Australian Government

Government of South Australia

MURRAY FUTURES
Riverine Recovery

Riverine Recovery

Monitoring and Evaluation Program - Technical Design

WATER FOR GOOD

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Contributions to the development of this monitoring program were received from the following:

David Deane, Kerri Muller, Nicholas Souter, Karl Hillyard, Kane Aldridge, Michelle Bald, Steve Barnett, Volmer Berens, Chris Bice, Tumi Bjornsson, Susan Gehrig, Kirsten Martens, Kate Mason, Sally Maxwell, Dale McNeill, Jason Nicol, Darren Oemcke, Richard Thompson, Rebecca Turner, Peter Waanders, Hugh Wilson, Qifeng Ye.

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

Table of Contents

List of Figures

List of Tables

Riverine Recovery Project

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

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

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

- Improve the health of wetlands, floodplains and the river (i.e. establish measurable and scalable environmental, social and economic goals and targets for riverine health and establish an adaptive management system for selected wetlands and floodplains to achieve these goals and targets).
- Save water for the environment and be climate change ready (i.e. establish a water use system that will allow use of available water for best environmental outcomes in wetlands and on floodplains).
- Give security to regional communities (i.e. relocate pumps from wetlands to the river channel to provide better water-quality and secure access for irrigators currently dependent on a stable water level in the selected wetlands) (DWLBC 2008).

Ecological goals of the Riverine Recovery Project

Prior to river regulation a gradient in wetland types defined by the hydrological regime (i.e. a gradient from temporary to permanently inundated wetlands) would have existed on the Lower River Murray. However, the maintenance of stable weir-pool levels due to river regulation has resulted in many wetlands now tending towards either of the two hydrological extremes. Wetlands with sill levels below normal weir-pool level tend to be permanently inundated with little water level fluctuation. Wetlands with sill levels above normal weir-pool level suffer from extended periods of dryness. The exceptions are those wetlands below Lock and Weir 1 that experience regular water level variations of approximately 0.6 m, due to wind driven movement and evaporation in this reach. Since water regime is a principal driver of wetland ecology, this has had major impacts on these wetlands.

The RRP aims to restore the gradient in hydrological regimes that existed prior to river regulation by creating greater water-level variability in individual wetlands and at the reach scale. This will be achieved through the operation of wetland regulatory structures and the manipulation of weir-pool levels. Whilst the manipulation of wetland water levels and weir-pool water levels operate at different spatial scales, the reasons for manipulation are similar. Water levels are varied to create areas that are inundated at varying frequencies and durations and in doing so create a diversity of habitats for wetland and floodplain biota.

These manipulations will be aimed at restoring the condition of the Lower River Murray ecosystems. Figure 2 shows the broad ecological goals of the RRP, which is explored in more detail in *Conceptual understanding of the ecological response to water level manipulation* (DEWNR 2012a).

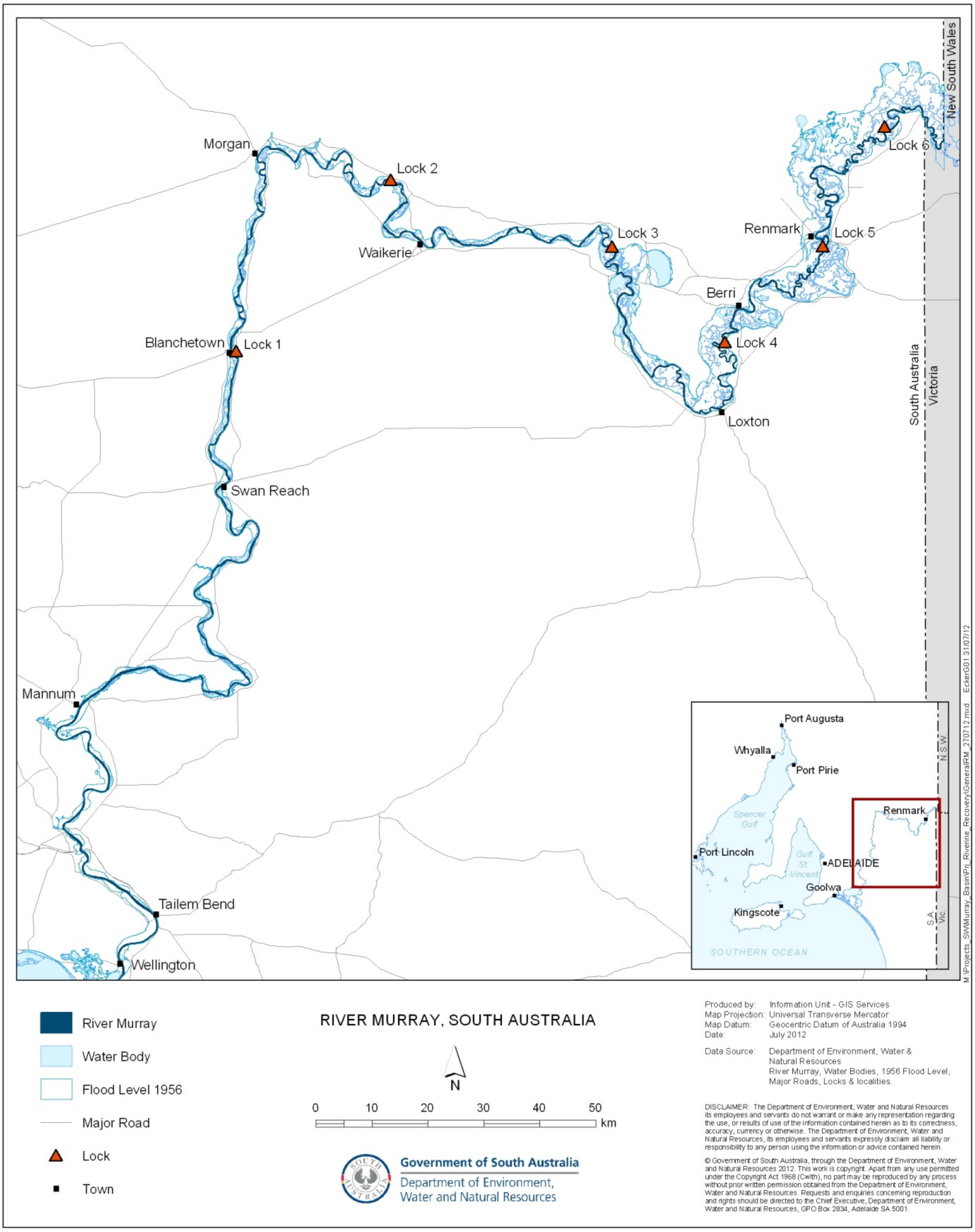

Figure 1: Map of the study region in South Australia showing the River Murray and area inundated by the 1956 flood. Major towns and the locations of the locks on the Lower River Murray area also shown

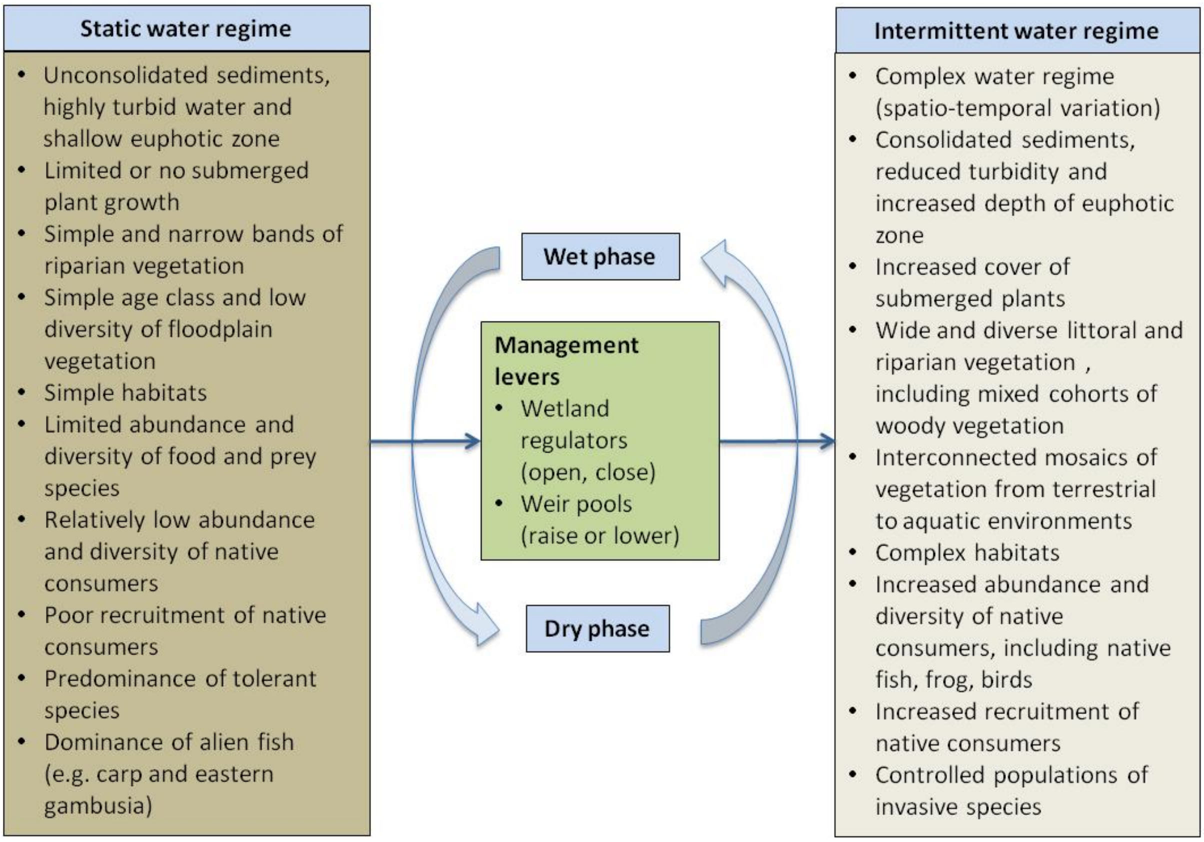

Figure 2: Broad goals of water level manipulation of the Riverine Recovery Project

Monitoring and evaluation program development

This program builds on previous work that has identified the processes influencing the ecological condition of the River Murray system. It also builds on previous work that has established the basis for monitoring and management of the River Murray system (e.g. Wilkinson *et al.* 2007a, 2007b). Wilkinson *et al.* 2007a identified four components that are essential to developing an effective monitoring and evaluation program:

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

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

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

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

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

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

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

Prior monitoring

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

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

Report description

This report and its companion report (DEWNR 2012a) focuses on the underlying conceptual understanding and technical aspects of data collection, analysis and interpretation of the RRP.

DEWNR (2012a) details the theoretical understanding of how it is anticipated that manipulating water levels will result in the desired ecological response. Possible risks that may influence whether or not management goals are achieved are also discussed. Current understanding is synthesised into conceptual models, illustrating the characteristics of wetland states expected under static and variable water regimes. This information allows management objectives to be accompanied with hypotheses. The conceptual models form a platform for planning water level manipulations and attendant monitoring and evaluation.

This report provides technical detail on the program design, the selection of indicators for the monitoring program, monitoring methods for selected indicators and guidance on how to evaluate and interpret data that are collected.

Monitoring and management framework

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

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

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

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

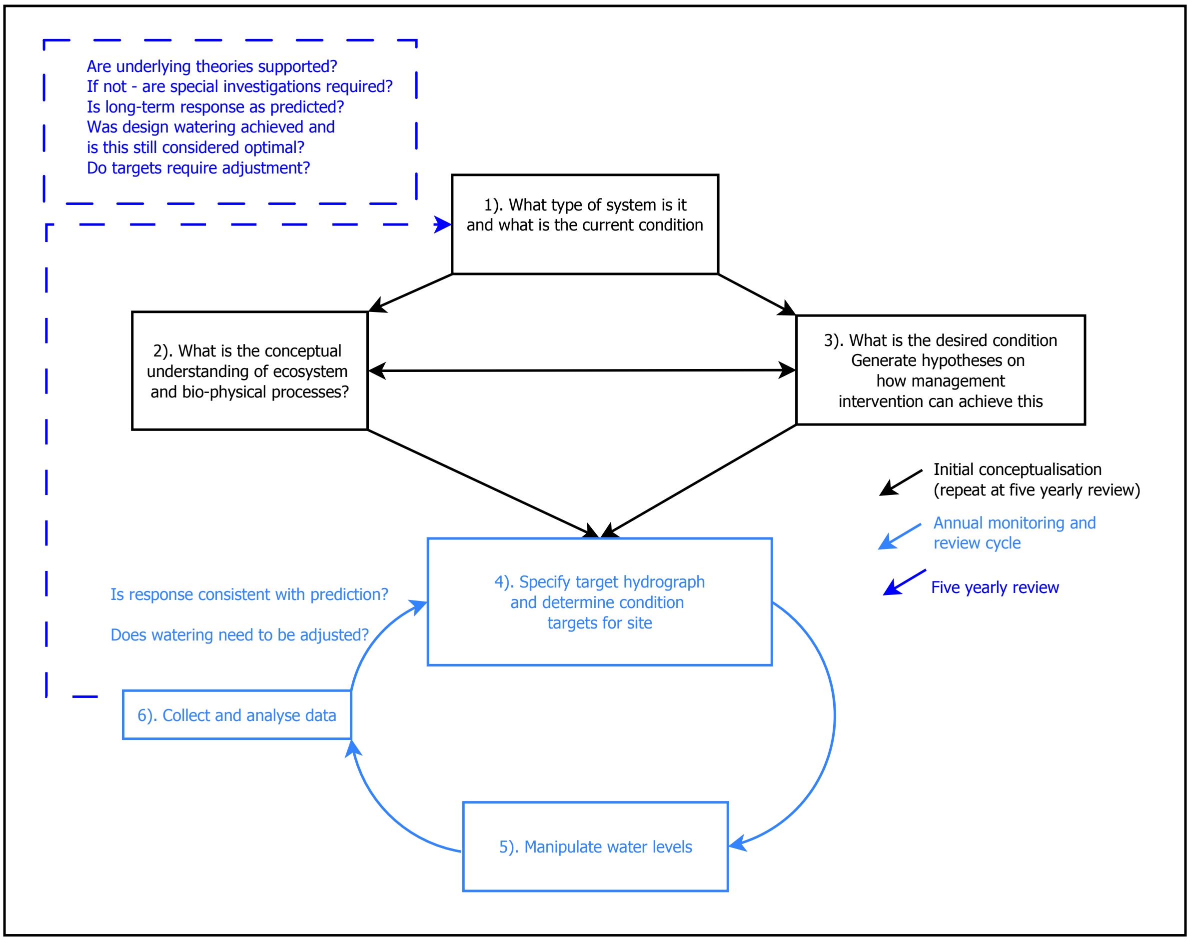

Figure 3: Monitoring and management program structure

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

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

The management plans for individual wetlands are written for a five-year cycle after which they will be reviewed and updated as required. Monitoring and adaptive management reviews will occur more frequently than every five years, ideally before and after every intervention. These reviews will be conducted to assess progress and refine the conceptual understanding, targets and monitoring effort. In addition to annual consideration of site level progress, adaptive management requires periodic review of the entire program at the landscape scale. Meta-analysis of monitoring data collected from all the RRP sites can improve our understanding of riverine processes at the landscape scale (the scale of the RRP program, Figure 1).

Overall, the monitoring program needs to provide answers to the following questions:

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

These questions have been fundamental in determining the technical aspects of program design.

Sampling design

Background

The sampling design of the RRP monitoring and evaluation program is based on the conceptual understanding of how water-level manipulation results in ecological changes (DEWNR 2012a).

Many of the program aims relate to trend detection, necessitating repeated measures and permanent plots (Austin 1981; Bakker *et al.* 1996; McDonald 2003). Managed wetlands, which represent most of the treatment sites, were not selected with randomisation in mind. These practicalities have implications for data analysis (see the *Statistical framework* section). However, where possible, randomisation has been incorporated in the selection of sampling units and in the site-scale configuration of sub-sampling protocols.

McDonald (2003) reviewed sampling design for environmental monitoring programs and described the following components:

- defining the scope, specifying target and sample populations, and sampling units
- membership design – the allocation and arrangement of sampling units
- re-visit design – the frequency of repeat sampling events
- response design – the actual measurements taken at sampling locations
- determine sample size – replication required for desired power to detect change.

This outline has been used to present the RRP monitoring program sampling design.

Geographical boundary and sampling populations

The RRP has a geographical boundary defined by the area of the River Murray 1956 floodplain, constrained by the South Australian border and the township of Wellington (Figure 1). As any location within this area may be of interest to the RRP, the target population for the monitoring program comprises all of the discrete environmental elements such as wetlands, anabranches or floodplains within this boundary.

For design purposes the sampling population is constrained to areas where the RRP management actions will alter the water regime, or similar unmanaged areas from which a comparison can be drawn. Inference for the RRP relates to establishing an improvement at individual sites where management has been able to change the water regime (fixed in space) over the population of wetlands or areas of floodplain not under management intervention. This includes sites with both wetter and drier hydrological regimes.

The study area is differentiated into three basic elements: wetlands and associated anabranches and riparian vegetation (wetland complexes); floodplains; and weir-pools. Wetland and floodplain sampling units are nested within weir-pool scale units. This is an important design consideration when selecting control sites and determining the ecological response at the weir-pool scale.

Wetland complex sampling units

The wetland complex is the main sampling unit for the program and is relevant to both wetland management and weir-pool manipulation. A wetland complex is considered to include the wetland, or anabranch of interest, or both and any riparian vegetation within the zone of influence of these features (see the *Zone of influence* section). Whilst the riparian vegetation in this zone can be thought of as part of the floodplain, it is connected to both temporary and permanent wetlands via the groundwater (Holland *et al.* 2006).

Stratification between wetlands

Many aspects of sampling design rely on stratification. Wetland complexes are the most common sampling unit, but a means for grouping these is required for control site selection. Whilst River Murray wetlands have been classified in a number of ways, the RRP uses South Australian Aquatic Ecosystem types (SAAE) (Fee and Scholz 2010; Jones and Miles 2009). River Murray wetlands can be classified according to five major SAAE types: permanent lakes and swamps, temporary wetlands, saline swamps, channels and floodplains (Souter 2009). These categories provide a means to group wetland complexes together for randomised selection of control sites.

Lakes, swamps and wetlands can be further classified according to the nature of their connection with the main channel. Terminal systems have a single connection between the river and wetland. Flow from the river replaces evaporative losses. Through-flow systems, if permanent, are connected at pool level both upstream and downstream of the water-body and water is lost through evaporation and natural drainage via the downstream connection to the river. The difference in pool-level hydraulic head between inflow and outflow channels dictates the volume of water movement, which is typically low.

Lakes and swamps were differentiated largely on depth. Due to their greater depth, lakes were unlikely to support submerged aquatic vegetation, whilst swamps are. Temporary wetlands have a wetting and drying cycle and receive significant volumes of water from non-channel sources (e.g. surface runoff; groundwater inflow). Saline swamps are saline with salinity exceeding 10,000 mg/L (~16,310 µS/cm).

River channels include permanent, seasonal and ephemeral watercourses (e.g. anabranches, flood runners). These are differentiated in the same manner as wetlands with permanent reaches being connected at pool level and the others flowing only seasonally or occasionally. Many managed wetland regulators are located on permanent reaches connecting the wetland with the river channel. This forms a distinct habitat type within a managed wetland complex.

The floodplain in South Australia is defined by the 1956 flood level. The floodplain is the flat or nearly flat land adjacent to the river. These areas are differentiated from swamps in the period of time that water remains pooled on their surface once a flood subsides.

Floodplain sampling units

Areas of non-wetland dependent floodplain are able to be affected by RRP management actions through inundation via weir-pool raising. The sampling population for floodplain elements is constrained to areas likely to be inundated by a weir-pool raising, but not located within the zone of influence around the four wetland inundation classes.

Weir-pool sampling units

Weir-pools are defined by infrastructure and are not subject to sampling design in the sense that wetland complexes and floodplain sampling units are.

Weir-pools require monitoring capable of informing decision-making and will form discrete units for reporting both condition and trend. Such requirements will rely on a combination of scaling-up wetland complex sampling-unit responses and the use of remote sensing to interpolate condition between floodplain areas and wetlands. For the cross-scale interpretation to be effective, the selection and location of control sites needs careful consideration.

Zone of influence

The ecological response to the creation of a more variable water regime will not be limited to the area influenced by surface water inundation. Instead, it may extend to areas that are influenced by bank recharge and groundwater movement that result from a more variable water regime. Bank recharge has been shown to be important for riparian vegetation growing within 50 m of temporary and permanent wetlands (Holland *et al.* 2006). Therefore, sampling design needs to account for the anticipated zone of influence around all wetlands, to assess all impacts that may occur as a result of water level manipulation. Although impacts may potentially extend further over time, any change in condition should be evident first within this zone and would indicate a possible need for further investigation at the site. Monitoring needs to consider this zone in both wetland drawdown and weir-pool raising. The zone of influence can be determined by baseline studies including bathymetry for managed wetlands, and through the use of modelling for weir-pool manipulations.

Membership design – allocation of sampling units

Membership design specifies the spatial allocation procedure of selecting sampling units (McDonald 2003). The overall design for the RRP is quite complex and the relationship between treatment and control sampling units will need to be clearly understood. Monitoring effort for the RRP is focused on managed sites (treatment sites) and sites that are physically comparable (not including hydrological characteristics), but are not affected by RRP water-level manipulation (controls).

Regulation has largely converted the wetlands into two types; those that are connected at normal pool level (essentially permanently inundated) and those that are filled infrequently by overbank flows. Drawing an inference from wetland-complex sampling units to the broader population requires that both permanent and isolated temporary systems be used as controls. Probabilistic sampling[1] is required to draw statistically defensible inference against the broader population (i.e. generalise the findings to all possible sties). However, it is anticipated that relocating sites each sampling round will not be realistic given the level of effort required. It also runs contrary to the need for permanent plots to determine trends (Austin 1981; Bakker *et al*. 1996; McDonald 2003). A compromise has been adopted for

[1] where each member of the sampling population has an equal probability of being selected for sampling

membership design where control and treatment sites are chosen via a stratified random approach.

For design purposes, wetlands are stratified into four categories of inundation frequency, reflecting different positions on the gradient (Table 1). Assigning wetlands to particular categories requires careful planning with consideration to the descriptions given in Table 1. Each category functions in one or more roles in sampling design. Note that managed wetlands and enhanced temporary wetlands are both achieved through RRP management actions.

Table 1: Wetland complex inundation classes and their role in sampling design

Category	Description	Role in sampling design
Permanent wetlands	Wetlands where sill level is below normal weir-pool level Permanently inundated, with water level variability insufficient for significant drying and rewetting of the wetland bed	Sampling population for controls on managed and enhanced temporary wetlands
Managed wetlands	Wetlands whose connection to the river is manually controlled by a regulating structure, allowing partial or complete drying	Treatment sites for RRP
Enhanced temporary wetlands	Temporary wetlands with a sill level above normal pool operating level but that can be filled through weir-pool raising	Treatment sites for RRP weir-pool manipulation Control site for RRP managed wetlands
Isolated temporary wetlands	Temporary wetlands with a sill level above pool level and are unable to be filled through weir-pool manipulation	Control sites for both weir-pool manipulation and managed wetlands

Monitoring effort will initially be intensive, with all control and treatment sites monitored every sampling round. If wetlands adapt to the altered water regimes this level of effort may be reduced, with data analysis used to inform appropriate sampling effort. At this point it will be possible to randomly assign controls and treatments within the initial total sampling population (McDonald 2003). These can then be visited on, for example, a five-year rotating schedule as required for condition reporting and management.

Control site selection

Wetland complex sampling units

Ideally reference sites (Downes *et al*. 2002) with natural water regimes would be identified to compare to those being implemented through management. Unfortunately this is not possible as these systems are very rare in the Lower River Murray.

Instead, to provide points of comparison for the full range of management objectives, it is necessary to have control sites that are permanently inundated and rarely inundated (and shall remain so). Managed sites will occupy the intermediate areas of the two extremes. This will allow the impacts of water-level manipulations at treatment sites to be drawn. As each wetland is likely to have a different water regime, the gradient in water regimes across the landscape may be used to assess the response to altered water regimes. Furthermore, while weir-pool raising operates at a larger scale, monitoring of this will include temporary wetlands as treatments. Assuming weir-pool raising becomes a regular part of river operations, the environmental values that develop at weir-pool treatment sites should be comparable to those at managed wetlands, but without the presence of a regulator. Comparisons between managed and temporary wetlands subject to weir-pool level driven inundation will allow for the effects of management to be assessed. Over time, such comparisons should provide valuable understanding to inform continual improvement in both wetland management and weir-pool manipulation.

