occur either suspended within the water column (phytoplankton) or attached to substrata (phytobenthos). Numerous studies have shown production by either fraction to be influenced by light availability, nutrients and by substratum quality (*e.g.* Turner *et al.* 1983, Cox 1988, 1990a,b,c, Reynolds and Descy 1996, Havens *et al.* 1998), although taxon specific responses do vary greatly (Reynolds 1997, Huszar and Caraco 1998).

We anticipate that algal production and the relative contributions of phytoplankton and phytobenthos to total algal production following the inundation of the Potterwalkagee Creek floodplain will be influenced principally by two factors; nutrient availability and the presence of carp.

As discussed earlier, the inundation of previously dry floodplain sediments is often associated with a short-lived pulse of nutrients derived from both the inflowing water and sediments releases (Baldwin 1996, McComb and Qiu 1998, Baldwin and Mitchell 2000, Scholz *et al.* 2002). We anticipate that such increases in nutrient availability will likely fuel an initial pulse of increased algal production/biomass. Subsequent declines in algal production/biomass are also expected once nutrient availability decreases (*i.e.* nutrients become more tightly coupled with biotic and abiotic uptake and release processes) and grazing pressure increases.

Several studies have linked the presence of carp with increases in turbidity and suspended nutrient concentrations (*cf.* King *et al.* 1997, Robertson *et al.* 1997). This is likely to directly influence the relative abundances of planktonic and benthic algae in several ways. Firstly, benthivory of carp increases the potential for benthic algae to become light limited by physically disturbing the sediments and reducing the penetration of light. And secondly, physical disturbance of the water column and sediments by carp maintains phytoplankton in suspension, increasing their retention within the photic zone, and increasing their access to nutrients and light. Accordingly, phytoplankton production is likely to be elevated and phytobenthic production inhibited (*cf.* Ogilvie and Mitchell 1998). As phytoplankton and phytobenthos are each likely to support different grazer communities, the relative distribution of production between these algal fractions is likely to influence wetland trophic structure. As Option 3 (or any of the other flow management options) will not prevent the movement of carp onto the Potterwalkagee Creek floodplain as it floods, assessing the impact of carp on the partitioning of production between phytoplankton and phytobenthic fractions will not be possible. However, this will be possible at Webster's and Horseshoe lagoons.

Invertebrates

Zooplankton

Hypothesis F1: Inundating the Potterwalkergee Creek floodplain (Option 3) will stimulate an initial short-lived pulse of increased zooplankton density lasting approximately 4 weeks.

Zooplankton provide major links in aquatic food chains by facilitating the transfer of nutrients, carbon and energy between bacteria, algae and higher consumers, such as fish and water fowl (*e.g.* Boon and Shiel 1990, Boulton and Jenkins 1998, Humphries *et al*. 1999). Because of this, zooplankton plays a key role in structuring ecosystem function (*e.g.* Shapiro *et al*. 1975, Lazzaro 1997).

Zooplankton community structure is influenced by many factors, including life history characteristics, food availability, predation pressure, water quality, habitat diversity and complexity, and exchanges between wetland and riverine environments (Wiggins *et al.* 1980, Shiel and Walker 1984, Williams 1985, Shiel 1985, 1986, 1995). Many of these factors are influenced by season and in ephemeral systems by drying and flooding (*e.g.* Scholz and Gawne 2004a). Despite potentially complex interactions between these factors, generalised responses of zooplankton to the inundation of the Potterwalkagee Creek floodplain are to be expected.

We anticipate that the inundation of the previously dry Potterwalkagee Creek floodplain will stimulate an initial pulse of zooplankton density driven initially by floodwater borne immigrants which are subsequently replaced by emergents from the sediments during the first month after flooding (Boulton and Lloyd 1992, Maher and Carpenter 1984, Jenkins and Boulton 2003). Such pulses in post-inundation zooplankton density are thought to increase feeding opportunities and the recruitment success of native fish species and waterbirds (*e.g.* Cushing 1975, Maher 1984, Maher and Carpenter 1984, Crome 1986, Cushing 1990).

The duration of the initial post-flood pulse of zooplankton production has been shown to vary between wetlands, but generally decline over the first month of inundation as the intensity of trophic interactions increase (*e.g.* Carpenter *et al.* 2001, Scholz and Gawne 2004a,c). The magnitude of initial increases in zooplankton density after flooding has been linked to both the frequency of inundation and to the duration of the preceding dry period. For example, Boulton and Lloyd (1992) reported that wetlands that experience frequent (annual) episodes of flooding tend to be more productive than those that flood only infrequently (once in 22 years). Whilst longer dry phases tend to reduce the viability of aestivating individuals and of eggs and cysts deposited in the sediments, studies in dryland systems indicate that resting eggs are extremely long lived and eggs can survive dry periods of 20, 50 and even 100 years and still emerge once the sediments are flooded (Hairston *et al.* 1995, Jenkins and Briggs 1997 Jenkins and Boulton 1998).

As the Potterwalkagee Creek floodplain has not been inundated for at least 10 years, we expect the magnitude of initial post-inundation responses in zooplankton to be lower than those that are likley to be encountered in more frequently inundated systems, such as Webster's and Horseshoe lagoons, although direct comparisons are confounded by differences between systems such as the strengths of trophic interactions and historical wetting and drying frequencies.

Macro-invertebrates

Hypothesis F2: Inundating the Potterwalkergee Creek floodplain (Option 3) will stimulate an initial short-lived pulse of increased macro-invertebrate density lasting approximately 8 weeks.

Macro-invertebrates are important consumers within wetland ecosystems, occupying a range of functional groups (*e.g.* grazers, detritivores, filter feeders, predators) and provide the principal food source for many vertebrates such as fish and birds (Bunn and Boon 1993). Macro-invertebrates have proven effective indicators of ecosystem health in stream environments (Cranston *et al.* 1986) and their use as indicators in wetlands has been advocated (*e.g.* Davis *et al.* 1993).

Macro-invertebrate community structure is influenced by many factors, including life history characteristics, water quality, habitat quality, and exchanges between wetland and riverine environments (Wiggins *et al.* 1980, Shiel and Walker 1984, Williams 1985), and by trophic interactions (*e.g.* Carpenter and Kitchell 1988, Carpenter *et al.* 2001). Numerous studies have also shown hydrology to be a major determinant of macro-invertebrate community structure and productivity (*e.g.* Wiggins *et al.* 1980, Bataille and Baldassarre 1993, Jeffries 1994, Leslie *et al.* 1997). Despite potentially complex interactions between these structuring forces, generalised responses of macro-invertebrates to the inundation of the Potterwalkagee Creek floodplain are to be expected.

We anticipate that the inundation of the previously dry Potterwalkagee Creek floodplain will stimulate an initial pulse of macro-invertebrate abundance driven by the two processes; the rapid recolonisation of newly created habitat and favourable trophic interactions. Firstly, recolonisation may occur via several pathways, such as emergence from the sediments of desiccation resistant eggs and larvae or desiccation resistant adults, passive movement with the incoming waters, active migration, chance introduction by other animals, and aerial dispersal (Talling 1951, Wiggins *et al.* 1980, Batzer and Wissinger 1996, Hillman and Quinn 2002). And secondly, trophic cascade or food web models suggest that increases in the availability of food resources (*e.g.* zooplankton; Maher and Carpenter 1984, Boulton and Lloyd 1992) combined with reductions in predation pressure that are commonly encountered during the initial post-inundation period (*e.g.* Lake *et al.* 1989, Batzer and Wissinger 1996, Battle and Golladay 2001) will favour the establishment of abundant macro-invertebrate communities.

The little data that is available suggests that initial post-flood increases in macro-invertebrates density may persist for as little as 1-2 months (*e.g.* Scholz and Gawne 2004c) or for as long as 2 years (*e.g.* Maher and Carpenter 1984). Subsequent declines in macro-invertebrate density are anticipated once the availability of food resources decreases and the intensity of biotic interactions (*e.g.* competition, predation) increase. The presence of carp has been shown to exert a significant negative pressure on invertebrate biomass, either directly through predation or indirectly through habitat modification (*e.g.* Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al*. 1994, Tatrai *et al*. 1994). As we anticipate that floodplain inundation will likely stimulate spawning by native small-bodied fish and that non-native fish, such as carp, will be present (refer to the following discussion of fish responses), it is likely that any observed post-inundation pulse of macro-invertebrate density will be relatively short-lived.

Fish

Hypothesis G1: Inundating the Potterwalkergee Creek floodplain (Option 3) will stimulate spawning in small-bodied native fish species.

Fish provide an important link in wetland food webs through their consumption of prey and their consumption by birds. Wetland hydrology plays an important role in structuring fish assemblages though its influence on ecosystem productivity, habitat availability and flow related cues for fish spawning. Native fish throughout the Murray-Darling Basin have been severely impacted by altered flow regimes, the loss of habitat, and barriers to passage (MDBMC 2002). Altered flow regimes, in particular, are thought to have impacted on the distribution of native fish species and to have favoured the expansion of alien species, such as carp (*Cyprinus carpio*) and mosquitofish (*Gambusia holbrooki*) (Harris and Gehrke 1997). Whilst fish distributions within the Basin have been reported (Llewellyn 1983, Harris and Gerhke 1997), and responses of fish communities to flooding in Australian floodplain systems have received some attention (*e.g.* Gerhke 1991, 1994, Gehrke *et al*. 1997, McKinnon 1997, Humphries *et al.* 1999, King *et al*. 2003, Scholz and Gawne 2004a,c), little information is available regarding the fish commumities of Mulcra Island.

In two surveys of Potterwalkagee Creek in 2001 and 2004 SKM (2004c) recorded nine species including the FFG 1988 listed Murray River rainbowfish (*Melanotaenia fluviatilis*) and the vulnerable (DSE 2003) golden perch (*Macquaria ambigua*). Despite the availability of apparently suitable habitat, no Murray cod (*Macculochella peeli peeli*) have been recorded from Potterwalkagee Creek. Fish catches were dominated by the exotic carp (*Cyprinus carpio*) and the native bony herring (*Nematolosa erebri*). Fish with no flow related spawning requirement, such as Australian smelt (*Retropinna semoni*), bony herring and carp gudgeon (*Hypseleotris* spp.) were more abundant relative to species whose recruitment is enhanced by overbank flooding such as golden perch and Murray cod.

The overall ecological objective for the fish community on Mulcra Island is to retain a diverse community of small native fish and provide increased opportunity for colonisation by large-bodied native fish. The main constraint for large-bodied fish in Potterwalkagee Creek is access to and from the Murray River. Removal of the culverts at the Stony Crossing inlet and at the outlet into the Lock 7 weir pool will likely facilitate fish passage. Increasing the frequency of spring/summer overbank flows has also been suggested to stimulate recruitment of larger native fish, however the occurrence of such flows is dependent on flows in the Murray River and can not be imposed using the control structures proposed for any of the flow management options. Whilst issues such as flow variability and increased fish passage/access to Potterwalkagee Creek are to be addressed principally under Option 2 (Stony Crossing option), we anticipate that the inundation of the Potterwalkagee floodplain under Option 3 (Breached Dam option) is likely to greatly enhance the availability and access to potentially critical spawning and nursery habitat for a range of small-bodied native fish species.

Although most small-bodied native fish exhibit seasonal preferences for spawning, recent work has shown that most (if not all) have the capacity to spawn opportunistically during an over-bank flood (Meredith and McCasker in prep.). This is also true for exotic species such as carp. This suggests that the initial flooding of the Potterwalkagee Creek floodplain wetland behind the breached dam will likley trigger spawning in most species. Whilst similar post-inundation responses are likley to occur at the other Icon Site wetlands, subsequent recruitment success of fish from larvae to juveniles and from juveniles to adults may be lower in this system relative to other Icon Site wetlands, such as Webster's and Horseshoe lagoons, due to two potentially important factors.

Firstly, the Potterwalkergee floodplain has not been inundated for at least the past 10 years. It is possible that this may have reduced the viability of invertebrate propagules deposited within the sediment (*e.g.* Hairston *et al.* 1995, Jenkins and Briggs 1997, Jenkins and Boulton 1998), thereby reducing the availability of this potentially critical food resources required by juvenile fish.