A stratified random sampling approach is used for initially selecting control sites. The aim is to ensure data are representative of non-managed assets, but to also allow inference about the relationships between water regime and changes between control and managed sites to be drawn. After pooling candidate-wetlands using selection criteria, randomisation should be used to select the final sites.

Stratification of control sites needs to consider wetlands in the same weir-pool and geomorphic reach and wetlands that have:

- the same class of South Australian Aquatic Ecosystem (SAAE) wetland type identified as being functionally equivalent in Souter (2009)[2]
- similar level of connection to the main channel (while regulating structure is open), including distance from the main channel, degree of regulator openness
- similar water-quality when connected (in particular salinity)
- comparable area (ideally within ±20–30%)
- similar exposure to natural flow variations (flooding, flow-driven fluctuations)
- similar solar and topographic exposure level (e.g. side of river, proximity to cliffs)
- similar bathymetry (if known – especially broad shape and total depth)
- close proximity, or at least similar distance, from weirs
- similar complexity of riparian habitat and cover
- similar anthropogenic impacts (not including water regime)
- ease of access and safety considerations

Temporary wetlands used as controls for both RRP approaches need to be beyond the reach of elevated river stage resulting from weir-pool manipulation. i.e. the isolated inundation class.

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[2] Jones and Miles (2009) can be used as an indication of SAAE wetland type, but this should be verified through field visits before site is selected

Floodplain sampling units

Floodplain sampling units will be primarily used to ground-truth remote sensing data collected to assess weir-pool raising. This will require the establishment of a condition gradient. Riparian vegetation data collected from wetland complexes will provide some points on the gradient. Additional sites are required to determine the change in condition resulting from weir-pool inundation. Vegetation mapping and scenario modelling of weir-pool inundation extent can be used to stratify potential sites. Final sites can be selected via randomisation and their suitability, verified by field visits.

Floodplain treatment sites used to assess weir-pool raising should be selected prior to them being matched against control sites. Criteria for stratifying floodplain-vegetation sampling units for selection as controls are as follows:

- not inundated by the highest potential weir-pool raising
- located within a short lateral distance from the river
- located outside the zone of influence of wetland hydrology to avoid confounding responses between bank flux and inundation
- similar patch size to treatment site, or at least large enough to provide adequate interior area to contain a sample plot – avoid the edge of patches by at least one quadrat width if possible
- similar vegetation association and same dominant vegetation to controls (ideally a similar tree-stem density as well)
- similar local topographic variation – small depressions will pond water longer than flatter or raised areas.

Weir-pool sampling units

At the scale of a whole weir-pool, it is not possible to set up a design with appropriate treatment and control sites due to operational constraints. Inter-pool comparison would be possible, with weir-pools with 'managed' weir levels acting as treatments and weir-pools with normal weir-pool levels acting as controls. However, this would be confounded by factors such as longitudinal variations in geomorphology, tail-water effects etc. This is true along any river (or linear environmental feature) where effects accumulate downstream. Time-series analysis of floodplain and temporary wetland vegetation responses provides the best opportunity for the spatial extent and magnitude of response to weir-pool manipulation to be identified.

Treatment site selection

Wetland complex sampling units

Managed wetlands act as treatment wetlands. These have been pre-selected and were not selected according to monitoring-program design criteria. As weir-pool raising is able to inundate wetlands above pool level (Cooling *et al.* 2010), the selection of temporary wetland complexes as treatment sites is required. Monitored using the same techniques as managed wetlands and controls, these will also provide an additional point of comparison to assess the effect of changed water regimes on managed wetlands.

Selection of wetland-complex treatment sites for weir-pools requires a stratified random selection process. Prior to raising a weir-pool, the anticipated area to be inundated by the proposed level of raising will be established using existing hydraulic models. This will provide an indication of which temporary wetlands will be inundated by the weir-pools raise. Other than being filled by the lowest likely weir-pool raising level, stratification of sites should be based on similar criteria to those in the previous section.

Based on a pilot study undertaken during a trial weir-pool raising in 2005, Souter and Walter (unpublished) provide some considerations for site selection where demonstrating a short-term response is important:

- The site should not be connected to the river at pool level, but likely to be periodically inundated by either rainfall or groundwater expression
- The connecting sill level should be within the range of weir-pool manipulation above normal pool level
- Basins or depressions capable of holding water are more likely to demonstrate a response than floodplain areas which shed water
- No evidence of salt intrusion should be present in order to manage associated risks
- Flood-dependent or non-salt-affected vegetation in good condition is desirable
- Sites should be within close proximity of weir-pool extent at normal operating level
- Accessible by vehicle or boat.

Such considerations may create bias in the sampling design, which needs to be accounted for in interpreting outcomes.

Floodplain sampling units

Floodplain sampling units will be primarily used to ground-truth remote sensing data. The criteria for stratifying sites for selection are the same as for controls, with the difference that a suitable number of sites should be inundated under the lowest likely weir-pool raising level. Depending upon the weir-pool operating regime, it may be possible to further stratify treatment sites. If for example the weir-pools are to be routinely raised to two different levels, then sites that are inundated by the lowest rise and those inundated by the highest should be chosen to determine the effect of the difference in water regime between the two levels of raising.

Response design – indicator selection

The selection of monitoring indicators has been linked to the conceptual models upon which management is based (*cf*. Saintilan and Imgraben 2012). Only those dependent variables (potential indicators) that are actually predicted to respond are included. Indicators were selected to provide evidence on progress towards management targets based on the conceptual understanding of the response to management actions (see DEWNR 2012a). Ideal ecological indicators also have the characteristics of being easily measured; follow a predictable response that provides an indication of how management should proceed for a given outcome; and feature low variability in the magnitude of response (Dale and Beyeler 2001). It is reasonable to assume that no one indicator possesses all of these qualities and a suite of indicators are necessary to provide an integrated picture of ecosystem condition and trajectory.

Indicators selected had to be able to inform management and allow comparisons to previous baseline surveys (*Monitoring and evaluation program development* section). The collection of data from a wide range of indicators provides additional information on ecosystem structure and function. In particular, it is acknowledged that indicators should include a combination of attributes and processes (Wallace 2012; Wallace *et al*. 2011). However, there is a trade-off between the costs of data acquisition and the additional information sampling a wide range of parameters will provide to managers. As well as the above criteria, the selection of indicators had to consider those that provided most value to answer the management questions. Given the scale of the RRP, some indicators were omitted as the value of the information obtained for making management decisions was low, when compared to the costs associated with collection.

To help address the needs of the RRP to both monitor and report on progress towards environmental targets and provide 'real time' information for management decisions, two classes of indicator have been included:

- Progress measures – these are generally continuous variables or indices that are used to track progress towards planned environmental goals. Trends in indicator status at sites can be established based on these data and they also form a basis for comparison between treatments and controls.

- Condition and Trend (CaT) – based on the current condition and future trend approach (Appendix 1 – Souter and Watts (unpublished)) the aim of CaT indicators is to assess current resource condition and inform short-term management decisions around watering. These can be viewed as early warning indicators of the need to change planned watering decisions and are critical to site management.

Wetting and drying large areas of wetlands and floodplain has water-quality risks that may limit the ecological response to management actions and these risks are considered in the monitoring program. However, there are also water-quality risks to the River Murray. The monitoring to address risks to the River Murray is not provided in detail here as the objective was to design a monitoring program that is able to assess the ecological response to management actions. However, some additional information on water-quality risks and monitoring are given in the *Response design – indicator selection* section.

Wetlands

Due to resources constraints described above, the program distinguishes between core indicators (Table 2) and targeted investigation indicators (Table 3). The recommended approach is to adopt core indicators to monitor progress towards targets as hypothesised. However, if management plans for individual wetlands do not specify actions associated core indicators then these indicators could be omitted from the monitoring program for the wetland. Alternatively, if management plans for individual wetlands specify actions-associated with other indicators then these indicators should also be monitored. In particular, if management plans specify actions associated with groundwater, then a groundwater-level monitoring network will be necessary. Expert hydrogeological advice should be sought on the most appropriate observation-well configuration to meet management needs in this case. Furthermore, whilst not considered a core indicator based on the criteria above, where there are existing groundwater piezometers, groundwater levels and salinity should be measured following methods of Tucker *et al*. (2004). This information should be used to inform the response of groundwater to management actions. Furthermore, these data could be used to calibrate electromagnetic measures of regional groundwater salinities and inform management at the landscape scale.

If hypothesised states are not achieved, targeted investigations should be undertaken to determine why. These targeted investigations include the monitoring of processes that have not been included as core indicators due to resource constraints.

Table 2: Core indicators and information generated by the indicator

Core Indicators	Information/understanding provided	Requirement for management
Hydrology	Changes in water regime associated with management actions	Provide evidence of management actions leading to desired water regime Link water regime to observed ecological responses
Plant functional group and vegetation cover	Changes in the extent, diversity and type of vegetation as a result of changed water regime	Identify optimal drawdown periods to facilitate recruitment Provide evidence of having achieved management objectives
Submerged vegetation cover	Changes in the extent and diversity of submerged plants in response to improved water clarity	Identify optimal drawdown periods to facilitate recruitment Provide evidence of having achieved management objectives
Tree and amphibious plant condition	Crown condition and future trend as a result of changed water regime Current amphibious plant condition and future trend	Assess immediate site watering requirements Determine the benefits of management actions and ensure no impacts
Frog populations	Changes in wetland use as a result of changed water regimes Changes in the presence of listed and/or endangered species	Decisions on timing and duration of wetland isolation to avoid impacts on frog breeding or dispersal processes Determine the benefits of management actions and ensure no impacts
Water-bird populations	Changes in wetland use for feeding or breeding across control and managed wetlands Changes in the presence of listed and/or endangered species	Decisions on timing and duration of wetland isolation to avoid impacts on bird breeding or dispersal processes Decisions on optimal management at landscape scale Ensuring drying a wetland will not adversely impact a listed species
Turbidity	How wetland turbidity changes over drawdown Spatial variation in turbidity	Determine whether drawdown has led to reduced turbidity Determine optimal and achievable target turbidity levels for future management planning
Salinity	How wetland salinity changes over drawdown through evapo-transpiration	Assess immediate site watering requirements

Core Indicators	Information/understanding provided	Requirement for management
	Spatial variation in salinity Groundwater discharge	Determine whether groundwater discharge is a major component of wetland water balance Ensure physiological thresholds are not exceeded (especially for fish)
pH	Determine whether acid sulfate soils pose a risk, on or off site	Optimising wetland drawdown cycle durations and timing to avoid acidifying sites
Dissolved oxygen	Determine whether deoxygenation is a risk, on or off site	Optimising wetland drawdown cycle durations and timing to avoid deoxygenation

Table 3: Possible targeted investigations indicators and applications for the RRP (note: other indicators may arise depending on the observed response to treatment)

Possible targeted investigation indicators	Possible investigation aims
Groundwater	Determine whether altered groundwater levels or salinity are preventing the recovery of vegetation (and so reducing habitat for higher trophic organisms)
Soil condition (soil moisture, salinity, sodicity, organic matter)	Determine whether unfavourable soil conditions are preventing recovery of the tree and amphibious plant communities (and so reducing habitat for higher trophic organisms)
Nutrients and chlorophyll (suspended or attached)	Determine whether increased nutrient concentrations and high phytoplankton and epiphytic biofilm biomass are preventing recovery of the submerged plant community (and so reducing habitat and food for higher trophic biota)
Zooplankton and macroinvertebrates	Determine whether low numbers of invertebrates or diversity are limiting the response of consumers (fish, water-birds, frogs)
Recruitment success of various biotic groups	Determine whether recruitment is limiting the observed response to altered water regimes

Core indicators

Based on the criteria above, core indicators selected to assess progress towards management targets were (Table 2):

- Hydrology – Even where a well-established eco-hydrological relationship informs a management plan hydrograph, it is necessary to determine how closely the manipulation matches the theoretical design. Ensuring that the proposed management action was successful is a form of implementation monitoring (Wilkinson *et al.* 2007a, 2007b). Without such data the success of an intervention cannot be evaluated. The duration, rates of change and spatial extent of inundation are effectively part of all management hypotheses under test. In addition, and depending on hypotheses or specific management aims, changes in velocity as water levels are manipulated may also be obtained.

- Water-quality – Water-quality is a major driver of the floristic and faunal composition of a wetland. For this program, turbidity and salinity are identified as the main water-quality factors that indicate whether there is a transition towards the desired wetland state (DEWNR 2012a). Due to the low additional cost of measurement, and the risk to the desired ecological response, measurements of pH and dissolved oxygen are also included (see DEWNR 2012a). Water-quality data are also needed to support a number of management targets.

- Vegetation communities – Considerable effort is directed towards monitoring vegetation. This is justified as vegetation targets are found in all wetland management plans, the response of vegetation to changes in water regime is predictable and plants have relatively non-motile life history strategies. Furthermore, it is anticipated that vegetation will respond directly to water level manipulations and the vegetation response is critical to the response of all consumer indicators (fish, water-birds and frogs; DEWNR 2012a). Data needs to be collected at sites that enable the complete picture of change at the site to be determined.

- Fish communities – Optimising native fish diversity is a common objective of wetland management plans. To inform targets and management decisions, fish community monitoring is required. In addition, where improved fish habitat is a management objective, habitat availability (obtained from vegetation data) also needs to be monitored. Another common management objective is the exclusion or removal of common carp. Fish are highly mobile opportunistic wetland users and there is a risk that isolation from the main channel may adversely impact on community composition, recruitment or condition in managed wetlands. Conversely it is also likely that cyclic water regimes will increase wetland productivity and habitat availability (submerged vegetation), thereby benefiting native fish populations. Over time, this dataset should provide evidence of the most successful management actions to improve native fish populations, potentially even allowing managed wetlands to be manipulated as source populations once re-connected.

- Water-bird communities – Water regime manipulation also often aims to create optimal conditions for water-bird use. Local variations in usage should reflect the success or otherwise of management actions. Where improved water-bird habitat is a management objective, the achievement of both improved habitat (obtained from vegetation data) and a community response needs to be monitored. This allows for a distinction between the successful creation of suitable habitat (an adjustable management objective) and the actual use of that habitat (outside RRP management influence) to be made. Furthermore, a wide variety of birds are connected to riverine habitats and food webs and trends in their populations can be used to evaluate the success of remediation in riverine environments (Vaughan *et al*. 2007). As with all biological indicators, the scale of response must be considered when determining what data reveals about site condition. For mobile taxa such as birds, processes that operate and large spatial scales present a challenging source of variability. While trends in water-bird population dynamics will be affected by processes operating at large scales, the effect of such processes are expected to be consistent between managed and control wetlands.

- <u>Frog communities</u> – As for fish and bird communities, optimising native frog diversity or habitat is a common objective of wetland management plans and so frog community monitoring is required. In addition, where improved frog habitat is a management objective, habitat availability (obtained from vegetation data) also needs to be monitored. However, current approaches to monitoring frog communities in the South Australian Murray-Darling Basin are difficult to assess quantitatively and it is recommended that a quantitative approach to monitoring and evaluation is developed. A standardised approach that has been used previously is suggested in the interim.

Possible targeted investigation indicators

Based on the selection criteria above, targeted investigation indicators are those that may limit the progression towards a desired state. It will be possible to determine which factors were limiting the observed response using data collected after intervention. This would require a comparison to appropriate controls and treatments where the hypothesised response has been observed. Based on the conceptual understanding developed by DEWNR (2012a) these may include (but are not limited to) (Table 3):

- <u>Groundwater</u> – The interaction between groundwater, surface water and floodplain vegetation will likely change following the introduction of a variable water level regime. Changing a wetland from a permanent to a variable water level regime may see increased saline groundwater flow into the wetland. This is because the former static elevated hydraulic head prevented such intrusions. However, for wetlands that will be considered for the RRP, the risk is thought to be generally low (DEWNR 2012a). Given the severity of potential consequences of saline groundwater intrusion, monitoring may be required. When salinity risk factors (e.g. deep wetlands; sandy soil profiles; shallow, saline groundwater) are present at a site then a full risk assessment should be conducted to evaluate whether saline discharge to the wetland is likely. If the risk assessment cannot rule out interactions, a network of groundwater piezometers may need to be constructed. Data from this network can then be used to manage drawdown levels so that they do not create a reverse head that would cause groundwater to enter the wetland. In the event that salinity provides too great a risk, then a variable water level regime will not be introduced.

- <u>Soil condition</u> – Soil quality has been identified as a factor that is most likely to limit the desired response to the altered water regimes, as it may limit the vegetation response upon which other ecological components are dependent (DEWNR 2012a). Decreased soil moisture and increased soil salinity and sodicity may need consideration. The impacts of altered water regimes on these parameters are not currently well understood, but targeted investigations could be conducted to improve this understanding and thus wetland management.

- Nutrients and chlorophyll – Re-introducing drying and wetting cycles to wetlands may result in the flux of large amounts of nutrients from the sediments to the water column. This may promote the growth of phytoplankton, thereby reducing light availability for submerged plants and limiting the ecological response to management actions. In this case, understanding the nutrient and chlorophyll (phytoplankton biomass) levels would provide insights. Insights may be provided by dissolved oxygen data, with large diurnal fluxes in dissolved oxygen suggesting an active and abundant phytoplankton community (photosynthesis during the day and respiration during the night).

- Invertebrates – Invertebrates are an important food source for higher trophic levels (e.g. fish, birds and frogs). If the anticipated response of higher trophic levels to management actions is not achieved, this may be due to either inadequate habitat or food availability. If habitat availability is considered adequate then an understanding of food availability would be warranted.

Weir-pool manipulation

A number of questions arise when assessing the effects of weir-pool manipulation that need consideration for evaluating the response. Here we consider weir-pool raising:

- Did the raising inundate the area that was expected to be inundated?

- Over how much of this area was there an ecological response?

- What was the nature of the ecological response, in both the short and long-term?

- For how long has this response lasted?

- When do we need to repeat the weir-pool management intervention (raising or lowering)?

Previous monitoring of weir manipulation has assessed the impact on replicated, but spatially small, sections of the floodplain (Siebentritt *et al.* 2004; Souter and Walter unpublished). Whilst these studies deliver useful information, they cannot provide all the information required for management at the reach scale. Weir-pool manipulation will affect the riverine environment at a range of spatial scales from the reach to habitat. Indeed, the main advantage of raising and lowering weir-pools is that large sections of the floodplain, wetlands and the river channel can be influenced with a single action. Thus, monitoring the effects of weir-pool manipulation at the reach scale is required, particularly for weir raising. Indicators that are able to be used to assess the response at large scales are necessary.

Remote sensing methods are most suited to determining the extent of the area inundated and ecological response via a weir-pool manipulation at the scale of the reach. Remote sensing of the response of floodplain vegetation is an appropriate measure of the ecological response. Remote sensing analysis will provide an indication of when the response of a previous weir manipulation is declining. This may provide a trigger for a future weir manipulation. Indicators of future trend measured via tree and sedge condition assessments and the rate of change in remote sensing indicators compared to non-inundated and pre-inundation measures will provide this information.

Indicators to assess risk to river water-quality

Weir-pool manipulation and drying and rewetting of wetlands may cause water-quality issues within wetlands, but also within the River Murray when wetland water is returned to the river (DEWNR 2012a). A range of water-quality parameters are routinely monitored along the Lower River Murray by the South Australian Environmental Protection Authority, SA Water and the Murray-Darling Basin Authority. Water-quality parameters are collected for a range of reasons including meeting the requirements of the *Water Act 2007* (Clause 45) (Australian Government 2007) and determining what level of treatment is required for water extracted for potable use. A range of parameters are analysed at varying frequencies and different locations. For example, the most comprehensive data are collected from Morgan with weekly samples taken of pH, turbidity, electrical conductivity, temperature, colour, oxidised nitrogen, total Kjeldahl nitrogen, total phosphorus, filterable reactive phosphorus, silica and soluble organic carbon (see Aldridge *et al*. 2012a) and quarterly collection of bicarbonate, chloride, sulfate, potassium, sodium, calcium and magnesium.

Whilst the impact of drying and rewetting individual wetlands may be a minor risk to water-quality of the River Murray, there may be a combined effect of multiple wetlands that needs consideration. It appears that there are adequate data available to assess the impacts of drying and rewetting wetlands on the water-quality of the River Murray, although appropriate data-sharing arrangements may be required.

Due to the large area of floodplain likely to be inundated due to weir-pool manipulations, more comprehensive data collection may be required. Parameters identified by SA Water as being of concern during the March–April 2005 Lock 6 raising were salinity, turbidity, cyanobacteria, 2-methylisoborneol and geosmin. It is recommended that these parameters are also collected in addition to the parameters described above. This may require current sampling to be supplemented and appropriate data-sharing arrangements may be required. Samples should be collected frequently before, during and after any weir manipulation at pre-existing monitoring stations. For each weir-pool that is being manipulated, at least two sites are required; one upstream and one downstream of the reach. Further refinements to the water-quality monitoring program are expected to be made as a result of the risk assessment of weir-pool manipulations (to be undertaken from July–December 2012).

Sampling rationalisation

The suite of indicators monitored at wetlands and/or frequency of monitoring will be rationalised as the monitoring program is implemented, with ongoing testing of the protocols and evaluation of the information generated. Initially, it is recommended that a full suite of indicators are assessed. Enough data for all indicators will need to be collected to calculate critical effect sizes. Once sufficient data have been collected to generate critical effect sizes, the monitoring effort can be reduced. As more sites are added, the level of monitoring effort per wetland should decrease as less informative techniques are excluded. Another important consideration when rationalising sampling is deciding how best to allocate sampling effort among spatial and temporal replicates (Rhodes and Jonzén 2011). This can be assessed by measuring the levels of spatial and temporal correlation in monitoring populations (Rhodes and Jonzén 2011) and is recommended as sufficient data are collected to make these assessments.

Revisit design – sampling frequency

Most indicators exhibit different dynamics over drawdown, flooding, seasonal or longer cycles and so their interpretation must relate to specific predictions for each circumstance of imposed conditions and timeframes. Optimising the timing and frequency of data collection to observe these varying responses is therefore desirable. It will be neither possible nor desirable to sample all indicators at the same time or place. Different sites may be used to test or control for varying responses at different times, leading to a relatively complex but necessary re-visit strategy.

Frequent sampling visits are required to coincide with either re-filling or maximum drawdown with consideration of predicted lag effects. Whilst this adds complexity to implementing the sampling program, in practice wetlands will have slight differences in the timing and duration of events. This will help to ease, but not remove, the organisational complexity associated with intensive sampling periods. To manage the complexity of data collection, careful work scheduling will be necessary, in particular as the network of managed and control sites grows over the life of the program. For the sampling frequency of individual indicators see the *Wetland monitoring* section.

Replication, sample size and power

Within versus between replication

Two levels of replication need to be considered during sampling design:

- The number of sampling units (wetland complexes, floodplains and weir-pools)
- The number of sub-samples collected at a site to estimate sampling unit population parameters.

To provide an estimate of the number of sampling units required to demonstrate a difference between treatments and controls, data collected from baseline surveys were analysed. However, these baseline surveys were mostly conducted during the recent extreme drought period and thus do not represent the ecological condition under the full range of water level and flow variations that have been experienced at these sites both before and since river regulation. Consequently, the variability in the ecological components measured do not represent the full variability likely to be observed at these sites under improved hydrological conditions. However, the baseline survey data are the only data available for this analysis. It is recommended that calculations of critical-effect sizes are repeated after more data becomes available and the program design is updated accordingly.

In the analysis, measures for several indicators were pooled to generate a single wetland value (fish, water-birds, and frogs). In this case each sampling unit acts as a single sample and the baseline analysis dictates the number of sites required.

Data were not available to determine the level of sub-sampling required to generate estimates of precision for parameters collected at a site to estimate sampling unit population parameters (e.g. plant functional group cover). Estimated sub-sampling requirements have been made in the relevant sections, but the initial phase of the program must be viewed as a pilot study when measuring vegetation zonation and water-quality. Over-sampling (collecting additional replicates for vegetation and water-quality) is advocated as a necessary component of the initial stages of program implementation.

Change detection between sampling units

The ability to detect change has historically been the realm of power analysis within the frequentist statistical paradigm. Power analysis is an important but often overlooked component of program design (Di Stefano 2003; Fairweather 1991; Osenberg *et al.* 1994; Peterman 1990; Saintilan and Imgraben 2012; Underwood and Chapman 2003). As an alternative to power analysis, planning using precision can provide a means to determine the necessary replication by explicitly designing for a specified margin of error (Cumming 2012).

To determine the number of controls necessary to detect critical effect sizes both power and precision were calculated where appropriate baseline information was available. Precision was calculated based on a margin of error less than half of the critical effect size estimated from observed repeated measures data.

Classical tests of statistical power require a number of assumptions or estimates as follows:

- Specification of α and β error rates
- An indication of the variability in the value of the true population parameter to be estimated. The usual approach is to use (via modelling) the standard deviation of pilot study data
- The effect size to be determined – the ecologically significant difference that is being detected

Error rates

Environmental impact assessments require an explicit consideration of the real world consequences of both α and β error rates and that these should reflect the relative costs of making a Type I or Type II error (Di Stefano 2003; Fairweather 1991). Whilst Mapstone (1995) advocates a flexible approach to α and β error rates, the consequences of either error type for the RRP are considered to be equivalent and so α and β error rates were set to 0.05. These values were adopted for power calculations. This decision is based on the following logic.

For a null hypothesis that water level manipulation would result in no environmental change over controls then:

- A Type I error (the rejection of the null hypothesis when it is in fact true – a false positive) would mean that the planned watering had not generated the desired outcome, but the analysis indicated that it had. Potential consequences for this scenario might include:
  - continued delivery of a sub-optimal water regime, resulting in long-term management objectives not being achieved with a sub-optimal use of water
  - the failure of management trigger value monitoring to initiate a management response (e.g. refilling a wetland) may result in ecological thresholds being exceeded (e.g. salinity levels may exceed lethal dose for a species of conservation importance)

- A Type II error (failure to correctly reject the null hypothesis – a false negative) would mean that an environmental impact (benefit) had in fact occurred, but had not been detected. The impacts of such an analysis for future water management might then include:
    - reduced confidence that the planned watering regime was effective in meeting the desired objective
    - unnecessarily changing the watering regime to create the benefit that had already been achieved, by either withholding or providing extra water

Baseline data analysis

Sampling effort should ideally be based upon pilot surveys from which statistical power to detect change for relevant effect sizes can be determined (Fairweather 1991; Munkittrick *et al.* 2009; Saintilan and Imgraben 2012). For the RRP, considerable data are available from baseline surveys, albeit collected during a drought period. This includes data for a range of variables collected from 2003-2005 from over 30 wetlands in the RRP study area. While many of the wetlands were visited more than once during this period, two distinct surveys were undertaken. Owing to some differences in the methodology, the ability to pool data for analysis was reduced, but estimates of critical effect sizes were in most cases possible (Appendix 2 – Critical effect size determination).

Critical effect size

To determine power it is necessary to understand the size of the change that is expected, or alternatively for environmental monitoring an effect that has ecological significance. This is referred to as the critical effect size (CES) and takes a value for each indicator measure. Critical effect sizes require specification of both form and magnitude (Munkittrick *et al.* 2009). Form is the actual state variable that is under investigation (e.g. individual- and/or community-level descriptors), while magnitude is the amount of change for the state variables under scrutiny (Mapstone 1995).

Munkittrick *et al.* (2009) recommend several approaches for establishing a CES from prior studies. Here developing CES was based on the assumption that through varying the water regime it should be possible to achieve greater variability than is observed under static water regime conditions. Therefore, the level of variability is of interest, in addition to the mean values.

Where it has not been possible to determine CES from existing data, the use of two standard deviations from controls, a value that has been adopted for a wide variety of monitoring programs (Munkittrick *et al.* 2009), is recommended. Whilst this value will not be known until data start to accumulate, it could be done with theoretical values obtained from the literature.

Replication for control wetland complexes

Power and precision analyses (Cumming 2012) were both used to gain an indicative range for the necessary replication for the RRP to demonstrate an environmental benefit (Table 4). The analysis was undertaken using critical effect sizes calculated from baseline survey data (Appendix 3 – Estimated replication for vegetation surveys) for some commonly employed environmental objectives in wetland management plans.

For some variables the number of controls required using this approach is considerable and likely beyond available resourcing. Moreover, comparison with controls is only one line of evidence for demonstrating environmental benefit. The final decision on how many controls are required must take into account resourcing constraints. Careful selection and distribution of controls both in comparison with treatment sites and considering longitudinal variation along the river may provide adequate lines of evidence. The implications of the fact that baseline data were collected during a drought also needs consideration when planning the number of control sites.

As a minimum, it is suggested three permanent and three isolated temporary controls are included for each weir-pool and below Blanchetown. This would lead to a total of 36 control sites; 18 in each category. This would address the replication level suggested in Table 4 for all but water-bird richness.

Table 4: Recommended numbers of controls necessary to demonstrate power (β = 0.05) and precision (margin of error shown)

Indicator	CES[1]	Standard deviation[2]	Number of controls based on Power analysis	Margin of error	Number of controls based on Precision[3]
Native fish richness	2 species	2.67	42	1 species	33
Native to exotic fish ratio	0.8	1.6	16	0.4	20
Frog richness	3 species	1.79	12	1 species	26
Water-bird richness	6 species	6.5	27	3 species	38

Note: 1. CES = critical effect size; 2. Estimated from baseline data; 3. Precision calculated is for margin of error shown

Data integrity, storage and analysis

Quality assurance and quality control

Ensuring data are of appropriate quality and consistency requires processes to be in place throughout the project, including planning, collection, storage and analysis. Considerations that are required during each of these components are described below.

- Planning
    - Since this monitoring covers a large number of wetlands, it is essential that consistent field methods (including data storage) are in place across the wetlands to ensure that data are comparable on spatial and temporal scales.
    - Field staff need to be suitably trained before collecting data.
    - All relevant legal sampling requirements (e.g. animal ethics, *Fisheries Management Act 2007* exemption (Government of South Australia 2007)) need to be in place well in advance of data collection.
    - Consistent and adequate datasheets (or portable data devices) need to be prepared prior to data collection (e.g. SKM 2006b).
- Collection
    - Field staff need to be suitably trained.
    - Field (and laboratory) equipment needs to be properly maintained, calibrated and any constraints on use are fully understood by field staff. Spare batteries and calibration equipment should be taken into the field.
    - Field (and laboratory) equipment used needs to be consistent between sites and sampling times. Where equipment is replaced or not consistent, appropriate cross-checking is required.
    - Data needs to be stored on consistent and adequate datasheets (or portable data devices).
    - Where uncertainty of species exist, record as 'Unknown 1' or similar and ensure vouchers are collected and provided to expert taxonomists and lodged with relevant museums and herbaria. Ensure field data are updated to reflect the correct name.
    - Data verification – after collection, ensure that data looks realistic based on what is known about the system. Are any outliers present? If so, do these represent a mis-entry or a true measurement?

- Storage
    - Consistent and adequate datasheets (or portable data devices) need to be prepared for data collection, including a description of who collected the data and where and when data were collected. These data sheets need to be stored securely and need to be readily accessible.
    - Raw data needs to be entered and stored electronically as soon as practically possible and backed up on multiple storage devices.
    - Data verification – after entry, ensure that data looks realistic based on what is known about the system. Are any outliers present? If so, do these represent a mis-entry or a true measurement?
    - Raw data are accompanied with appropriate metadata (who collected, where raw data are housed, contact details for raw data etc.).
    - All raw data are stored in a common database.
    - Data storage, security and accessibility – data should be stored in a secure, backed-up, maintained database and be readily available in forms required for analysis.
    - All data needs to be checked for quality after storage in common database.
- Analysis
    - All data needs to be checked for quality with possible outliers identified. If outliers are identified then cross referencing to raw data needs to be done (using appropriate metadata described previously).
    - For this program, monitoring data will be stored in the Management Action Database (MAD). This will include a holding point after the data are first entered. At this point, data will be reviewed before it gets accepted into the database. Appropriate metadata will also need to be made available to MAD.

Statistical framework

The RRP's monitoring aims have parallels with environmental impact monitoring (Downes *et al.* 2002), albeit with the impact in this case an attempt at restoration. Green (1979) introduced the need to monitor both control and impact sites in environmental impact monitoring and subsequent work saw a progression in impact study design complexity (Downes *et al.* 2002; Fairweather 1991; Ellis and Schneider 1997; Green 1993; Mapstone 1995; Underwood 1991; Underwood 1994; Underwood and Chapman 2003). The RRP needs to demonstrate the nature and magnitude of any benefits associated with management and provide information suitable for use in adaptive management.

By manipulating the water regime of managed wetlands the RRP hopes to promote the development of distinct communities, which differ from those present in the permanent and seldom flooded wetlands. The variability in species composition, abundance and community structure in permanently flooded and isolated temporary wetlands provides a template against which the managed wetlands can be judged – that is control sites represent end points on an inundation duration/frequency continuum, bracketing the ecological possibilities. Control systems establish the bounds for CES for a range of parameters, within which we wish to maintain treatment sites.

The monitoring program will take measurements from a selection of unmanaged wetlands (controls), including both permanently inundated and seldom flooded wetlands. This is required to establish the characteristics of such sites, including CES. At the same time, managed wetlands will be monitored to determine their response to management and how effective the suite of proposed indicators are for assessing performance against CES or defining community composition. For the managed wetlands, changes in parameter effect sizes and community composition in response to the water regime will be important in determining how the wetland biota has responded to management. An examination of these data will help managers decide when, and what type of, future management is required.

Plant functional group zonation will provide information on when managed wetlands are approaching the CES (derived from the permanently inundated and seldom flooded wetlands) and need to be drawn down or refilled. Other indicators such as water-birds and frogs are better used as indicators of wetland status. As both species are mobile, they are able to choose (to a certain degree) between wetlands. Thus by comparing managed and unmanaged wetlands it is expected that the managed wetlands will provide superior habitat and thus see increased richness and abundance of frogs, fish and water-birds. Evidence for this would come from the mean frog or bird richness and species identity of wetlands with different water regimes.

Many of the program aims relate to trend detection, necessitating repeated measures and permanent plots (Austin 1981; Bakker *et al.* 1996; McDonald 2003). Managed wetlands, which represent most of the treatment sites, were not selected with randomisation in mind. This violates assumptions of independence of samples of many statistical approaches and necessitates a model-based, rather than designed-based, approach to inference. To address some of the weaknesses of a model-based study design, where possible, randomisation has been incorporated in the selection of sampling units and in the site-scale configuration of sub-sampling protocols.

Statistical models are representations of the biological relationships between response and predictor variables in a mathematical form (Downes *et al.* 2002). A range of potential statistical models can be used to analyse the range of parameters identified for monitoring. Some examples are given in Table 5, but this detail is provided for background information only: it does not include all variables able to be analysed; it does not detail all possible approaches and it does not consider relationships between variables. Moreover, statistical approaches are constantly improving and therefore future analyses may differ.

Evaluation

Data will be collected over one or more sampling visits each year at both managed sites and wet and dry controls. Once the program is established, data analysis and interpretation can be timed to coincide with operational and reporting needs. This is necessary for future trend indicators which provide an important line of evidence for informing watering decisions.

Whilst all data can be analysed and presented each year, some indicators may take time to respond. Environmental variability means that time series data will provide the clearest indication of progress towards management objectives. A comprehensive data review should occur every five years to inform management plan revision. This analysis and review will focus on whether, and to what extent, hypothesised responses have occurred.

In this report, the estimates of the required number of replicates needed to achieve the necessary precision were based on baseline data. These data were collected from wetlands maintaining a permanent connection to the channel. Variability in monitoring data from managed wetlands will likely be different, requiring a re-assessment of the level of replication. During the initial stages of program implementation it is important that data are inspected immediately after sampling, and certainly no later than the next round of sampling. This should include an assessment of whether adequate precision is being obtained, and if not then, steps should be taken to increase sampling effort.

Table 5: Statistical framework for possible analysis of some indicator measures

Characteristic	Variable	Sampling population	Effect size	Analysis approach	Sampling unit/reps
Community composition (for vegetation, fish etc.)	Community composition	As above	Community composition (species or functional group) relative to permanently inundated and seldom flooded wetlands.	Multivariate statistics. Calculate distance matrix for sampling units (either quadrats or pooled transects). Classification (UPGMA) to determine clustering. Ordination (NMDS) trajectory plots. Bray-Curtis distance between samples demonstrates change. Characterise community composition using SIMPER. Difference between managed and control sites via PERMANOVA, ANOSIM Determine whether taxa groups relate to particular environmental variables present in control or managed sites Indirect gradient analysis (such as PCA) of environmental variables e.g. drawdown duration, max-min EL, EC, etc.) Canomical analysis of principal coordinates (CAP) to relate multivariate species data to environmental variables or gradients	Wetlands as replicates
Littoral plant zonation	Mean cover of various functional groups	Plants occurring between 0.9 m below and 0.2 m above normal pool level	Mean Plant Functional Group (PFG) cover over time for managed wetlands assessed against CES generated from permanently inundated and seldom flooded wetlands.	Reduce species data to PFG. Time series of mean cover ±CI for PFGs. Model response of changes in cover against hydrological and ecological variables (e.g. regression tree, MARS)	Replicate samples within each wetland. Representative radial transects across elevation gradient, with perpendicular 15 x 1 m quadrats at regular elevations Replicates determined from oversampling and

Characteristic	Variable	Sampling population	Effect size	Analysis approach	Sampling unit/reps
					a minimum variance according to J. Nicol data analysis.
Littoral plant diversity	Vegetation diversity (Simpsons index)	As above	Mean diversity over time for managed wetlands assessed against CES generated from permanently inundated and seldom flooded wetlands.	Calculate Simpson on proportional species data pooled to provide a single quadrat estimate Time series of mean ± CI Model response of changes in cover against hydrological and ecological variables (e.g. regression tree, MARS)	As above
Fish survivorship	Proportion in age classes OR Mean growth increment in Young of the Year (YOY) OR Condition indices	As above	Mean annual growth estimated as the difference between the mode in the distribution of YOY in autumn and 1+ in spring. Mean condition as length/weight ratio ES recorded from a managed wetland assessed against CES generated from permanently inundated wetlands.	Time series of mean growth ±CI for separate species. Model response of mean growth against hydrological) and ecological variables (e.g. regression tree, MARS)	Wetlands as replicates
Fish diversity	Simpson (or other diversity index)	As above	As for littoral plants	As for littoral plants	Wetlands as replicates
Turbidity	Turbidity	Wetland water column	Mean turbidity over time for managed wetland assessed against CES generated from permanently inundated wetlands.	Time series of mean turbidity ±CI. Turbidity expected to be less than permanent wetland derived CES	Multiple turbidity measurements per wetland, NTU

Characteristic	Variable	Sampling population	Effect size	Analysis approach	Sampling unit/reps
Frogs	Richness and species identity	Frog species presence/absence	Mean frog richness over time for managed wetlands assessed against CES generated from permanently inundated and seldom flooded wetlands. Expect >3 species to be present in managed wetlands	Comparisons of mean frog richness ± CI between managed, permanently inundated and seldom flooded wetlands. Showing difference in effect size. Also effect before and after management for managed wetlands. BACI using wetlands as replicates BACI design using PERMANOVA to include species identity information	Wetlands as replicates
Water-birds	Water-bird abundance, richness and identity, including functional groups	Water-birds	Mean abundance and richness for managed wetlands compared with permanently inundated and seldom flooded wetlands.	Comparisons of mean bird abundance and richness ± CI between managed, permanently inundated and seldom flooded wetlands. Showing difference in effect size. Also effect before and after management for managed wetlands. BACI using wetlands as replicates	Wetlands as replicates
Tree condition	River red gum, black box, river cooba	Canopy spp within 50 m buffer of wetlands Floodplains inundated by WPM	Condition and future trend indices assessed against CES generated from permanently inundated and seldom flooded wetlands and other tree condition datasets.	Time series of mean condition and future trend indices ±CI. Current condition future trend graphical analysis.	TLM tree condition method (Souter *et al.* 2010). Trees sampled in areas likely to be influenced by wetland mgmt, or weir-pools.
Sedges	Sedge species Qualitative condition classes CAP based on presence absence data	As for littoral vegetation	Condition as sedge cover and future trend index assessed against CES generated from permanently inundated and seldom flooded wetlands.	As above	Littoral vegetation quadrats.
Water-birds	Binary – breeding/not	Water-birds within, or dependent upon	Presence of nesting water-birds	Breeding would be a trigger for management actions that may differ from the management plan.	Wetland

Characteristic	Variable	Sampling population	Effect size	Analysis approach	Sampling unit/reps
	breeding	wetland			
Salinity	Salinity	Salinity measures within isolated wetland	Mean salinity ±CI, or wetland salinity profile, compared against CES derived from literature.	Mean salinity ±CI comparison against CES.	To be determined from baseline data.

Review of progress against overall management objectives

With clearly defined management objectives and targets selected, and monitoring methods in place, it is essential that resources are made available for progress to be demonstrated. It will be necessary to periodically review what the data reveal about progress towards the intended state of the managed system. Five-yearly management plan reviews are recommended to demonstrate whether water regime manipulation is achieving the intended goals. If no progress is being made, then reasons why need to be considered and targeted investigations may be required.

As data accumulate at multiple sites over a number of years, a range of manipulations will have occurred at multiple sites. Whilst different responses will have been observed, they will have been measured in a comparable manner. This provides an opportunity to undertake a meta-analysis, the most powerful means to generate inferential understanding from any investigation (Cumming 2012). Meta-analyses will provide a level of understanding of effect size in response to water level manipulation that will improve current management practices and increase the confidence in predictions for newly managed sites. As this understanding increases, the level of effort necessary for monitoring can also be greatly reduced, or re-directed.

At the time of management plan review, these data may indicate:

- interventions are working as the system is moving towards the desired state; or
- interventions are not working as the system is either not changing or moving in a different direction to that anticipated.

In the first situation, the main questions of interest are the size of beneficial impacts relative to watering decisions (i.e. can water-use efficiency be improved) and whether the variability in responses is such that the monitoring effort can be reduced or re-directed.

The latter situation is more complex. Questions for managers in such a situation include:

- Is more time required for a response to be observed?
- Are data collected adequate to demonstrate change over natural variability?
- Have other similar sites successfully achieved the same result, and if so, how did interventions at the sites compare?
- What possible confounding factors might be preventing the desired response?

In some cases data analysis may have shown the management objective to be inappropriate or unachievable owing to unforeseen circumstances or confounding factors. In other situations, the reasons may not be evident and unless the original management objectives are abandoned and new ones developed, additional investigations will be required to determine why the hypothesized response has not yet been produced.

Management of a large database is an essential component of a monitoring project that is often overlooked. If not done properly, this can severely inhibit the successful application of the database. In particular, data needs to be of appropriate quality and consistency, be stored securely, be accessible and data management processes should be in place to enable checking of stored data.

Wetland monitoring

Stratification within wetlands

The sampling design for indicators at wetland scales will rely on stratification – the grouping of areas of similar habitat for biota for sub-sampling. The link between wetland morphology and hydrology creates a mosaic of habitats. In order to capture the full range of variability in response to a managed water regime, the environment needs to be stratified such that proportional sampling is possible. Managers can use the baseline survey and management plan objectives to determine zones of direct and indirect impact and stratify monitoring effort according to the range and relative areas of habitat.

Two decisions are required when establishing a new site:

- What habitat types need to be considered?
- Where should sampling units be placed to obtain the best estimate of sampling population parameters of interest to managers?

The first question requires habitat stratification. For managed wetlands there are three broad habitat types to consider:

- wetlands
- anabranches
- floodplains.

The preceding sections provide an example of how to stratify sites within the sampling population based on habitat type, water regime class and impact zones. Whilst presented for vegetation here, a similar approach is required for all indicators.

An example case for site-level design for sampling amphibious vegetation

The different morphologies of the broad habitat types has implications for placement of transects. Wetlands are typically bowl shaped features in cross-section with shallow sloping banks and tend to be closer to circular in plan form. It is anticipated that a more or less continuous amphibious plant community will establish along the direct impact zone (i.e. the area between normal pool operation and maximum drawdown level). Anabranches and flood runners are alternative or former flow paths for the river. They tend to be linear features and may have steeply sloping banks. In features with steep sides, amphibious plants are likely to colonise the flattened bed areas and the break of slope at the top of the bank, where terrestrial cover should decline. Floodplains assessed as likely to be affected by management are constrained to be patches within 50 m of a wetland pool-level boundary, or areas directly inundated through weir-pool manipulation.

An example case study illustrates how to establish monitoring at a new site. Note that although the site presented is an actual managed wetland complex (Kroehn's Landing), discussion refers to generalities about the features, rather referring to physical realities potentially present at the site.

Both vegetation zonation and wetland bathymetry are determined as part of baseline data collection and can be used for stratifying managed wetland sites into habitat-level sampling units (Figure 4 and Figure 5). Site familiarisation visits should confirm patterns evident in baseline mapping and note any habitat features that have not been recorded.

The wetland complex is delineated into areas where the impact of a change in water regime may be expected and habitat zones within these areas (Figure 4). The complex includes wetlands and channels where water level is manipulated (direct impact zone). A hypothetical planned minimum-drawdown level for the wetland based on bathymetry is shown by the black dotted line. Riparian vegetation within 50 m of the wetland comprises the indirect impact zone, indicated by the red dotted line (Figure 4). Impacts from water level variation are predicted to occur at the site within the red line. The great majority, if not all, of the monitoring effort for this site therefore should occur within this area.

The nearest edge of the wetland is around 200 m from the main channel (Figure 4). Two (formerly) permanent anabranches connect a large permanent through-flow lake to the main channel. A vegetated island is present within the main wetland. A broad shallow zone is present to the north of the island, whilst the rest of the wetland is steeper. Both of these areas may develop different patterns of vegetation zonation once the hydrologic regime is changed and can be considered sub-habitats for stratification purposes. This highlights the value in using bathymetry data to help plan sampling strategies.

Riparian vegetation within the zone of influence (red dotted line) is dominated by river red gum. Two main features may influence the condition of these trees. The presence of the channel may help to maintain higher soil moisture levels on the river side of the wetland than the outer side. In addition, a cliff is present to the south-east of the site, which in places constrains the extent of the indirect impact zone. Depending on what area of the river the wetland and cliff were located, there may also be saline groundwater discharges to consider in planning. When sampling river red gum condition two linear quadrats should be established to account for effects of the river and cliffs, with one on the channel (western) side of the wetland and the other on the far (eastern) side of the wetland below the cliff.

Main wetland transect locations

Replication is based on having a minimum of 12 quadrats (3–4 per transect depending on the elevation gradient). Determining the randomised locations of permanent transects in major sub-habitat types was undertaken using a GIS and random number generator in Excel. The randomisation was done on the length of the wetland perimeter for the main wetland and on the perimeter of the maximum drawdown level for the island habitat. That is, the total length of each sub-habitat area was measured and random points were selected for survey. A constraint was applied that required transects be at least 5% of the relevant length apart to ensure reasonable coverage of the total area without introducing unrealistic levels of replication. The determination of minimum distances should be based on estimates of spatial auto-correlation which should be collected during the initial phase of the program and used to inform the minimum distances between quadrats.

The lines at each transect marker provide an indicative direction, but in reality these would be positioned across the elevation gradient observed during site establishment (Figure 5).

A similar approach was used to select sample locations for the anabranches and backwater pool, with length rather than perimeter the randomising factor (Figure 6).