And secondly, recruitment success of small-bodied native fish within the Potterwalkagee Creek system may be compromised by the presence of exotic fish (*i.e.* carp, goldfish, mosquitofish). Under Option 3 (or any of the other flow management options) it will not be possible to exclude the movement of exotic fish onto the floodplain. Although the evidence is fragmented and sometimes contradictory, carp have been implicated in the demise of native fish stocks by reducing spawning site availability (*i.e.* destruction of macrophytes; Hume *et al*. 1983, Brown 1996), by reducing the efficiency of visual feeding (*i.e*. increasing turbidity; Lamarra 1975 Roberts *et al.* 1995), and by competing for food resources (Hume *et al.* 1983, Fletcher 1986, Richardson *et al.* 1990, Wilcox and Hornbach 1991, Cline *et al*. 1994, Tatrai *et al*. 1994).

Waterbirds

Hypothesis H1: Inundating the Potterwalkergee Creek floodplain (Option 3) will result in an increase in the cumulative species richness of waterbirds.

Hypothesis H2: Inundating the Potterwalkergee Creek floodplain (Option 3) will result in an increase in abundances of generalist filter feeding and dabbling duck birds (all birds of the genus Anas, pink eared-duck, hardhead).

Hypothesis H3: Inundating the Potterwalkergee Creek floodplain (Option 3) will result in an increase in the abundance of piscivorous birds that prefer smaller fish (Caspian tern, white-faced heron, great egret, little egret).

Hypothesis H4: Bird assemblage composition of the inundated Potterwalkergee Creek floodplain (Option 3) will change over time, becoming increasingly dominated by piscivorous species.

Waterbird abundance is determined by the reproductive success of adults and the survival of post-juvenile birds. The breeding and survival of most waterbirds in the Murray-Darling Basin have been linked to cycles of flooding and drying (Briggs 1990); with most species being capable of moving large distances between wetlands as local breeding and feeding opportunities fluctuate. Almost all common waterbirds of the Murray-Darling Basin breed following wetland flooding, with some also breeding seasonally, usually in spring. In only 2 species (both ducks) is breeding always seasonal and apparently unaffected by wetland hydrology (Briggs and Lawler 1991).

Flow regulation within the Murray-Darling Basin has altered the temporal and spatial distribution of inundated floodplain wetlands available to waterbirds. Regulation has permanently inundated many formerly intermittent wetlands, and reduced the average duration of inundation for many others. Whilst this has increased the availability of survival habitat for waterbirds at the landscape scale, it has effectively reduced the availability of breeding habitat. Thus, the overall effect of river regulation on waterbirds is likely to be reduced recruitment of young, but enhanced survival of adults (Briggs and Lawler 1991). Because of this, it is important that the number of breeding opportunities for waterbirds within the Murray-Darling Basin be increased. As is discussed below, inundating the Potterwalkagee Creek floodplain is likely to achieve this objective.

The nesting requirements of Australian waterbirds vary (Serventy 1985), but all require adequate food to satisfy the energetic and other costs of nest finding or building, the laying and incubating of eggs, and the rearing of young. Whilst these costs may be partly provided for by the accumulation of body fat and/or protein reserves prior to laying (*e.g.* Thomas 1988, Norman and Hurley 1984, Briggs 1988), most waterbirds of the Murray-Darling Basin require readily available food of high quality at nesting time for successful breeding (Briggs and Lawler 1991). Many studies have shown the inundation of previously dry wetlands to initiate a highly productive succession of potential food resources (*e.g.* invertebrates and fish), which attracts many waterbird species and stimulates breeding (*e.g.* Maher 1984, Maher and Carpenter 1984, Crome 1986, 1988). Recently flooded wetlands thus provide a potentially important window of opportunity for breeding by many waterbird species. Over time after inundation and as wetlands begin to dry; changes in the composition and availability of food resources tend also to stimulate changes in the structure of waterbird assemblages (Scott 1997), increasing the cumulative diversity of species utilising the wetland.

In addition to food availability, post-inundation waterbird breeding success is reliant on the persistence of adequate food resources and of nesting habitat throughout the breeding (rearing) cycle. The current regime of floodplain inundation at Potterwalkagee Creek is not tailored for colonial waterbird breeding. Many colonial water birds require inundation under mature red gum trees for between 3-6 months and up to 10 months for successful breeding (Briggs and Thornton 1999). This does not currently occur. Creating suitable nesting would require the inundation of suitable vegetation (*e.g.* red gums, lignum). At this stage, hydraulic modelling is required to establish the capacity of the regulating structure to achieve this. Consequently no objective for providing for the breeding requirements of colonial waterbirds has been included.

A large number of waterbird species have been recorded on Mulcra Island. Many of these are considered threatened (refer to Appendix C in SKM 2004c). SKM (2004c) identified ecological objectives for three groups of birds; colonial nesting birds, waterfowl and grebes, and other water dependent birds. However, for purposes of developing testable hypotheses relating to the responses of waterbirds to the inundation of the Potterwalkagee floodplain (Option 3), bird species were divided into guilds (groups of species that exploit similar resources in a similar manner, but that are not necessarily closely related taxonomically). Two guilds were identified; the generalist filter-feeders and dabbling ducks and the piscivores that prefer smaller fish.

We anticipate that the creation of additional nichès in response to the inundation of the Potterwalkagee floodplain will stimulate an increase in the cumulative species richness of waterbirds utilising the wetland. Further, successional shifts in the availability and distribution of major food types and habitat throughout the wet/dry cycle will be reflected by shifts in the structure of waterbird assemblages. We anticipate that on re-flooding of the wetland, an initial burst of wetland productivity (0.5-2 months) will stimulate increases in the abundance of waterbirds, particularly the generalist filter-feeding and dabbling duck guild. This guild comprises all native species of Anas and pink-eared duck (*Malacorhynchus membranaceus*) and the hardhead (*Athya australis*) (*cf.* groupings of Roshier *et al*. 2002). We also anticipate a directional shift in waterbird assemblage composition over a period of 6-8 months of inundation, with the establishment of species that eat small fish as a major part of their diet, particularly Caspian tern (*Sterna caspia*), white-faced heron (*Egretta novaehollandiae*), great egret (*Egretta alba*), and perhaps little egret (*Egretta garzetta)*. We also anticipate the development of a longer term shift in assemblage composition if the 'structural biophysical' nature of the Potterwalkagee Creek wetland steadily changes, *e.g.* cover-dependent wetland-dependent species may colonise if dense reed beds and sedge beds develop. Species likely to establish as a consequence of such changes include; the clamorous reed warbler (*Acrocephalus stentoreus*), little grassbird (*Megalurus gramineus*), crakes and rails, and grazing waterfowl.