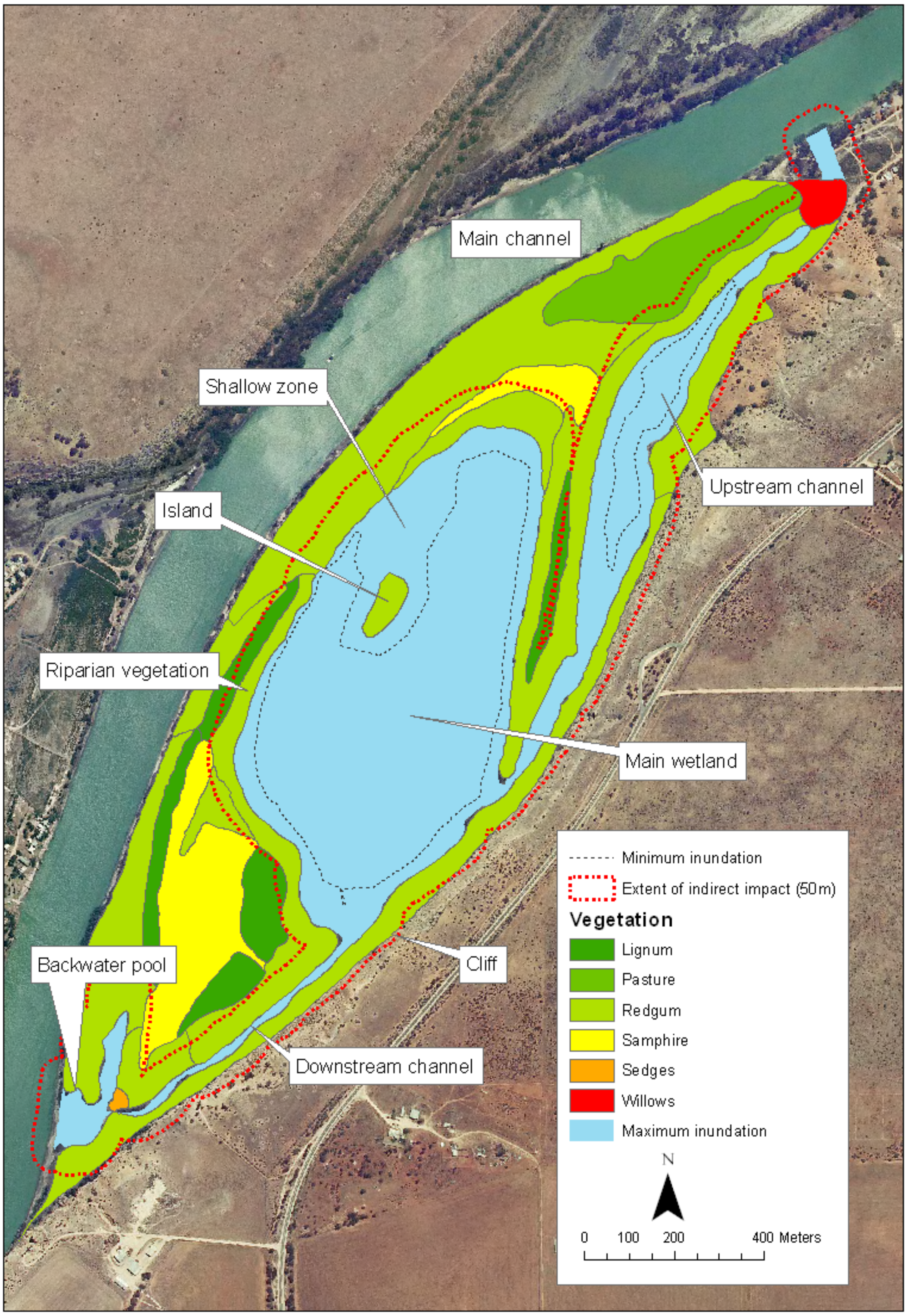

Figure 4: Kroehns Landing site with main planning considerations for RRP monitoring indicated

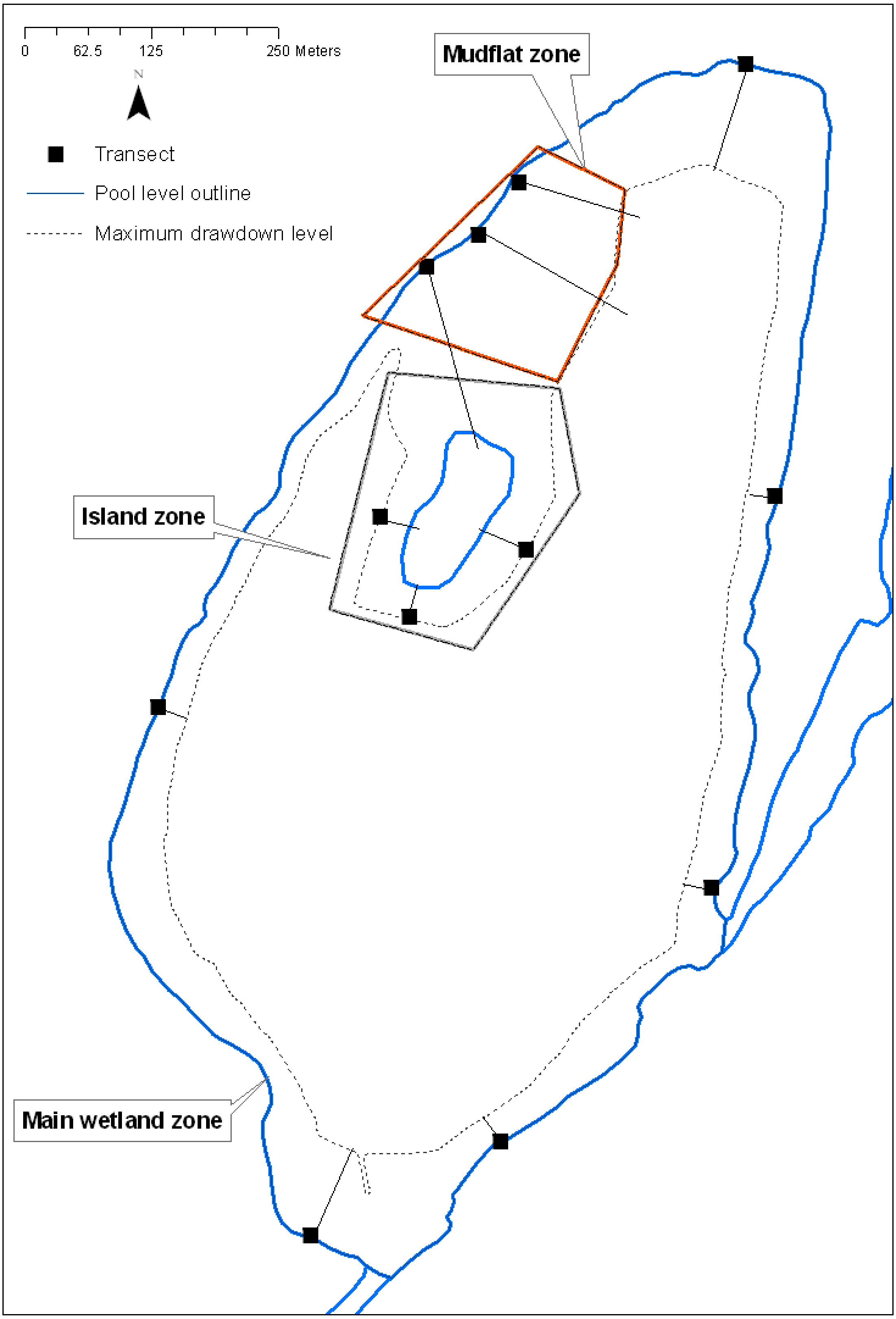

Figure 5: Transect placement in the Main wetland stratified by sub-habitat. Orange polygon represents the Mudflat zone, grey polygon the Island zone and blue outline the Main wetland zone

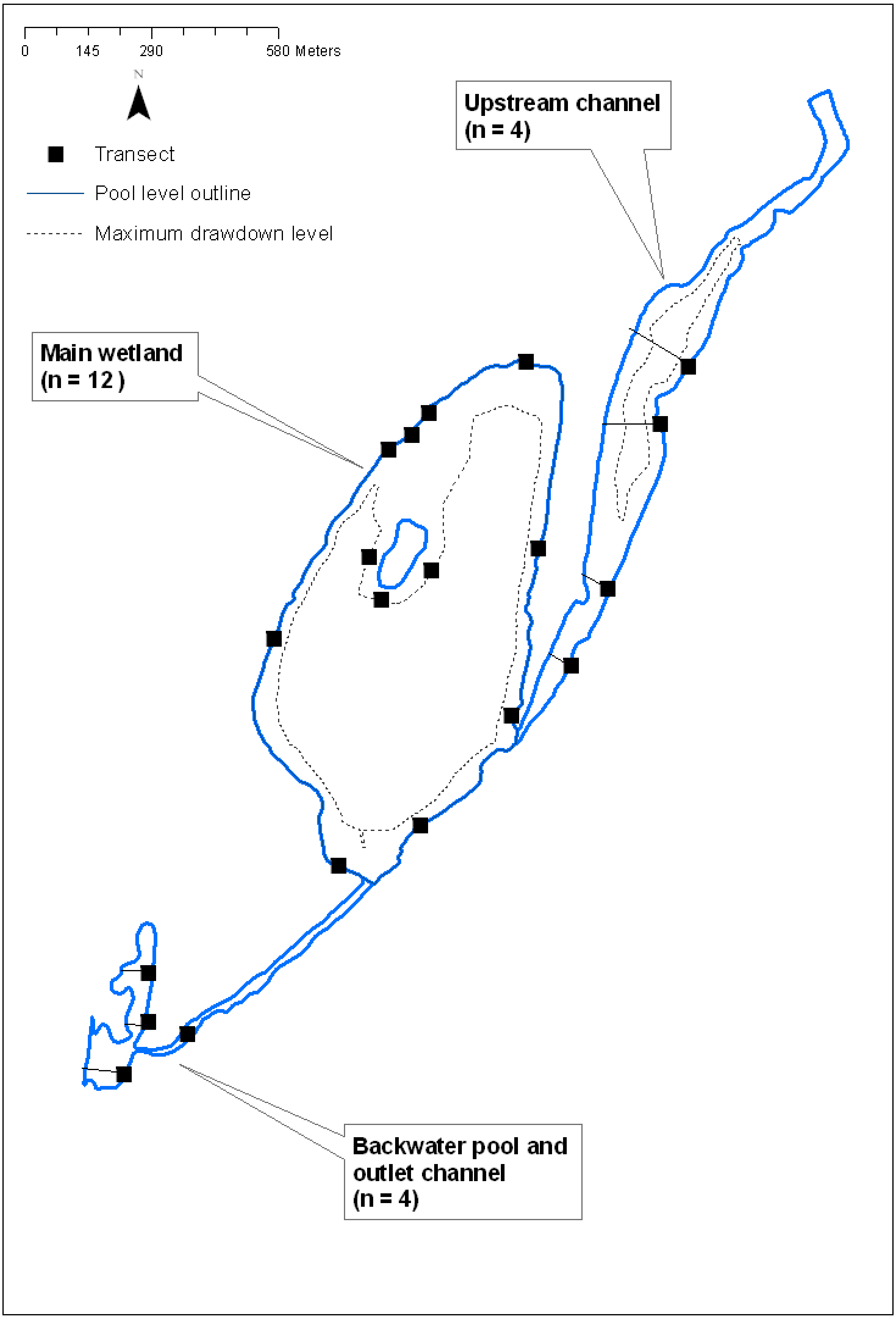

Figure 6: Anabranch and backwater pool sub-habitat sampling sites

Rationalising site level monitoring effort

The wetland area is likely to drive management actions, although the upstream channel may also be of interest. The downstream channel appears narrow and contributes little to the total area. If achieving standard replication levels meant that monitoring effort needed to be reduced at the site, the backwater pool and downstream channel would be the most logical component to omit. Initial site inspection and aerial imagery may determine whether or not it is worth sampling. This decision should be based on whether information derived from the sub-habitat would be used to inform management decisions or if it has any unique values of interest.

Over time, a decision on whether to reduce transect numbers can be informed by variability in the zonation and condition data. If management objectives have been achieved the minimum number of transects can be maintained for annual condition and trend monitoring, with two five-yearly composition-and-cover monitoring of the full set of transects undertaken to determine long-term changes.

Adapting to changes in zonation

Over time the distribution of a range of amphibious-plant functional group life forms, such as sedges, forbs and trees, can be expected to change. River red gums are expected to be suited to conditions at managed wetlands subject to drawdown, but when they reach a reasonable size amphibious tree species are not suited to monitoring with the composition and zonation methodology. It is anticipated that newly established river red gums will develop in clumps. The proposed solution is to record the presence of red gum as usual in the composition and zonation quadrats until the minimum size for monitoring via The Living Murray (TLM) methodology is attained. Furthermore, visual inspection should be used to examine the presence of juveniles. Once large enough, patches can be selected randomly for the establishment of new quadrats that use the TLM methods. Patches should be monitored annually at the time of amphibious plant monitoring.

Indicator: hydrology

The management targets for each site will dictate the level of detail required when collecting the two hydrological variables: change in water depth and spatial extent of inundation. Since management targets do not relate to flow volumes, these are not included. Hydrographs tend to be based on variation from pool level (depth). Most parameters of ecological interest (e.g. changes in vegetation zonation) are measured in the region of changing water level. When collecting water level parameters, determining the spatial location (e.g. relating water-level measurements to water regime at the site of vegetation data collection) and frequency of measurement is important. Where the rate of change in water level is critical to management, continuous automated logging may be required.

Sub-indicator: surface water depth

Methods for surface water depth data collection

Surface water depth can measured as a time series by one or more of:

- Manual gauge board reading
- Installation of a water level logger at the deepest point in the wetland. This is likely to be necessary where rates of rise or fall are important.
- Field survey and/or remote sensing methods e.g. SPOT, IKONOS, GEOEYE imagery.

Where to collect surface water level data

For all managed wetlands bathymetric data should be available. This will enable a single depth measurement to be extrapolated across the wetland to determine inundation at any location. All measurement infrastructure needs to be referenced to Australian Height Datum (AHD) to allow comparison with river level. It should also be referenced to a known elevation on the bathymetry (if the bathymetry data itself is not referenced to AHD).

Ideally the location selected for the measurement infrastructure would cover the full range of depths (from the wetland sill to the deepest section of the wetland). However, the placement of infrastructure will be subject to logistic constraints. More frequent measurements that cover most of the water level range will be more useful than a gauge board that extends to the full depth but is rarely read due to inaccessibility.

With a good understanding of wetland bathymetry, a single depth measurement point, referenced to the AHD, can be used to extrapolate depth across the wetland complex. If multiple large water bodies are included in the wetland complex, then a gauge board should be installed in each.

When to collect surface water level data

For managed wetlands the minimum amount of information required is the dates of closing and opening of the regulator, the lowest level that the water reached (in m AHD) and the date this occurred. This approach requires a measurement of weir-pool level and good wetland bathymetric data.

Additional measurements should be taken during site visits, but in particular when determining changes in wetland water-quality over the first few drawdown refill cycles.

Sub-indicator: spatial extent of inundation

Methods for spatial extent of inundation data collection

A range of options exist for determining spatial extent of inundation, from linking water level data to spatial extent via a digital elevation model through to remote sensing data. Combinations of these methods are ideal and provide a means of extrapolating point measures to establish areal coverage.

At wetland sites where local-scale monitoring is to be undertaken, installation of a gauge board referenced to the m AHD will allow manual readings to be taken opportunistically and during sampling visits. Where bathymetry data exists, it is a simple matter to determine the spatial extent for different water levels, allowing extrapolation of point measures to sites of biological data collection.

For remote sensing (see the

Weir-pool monitoring section), data will be required to determine inundation area. On-ground data collection will also be required for ground-truthing and calibration. Remotely-sensed images can be digitised and used to generate polygon feature classes in a GIS, which represent the area inundated for different water levels.

Indicator: surface water-quality

Water-quality data need to be:

- representative of the water-body
- collected with adequate replication to ensure appropriate precision
- collected in a manner which informs management decisions (e.g. to identify when trigger levels or thresholds are exceeded).

By far the simplest means to measure water-quality (physico-chemical conditions) of surface water is through the use of multi-parameter meters. Operators need to be familiar with the instruments and with calibration procedures, as these are critical for many parameters and uncalibrated instruments are a source of error that can not be compensated for.

Spatial and temporal variability are important considerations for all water-quality parameters (Table 6). It is important to ensure that these are random, rather than systematic errors, by ensuring measures are taken according to measurement protocols, as some parameters such as dissolved oxygen and pH change over daily time scales.

Sub-indicator: salinity

Salinity data collection has two main aims:

- To ensure critical biological thresholds are not exceeded (that is, to inform management decisions)
- To establish whether wetland salinity dynamics follow a simple evapo-concentration response and in doing so assess the likely contribution of groundwater inputs.

Methods for salinity data collection

The proposed method for measurement of all water-quality parameters is by the use of calibrated hand-held multi-parameter water-quality meters. Salinity is measured through collection of electrical conductivity data. Wherever electrical conductivity measurements are obtained, operators should collect data on all parameters for which the meter is capable of reading. Appropriate calibration is essential and should be conducted in accordance with the operating manual. Conversion of electrical conductivity to salinity requires an appropriate relationship to be determined or applied (e.g. Aldridge *et al*. 2012), with recognition that the relationship is non-linear.

Table 6: Technical considerations for common water-quality parameters (adapted from Baldwin *et al.* 2005)

Parameter	Source of variability	Controls
Dissolved oxygen	Daily changes in community metabolism lead to variations, with lowest values typically early morning High metabolic demand in poorly mixed or stratified water bodies lead to anoxic bottom layers	Measure at the surface and bottom of the water-body Measure at the same locations Measure at the same time, preferably early morning
pH	Daily cycle in values Groundwater influx Stratification	Measure the surface and bottom of the water-body Measure at the same locations Measure at the same time, preferably early morning
Salinity	Stratification may result in large differences between surface and bottom waters Gradients may exist across or along a water-body depending on mixing and the spatial dynamics of groundwater discharge High salinity readings tend to be more highly variable	Measure at the surface and bottom of the water-body Measure at the same locations When salinity exceeds normal river level, additional samples may be required
Turbidity	Variation in turbidity across the water column is unknown External inputs such as the proportion of Darling River water in River Murray flows High wind periods causing high rates of sediment resuspension High nephelometric readings tend to be more variable	Measure at the surface and bottom of the water-body Measure at the same locations When turbidity exceeds normal river level, additional samples may be required
Temperature	Thermal stratification may occur, even in shallow water bodies Temperature varies over daily and longer timescales and potentially according to groundwater input Many other parameters are only valid for a given temperature, and correction is required if the temperature of measurement differs	Measure at the surface and bottom of the water-body Measure at the same locations Measure at the same time, preferably early morning

Note: Data analysis demonstrating the increasing variance of mean turbidity and salinity with increasing concentrations are presented in Appendix 3 – Estimated replication for vegetation surveys

Where to collect salinity data

Salinity is likely to be highly variable in an isolated water-body, changing with depth, location (especially where groundwater inflows are occurring or at the blind end of a terminal wetland) and over time. Without characterising the variability of a water-body, it is not possible to estimate the mean salinity. Thus replication in space, including depth, is required to provide a point estimate in time and confidence intervals (Baldwin *et al*. 2005). A pilot study with intensive sampling can determine the salinity distribution across a wetland and this data can be used to randomly stratify future salinity data collection.

The number of salinity measurements is set by the desired precision of the estimated mean salinity. Appendix 2 – Critical effect size determination – presents an analysis of the baseline data for River Murray wetlands and indicates that variability increases linearly with conductivity, even when conductivity is low. The basic replication required for water-quality parameters is eight sites. If the mean differs by more than 40% in either direction from any individual measurement (Appendix 2 – Critical effect size determination), additional measurements should be taken up to a maximum of 20.

When to collect salinity data

Determining high resolution temporal dynamics are only possible with the installation of continuous recording and logging instruments. This level of effort would only be warranted if salinity data indicated increased salt loads were occurring at the site. This level of complexity is not required for the purposes of RRP monitoring and a suitable precision can be obtained through the use of a hand-held electrical conductivity meter. Repeated manual readings at numerous depths and locations within the water-body will establish three-dimensional variability in the system.

Salinity needs to be monitored with a resolution suitable to allow action to be taken prior to the threshold being exceeded. During the initial management period (first 1-5 drawdown-refill cycles), the aim of electrical conductivity monitoring is to confirm the theoretical volume-salinity relationship for the wetland (see below). Multiple salinity readings from a range of depths and locations across a wetland taken periodically at consistent stages of the wetting-drying cycle are required to determine this relationship (and also to determine whether management action thresholds have been attained). Revisit frequency for water-quality depends on the rates of water level change. During drawdown and refill, measurements need to be taken at a frequency that is able to adequately determine a volume-salinity relationship, noting that the relationship may be non-linear (see below). During stable water periods monthly measurements are adequate. Sampling of submerged vegetation and fish communities should be accompanied by measurements of salinity.

Evaluation

Salinity monitoring for trigger levels

Monitoring data will determine when the trigger level is reached and a management action is required, most likely reconnecting a wetland to the river. When assessing salinity data against thresholds, an understanding of variability in the mean is required.

Figure 7 shows four hypothetical samples taken from the same water-body but with different levels of precision. In Case a, the estimated interval for mean conductivity is clearly below the threshold value of 3000 µS/cm. Managers are able to delay re-filling for the time being. Case b provides equally clear guidance: the trigger level has been exceeded and re-filling should be considered as a matter of priority. Cases c and d provide equivocal evidence. In

Case c, the mean estimate appears to be well below the threshold, but data variability is such that the 95% confidence interval around the mean includes the threshold. Managers now may have to decide on a precautionary re-filling of the wetland, when conductivity may be well below the threshold. Similarly in Case d, the threshold appears to have been exceeded, but confidence intervals include a range that is below the threshold. Here managers are faced with a clearer decision in the case of salinity management, but the information at their disposal is not ideal. The precision of salinity data can be improved by taking more samples. By knowing the trigger level in the field this can be readily accomplished.

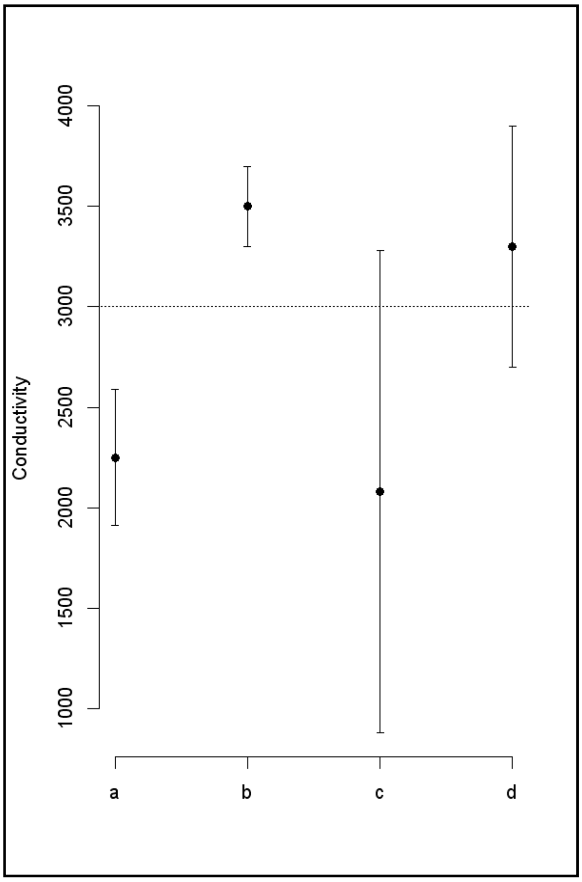

Figure 7: Influence of measurement precision in evaluating data for trigger levels

Salinity monitoring to assess groundwater inputs

The salinity of isolated wetlands is expected to increase during drawdown due to evapo-concentration. In the absence of external inputs (e.g. rainfall, groundwater) this will happen in a predictable manner as determined by the mass balance of salt loads. This relationship is unique to wetland bathymetry and may well be non-linear (Figure 8). Provided the wetland volume to depth relationship is well known, the contribution of groundwater to the salt balance can be obtained using a mass balance approach (see Aldridge *et al*. 2011). Comparison of observed salinity with salinity predicted from a wetland-specific volume to salinity relationship will provide an indication of whether groundwater is entering the wetland.

The volume to salinity relationship is determined by measuring the mean salinity of the water in a wetland after refilling, with adjustments made for rainfall inputs over the period of interest. Rainfall can be determined by installation of rain gauges or from surrounding rainfall gauged sites of the Bureau of Meteorology. Based on a mass balance, the concentration for a given wetland volume (depth) can be determined as the ratio of the fraction of water from the initial volume remaining and the salinity of the fill water according to the relationship:

Predicted salinity = initial salinity x 1/(remaining volume / initial volume)

where mean salinity follows the predictions from the evapo-concentration curve, groundwater discharge is not of concern. If the mean measured salinity differs significantly from this value then it is likely that groundwater inflow is significant. Monitoring and analysis of the salinity data should be undertaken frequently over the first few fill cycles in order to develop a the volume/salinity relationship (e.g. Figure 8) and verify this behaviour on subsequent cycles.

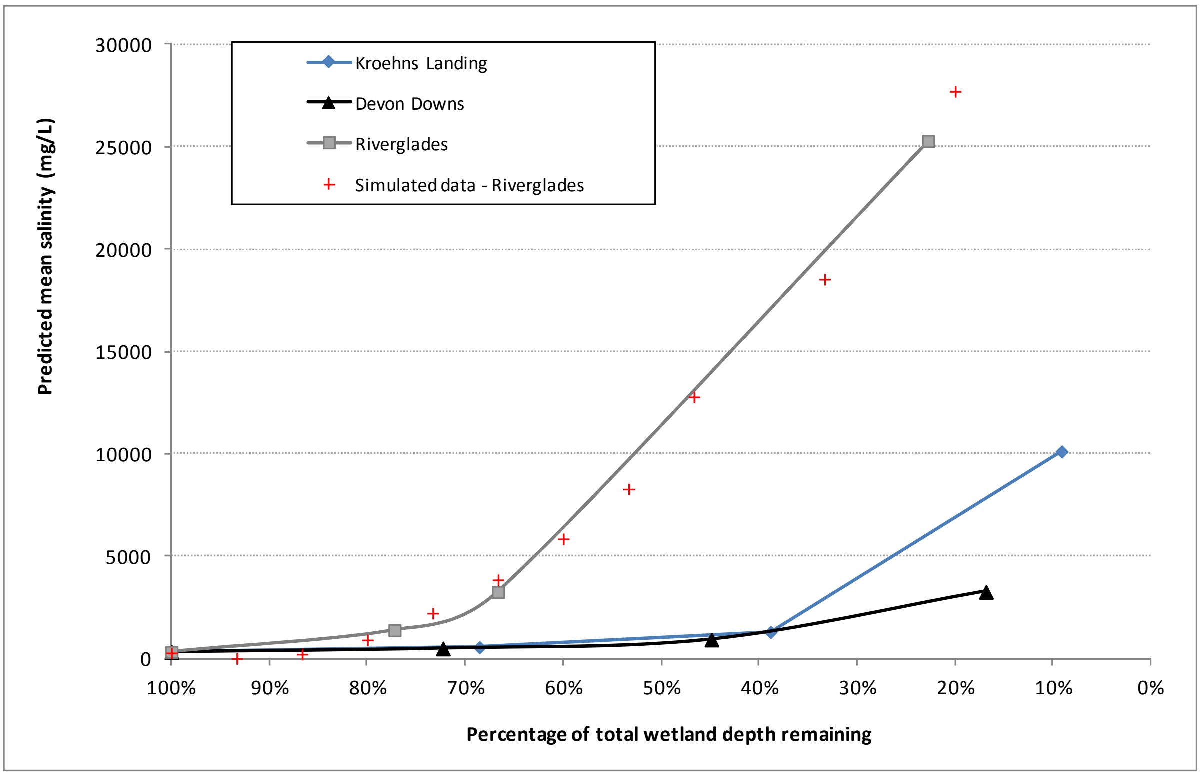

Figure 8: Theoretical mean wetland salinity as a function of remaining depth during a drawdown for three wetlands. Salinity is predicted to vary as a function of the percentage of total wetland depth as the wetland evapo-concentrates during periods of isolation. Negligible rainfall is assumed. The depth to volume relationship was obtained from SKM (2006a). The curves assume an initial salinity on refilling of 320 mg/L (500 µS/cm). Red crosses illustrate how observed salinity might vary if evapo-concentration is the only process affecting concentrations

Sub-indicator: turbidity

Methods for turbidity data collection

The two methods for measuring turbidity in common use are the secchi disk and electronic devices which record nephelometric turbidity units (NTU). The use of NTU as a measure is recommended for the RRP as the data can be collected at the same location and time as other water-quality parameters and it is an objective measure not dependent upon ambient light.

All water-quality parameters will be measured using a calibrated hand-held multi-parameter water-quality meter. Wherever electrical conductivity measurements are obtained, operators should collect data on all parameters for which the meter is capable of reading. Appropriate calibration is essential and should be conducted in accordance with the operating manual.

Where to collect turbidity data

Turbidity measures can be co-located with those for electrical conductivity. The aim is to obtain the best indication of the mean turbidity of the water-body as a whole. Sampling at a number of positions around the wetland and different depths is ideal.

As with electrical conductivity, the aim of the turbidity monitoring is to estimate a mean turbidity value and suitable confidence intervals across the wetland. Multiple measurements across a wetland at a single time represent a sample. Repeat samples taken over a period of time can be compared to demonstrate dynamics in turbidity.

Guidance on the normal range for turbidity and effect sizes to manage for were obtained from baseline survey data (Appendix 2 – Critical effect size determination). As precision is largely a function of variability, to maintain the same precision the required number of replicate measurements increases rapidly as variability increases.

Measurement error for the range of replication in baseline data (n = 3–5, Figure 9) indicates that confidence intervals increase linearly with mean turbidity (Figure 10). Four replicate measures were adopted as the measuring protocol for baseline surveys, but measurement precision was too unpredictable at this replication to adopt this value for the RRP. Exact guidelines for the number of replicates required can only be established after sampling at the recommended replication is undertaken and the distribution of sampling values can be ascertained. However, given the ease with which turbidity can be measured the collection of additional samples is not likely to be unduly costly.

A doubling of this level of sub-sampling (eight replicates) is recommended. However, variability in turbidity measurements increases linearly with the mean (Appendix 2 – Critical effect size determination), resulting in wide confidence intervals. Monitoring staff will need to be able to adjust according to observed variability and collect extra samples as required. As a guide, additional samples should be collected when any of the eight recommended sub-samples vary by more than 30% in either direction from the mean value calculated on the day (Appendix 2 – Critical effect size determination).

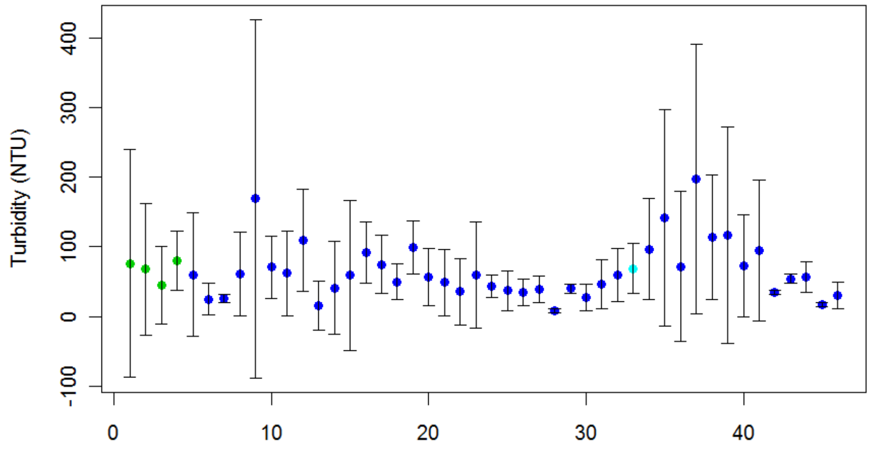

Figure 9: Mean and 95% CI for baseline survey turbidity data for various wetlands (x-axis). Colours indicate replication: Green = 3; Dk blue = 4; Lt blue = 5

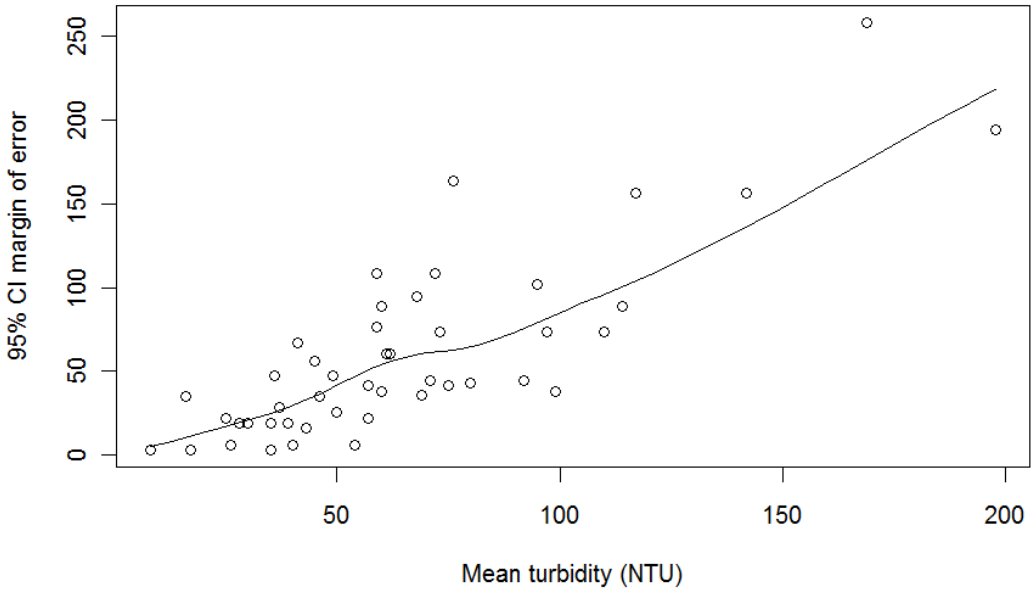

Figure 10: Margin of error for 95% confidence intervals as a function of mean turbidity estimate for baseline surveys. Curve is a lowess smooth of the data

When to collect turbidity data

Turbidity in the river is highly variable and depends on the proportion of water sourced from the Darling River. Baseline data suggests minimum turbidity occurs in late winter (August, although no data were collected for May to July) and maximum in early summer (Figure 11). However, turbidity is still extremely variable (Figure 11). Sampling times are recommended for early spring and late summer, but relate more to drawdown and re-filling cycles. Turbidity measurements should occur periodically, including measurements as soon as practical after closing the regulator (not beyond a month after closing). The timing of turbidity measurements should coincide with those of electrical conductivity.

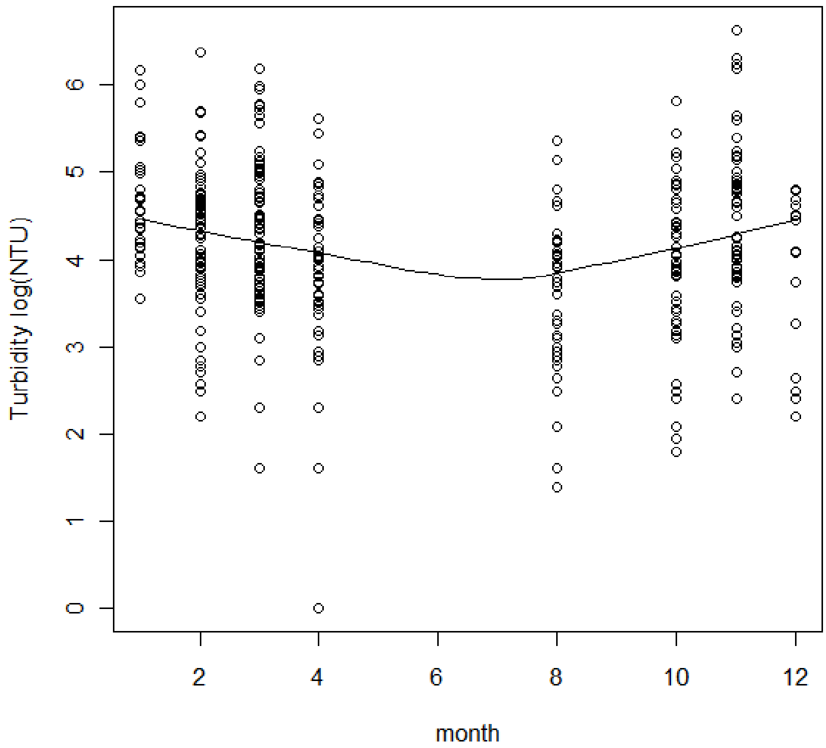

Figure 11: Log transformed turbidity as a function of sampling month (of the year). Curve is a lowess smooth of the data.

Evaluation

Turbidity data are collected to assess the management aim of improving water clarity. Data are analysed by comparing mean turbidity pre- and post-treatment. The main consideration when collecting turbidity data is the precision of the mean estimate for the number of samples collected.

The distribution of baseline survey data shows the majority of turbidity measures were below 121 NTU and the presence of extreme outliers increases the mean well above median values (Table 7; Figure 12).

Table 7: Summary statistics on baseline turbidity measures (n = 463)

Statistic	Value (NTU)
Median	70
25th percentile	39.5
75th percentile	121.5
Interquartile range	82
Mean	103.8
Standard deviation	114.3

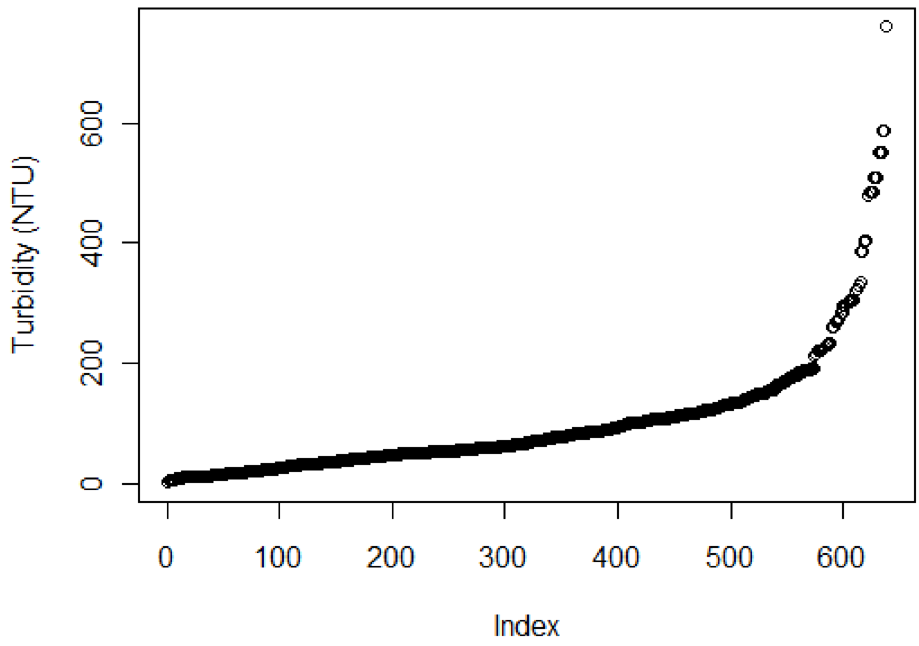

Figure 12: Rank order (index) for turbidity – baseline surveys 1 and 2 (n = 643)

Sub-indicator: pH

Under typical conditions pH is not a parameter that river managers would focus on. However, in situations where acid sulfate soil formation is a risk, wetland acidity requires monitoring. Sites at risk of acid sulfate soil formation will not knowingly be incorporated into the managed wetland program, and are not expected to be of concern. However, while likelihood is low, the consequences to the wetland itself would be high and it is advisable to collect data to analyse trends in pH.

Methods for pH data collection

A calibrated hand-held multi-parameter water-quality meter should be used for collecting pH. Wherever electrical conductivity and turbidity measurements are obtained, operators should collect data on all parameters, including pH. Appropriate calibration is essential and should be conducted in accordance with the operating manual.

Where to collect pH data

As the aims of salinity and turbidity monitoring are to characterise the mean values for the water-body, no specific additional considerations are required for pH.

No additional data collection is suggested for pH in addition to that of salinity and turbidity, unless data indicate additional investigation is required. For inland waters, it has been recommended that changes of more than 0.5 pH units from the natural seasonal maximum or minimum warrant additional investigation (ANZECC 2000). Baseline data may not provide a good estimate of seasonal extremes under natural conditions, as these were effectively opportunistic and only collected for a single year, although these represent the best data available. Where long-term time-series data have established the natural range of pH, then it is possible to adopt the (ANZECC 2000) suggestion.

Baseline data indicate the mean difference in pH units between seasonal maximum and minimum is 1.1 pH units [95% CI: 0.87–1.35] and a difference below the baseline data minimum value at a site exceeding 1.4 pH units requires investigation. In standard deviation units, this represents an effect size in excess of 2. Replication for water-quality measures of eight gives power to detect a change of 0.99.

When to collect pH data

The main risk of acid soil formation is during the refill after extended drying where there are large quantities of sulfidic minerals (see DEWNR 2012a). Any issues associated with acidity should be evident at the time of sampling for turbidity and salinity. No additional data collection is suggested for pH unless data indicate additional investigation is required.

Evaluation

The main aim in collecting pH data is to ensure that no adverse changes in pH have occurred on re-filling, due to the exposure of acid sulfate soils during the drying phase. Data analysis involves calculating mean pH from samples. Since pH units are a negative logarithm of hydrogen ion concentration, calculation of average pH units requires calculation of the hydrogen ion concentration of replicates, calculation of mean hydrogen ion concentration followed by calculation of mean pH. Based on baseline data, a change in pH values exceeding 1.4 pH units below the minimum in baseline data indicates an unusually large variation and requires additional investigation.

Sub-indicator: dissolved oxygen

A major water-quality risk associated with implementing drying and rewetting cycles in wetlands is the development of persistent hypoxic or anoxic conditions resulting from enhanced rates of organic matter decomposition, which consumes oxygen through microbial respiration (see DEWNR 2012a). This may reduce dissolved oxygen levels beyond the physiological tolerances of biota.

Methods for dissolved oxygen data collection

All water-quality parameters will be measured using a calibrated hand-held multi-parameter water-quality meter. Wherever electrical conductivity and turbidity measurements are obtained, operators should collect dissolved oxygen data. Appropriate calibration is essential and should be conducted in accordance with the operating manual.

Where to collect dissolved oxygen data

As the aims of salinity and turbidity monitoring are to characterise the mean values for the water-body, no specific considerations are required for dissolved oxygen.

When to collect dissolved oxygen data

The main risk of oxygen depletion is during the refill after extended drying where there are large quantities of organic material available (see DEWNR 2012a). Any issues associated with hypoxia or anoxia should be evident at the time of sampling for turbidity and salinity. No additional data collection is suggested for dissolved oxygen unless data indicate additional investigation is required.

Evaluation

The main aim in collecting dissolved oxygen data is to ensure that no adverse changes in dissolved oxygen have occurred on re-filling. Data analysis involves calculating mean dissolved oxygen and should be related to the physiological tolerances of biota.

Indicator: vegetation

Sub-indicator: photopoints

Photographic time series are a useful visual tool to demonstrate change through time. At least one photopoint should be established at each surveyed wetland. Photopoint monitoring should be continued at wetlands where it has already been established.

With the advent of digital photography, the use of photopoint monitoring has the potential to provide quantitative outputs (Crimmins and Crimmins 2008). Detailed guidelines in the theoretical aspects for establishing (Hall 2001a) and analysing (Hall 2001b) traditional photo survey points for environmental monitoring should be consulted when setting up new photopoints. However recent advancements include the GigaPan system (http://www.gigapan.com), which uses a robotic camera mount to automate the position and shutter release of a standard digital camera. Panoramic images are produced by stitching together a series of photographs taken as the robotic camera mount rotates and operates the camera. Custom software stitches the images together creating a very-high-resolution gigapixel panorama. The use of such techniques is in its infancy but offers great promise for monitoring (Nichols *et al*. 2009). Such images could be taken along selected transects so that the vegetation response can be correlated to changes in imagery. If such correlations prove reliable then the effort directed towards detailed on-ground monitoring can be reduced as sufficient information is gained through photographic methods. However the use of such a system requires adequate testing and evaluation.

Sub-indicator: amphibious plant functional group cover and composition

Methods for amphibious plant cover and composition data collection

Amphibious plant cover, composition and zonation are assessed under this program using existing methods that have been successfully applied to monitor responses to inundation (Nicol 2010; Zampatti *et al.* 2006). In particular, the methods are a direct adaptation from work used for *The Living Murray* monitoring and has been successfully developed over a number of years for which a considerable database is accumulating (see Nicol and Weedon 2006; Nicol *et al.* 2010b; Zampatti *et al.* 2006).

Permanent vegetation plots are a necessary technique for investigating vegetation succession (Austin 1981; Bakker *et al.* 1996). This approach is adopted here, with permanent transects and quadrats established to monitor the development of amphibious plant zonation in direct impact zones (i.e. the area between normal pool operation and the lowest drawdown level).

The basic approach is to establish transects perpendicular to the water line and contours along elevation gradients. Sampling units comprise 15 contiguous 1 x 1 m cells that form a linear quadrat, perpendicular to the transect (parallel with contours). Quadrats are positioned at regular increments in elevation along each transect, covering the full range of elevation within the direct area of influence of the water level manipulation.

To survey a quadrat, a tape measure is used to mark the boundary and the number of cells in which vegetation species are recorded. For each cell in the quadrat, vegetation presence/absence is recorded. Data collected are species abundance determined as frequencies – the number of cells within each transect where the species is present. Hence abundance values for each species in each quadrat vary between 0 and 15. Note that cells with no living plants will be recorded as bare soil and given a bare ground score of one.

Wherever possible plants should be identified to species. Nicol *et al*. (2010b) recommend using Sainty and Jacobs (1994), Cunningham *et al*. (1981), Jessop and Tolken (1986), Romanowski (1998), Jessop *et al*. (2006) and Dashorst and Jessop (1998), and use nomenclature following Barker *et al*. (2005). Any unknown species encountered should be taken as a voucher sample and referred to on datasheets as Unknown species x (where 'x' reflects the number of unknown species recorded i.e. Unknown species 1, Unknown species 2 etc.). Voucher specimens should be referred to expert taxonomists (e.g. at South Australian Research and Development Institute or the State Herbarium of South Australia) for positive identification. Field data should be updated with the correct species name prior to final archiving in secure databases.

This method provides a quick, quantitative estimate of the distribution and abundance of species and is sufficiently sensitive to detect differences in floristic composition through time and in response to management actions (e.g. Gehrig *et al*. 2012). Furthermore, there is no discrepancy between observers, which can occur when visual estimates of cover are used, such as estimates of percentage cover or Braun-Blanquet (1932) cover abundance scores. However, there are instances where frequency and percentage cover are poorly correlated. For example, a species that has 100% cover in a quadrat and species that has less than 1% cover but one individual in each cell will both get a score of 15.

Where to collect vegetation cover data

Both wetland and anabranch habitat types need to be sampled. If multiple wetlands exist, each needs to be sampled and analysed separately. Multiple channel reaches may be pooled for analysis as the area concerned is much lower.

As described in the *Stratification within wetlands* section, wetland bathymetry should be used to plan transect locations. Transects should extend to incorporate the area over which water level variation will occur, provided this can be well predicted. If the exact area of inundation cannot be predicted, then additional quadrats should be measured and the actual extent of inundation recorded, with the inundated quadrats retained for future monitoring.

Within areas of relatively uniform conditions, randomisation should be used to select transect locations. A minimum spacing requirement should be imposed to ensure reasonable areal coverage, with no transects located within 50 m of another (as applied in the Nicol *et al.* (2006) sampling methodology). However, this value may increase when the effects of spatial autocorrelation are determined using pilot data. Depending on the size of the wetland and the intended drawdown, the minimum distance criteria may need to be applied at the lowest elevation, as transects will be radial, reducing inter-quadrat distance with movement down the elevation gradient. Any method that allows for a random number to be selected that can be related to a real location in the wetland is acceptable. For wetland habitat, transects will usually be radial (as in the spokes of a bicycle), while in anabranches transects will be perpendicular to the long axis.

Any variation at sub-habitat scales will be averaged out over the entire sample. It may be necessary to review this if wetlands vary dramatically – perhaps one side of the wetland differs in soil texture, or an adjacent cliff may provide high levels of shade to one area of a wetland. In such cases riparian vegetation or lack thereof should provide some indication as to whether the variation will affect the growth of amphibious plants. If such impacts are suspected, then stratifying the wetland to sub-habitat level will be necessary prior to selecting transect locations.