As wetland waterbird populations are highly mobile and are influenced by processes occurring at landscape scales, they are inherently very variable. Whilst we anticipate that we will be able to detect short-term responses in waterbird numbers between dry and flooded phases, the ability to reliably detect such responses in waterbird assemblages to changes in hydrology will likely require the collection of long term data extending over multiple wet/dry cycles.

Waterbird abundance is particularly expensive in both time and resources to monitor adequately. As no control site exists for Potterwalkergee Creek, we recommend that the monitoring of waterbirds at Potterwalkergee Creek be foregone in favour of concentrating effort at Webster's Lagoon for which a suitable experimental control exists permitting the establishment of cause and effect relationships, and for which we would expect similar post-inundation responses of waterbird populations.

6.3 Response variables

Response variables to be measured as part of the monitoring program were identified for each of the hypotheses stated in the previous section. Standard sampling and analytical protocols for each response variable are listed in Table 6.2.

Response variable	Field and laboratory method
Surface water	
Height (mAHD)	Automated logging using Odyssey pressure sensors (Dataflow Systems Pty Ltd, Christchurch New Zealand).
Area and volume	Capacity tables constructed from profile surveys.
Groundwater	
Depth (m)	Measured from piezometers located along transects radiating outward from the wetland (locations yet to be determined based on hydraulic modelling). Surveying of piezometers to mAHD will enable comparisons of water tables.
Electrical conductivity (EC; µS cm^{-1})	EC (standardized to 25 ^oC) of water samples extracted from piezometers measured in the field using a U-10 multi-probe (HORIBA Ltd., Australia).
Water quality	
Turbidity (NTU)	Turbidity of 500 ml sub-surface water samples determined using a U-10 multi-probe (HORIBA Ltd., Australia). Samples diluted 1:3 with distilled water where turbidity exceeds the max. level of detection of 1000 NTU.
Total nitrogen (TN; mg N l^{-1})	Unfiltered sub-surface 200 ml samples stored frozen. TN determined colorimetrically after pre-digestion in NaOH-$K_2S_2O_8$ and oxidation to nitrate (APHA 1995). Detection limit ± 0.019 mg N l^{-1}.
Total phosphorus (TP; mg P l^{-1})	Unfiltered sub-surface 200 ml samples stored frozen. TP determined colorimetrically using the phosphomolybdate-blue method after pre-digestion in NaOH-$K_2S_2O_8$ and oxidation to orthophosphate (APHA 1995). Detection limit ± 0.0025 mg P l^{-1}.
Oxides of nitrogen (NOx; mg N l^{-1}),	10 ml of 0.2 µm filtered water samples stored frozen. NOx determined colorimetrically after its reduction to nitrite using a cadmium column (APHA 1995). Detection limit ± 0.003 mg N l^{-1}.
Filterable reactive phosphorus (FRP; mg P l^{-1})	10 ml of 0.2 µm filtered water samples stored frozen. FRP determined colorimetrically using the phosphomolybdate-blue method (APHA 1995). Detection limit ± 0.001 mg P l^{-1}.
Wetland vegetation	
Aereal cover (ha) and distribution	Survey protocol to be determined. This will be consistent across Icon Site wetlands to permit comparisons at the regional scale.

Table 6.2: Response variables, measurement units and analytical methods used to test hypotheses developed for Potterwalkagee Creek under Option 3 (‘Breached Dam’ option).

Response variable	Field and laboratory method
Algae	
Phytoplankton biomass (CHLp; µg CHL l^{-1}, CHL m^{-2})	Phytoplankton biomass measured indirectly as chlorophyll *a*. 500 ml of sub-surface water passed through glass fibre filters (Whatman Pty. Ltd. GF/C). 10 ml 100 % ethanol added to filters and chlorophyll extracted for 24 hours in the dark at 4°C then at 80°C for 10 minutes. Supernatant centrifuged and chlorophyll *a* measured spectro-photometrically at 665 nm and 750 nm without acidification (APHA 1995). Chlorophyll concentrations expressed per unit volume and, by recording water depth at time of sampling, per unit area.
Phytobenthos biomass (CHLb; µg CHL m^{-2})	Phytobenthic biomass measured indirectly as chlorophyll *a*. 100 ml 100 % ethanol added to 18 cm^{-2} sediment surface cores (1 cm depth) chlorophyll extracted for 24 hours in the dark at 4°C then at 80°C for 10 minutes. Supernatant centrifuged, and chlorophyll *a* measured spectro-photometrically at 665 nm and 750 nm without acidification (APHA 1995). Chlorophyll concentrations expressed per unit area.
Invertebrates	
Zooplankton density (individuals l^{-1}) and species richness	75 l water passed through 50µm mesh size net and concentrates preserved in 70 % ethanol. A minimum of 100 individuals counted from at least 4x1 ml sub-samples using a Sedgewick-Rafter Cell (APHA 1995). Identifications to at least genus according to Shiel (1995) and Ingram *et al*. (1997).
Macro-invertebrate density (individuals l^{-1}) and species richness	20 m benthic sweep samples collected using a 250 µm mesh size net and concentrates preserved in 70 % ethanol. A minimum of 100 individuals or whole sample counted and identified to genus level according to Williams (1980), Hawking and Smith (1997) and Anderson and Weir (2004). This is the rapid bio-assessment protocol of Tiller and Metzeling (1998) (see also Coysh *et al.* 2001). Compared with other commonly used active and passive sampling methods, sweep netting provides similar diversity though lower abundances (Humphries *et al*. 1998).
Fish	
Relative abundance (fish $CPUE^{-1}$) and biomass (kg $ha^{-1}CPUE^{-1}$)	For each sampling event 6 large fyke nets (LFN) (set tangential to shore with cod end away from shore), 6 small fyke nets (SFN) (set parallel to shore with pairs set in opposite directions) and 6 larval light traps (LT) set overnight. Set and pull times recorded and total catch adjusted to 24 hours (= 1 CPUE). Standard lengths (±0.05 mm for LFN and ±0.005 mm for SFN and LT) and weights (±0.5 g for LFN and ±0.0005 g for SFN and LT) recorded for the first 30 individuals (including the largest and smallest individuals caught) per species. Fish identifications follow McDowall (1996). Carp gudgeons identified to genus level only (*i.e. Hypseleotris* spp.) owing to the current taxonomic uncertainty at the species level (Bertozzi *et al.* 2000). All large native fish returned. Ethics approval obtained prior to sampling.
Waterbirds	
Abundance (birds $wetland^{-1}$)	Survey protocol to be determined. This will be consistent across Icon Site wetlands to permit comparisons at the regional scale.