Locating quadrats

Anabranches and flood runners are likely to have steeply sloping banks and a linear, planform geometry. The distribution of amphibious plants expected to develop in steep sided features is anticipated to be limited to flattened areas of the bed and top of the bank. Quadrats are placed to focus on these areas (Figure 13). For wetlands, it is anticipated that a relatively continuous amphibious plant community will establish along the direct impact zone. Regular quadrat placement is required (Figure 14).

Information is also provided here for floodplain monitoring as methods for on-ground weir-pool monitoring of floodplains will be the same. Where weir-pool raising will inundate an area of floodplain, regular quadrat placement is required. However, quadrat placement will be at smaller increments than for wetlands as floodplains areas tend to have a shallower slope than wetlands (Figure 15).

In all cases, transects are oriented to traverse the elevation gradient (Figure 5, Figure 13, Figure 14 and Figure 15). Quadrats sample individual elevations and are placed along contours at regular intervals perpendicular to transects.

Quadrat set-up will require the use of a surveyor's level. Placement quadrats along transects according to elevation increments are as follows:

- Normal pool level – this is the zero elevation datum for all sites.
- For raising – locate one quadrat at the anticipated uppermost extent of inundation and two further quadrats at equally distributed elevation mid-points between the uppermost and normal pool levels. Elevation increments between quadrats should not exceed 0.2 m. If so, quadrats should be positioned at 0.2 m increments. Hence a minimum of four quadrats should be located on each transect.
- For lowering – locate one quadrat at the lowest point that water level will be drawn down to. Place additional rows of quadrats at 0.3 m vertical increments along the elevation gradient. If the difference in elevation between the upper and lower extent quadrats does not exceed 0.3 m, place two additional quadrats at equal differences in elevation between normal pool and minimum levels.

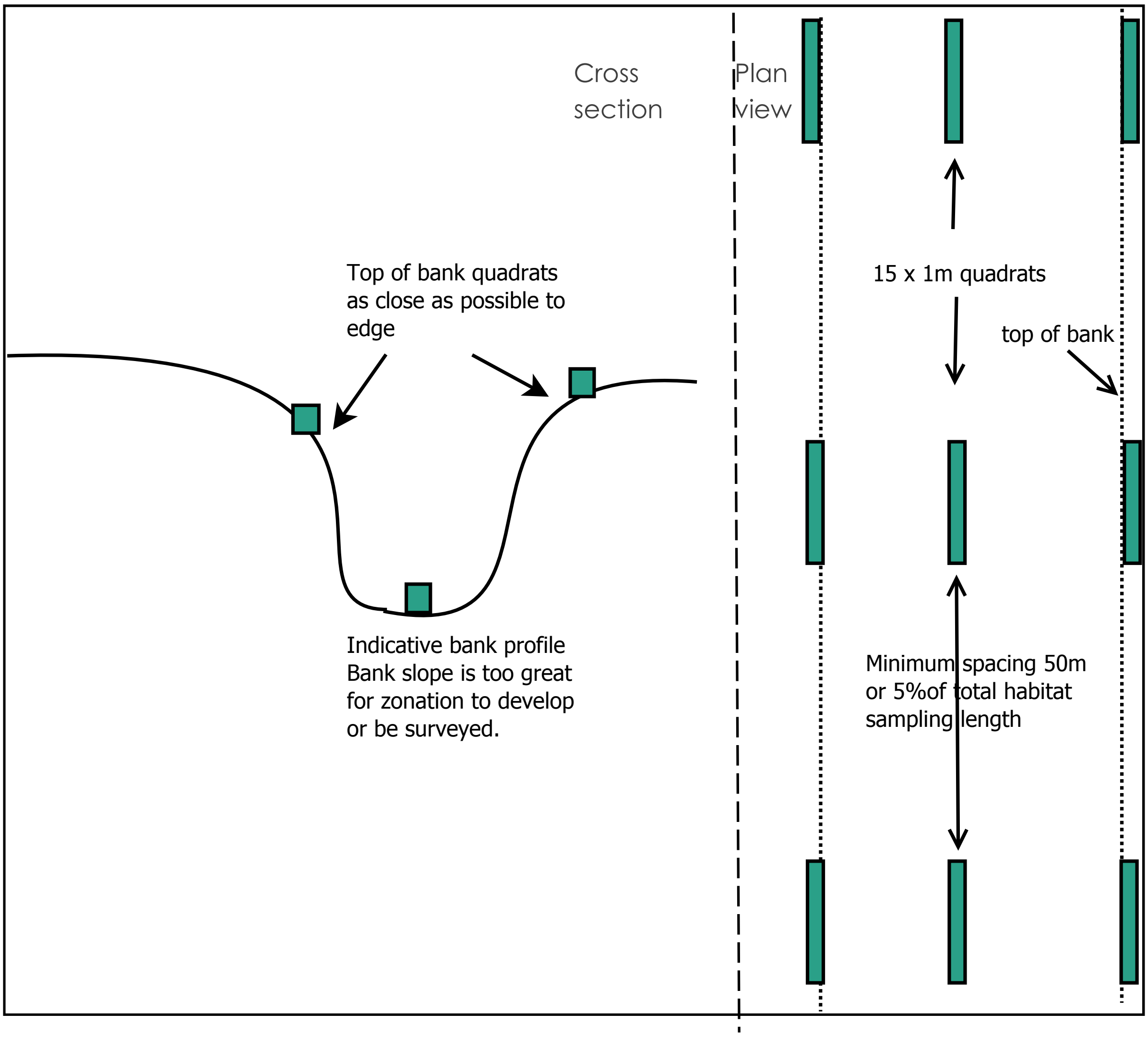

Figure 13: Quadrat orientation for steep bank profile channels (adapted from Nicol *et al.* (2010b))

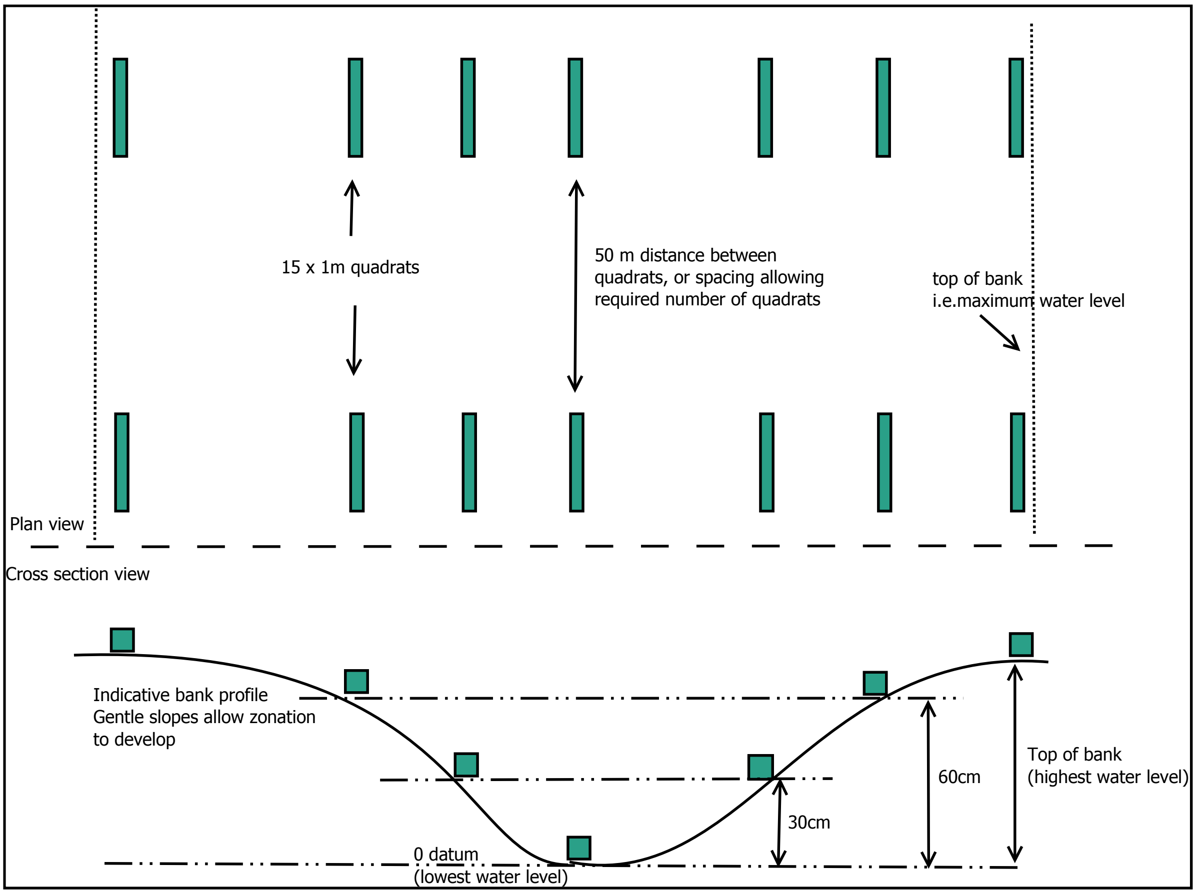

Figure 14: Quadrat orientation for gently sloping banks of wetlands (adapted from Nicol *et al.* (2010b))

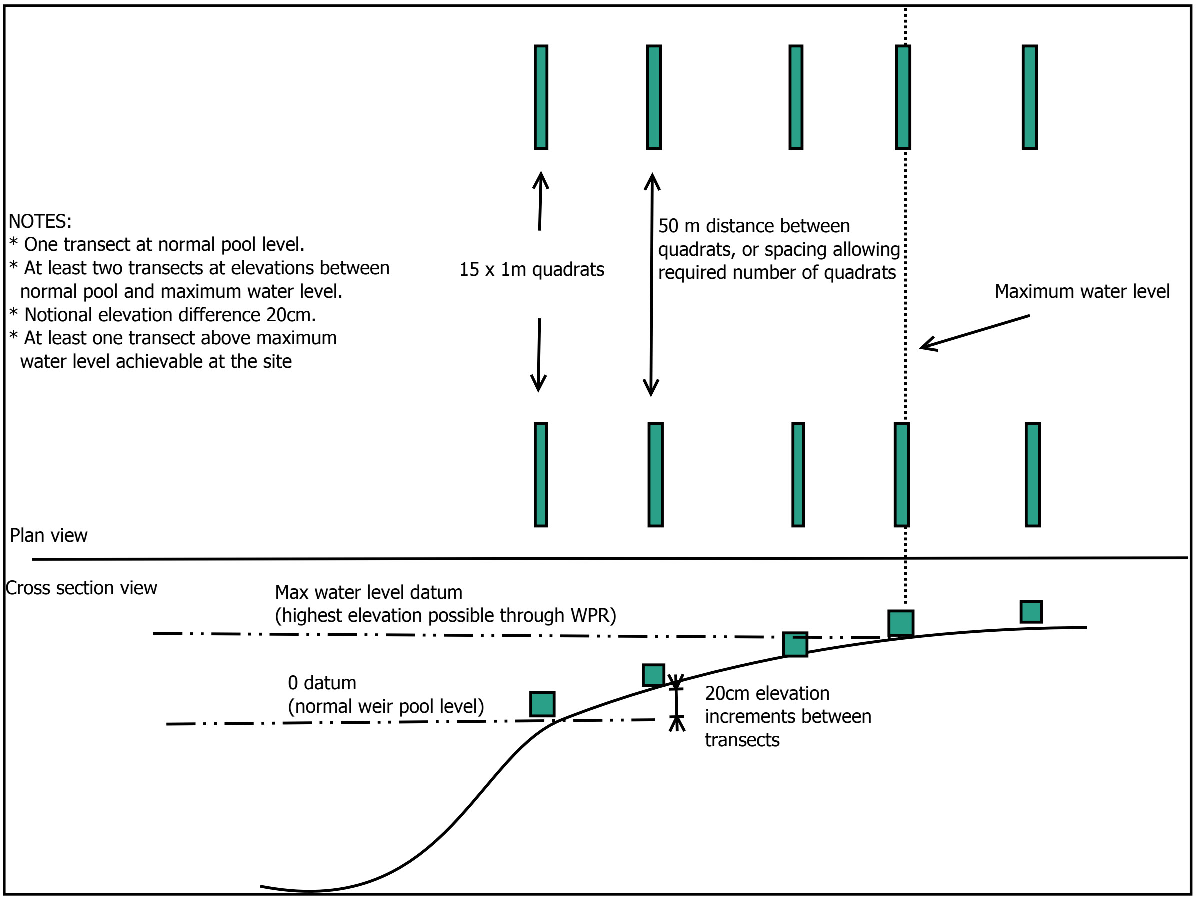

Figure 15: Quadrat orientation for riparian zones adjacent to wetlands

Baseline data analysis (Appendix 2 – Critical effect size determination) indicates that 12 replicate quadrats should provide adequate precision for estimates of site diversity at the most variable of sites. Assuming that responses do not exhibit major differences across the elevation gradient, then this is the total number of quadrats required to estimate plant community character. If over time changes are observed in the cover and composition of quadrats at different elevations, then it may not be possible to pool these to form the sample and at least 12 replicates may be required at each elevation.

Each transect should be marked by a 1.5 – 2 m star picket with a high visibility cap and be located high enough above maximum pool level to avoid loss. If permanent star picket placement will create a safety or other issue, then shorter pickets can be used. These should be driven in flush with the ground surface and in this situation it will be necessary to use a metal detector to re-locate the site marker. Transect locations need to be recorded using a hand-held GPS.

A compass bearing in the direction of the transect should be taken and the linear distance from the marker to the elevation of each quadrat can be determined using a tape measure. This allows for future re-location of quadrats without individual marking. On datasheets (or PDA) transects and quadrats should be identified with unique codes that should reflect the site (e.g. first three letters of the wetland name), the habitat type, a unique transect number and the elevation of the quadrat.

Digital photographs should be taken at each transect orientated along the transect direction of survey following the methods presented in Hall (2001a) or similar reference guide. Note that this requires the use of additional markers allowing later analysis of the photographs in a semi-quantitative manner (methods are presented in Hall 2001b). Details of the photo reference should be documented in datasheets (or handheld PDAs).

When to collect vegetation cover and composition data

Amphibious vegetation should be surveyed shortly after wetlands reach minimum water levels and access is available to the full area of direct impact. If wetlands remain drawn down beyond a season (including partial drawdowns), data should be collected seasonally to determine changes in cover and composition over annual cycles.

Data from permanent and temporary wetland control sites should be collected as near as possible to the same time as from sites where intervention has occurred.

Evaluation

It is expected that amphibious plants will begin to colonise the exposed wetland bed as the water level is first drawn down. Over time, the vegetation within a managed wetland should further respond to the altered water regime through changes in zonation. This should proceed under a reasonably predictable trajectory, and an illustrative example is presented in Figure 16. Although aspects of the shape of the response may change (for example the speed of establishment and maximum cover), it is expected that an initial phase of establishment and expansion will occur (the positive slope in Figure 16 for years 1–6). Following the expansion phase, areal cover can be expected to reach a relatively stable limit, although it will still reflect natural variability. This is represented in Figure 16 as the flattening of the curve in latter years of data collection.

Once the new zonation has been established, the management objectives should be reviewed and changed to reflect the new state. The objective could now switch to maintenance of condition. Part of the adaptive management review process will be assessing the required monitoring effort. For the hypothetical situation shown in Figure 16, managers are faced with an opportunity to reduce monitoring effort, potentially focussing on a current condition indicator to provide guidance on the water requirements needed to maintain the site.

An alternative situation may present where the density of amphibious plants has stabilised at a lower density than desired – for example 20% cover. In this case a review of the duration of inundation could be considered, with the aim of increasing density. Furthermore, a review of the desired water regime may be required.

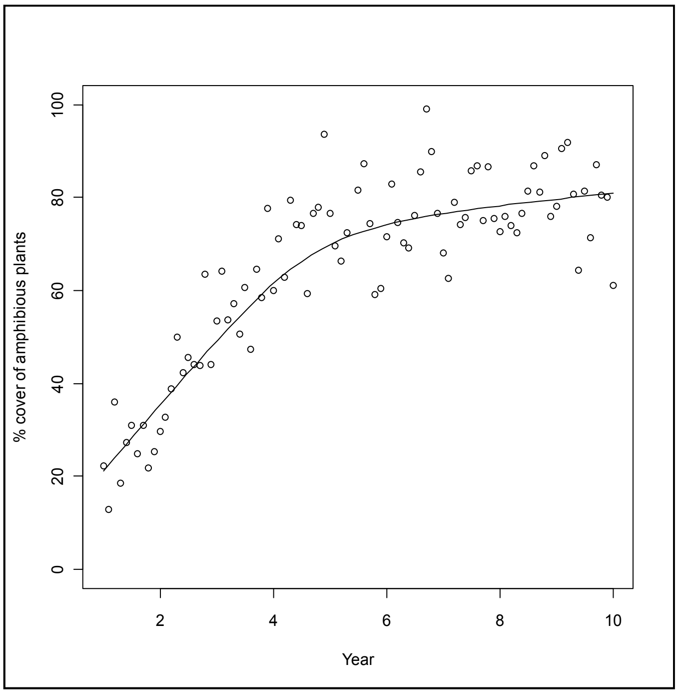

Figure 16: Theorised data representing the predicted increase in amphibious plants over time. A more or less linear increase is predicted to occur initially, followed by a flattening of the smoothed curve associated with a new equilibrium density being achieved around year 5

The statistical population of interest for littoral plant zonation in managed wetlands is all plant functional groups classed as amphibious[3] that are growing in the area of wetlands within the direct impact zones. The state variable is the mean estimated cover obtained from pooling all quadrat data for a site.

In discussing the rate at which successional change occurs, Bakker *et al*. (1996) distinguished between an incident rate observed in a given year and an overall rate (net result) over a certain period. The Bray-Curtis between subsequent years in an ordination provides one measure of the annual rate of change at the community level (Bakker *et al.* 1996; Munkittrick *et al.* 2009; Myster and Pickett 1994).

When the multivariate species (or functional group) trajectory is plotted over a five-year period for a management plan, the overall rate of change can be calculated using the same approach. Inspection of individual incidental rates against other variables (e.g. period of inundation or exposure) can help optimise future management actions. Both incidental and overall rates can be compared between sites as part of periodic meta-analyses.

[3] Plant functional group designations are assigned according to Nicol *et al*. (2010a)

Sub-indicator: vegetation condition and trend (CaT)

Methods for vegetation CaT data collection

Established CaT methods are available for floodplain tree species (Souter *et al*. 2010). This method has been adopted for use throughout the Murray-Darling Basin and should be employed for riparian tree crown condition at each treatment and control site.

Methods developed for CaT assessment of sedges are also presented for trial in the monitoring and evaluation program.

The basis of the current condition–future trend model is the partitioning of a suite of parameters into those that measure current condition and those that indicate a likely change in condition (future trend). Whilst condition is a longer term measure, future trend parameters signal either a likely improvement or decline in condition. The basis of the river red gum model is that tree condition (the amount of foliage in the tree crown) will change according to environmental conditions. However, observing a change in condition requires multiple site visits. The current condition–future trend model presumes that prior to a change in condition being observed, a range of indicators will reveal the direction of a tree's likely future change in condition. These indicators of future trend either measure the growth (new tip growth, epicormic growth) or loss of foliage (leaf die-off) or reveal the capacity of the tree to produce new foliage (reproductive capacity, mistletoe load, bark condition).

The period of most likely stress for riparian trees and sedge communities is at the end of summer as drawdown levels reach their lowest and soil stores are exhausted. Sampling should be done during this period prior to re-filling (if planned) to determine any detrimental impacts of drawdown and whether the wetland needs to be refilled if not already planned. Monitoring should be done annually whether or not a drawdown and refilling is planned or not.

Condition and trend for sedges will be collected concurrently with the amphibious vegetation zonation and composition data from the same 15 x 1 m quadrats. Quadrats for floodplain tree crown condition should be collected from within 50 m of the pool level elevation, with quadrats aligned along the edge of the wetland (Souter *et al*. 2010).

Methods and analysis of tree CaT indicators

Tree crown condition data will be collected using *The Living Murray* method (Souter *et al*. 2010). The TLM method assesses eight tree condition parameters, each of which receive an index score. For extent and density, the index scores are the midpoint percentage values for each category expressed as a number between 0 and 0.95 (Table 8). For the rest of the indicators, the values are the category scores assigned to the level of each parameter. However, the index scores are positive for indicators which show a likely future improvement in condition and negative for those which show a likely decline (Table 9 and Table 10).

The current condition-future trend (CaT) methodology (Souter and Watts unpublished) generates two indices from the tree condition data collected by the TLM method (Souter *et al*. 2010). These are condition and trend indices, which are used to determine time-series response of the vegetation and determine the likely response to different watering scenarios.

Table 8: Category scale and index score used to assess crown extent and density in the TLM free condition assessment method

Index score	Category and description	Percentage of assessable crown (for extent) and the foliated portion of the crown (for density)
0	0 – None	0 %
0.5	1 – Minimal	1-10 %
0.15	2 – Sparse	11-20 %
0.30	3 – Sparse - Medium	21-40 %
0.50	4 – Medium	41-60 %
0.70	5 – Medium - Major	61-80 %
0.85	6 – Major	81-90 %
0.95	7 – Maximum	91-100 %

Table 9: Category scale and index score for reporting positive future trend indicators (new tip growth, epicormic growth, reproduction) and negative trend indicators (mistletoe and leaf die-off) in the TLM free condition assessment method

Positive index score	Negative index score	Category and description	Definition
0	0	0 – Absent	Effect is not visible
1	-1	1 – Scarce	Effect is present within the assessable crown but not readily visible
2	-2	2 – Common	Effect is clearly visible throughout the assessable crown
3	-3	3 – Abundant	Effect dominates the appearance of the assessable crown

Table 10: Category scale and index score used to assess bark condition in the TLM tree condition assessment method

Index score	Category and description
0	0 – Intact bark
-1	1 – Minor cracking - cracks limited in number and bark still held in place
-2	2 – Moderate cracking - numerous cracks but bark still held in place
-3	3 – Extensive cracking - numerous deep cracks which are lifting the bark off the sapwood
-4	4 – No bark (long-term dead tree)

Condition is the product of the crown extent and density index scores. This gives a range of values from 0 to 0.9025. This value is range standardised so that it lies between 0 and 1 using the ranging formula:

$$y_i' = (y_i - y_{min})/(y_{max} - y_{min})$$

where y_i' is the standardised value for the condition score (between 0 and 1), y_i is the non-standardised score, y_{min} = 0 and y_{max} = 0.9025.

The 0-1 Condition score scale can be divided up into a series of condition ratings (Table 11). These divisions are based on the products of the different extent and density categories (Table 12).

Table 11: Tree condition rating as derived from the Condition score range

Condition score range	Condition rating
0-0.01	Extremely poor
0.01-0.1	Very poor
0.1-0.4	Poor
0.4-0.7	Good
>0.7	Very good

Table 12: Matrix showing tree condition scores as derived from all combinations of the product of extent and density. Red, extremely poor; orange, very poor; yellow, poor; light green, good; dark green, very good

Extent/density index scores	0	0.05	0.15	0.3	0.5	0.7	0.85	0.95
0	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
0.05	0.000	0.003	0.008	0.017	0.028	0.039	0.047	0.053
0.15	0.000	0.008	0.025	0.050	0.083	0.116	0.141	0.158
0.3	0.000	0.017	0.050	0.100	0.166	0.233	0.283	0.316
0.5	0.000	0.028	0.083	0.166	0.277	0.388	0.471	0.526
0.7	0.000	0.039	0.116	0.233	0.388	0.543	0.659	0.737
0.85	0.000	0.047	0.141	0.283	0.471	0.659	0.801	0.895
0.95	0.000	0.053	0.158	0.316	0.526	0.737	0.895	1.000

Future trend is calculated as the sum of the future trend parameters. It is range standardised using the formula:

$$y_i' = (y_i - y_{min})/(y_{max} - y_{min})$$

where y_i' is the standardised value for the condition score (between 0 and 1), y_i is the non-standardised score. Also, y_{min} = -10, which is the maximum negative future trend score a tree can receive if it had no positive future trend parameters, all negative parameters were abundant and the tree had no bark. Also, y_{max} = 9 as this is the maximum positive future trend score a tree would have received had it no negative future trend parameters and all positive future trend indicators were Abundant.

The final future trend index score is calculated by subtracting 0.526 (which occurs when y = 0 and neither positive or negative trend indicators are dominant) from the range standardised value. This is done so that the likely future direction of change is evident. A positive value indicates a likely future improvement and a negative value a decline. The lowest future trend value is -0.526 and the highest future trend value is 0.474.