Table 6.2 cont.: Response variables, measurement units and analytical methods used to test hypotheses developed for Potterwalkagee Creek under Option 3 ('Breached Dam' option).

6.4 Experimental Design

6.4.1 Finalization of design process

The conceptual models, hypotheses and response variables outlined above for the future management of Potterwalkagee Creek represent the refinement of conceptual models developed as part of the expert panel approach to hypothesis development undertaken at Webster's Lagoon (see Section 3.2). They are based on a conceptual understanding of what might be achievable with the instigation of the 'Breached Dam' option (installing regulators on the Lower Potterwalkagee Creek to enable wetlands to be inundated by backing water up in the creek). The extent of floodplain inundation, and therefore the spatial context for this conceptual understanding, will only be known once hydraulic modelling has been completed. Without explicit knowledge as to which sections of the floodplain will be inundated, and the effect future inundation will have on key biota, it is therefore not currently possible to further the monitoring design process.

It is envisaged that hydraulic model development will be completed by October 2005. Incorporated within the development of the model there will need to be a 'test case' scenario run, and to accelerate the development of the monitoring program, it has been agreed that the Breached Dam option will provide this test case.

Once the hydraulic model has been completed and the Breached Dam option run, finalisation of the monitoring design process and sampling protocols will consist of the following steps (Table 6.3):

Date	Activity
October 2005	Finalisation of hydraulic model, including model run of 'Breached Dam' option at Potterwalkagee Ck
November 2005	Key members of the monitoring design expert panel will be engaged to re-examine existing conceptual models using spatially explicit hydraulic model data
December 2005	New spatially explicit hypotheses will be developed with particular relevance to key biota that will/will not be inundated
December 2005	In accordance with finalised hypotheses, experimental design and sampling protocols will be defined
December 2005	Monitoring conceptual models, hypotheses, experimental design and sampling protocols reviewed by the Technical Review Committee of the Mallee CMA
January 2006	Pre-intervention (*i.e.* 'Before') monitoring will commence

Table 6.3: Timeframe for finalising the monitoring design process.

As the hydraulic modelling of the breached dam option will be used to inform key structure design and implementation requirements, it is envisaged that construction and operation of the final structure will not be complete until after January 2007. As such, the completion of at least 12 months of pre-intervention (*i.e.* 'Before') data will have occurred. Although 24 months would be ideal, 12 months is considered sufficient for the analysis of monitoring data using a BACI type design, as per Webster's Lagoon.

Independent of future monitoring developments for the 'Breached Dam' option, there is currently being undertaken a feasibility investigation into the upgrading of the Stony Crossing inlet (Option 2). Associated with the increased capacity for inflows that an upgraded structure would provide is an increased potential for instigating water level variability (independent of base flow operations at Lock 8), and the improvement of fish passage into Potterwalkagee Creek. In line with the process that has been undertaken for the development of such options at Webster's Lagoon, Horseshoe Lagoon and Lake Wallawalla (Zukowski and Meredith draft), this Stony Crossing feasibility study will identify site specific ecological objectives to be met by the installation of an ungraded structure. It is from these objectives that the development of specific monitoring hypotheses will be possible. Under the current monitoring program outlined in this report, the development and monitoring of such hypotheses will be negotiated with the Mallee CMA and the MDBC as appropriate.

7 Hattah Lakes

7.1 Background

The Hattah Lakes Significant Ecological Asset (SEA) lies within the Hattah-Kulkyne National Park, which covers 48,000 ha of River Murray floodplain. The park was designated as a Biosphere Reserve in 1981 and 12 of the lakes were listed under the RAMSAR Convention in 1982. The Hattah Lakes complex consists of 17 perennial and intermittent freshwater lakes that fill only during high flow episodes on the Murray River via Chalka Creek (Figures 7.1 and 7.2). Their hydrological regimes vary widely, ranging from lakes which used to hold water almost constantly; to those with inflows averaging 1 year in 4 and with dry spells of 4 to 12 years.

The hydrological regime of the Hattah Lakes complex has changed substantially as a result of regulation and modification of both the Murray River and Chalka Creek, an anabranch of the Murray River and the primary source of flows into the lakes. Modifications to Chalka Creek include deepening and regrading of the channel and the installation of a regulator at Messengers Crossing to prevent floodwater from receeding. Internal modifications to the Hattah Lakes system include the construction of a channel between Lockie and Hattah lakes, and the installation of an earthen bank and drop-board between Hattah and Little Hattah lakes. As a result of these changes, the frequency, magnitude and duration of lake inundation have been reduced, so that the lakes are flooded for approximately half the time that they would have been under natural conditions. Also, the timing of flood pulses into the system has been delayed by several months; the most common month of flood initiation changing from August (pre-regulation) to October (post-regulation). These changes to the hydrology of the Lakes complex have led to a perceived reduction in the ecological health and biodiversity of the region (Cumming and Lloyd 1993, SKM 2002c).

The Integrated Water Management Plan (IWMP) for Hattah Lakes (SKM 2002c, 2003b), proposed a set of ecological and flow objectives for the Lakes (Table 7.1), and recommended the following management options: the construction of 4 regulators within the Lakes complex, between lakes Hattah and Little Hattah, lakes Bulla and Arawak, lakes Yerang and Mournpall, and lakes Mournpall and Konardin, the installation of pumping stations for short and long term pumping operations, and the excavation of Chalka Creek to lower commence to flow thresholds (Figure 7.2) (SKM 2003b). These flow objectives and recommended actions were developed prior to the Living Murray Initiative and hence on the assumption that no additional inflows to the Lakes system were available.

The vision for the Hattah Lakes under the first step of the Living Murray Initiative is the maintenance and restoration of a healthy mosaic of floodplain communities. Broad ecological objectives are to have 80% of permanent and semi-permanent wetlands and 30 % of red gum forests in healthy condition, and the successful breeding of colonial waterbirds in at least 3 years in 10. While the IWMP detailed ecological, flow and management options, the appropriateness and feasibility of these will need to be reassessed in light of the potential for increased environmental flows under the Living Murray Initiative. Further, the lack of lake specific ecological objectives means that the ecological rationale for proposed flow objectives and management options for each lake are not clear. A feasibility investigation for water management of the Hattah Lakes that will address these issues is currently in progress (due December 2005).

Ecological objectives
Restore a mosaic of hydrological regimes which represent pre-regulation conditions (to maximise biodiversity).
Restore the lake macrophyte zone around at least 50 % of the lakes (to increase fish and waterbird habitat).
Improve the quality and extent of freshwater meadows so species typical of this ecosystem are represented.
Protect the ecological character of the RAMSAR site.
Increase successful breeding events of colonial waterbirds to at least 2 in 10 years.
Provide suitable habitat for migratory bird species.
Increase distribution, numbers and recruitment of local wetland fish by providing appropriately managed habitat.
Maximise use of floodplain habitat for recruitment of all indigenous freshwater fish.
Flow objectives
Increase the level of inundation permanence in lakes Hattah, Bulla, Arawak, Marramook, Brockie, Boich, Lockie and.Mournpall.
Decrease the duration and frequency of dry spells in lakes Hattah, Bulla, Arawak, Marramook, Brockie, Boich, Lockie and.Mournpall.
Maintain the current frequency and duration of inundation experienced by lakes Yerang, Yelwell, Konardin, Bitterang and Nip Nip.

Table 7.1: Ecological and flow objectives developed for the Hattah Lakes under the IWMP (source SKM 2003b).

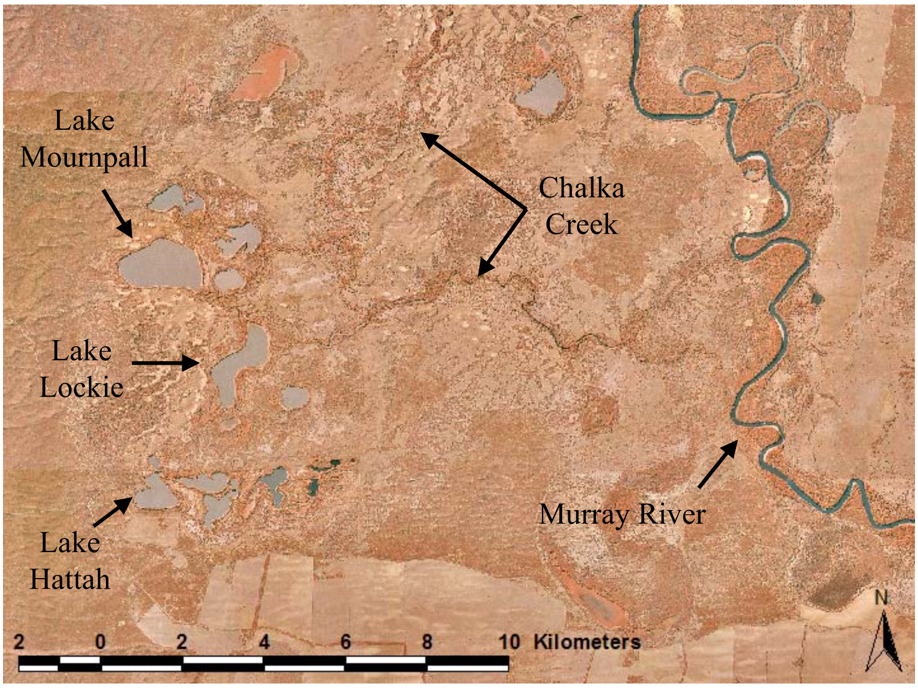

Figure 7.1: Hattah Lakes SEA (source MDBC River Murray Mapping 2nd edn. May 1996).