Condition and future trend are depicted together. Condition is presented as the frequency of trees in a range of condition classes and describes the stand's current status. Future trend provides an indication of the likely future condition of trees in each condition class and is calculated as the mean of the future trend scores for each tree in that class. Using trees surveyed on two occasions from Yatco as an example (Figure 17) the frequency of trees in the 0.3–0.4 class has increased from the first to the second survey, whilst the frequency of trees in the lower condition classes has decreased. In the second survey only the trees in the 0 class show a negative mean trend, with no trend or a positive trend evident for all other condition classes in the second survey. The response as measured by mean future trend has increased from the first to the second surveys for the majority of condition classes.

Alternatively the frequency of trees in each of the five condition classes can be presented (Figure 18). Here there appears to be little difference in condition between the two dates. However there is a strong positive future trend response for the good and very good trees, whilst the extremely poor and poor trees show neutral future trend.

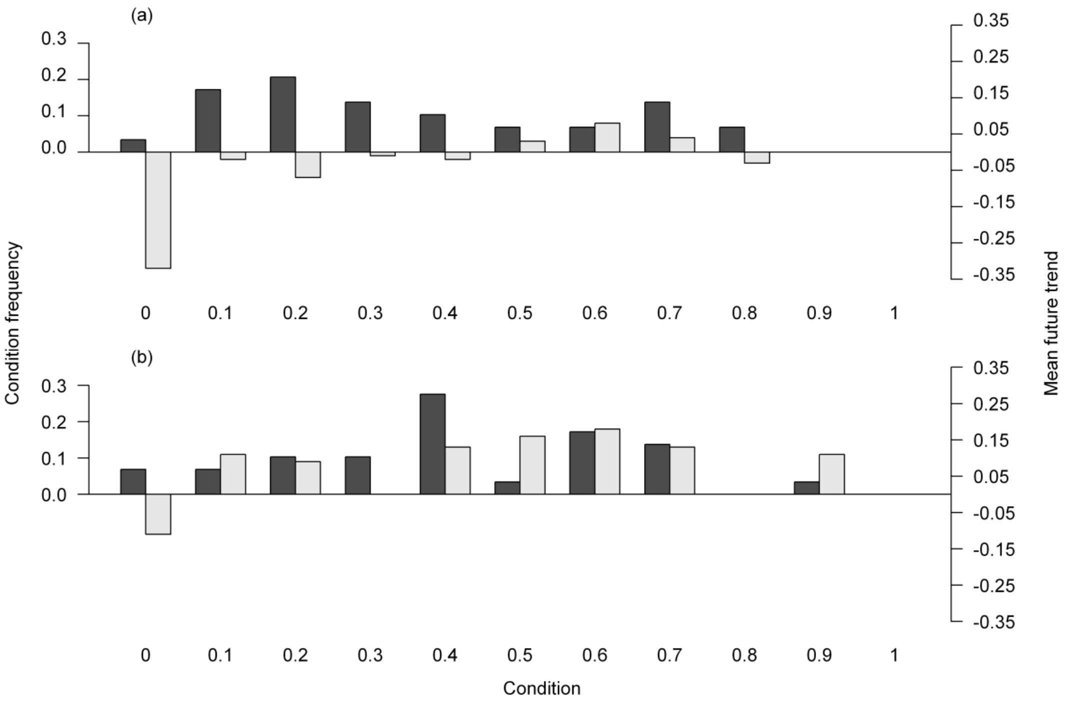

Figure 17: Current condition and future trend of river red gum at Yatco (a) 14/9/2010, (b) 12/4/2011. Dark grey bars are frequency of trees in each condition class, light grey bars are mean future trend score for trees in each condition class. The condition values depicted represent ranges (with the maximum value that of the depicted value and the minimum greater than the adjacent (lower) condition value) except for 0, which only includes 0 values. Sourced from Souter and Watts (unpublished)

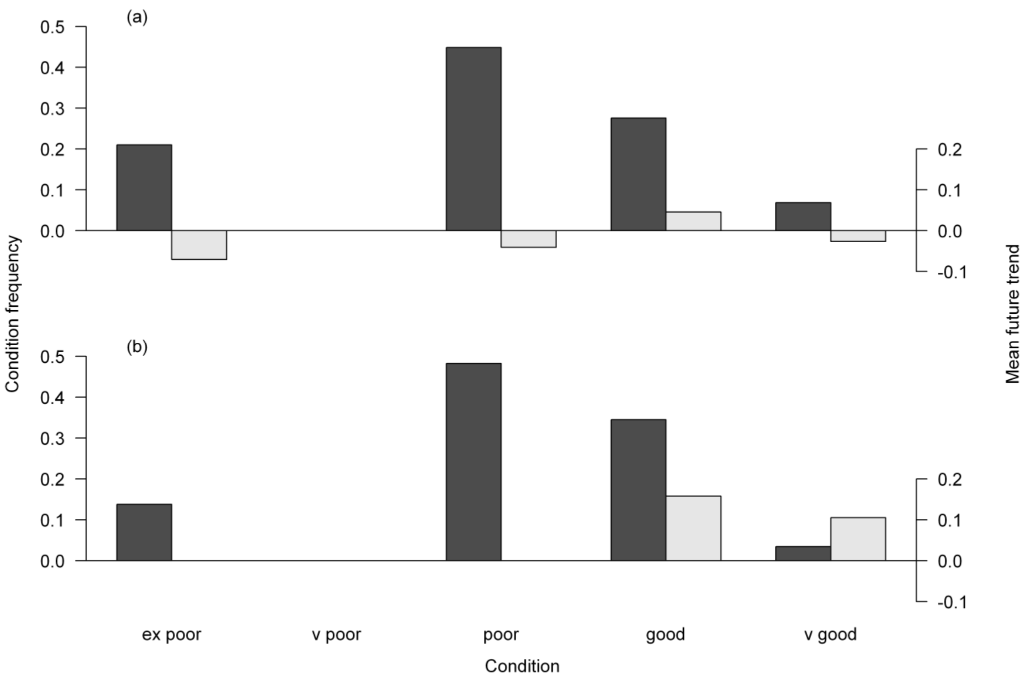

Figure 18: Current condition (five classes) and future trend of river red gum at Yatco (a) 14/9/2010, (b) 12/4/2011. Dark grey bars are frequency of trees in each condition class, light grey denote mean future trend score for trees in each condition class Sourced from Souter and Watts (unpublished)

It is of interest whether or not the condition and future trend of trees has changed over time and the size of the change in these indices. The mean ± 95% confidence interval values of the current condition and future trend indices can be estimated for each survey period and the difference between them calculated to give an effect size (Figure 19 and Figure 20, respectively). This is the change in the parameter value between two surveys. In the case of tree condition at Yatco we have a paired design as trees are repeated measures. The mean condition on 12/04/2011 (0.388) is greater than that recorded on 14/09/2010 (0.337). The mean condition of these trees in both cases would be regarded as poor. The difference between the two surveys 0.051, [-0.30, 0.132] is minor given that the condition scores range between 0–1. Also, as the 95% confidence is less than zero we cannot be confident that there is a real difference in condition between the two dates.

The difference in mean future trend between the two survey dates is 0.123, [0.071, 0.177]. As the 95% confidence interval is greater than zero, we can be confident that there was a real increase in the future trend index from the first to the second survey. This would suggest that that future trend increased over time and that condition is likely to improve.

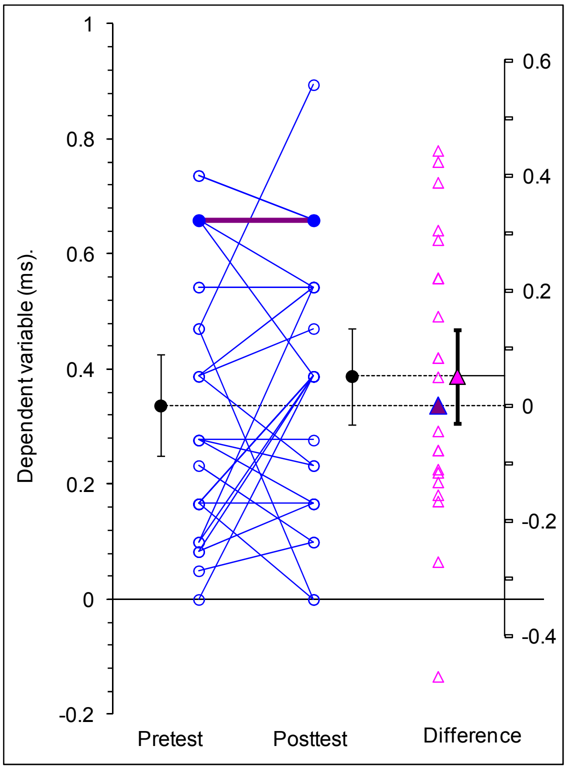

Figure 19: Mean and 95% confidence intervals of tree condition from Yatco[4]. The Pretest measure (left hand black dot with error bar) is for tree condition measured on 14/9/2010. The Posttest measure (middle black dot with error bar) is for tree condition measured on 12/04/2011. The mean paired difference is shown with its 95% CI as the closed pink triangle against a floating difference axis, whose zero is lined up with the pretest mean. The paired data (each of the 29 trees) are shown as small circles joined by lines. The differences are shown as triangles on the difference axis. An example tree is denoted by the purple line and triangle

[4] Figure created using 'Exploratory Software for Confidence Intervals' (ESCI) supporting software for Cumming (2012)

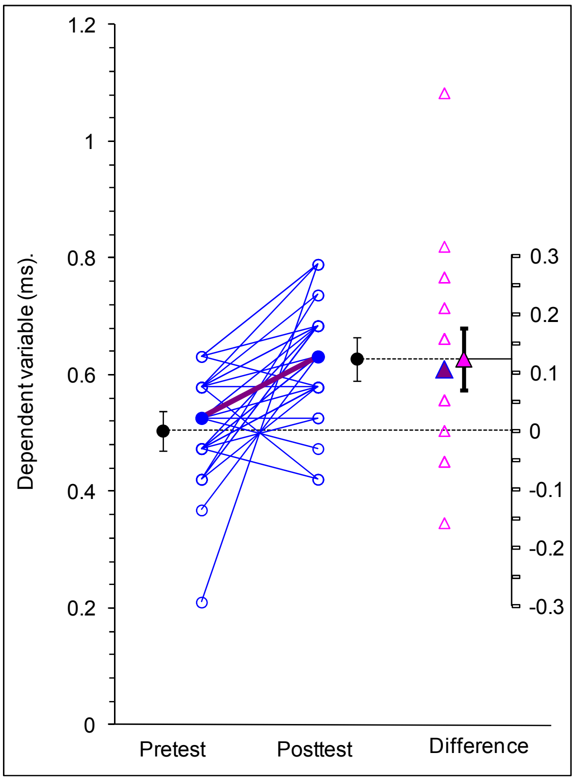

Figure 20: Mean and 95% confidence intervals of tree future trend from Yatco[5]. The Pretest measure (left hand black dot with error bar) is for future trend measured on 14/9/2010. The Posttest measure (middle black dot with error bar) is for future trend measured on 12/04/2011. The mean paired difference is shown with its 95% CI as the closed pink triangle against a floating difference axis, whose zero is lined up with the pretest mean. The paired data (each of the 29 trees) are shown as small circles joined by lines. The differences are shown as triangles on the difference axis. An example tree is denoted by the solid blue circles and purple line solid purple triangle

Methods and analysis of sedge CaT indicators

Sedge condition can be assessed using a range of parameters. Each 15 x 1 m quadrat provides a single estimate of condition measured as the frequency of occurrence in the fifteen 1 m^2 quadrats. There are three future trend parameters: new shoots (Table 13), greenness (Table 14) and reproduction (Table 15). Each of the three future trend parameters is measured at this time by assessing sedges in the entire 15 x 1 m quadrat.

A combined future trend index similar to that used for river red gum will be trialled for the sedge community. Future trend is calculated as the sum of the future trend parameters (new shoots, greenness and reproduction). It is range standardised using the formula:

$$y_i' = (y-y_{min})/(y_{max}-y_{min})$$

where y_i' is the standardised value for the condition score (between 0 and 1), y_i is the non-standardised score. Also, y_{min} = -4, which is the maximum negative future trend score a quadrat can receive if it had no positive future trend parameters and all negative

[5] Figure created using 'Exploratory Software for Confidence Intervals' (ESCI) supporting software for Cumming (2012)

parameters were *Abundant*. Also, y_{max} = 8 as this is the maximum positive future trend score a quadrat would have received had it no negative future trend parameters and all positive future trend parameters were *Abundant*.

Table 13: New shoots future trend indicator

Score	Condition class	Description
-1	Absent	No new shoots
1	Scarce	New shoots on less than 25% of plants
2	Common	New shoots or tillers on between 25-75% of plants
3	Abundant	New shoots on more than 75% of the plants

Table 14: Greenness future trend indicator

Score	Condition class	Description
-2	Largely brown	>75% of shoots brown; <25% shoots green
-1	Brown/green	<75%, >50% of shoots brown; <50%, >75% shoots green
2	Largely green	>25% of shoots brown; <75% shoots green

Table 15: Reproduction future trend indicator

Score	Condition class	Description
-1	Absent	No fruits or flowers;
1	Scarce	Fruits and flowers on less than 25% of sedges
2	Common	Fruits and flowers on between 25-75% of sedges
3	Abundant	Fruits and flowers on more than 75% of sedges

The final future trend index score is calculated by subtracting 0.333 (which occurs when y_i = 0 and neither positive or negative trend parameters are dominant) from the range standardised value. This is done so that the likely future direction of change is evident, as a positive value indicates a likely future improvement and a negative value a decline. Thus the lowest future trend value is -0.33 and the highest 0.67.

These data will be presented and interpreted in the same manner as the tree condition data. However, as this method is yet to be trialled, examples cannot be provided. In the interim, as a guide to management low future trend indicators and low sedge cover will suggest a management review is required and potential change in operation.

As this indicator is untested, initial data collection should be viewed as a pilot. It is proposed that any class into which more than 25% of samples falls should be considered in management decisions. Note that the aim of future trend indicators is only to advise where management may need to change from the planned hydrograph to avoid possible impacts at managed sites.

Sub-indicator: submerged vegetation cover

In wetlands with low turbidity (and high visibility) transects crossing wetlands, can be employed to quantitatively survey submerged vegetation (Henry and Amoros 1996). While in future this may become possible at some sites, generally in the Lower River Murray this is not likely to be feasible due to the high turbidity levels.

Turbid water makes monitoring submerged vegetation difficult, but quantitative sampling of submerged vegetation is required. Prior attempts to survey ribbon weed (*Vallisneria americana*) while investigating the effects of a weir-pool surcharge in the Lower River Murray proved unsuccessful (Souter and Walter unpublished). This was attributed to the turbidity of the water and difficulties in determining whether ribbon weed had actually grown at the survey sites or been detached from elsewhere and floated in.

Methods for data collection

This monitoring should be seen as a pilot study and only limited guidance can be given until some trials of the methods are undertaken. The review process will initially be undertaken as part of the RRP delivery and will be maintained as part of operational procedure.

Sampling will follow guidelines produced to advise the *Sustainable Rivers Audit*, which suggests that grapnels, rakes and grabs can be used for the remote sampling of habitats that are otherwise inaccessible or, due to turbidity, unobservable. The effectiveness of these methods does however depend on the plant species being surveyed (i.e. some must break off and be collected), substrate type and the presence of in-stream obstructions such as snags or boulders.

It is proposed that the sampling and abundance scoring methods presented in Rogers and Owens (1995) be adopted. This involves using three rake pulls to sample the vegetation, with abundance recorded as an ordinal density rating as shown in Table 16 (Rogers and Owens 1995).

Table 16: Density rating for recovered vegetation using rake pulls (Rogers and Owens 1995)

Recovery of species	Density rating
Teeth full on all three casts	4 (abundant)
Teeth partly full on all three casts	3 (common)
Taken on two casts	2 (occasional)
Taken on one cast	1 (rare)
No submerged vegetation	0 (bare)

This method should be used with caution, particularly where invasive macrophytes are present or suspected. Species such as *Elodea* spp. and *Egeria* spp. reproduce via fragmentation (Sainty and Jacobs 1994) and sampling may increase their spread. If suspected invasive plants are sampled, then monitoring should be extremely cautious to retrieve all vegetative fragments and obtain a positive plant identification. Furthermore, relevant authorities and future sampling teams need to be notified in order to manage and avoid risk.

Where to collect data

Owing to the destructive nature of sampling, randomisation is essential. Wetlands should be stratified according to 0.3 m depth classes and random sampling locations selected within these (for each sampling trip). The sites for sampling then need to be located in the field, which can be done using a hand-held GPS unit.

No data are available to determine an adequate replication level, so 12 samples at each depth class are recommended as a value for the pilot study, which is consistent with amphibious zonation replication. Analysis of data variability and observed effect sizes should be undertaken after 1–2 years data are available to estimate optimal rates.

When to collect data

Data should be collected during spring at least one month after refilling the wetland.

Evaluation

The statistical population for analysis are submerged plants in control and treatment wetlands that are permanently or temporarily under water. The state variables that are being estimated from sampling are the areal cover and species richness of the sampling population at wetland scale.

Theorised responses for submerged vegetation are largely speculative as they are based on only a few anecdotal observations. Data collected from wetlands with a changed water level regime will need to be analysed and the results used to better plan management actions that aim to improve the cover or diversity of submerged species. However, it is anticipated that a submerged community of modest species diversity may develop. Submerged plants are expected to colonise areas of the wetland that are maintained within euphotic and water-quality constraints.

As the submerged vegetation data are collected as an ordinal categorical variable, appropriate analytical techniques include contingency tables (to compare state variables between control and managed sites), the use of Markov models to generate transition probabilities and ordinal regression. In the latter case, continuous variables might include turbidity, depth or duration of drawdown. If abundance scoring proves unreliable, the information can be reduced to presence-absence at each sampling point and used to estimate cover (as a frequency of occurrence) and richness.

Indicator: fish communities

Methods for fish community data collection

Whilst a wide range of sampling methods are available (e.g. Murphy and Willis 1996), a standard technique has been developed for the RRP based on proven methods used in the Lower River Murray. It is essential that methods are standardised within and among wetlands (including previous baseline surveys, as far as practical) to maintain a level of consistency in these data and to continue to allow local- and regional-scale analysis of the wetland fish community.

Sampling methods are based on previous baseline surveys as reported in SKM (2006b), Smith and Fleer (2006) and Aldridge *et al*. (2012). However, these projects all used slightly different gear types, which will need to be accounted for when comparing with RRP monitoring data. The development of methods different from those used for *The Living Murray* and *Sustainable Rivers Audit* was unavoidable as the aims of the RRP differ and because of its focus on wetlands.

Organisations carrying out fish monitoring are responsible for ensuring that they have all appropriate permits (e.g. ethics, research, etc.) prior to sampling. Furthermore, they must adhere to all relevant guidelines when carrying out monitoring, including the treatment of alien species. As requirements may differ between organisations they are not detailed here.

Standard gear types and deployment

For the RRP the aim of fish monitoring is to obtain fish assemblage composition, condition and size distribution data. These descriptors, particularly size distribution, will be dependent on the sampling methods (i.e. net type, mesh size) employed, hence the necessity of a standardised methodology. A variety of gear types is suggested to maximise the range of species and sizes sampled. The gear selection has been based on those options likely to adequately describe each wetland's fish community within realistic sampling effort and logistical limitations.

To standardise fishing effort as much as possible, a standard composite set of fishing gear is recommended, consisting of three fyke nets, one gill net and three box traps. A minimum of three sets are required per wetland. The number of sets should increase at large wetlands, though not beyond a realistic sampling effort. Seine netting may be used where wetland morphology allows. Apart from seines all nets are set overnight with setting and hauling times recorded to calculate total soak time and allow Catch Per Unit Effort (CPUE) calculations. In addition the spatial coordinates and description of the site including the habitat type and substrate, depth at the point of sampling and estimated average depth are recorded. Water-quality measurements should be co-located with fish sampling sites.

The standard composite set of fishing equipment is:

- 3 small singled-winged fyke nets:
    - 7 m wing, 0.7 m drop, with 0.7 m high 'D' and 3 compartments (funnels) 6 hoops with 6 mm mesh without exclusion grills.
    - Set in littoral or complex habitats (e.g. macrophytes, snags) at a 45–90° angle to the wetland shore.

- 1 multi-panel gill net:
    - o 15 m long, 3 panels per net including 45 mm, 75 mm and 115 mm inch stretched mesh.
    - o Set in deep or open water habitats adjacent to the three fyke nets.
- 3 bait traps:
    - o 400 mm in length with two square ends (250 x 250 mm) with 70 mm openings at each end.
    - o Brown in colour.
    - o Targeted to complex habitat where other nets cannot sample (e.g. complex snags, macrophytes) near the three fyke nets.
- Seine nets may be employed where it is believed species diversity may be increased:
    - o Only employed where substrate is firm and free from snags.

No quantitative assessment of these data should be attempted as sampling effort cannot be adequately quantified.

Sample processing

Captured fish are identified to species level (with the exception of the unresolved carp gudgeon species complex: Bertozzi *et al*. 2000) and counted to determine relative abundance. From each standard net set, the catch from each net type should be pooled and data recorded on a per net type basis. In the event that a within net type sample is so large that processing time will cause fish death, sub-sampling should occur (Smith and Fleer 2006). In this situation catches are divided into halves or thirds by placing the catch into storage containers to approximately equal volumes and randomly selecting containers to be assessed. The selected sub-sample for processing can be processed as usual, while the other proportion of the sample is inspected for rare species and returned to the water without delay. When undertaken the sub-sample fraction size should be noted on field datasheets and the adjusted total abundance recorded to reflect the total sample size.

From each of the net types within a standard set a subsample of each species will need to be measured and weighed. As random sampling is time consuming and places undue stress on the captured fish, the first 50 fish of each species collected per net type within each set are measured (i.e. within one standard net set, up to 50 from the three fyke nets, up to 50 from the one gill net and up to 50 from the three box traps for each species). Remaining fish should be visually inspected to identify any native rare/threatened species and released alive. Alien species should be humanely destroyed.

Length measurements are standardised to the morphology of the different species (Fork Length for fork-tailed species, Total Length for round- or truncate-tailed species). Total Length is measured from the most anterior (front end) point of the fish to the tip of the longest caudal fin (tail) ray. Fork Length is measured from the most anterior (front end) point of the fish the end of the middle (inside) caudal fin rays.

Where time allows, fish weight should be measured in tandem with length measurements to enable biomass, length-weight regression and condition indices to be calculated. Weight measurements for live fish can be made with either spring or electronic scales, but field measurements are often of low precision because of factors such as varying fish surface wetness (Gutreuter and Krzoska 1994), the relative importance of which depends on body size. Ideally scale accuracy should be within ±1% of body weight (Wege and Anderson 1978), meaning a 1 g fish (typical of small-bodied species found in the Lower Murray) would require a scale with an accuracy of at least 0.01 g. A minimum guideline for scale capacity is that the fish weight should be more than 10% of the instrument range full scale[6] (Gutreuter and Krzoska 1994). It may be necessary to have multiple scales available to cover the range of body weights (e.g. 1 g to 5000 g) to required precision.

Where to collect fish data

Sampling of all habitat types is required to ensure the most reliable description of the wetland fish community. At least one standard composite net set should be employed per habitat type, e.g. one set on sandy shore, one set on reedy shore, one set on rocky shore. Site establishment will involve determining which habitat types exist and where best to sample each of these. When the full range of potential sample sites has been determined, those to be sampled should be randomly chosen. Field personnel may need to adjust the pre-determined sampling site following site inspection (e.g. adjusting location to maintain nets in appropriate water depth). Once established, it is recommended that the same locations are sampled each time. Should site character change, either through wetland management or natural processes, the site should be moved to a location of equivalent original habitat. Table 4.6 in SKM (2006b) provides guidance on which gear is best suited to different habitat types and is summarised below for recommended RRP gear types (Table 17).