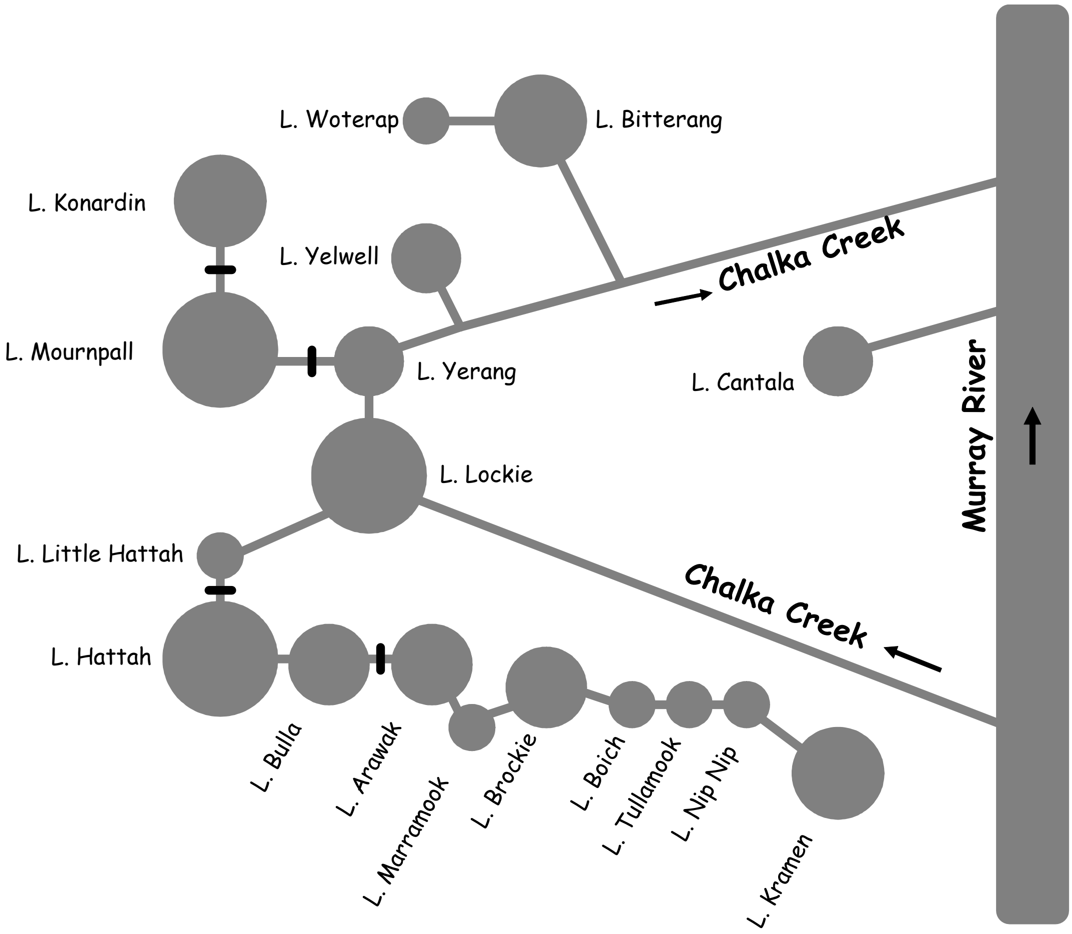

Figure 7.2: Schematic diagram of wetlands and their connections within the Hattah Lakes SEA, showing the location of regulating structures (black bars) proposed under the IWMP (source O. Scholz).

7.2 Conceptual framework

Assessing the impacts of imposed hydrological changes will, to a large extent, depend on the development of an appropriate conceptual model of ecosystem function. Such a model is needed to identify key processes and the spatial and temporal scales over which they operate, to develop key testable ecological response hypotheses, and subsequently to maximise the interpretation of monitoring data. At present, the conceptual understanding of the ecology of the Hattah Lakes is relatively poor. Conceptual models that have been developed for ephemeral semi-arid/arid systems within the Murray-Darling Basin to describe changes in ecosystem processes in response to both short- and long-term cycles of wetland wetting and drying (*e.g.* Puckridge and Walker 1996, Puckridge 1999, Scholz and Gawne 2004a,b,c) are likely to provide the most appropriate basis for developing an ecological response conceptual model for wetlands within the Hattah Lakes complex. However, this will only be possible once lake-specific flow and ecological objectives have been identified and agreed upon.

7.3 Proposed monitoring program

Monitoring being undertaken at Significant Ecological Assets (SEAs) as part of the Living Murray program is currently being undertaken at three different scales;

Monitoring of the River Murray at a Systems Scale: This will give an overview of the success or otherwise of the Living Murray Initiative. It depends on an ability to meta-analyse data from all the SEA sites, and the co-ordination of this will be the responsibility of the Murray Darling Basin Commission (MDBC).

Monitoring at the Whole-of-Asset Scale: This component of the monitoring program needs to assess change in condition of the whole asset over time, thus necessitating the use of monitoring techniques practical over large spatial scales (*e.g.* remote sensing). It is critical that the design of monitoring programs at the whole-of-asset scale incorporates the key concept of statistical power such that changes in response variable condition are measurable and interpretable. Importantly, however, monitoring being undertaken at this scale does not attempt to determine causality of any measured changes, largely because there is no 'control' site suitable at a whole of floodplain scale. Monitoring at the Whole-of-Asset scale therefore relies on 'Intervention Monitoring' (see below) to interpret observations.

Intervention Monitoring: This component of the monitoring program examines the causes and effects of managed interventions (*e.g.* re-instatement of a dry phase into wetlands, inundation of low-lying floodplain) on key biota and ecological processes. Cause-effect relationships are established through a rigorous hypothesis based scientific monitoring program such as that developed for Webster's Lagoon. A key component of this level of monitoring is the establishment and testing of expert derived conceptual models that outline the expected direction of ecological response and causal factors that are expected to elicit changes in the abundance, density, vigour or 'health' of key biota and processes. Such conceptual models are a necessary component of any adaptive management program as they can provide explanations for unpredicted or adverse ecological responses and can also be used to determine appropriate remedial (*i.e.* 'adaptive') management actions (Nyberg 1999a,b).

Previous monitoring designs described in this report target Intervention scale monitoring for Webster's Lagoon, Lake Wallawalla, Potterwalkagee Creek and Horseshoe lagoon. At each of these sites, considerable previous work has gone into identifying ecological and flow objectives, and into designing regulatory structures and operational rules that are capable of delivering an appropriate managed flow regime (Egis 2001, SKM and Roberts 2003, SKM 2004a-h). For Hattah lakes, however, this work is still not complete. It is therefore not possible to begin intervention scale monitoring at this site as was originally planned. Instead, the immediate focus for the monitoring program at Hattah Lakes will be at the whole-of-asset scale.

Currently a broad range of ecological objectives (also referred to as 'targets' in South Australia) are being monitored at the whole-of-asset scale across different SEA sites. At Chowilla, for example, monitoring has (at least in part) commenced, and is being undertaken by a range of state agencies and PhD students for the 24 preliminary 'targets' listed in Table 7.2. At Gunbower-Koondrook-Perricoota, the whole-of-asset monitoring program is set up somewhat differently. Here 27 ecological objectives have been derived from a set of 22 targets (Table 7.3). Eighteen Programmed Monitoring Activities (PMAs) then collect data to inform a tier of 50 'indicators' below these ecological objectives, and these 'indicators' are then used to assess trends in a meaningful way and inform the ecological objectives. As part of this process, whole-of-asset scale data is being collected at Gunbower-Koondrook-Perricoota to examine wetland condition, understorey condition and crown condition/tree health. Also, satellite imagery is being used to determine flood extent (and, in the future, possibly vegetation health), and the Murray Darling Freshwater Research Centre has recently been engaged to monitor fish at the whole-of-asset scale (monitoring to commence in June 2005).

In each case, however, whole-of-asset scale monitoring has not yet been subject to the definitional and statistical rigor (*i.e.* power analysis) required of the intervention scale monitoring (as outlined in the seven step process, Section 2). Formal guidelines for the application of this or a similarly rigorous process across all monitoring scales are currently being defined by the Murray-Darling Basin Commission's (MDBC) River Murray Environmental Management Unit, and will be available in the coming months.

Our aim for the Hattah Lakes SEA, is to develop a whole-of-asset scale monitoring program within the next 6 months. In achieving this, the following principles will be adhered to:

- There will be strong liaison with the MDBC's River Murray Environmental Management Unit to ensure adoption of the forthcoming best practice monitoring guidelines for the Living Murray project.
- There will also be strong liaison with the monitoring coordinators at other SEA floodplain sites (including Barmah-Millewa) to ensure consistency of methods and to exploit any economies of scale.
- Monitoring methods will align, where possible, with the soon to be published 'Recommended Methods for Monitoring Floodplains and Wetlands' (MDBC publication no. 72/04).
- The final monitoring program will allow for the statistical detection of change in an ecological response variable (*e.g.* 'tree health') over time (*i.e.* there will be a measure of variability around the central tendency score for each ecological response variable).
- Power analysis will be undertaken in some form (*e.g.* Bayesian, minimum detectable difference, precision) to ensure the detectable change in the response variable is appropriate.
- Appropriate time scales for the repeatable monitoring of each response variable will be outlined and adopted.
- Further monitoring at the intervention scale may be designed and undertaken (as negotiated with the Mallee CMA) as operating rules and procedures for water delivery at Hattah are refined into the future.

Clearly, the current lack of any water in the Hattah lakes will restrict early whole-of-asset monitoring to terrestrial biota and other non- water dependent response variables (*e.g.* vegetation health, groundwater, birds, terrestrial vertebrates). The monitoring design for other wholly water dependent ecological response variables (*e.g.* fish abundance) at the whole-of-asset scale will be undertaken and instigated once wetlands begin to fill (either naturally or artificially).

It is planned that initial monitoring at the whole-of-asset scale within the Hattah Lakes SEA will commence in December 2005.

Chowilla SEA Preliminary targets	
Vegetation	Maintain or improve tree health within 70% of the mixed red gum-black box forest and woodland areas.
	Maintain or improve tree health within 45% of black box woodland areas.
	Maintain or improve tree health within 40% of River Cooba woodland areas.
	Improve the health and conservation value of 40% of the areas* of lignum.
	Improve the area and diversity of grass and herblands.
	Improve the area and diversity of flood dependant understorey vegetation
	Provide conditions suitable for regeneration and seedling survival of all vegetation targets including (but not limited to) River Red Gum, Black Box, River Cooba and Lignum.
	Maintain or improve the area and diversity of grazing sensitive plant species.
	Limit the extent of invasive (increaser) weed species.
Fish Populations	Maintain the diversity and extent of distribution of native fish species throughout Chowilla.
	Reduce the barriers to fish passage throughout the floodplain creek system.
	Maintain successful recruitment of small bodied native fish every year.
	Maintain successful recruitment of large bodied fish at least once every 5 years.
Frog populations	Maintain or improve the distribution, abundance and biodiversity of the (8) riparian frog species.
Waterbird populations	Provide conditions conducive for successful breeding of colonial waterbirds in a minimum of 3 temporary wetland sites at a frequency of not less than 1 in 3 years.
Threatened Species	Maintain the relative abundance and distribution of the Southern Bell Frog, Stone Curlew and Broad Shelled turtle.
	Maintain the current nesting locations of Regent Parrots.
Aquatic Habitats	Increase the frequency and duration of floodplain river channel connectivity.
	Maintain or improve the area and diversity of submerged and emergent aquatic vegetation.
Groundwater	Lower groundwater by 2 meters (on average) across 8000 ha of floodplain.
	Reduce saltloads to the River by 8 EC.
Land Management	Maintain or improve Land Function for respective vegetation types and season.
	Limit mean total kangaroo density to a level of 2 per km^2 on the Game Reserve (floodplain).
	Limit mean feral animal (*e.g.* pig, rabbit, goat, fox and cat) densities to <1per km^2.

Table 7.2: Preliminary ecological objectives ('targets') being monitored at the whole-of-asset scale for the Chowilla SEA.

Gunbower-Koondrook-Perricoota SEA		
	Target	Objective
Water Management	Flow delivery plan	There is a plan.
	Flow delivery plan	The flow plan was delivered.
Water Regimes	Permanent Wetland.	Reinstate area to 50% natural.
	Semi-permanent Wetlands	Reinstate area to 50% natural.
	Temporary Wetlands	Restore the natural pattern of temporary wetlands within the forest.
	Red Gum FDU	Restore 50% of area that has been lost since natural conditions.
	Red Gum FTU	Reduce extent to allow for expansion of other wetland types.
	Black Box	Maintain extent.
	Grey Box	Maintain quality and extent.
	Connectivity	Restore connectivity between river and floodplain and between floodplain components.
	Permanent Wetlands	Reinstate habitat quality so that species typical of permanent wetlands are present.
	Semi-permanent Wetlands	Reinstate habitat quality so that species typical of semi-permanent wetlands are present.
	Red Gum FDU	Maintain habitat quality so that species typical of Red Gum FDU are present.
	Red Gum FTU	Maintain habitat quality so that species typical of Red Gum FTU are present.
	Black Box	Restore habitat quality so that species typical of Black Box wetlands are present.
	Watercourses	Restore habitat quality of Gunbower Creek.
Birds	Colonial Breeding Waterbirds	Restore colonial breeding waterbird populations by ensuring successful breeding events (maybe 4 events in 10 years, successful event is > 50 species, several species in colonies > 50??
	White bellied sea eagle	Restore White bellied sea eagle populations by providing breeding and foraging habitat (maybe just suitable nest trees, maybe presence of sea eagles each flood??)
	Birds needing scrubby habitat to live in possibly all year round.	Restore habitat for snipe.* Restore habitat for crakes.* Restore habitat for superb parrots.*
Fish	Fish (immigrant fish, opportunistic fish)	Restore populations of cod, perch to Target?? By providing opportunities for floodplain access .
	Gunbower Creek (cod, perch etc.)	Restore self-sustaining populations of cod, perch etc. in Gunbower Creek.
	Floodplain wetland fish	Restore self-sustaining populations of Pygmy Perch, Gudgeons, and other small indigenous fish.
Amphibia	Specialised frogs	Restore populations of Growling Grass Frogs and Giant Banjo Frogs.
	Other frogs	Maintain sustainable populations of other indigenous frog species.
Invertebrates	Aquatic invertebrates	Maintain/Restore diverse population of invertebrates.
	Yabbies	Maintain sustainable population of yabbies.
	Invertebrate Successions	Maintain invertebrate succession.

Table 7.3: Ecological targets and objectives developed for whole-of-asset scale monitoring for the Gunbower-Koondrook-Perricoota SEA. *-low priority (source Crome 2004).

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