Table 17: Gear types for different habitat types and sampling effort

Gear type	Habitat sampled
Fyke net	Muddy, deep or heavily snagged or vegetated areas
Bait trap	Fringing or overhanging riparian vegetation, grasses, snags, submerged vegetation
Gill net	Open or deep water areas
Seine net	Relatively firm substrate, delicate submerged vegetation or fringing grasses May be used solely in open water

Note: Adapted from baseline survey fishing methods (Table 4.6 SKM 2006b)

[6] Note that electronic scales typically have a range of measurement scales which are auto-selected by the instrument

The methods adopted for baseline data collection provide for all catch data to be pooled, creating a single sample if effort is strictly adhered to. Comparison of the variability in data after 1–2 years of sampling is recommended to ensure this level of replication is of adequate precision for the analyses suggested.

When to collect fish data

In the Lower River Murray wetland fish are sampled in the spring, to describe the juvenile and adult community and autumn, to detect recruitment. Samples are not collected in winter as fish are less active and unlikely to be caught, thus leading to an underestimate of population size. Nor are fish sampled in summer as it adds little additional information to that collected in spring and autumn. As the differences in fish catch between season (e.g. winter vs. spring), are expected to be greater than those due to management, management should be treated as a covariate to season. Thus season rather than management actions should determine the timing of sampling.

Ideally sampling would occur immediately prior to, or after, the closing of a wetland regulator (±2 weeks) to describe the captive fish assemblage. Then immediately prior to reopening the wetland the fish community should be re-sampled so that any changes since isolation from the river can be quantified. However this only holds if wetland closing and opening occur in either spring or autumn. Sampling in winter and the likely reduced estimates of fish presence and abundance can lead to erroneous results. For example if a wetland is closed in winter and reopened in autumn the initial sample is likely to underestimate the fish fauna and could lead to the conclusion that wetland isolation has been beneficial to the fish community, when in fact the change was only due to seasonal differences in detectability. Conversely a closure in spring and reopening in winter could lead to the opposite conclusion, that wetland closure caused the fish community to decline when it did not. Whilst sampling in summer is less likely to cause such problems, for consistency spring and autumn sampling is preferred. When wetlands are managed in either winter or summer the fauna should be sampled as early, or late, in the following or preceding season as possible. For example if a wetland is closed in winter, fish should be sampled early in spring to characterise the captive fauna, if the wetland is to be opened in winter it should be sampled as late as possible in autumn to detect the impact of management.

Sampling after wetland closing has a further purpose to identify the presence of threatened species that may govern management actions. If rare or threatened species are identified, relevant authorities should be notified as soon as possible and management reviewed and modified as required (e.g. a programmed complete-dry may be changed to a partial dry).

All wetlands should be sampled twice a year in autumn and spring to assess the effect of management on the fish community. It is likely that once sufficient data are collected the frequency of data collection at control sites and sites that will be completely dried can be reduced. Once fish community dynamics at control sites are quantified, less sampling may be required. Sites that will be completely dried should only need to be sampled either immediately prior to, or after, wetland closure to detect threatened species. After this, no further sampling should be required given that the fate of the trapped fish is certain.

Evaluation

Increases in habitat complexity from development of amphibious and submerged vegetation should provide ideal nursery conditions for native fish. Increases in overall productivity on re-filling drawn-down systems should also create good conditions for zooplankton populations, providing food for larvae, hence supporting recruitment (Schiller and Harris 2001). This should be reflected in the increased use of managed wetlands by native fish over permanent controls. Breeding success (e.g. young-of year) and condition are other variables which may demonstrate a response, although condition indices are better suited to targeted investigations. Hence a lag period between the initiation of managed regimes and an observable change in fish populations is anticipated.

For fish populations, these benefits must be traded off against decreases in water-quality as wetlands are drawn down and evapo-concentrate. Deteriorating water-quality is likely to be a problem as there will be no connectivity with the main channel and thus fish cannot escape. The analysis of fish data needs to provide information that guides management to produce conditions leading to the increased abundance and diversity of native species.

The sampling population for managed wetlands is the fish that are isolated from the main channel once regulators are closed. These populations are compared with fish in permanent wetlands, or temporary wetlands during natural floods or weir-pool raisings. Each survey will constitute a single sample and to calculate a mean, sites will need to be used as replicates. Alternatively a mean value for a wetland can be calculated over time for comparison against a baseline value (but caution needs to be exercised as the parameter value is expected to change over time as a result of management).

There are many variables that could be investigated but some examples are given below. State variables for fish analysis include:

- community structure
- community composition
- relative species abundance and biomass
- native and alien species richness or condition indices
- ratio of native to alien species
- population structure
- age/size classes (e.g. comparing the abundance of young of the year to adults for different species)
- condition (in particular weight as a function of length).

Fish diversity in the Lower River Murray is modest and hence species richness may not change greatly. Species richness is estimated from the pooled data from a wetland. As data from all gear types are pooled to estimate richness at each site, only one sample is obtained from each survey. This single estimate of richness can be compared against baseline data with multiple years used to estimate the mean value and variability. Where different sampling methods have been used, adjustments will need to be made for CPUE. The mean difference between treatment and control sites can be examined using a t-test ANOVA type model, or equivalent non-parametric tests, or by determining an effect size. Given the low number of species within these wetlands, species richness may not be a suitable measure. If this is the case, more detailed indices should be considered that account for the community structure. Local examples of these have been used (e.g. Davies and Jackson (2006) and McNeil *et al.*

(2011)). However, the choice of index requires careful consideration and perhaps even investigation to determine the usefulness of the index.

Changes in relative abundance and other community-structure indicators may provide more reliable evidence of preferential habitat in managed systems. The community structure should be analysed through appropriate multivariate statistical techniques. This could involve analysing spatial and temporal differences in community composition (e.g. ordinations, see Table 1).

Both population structure and condition are also likely to provide evidence of differential habitat suitability between managed and permanently flooded wetlands. Population structure is the proportion of the individuals which fall within a given age or size class (Anderson and Neumann 1996). The relative distribution between species or sites is a measure of success and survivorship. Figure 21 shows the type of comparison that can be drawn between autumn and spring age-structure using length data collected with the sampling methods proscribed for this program. The top row indicates a clear young of the year class for flathead gudgeon in autumn, with a mode of around 40 mm. By spring, the growth in the cohort is evident with a shift in the mode to around 60 mm. These values can be compared between treatments (e.g. drawdown-refill cycle duration) and between managed and control sites. The presentation of such graphs should also display the size ranges of recruits and mature fish. Note that to analyse data in this manner a consistent length-frequency histogram bin size is required (Anderson and Neuman 1996). For example, as the 20 mm class could be measured from 20–29 mm, or from 15–24 mm; a consistent approach is required. Whilst Kolmogorov–Smirnov tests can be applied, they should be interpreted with caution as although they detect differences, they are subjective to where differences exist.

When length-weight data are collected they can be used to determine fish condition. The definition of condition is fish body weight as a function of length, which generally follows a power function:

$$W = a.L^b$$

where W is weight, L is length and a and b are parameters to be fitted, b typically taking a value near to 3.

Logarithmic transformation results in a linear form that can be used to solve for a and b:

$$\log(W) = \log(a) + b.\log(L)$$

There are a great number of uses of length and weight data:

- Calculation of indices of condition (e.g. Fultons condition factor – Anderson and Neuman 1996; Froese 2006) can be used to compare different species and determine if management differentially impacts different fish species.
- Comparison of the value of the model parameters when fitted to fish data for managed and control wetlands allows for differences between sites to be determined.

- Calculation of empirical length-to-weight relationships allows biomass to be estimated, which can be converted to areal units for comparison between sites and thus management.

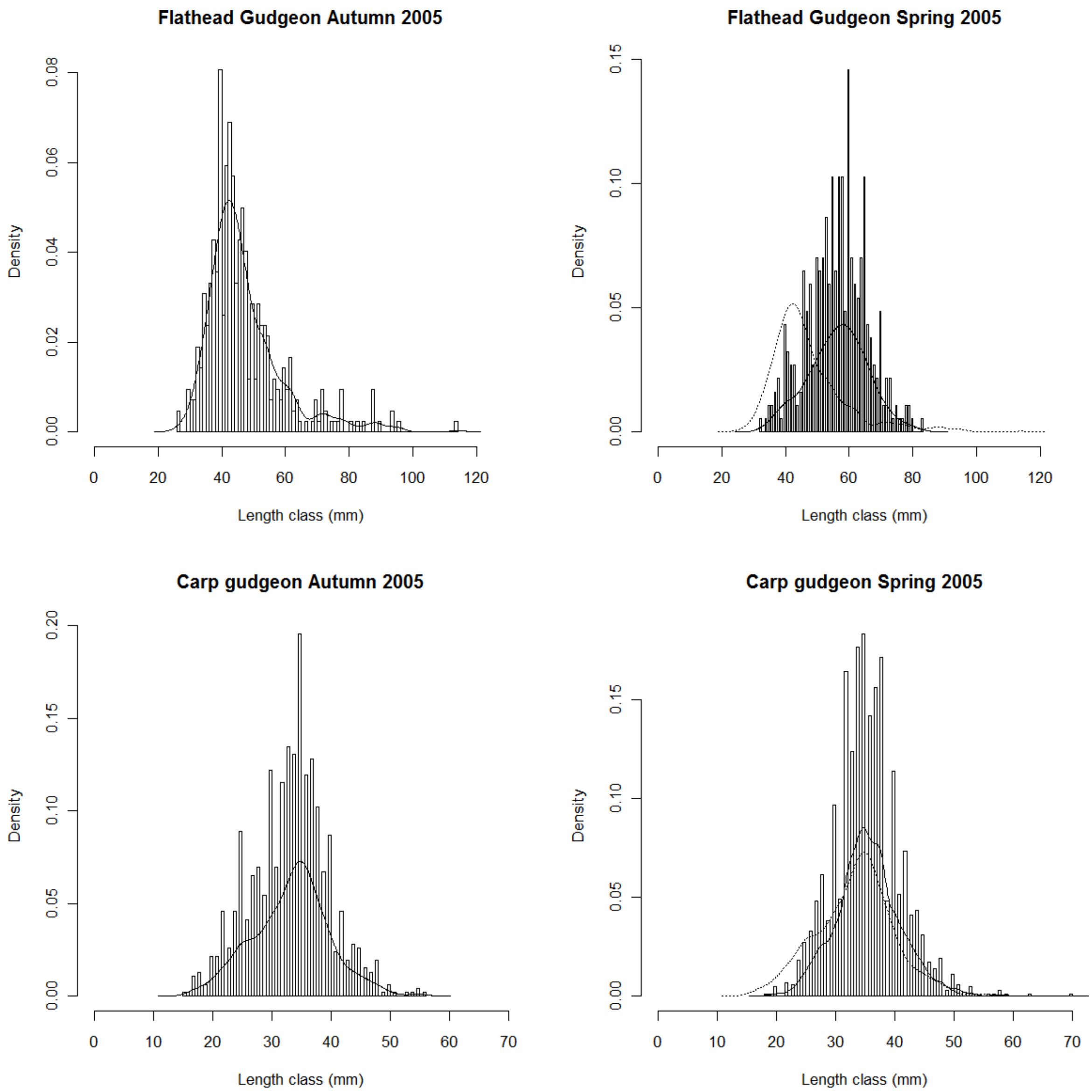

Figure 21: Autumn (left) and spring (right) length histograms from baseline surveys – all sites. Top row, flathead gudgeon, bottom row carp gudgeon complex (kernal density shown as a line – spring samples also have density for autumn superimposed for comparison)

Indicator: water-bird communities

Methods for water-bird data collection

Methods should follow those adopted for baseline surveys as presented in Tucker *et al*. (2004) or SKM (2006b).

Previous sampling effort has been determined by expert opinion, with the aim to sample the water-bird population at the site. Rather than stratify sites based on habitat, emphasis has been on survey points or methods that provide visibility over a defined area. This introduces an element of subjectivity which may affect measurement precision where different surveyors adopt different survey points or combinations of methods. It also precludes power analysis and limits the statistical inference which can be drawn from the data within and between sites. Similar to fish monitoring, a standardised approach is suggested.

Four different sub-sampling approaches have been commonly used:

- fixed area search - standing at one point in the wetland or from a vantage point outside of the wetland, and counting the birds within a defined area. Recommended for mudflats
- transect - undertaken on the larger more complex wetlands where visibility is difficult due to the terrain or vegetation cover. These are not of a fixed length, but adjusted as required to adequately cover a target area depending on the size and nature of the wetland
- wetland circumnavigation – undertaken when wetlands are small enough to easily walk around
- call and response recordings – the detection of cryptic species in dense habitat such as reeds or sedges requires the use of recordings. These should be played for 3-5 minutes with a range of species and their different calls included. If the identity of any responders cannot be determined with certainty, a recording should be made and referred to local ornithologists. An abundance estimate should be made according to the same ordinal scale used for frogs (*Indicator: frog communities* section).

To allow comparisons to be made, it is recommended that wetland circumnavigation be replaced by a combination of fixed area search or transects. Allowing for minimum replication requirements, the number of survey points should be based on the size, shape, complexity, access and visibility of the wetland complex. Select survey sites that give maximum coverage of a water-body without any overlap in the field of vision. Spend at least 20 minutes at each survey point.

Data to be recorded are species identity[7], estimated abundance and behavioural and habitat variables, providing greater information than other monitoring programs such as the Biological Survey of South Australia (Owens 2000). These are summarised on the field sheets

[7] Identification shall be at species level using nomenclature as defined by Christidis and Boles (2008).

presented in Waanders and Kuchel (2011), which can be adapted for the RRP. Special attention should be given to the activity that the bird is engaged in (in particular foraging or nesting), age class (if possible) and breeding activity (mating, nest building, nest/eggs, nest/young, dependent young).

Prior to a water-bird survey, it is recommended that surveyors use the online Birds Australia Atlas (http://www.birdsaustralia.com.au/our-projects/atlas-birdata.html/) and site species list from baseline surveys to familiarise themselves with the species likely to be present at each site.

Where to collect water-bird data

Prior to site visits, the wetland bathymetry at maximum planned drawdown should be determined to indicate likely variations in habitat availability over current pool level conditions. Areas where water depth is likely to be less than 100 mm may support waders. It should be possible to predict via desktop study where these areas will be located. Digital elevation models should help surveyors identify areas with an optimal field of view, although this will need to be verified on site.

In the field, site location is confirmed with consideration given to the visibility of the main habitat types for water-birds. The location of each site is then recorded using a GPS and the field of view documented with compass bearings and mapped. A detailed description of each site is required to create a formal sampling protocol for each site, thus ensuring repeatability.

Pre-existing data were analysed to determine the replication required (Appendix 2 – Critical effect size determination). For this, all observations during a survey were pooled to generate a wetland estimate of water-bird-species richness. This was necessary as prior work has not surveyed replicate sub-samples. Power analysis of paired samples indicates that six sub-samples should provide adequate precision to detect the difference in species richness identified as a critical effect size.

Although the different methods confuse the issue of sub-sampling, it is recommended that a combination of at least six fixed-area searches, transects and call/response surveys are undertaken at each site. In a large wetland complex additional sub-samples can be taken as required.

When to collect water-bird data

Each treatment and control site should be visited once during spring to detect breeding activity and at the end of summer to observe any mudflat use as wetlands reach full drawdown. Surveys should commence at first light.

Evaluation

Riverine condition indicators involving bird populations are of most value at or above the reach scale (Bryce *et al.* 2002; Vaughan *et al.* 2007). For RRP monitoring, spatial scales of interest range from a managed wetland complex to an entire weir-pool and beyond. Water-birds will not only provide an indication of the success of changes in wetland management in meeting management objectives, but also whether watering plans need immediate adjustment. As such, water-bird monitoring needs to assess bird behaviour (nesting) as well as species composition and abundance. The response of individual species that are described in wetland management objectives, will also need to be considered.

The sampling population is water-bird species within wetlands of all water regime classes. State variables at this scale include species richness, abundance, functional group structure, community level composition and diversity. The abundance of individual bird species will also be of interest.

Water level manipulation is expected to provide high quality bird habitat (e.g. submerged plants, fish, and invertebrates) in both saline and freshwater wetlands. Comparison of species composition and abundance between managed wetlands and controls provides a wetland scale indicator.

Species richness of sites can be compared with those observed during baseline studies (a repeated measures assessment). The species richness critical effect size was estimated from paired site-comparisons pre-treatment, where the range of natural variation between mean minimum and maximum observed diversity (plus confidence intervals) at a given site was less than six species. A change in mean species richness between either control and treatment wetlands, or pre/post water regime manipulation, will provide evidence of an environmental benefit.

Comparison can be drawn between control and treatment wetlands for functional group composition, diversity or other univariate indices and multivariate community composition.

Indicator: frog communities

Methods for frog population sampling

Baseline data (collected during a drought period) indicates that regional frog species richness is low and that autumn sampling is unlikely to produce a reliable estimate of species richness (Appendix 2 – Critical effect size determination).

Similar to fish and water-birds, the aim of frog sampling is to determine the diversity and abundance of the frog fauna present at each wetland. Call monitoring is the most effective means to sample frog populations, however, calling behaviour is highly variable and more than one technique is necessary (Mason and Hillyard 2011; Tucker *et al*. 2004; SKM 2006b):

- Call recording and recognition (listening for and recording vocalisations by calling males of each species).
- Call response – pre-recorded calls played on site to stimulate a response from male frogs present but not calling.
- Active searching – scanning fringes of water-body with small spotlight over a standard area.

At each sample site 15 minutes will be spent listening for and recording the calling males of each frog species heard (SKM 2006b). Where possible identification should be made in the field. If call identification is equivocal, recordings should be verified by an expert herpetologist. Call recordings need to be of high quality and made using a digital recorder and external microphone. Opportunities to apply emerging technologies to identify and quantify frog abundance, such as automatic audio file analysis (e.g. Song Scope ™ (Waddle *et al*. 2009) and SoundID ™ (Boucher *et al*. 2012), should be explored.

Call playback methods should also be employed at each site to further increase the potential to detect frog species. This method involves broadcasting frog-species calls to the surrounding area for two to five minutes per call. The number of responding males will then be recorded. Whilst call response was found to be unsuccessful by Mason and Hillyard (2011) for Southern Bell Frog, it is recommended that this method is continued to assess its suitability for other species.

As frogs are difficult to count in high numbers (Mason and Hillyard 2011), the identity of species and an ordinal scale abundance class should be recorded (Mason and Hillyard 2011) (Table 18). Data are recorded using either datasheets or PDA's and include information on habitat type, weather conditions, time of recording and any recent management changes. Any incidental observations regarding frogs (including frog spawn) that are made during other biological surveys need to be recorded.

Table 18: Abundance scores for frog surveys

Abundance score	Abundance (number of frogs)
0	0
1	1
2	2–9
3	10–50
4	>50

Note: Adapted from Mason and Hillyard (2011)

Where to collect frog data

Frog communities should be sampled along wetland margins at five sampling points per wetland complex. Suitable frog habitat includes areas that provide shelter for frogs and access to food and water. Shelter is often associated with vegetation, but the form and type varies between species (Wassens 2011). Consequently, it is important that different types of habitat within a wetland are examined. Whilst frogs are typically associated with habitat complexity, this may include trees (including dead), emergent macrophytes, submerged macrophytes and loose bark.

The wetland should be first explored to identify suitable frog habitat that is accessible. Within these areas, five sites should be randomly selected with stratification of habitat types to ensure all different habitat types are surveyed. To avoid overlap of sampling sites maintain a minimum distance of 400 m of wetland edge between sample sites. Where recently inundated emergent vegetation is present, this should be the location for at least two replicates. Different sites should be selected each sampling time, with consistent habitat stratification maintained. Record the spatial coordinates and precision at each site using a hand-held GPS unit.

When to collect frog data

Baseline survey data suggests autumn surveys have limited value (Appendix 2 – Critical effect size determination). Sampling should occur in spring to mid-summer each year. Initially, repeat surveys may be conducted to determine the optimal sampling period and measure any change in community composition over spring to mid-summer. It is recommended that sampling is conducted after dusk during calm, warm to hot nights. Ideally this would be when rain is forecast or after recent rain and not when ambient temperatures fall below 12° C or during strong winds. Frogs should be surveyed for 10–15 minutes at each sampling point.

Evaluation

Riverine condition indicators involving frog populations are of most value at or above the reach scale. For RRP monitoring, spatial scales of interest range from a managed wetland complex to an entire weir-pool. At the wetland scale, frog populations will not only provide an indication of the success of changes in wetland management, but also whether watering plans require immediate review.

The sampling population is frog species within wetlands of all water regime classes. State variables are species richness and abundance and community level composition and diversity. Water level manipulation is expected to provide high quality frog habitat (e.g. submerged plants) and thus an increase in species richness and abundance at treatment wetlands is expected.

Site species richness can be compared to baseline studies (a repeated measures assessment). The species richness critical effect size has been estimated from paired site comparisons pre-treatment. The range of natural variation between mean minimum and maximum observed diversity (plus confidence intervals) at a site was less than six species. A change in mean species richness between either control and treatment wetlands, or pre/post water regime manipulation, will provide evidence of an environmental benefit.

Comparison can be drawn between control and treatment wetlands' diversity or other univariate indices and multivariate community composition. Comparison of species composition between managed wetlands and controls provides a wetland scale indicator.

Weir-pool monitoring

Indicator: inundation extent

It is important to evaluate whether or not the weir-pool raising inundated the area of floodplain (or wetlands) predicted by hydraulic modelling. This will develop confidence in model predictions or identify areas where the models may need improvement. This information is also needed so that the ecological response to the weir raising can be compared against areas that remained dry.

Three approaches may be used to map inundation:

- Comprehensive assessment of inundation via field survey during raising or lowering
- Hydraulic modelling, with opportunistic point sampling of inundation to verify modelled extents (this can involve either a GPS or even simple translation of observed levels to aerial photographs, which can later be digitised)
- Remote sensing, supported by ground-truthing.

Remote methods supported by field measurements have the greatest potential as they can survey the entire affected area. If field survey is preferred, these could be conducted via boat or vehicles with inundation areas recorded with GPS. Any data collected should be stored in a digital spatial database. Initially, weir-pool manipulation will rely on modelled predictions of inundation extent to locate suitable sampling sites. Over time, it is anticipated that remote sensing data collected from previous events may be employed along with modelling to map the predicted extent of inundation for each anticipated event.

Remote sensing methods are most suited to determine the extent of the area inundated via a weir-pool manipulation. A range of methods can be used to determine inundation extent including radar altimeters, synthetic aperture radar or spectral imaging (Schumann *et al*. 2009; Smith 1997). However, the relative merits of these different systems for assessing flood inundation along the Lower River Murray have not been assessed. Such an assessment is required before a suitable remote sensing technique is recommended for monitoring inundation extent.

Indicator: vegetation

Weir-pool raising is expected to deliver a range of benefits to floodplain vegetation. This will occur as the weirs are used to raise the river level and inundate sections of the floodplain that would have remained dry during normal operations. Previous monitoring assessed small scale changes in the response of individual organisms (e.g. river red gum (*Eucalyptus camaldulensis*) and lignum (*Muehlenbeckia florulenta*)) or species/genera of interest (e.g. Cumbingi (*Typha domingensis*), common reed (*Phragmites australis*), river clubrush (*Schoenoplectus validus*), sedges (*Bolboschoenus* spp.), Lippia (*Phyla canescens*) and common sneeze weed (*Centipeda cunninghamii*)). Changes assessed were cover, condition, flowering and spatial distribution. An assessment of the floodplain vegetation community composition was also made (Souter and Walter unpublished). These methods were adopted as the species selected were thought likely to respond to weir-pool manipulation as they were either abundant species that provide important habitat (e.g. lignum) or were representative of broader groups (e.g. common sneezeweed represented floodplain herbs). However, not all plants at the surveyed sites responded rapidly to the single weir-pool surcharge in 2005–06, probably because the vegetation at these sites was well-established and resilient to change.

Determining the area over which vegetation responded

The monitoring for the 2005 inundation was designed to detect a range of specific changes in vegetation condition brought about by weir-pool manipulation. These were related to the response of individual species as representatives of larger groups. However, this level of monitoring is not able to report on responses at the reach scale, the scale at which weir-pool manipulation has its greatest effect.

In order to determine the area over which vegetation responded, the spatial extent of the floodplain inundation caused by weir-pool raising needs to be determined. Following this, a measure of vegetation response appropriate to the weir-reach scale is required. Remote sensing methods are suitable as they are able to collect data across large areas. Multi-spectral images of the earth's surface can be taken by a variety of orbiting satellites. The information contained within these images has been extensively used to assess vegetation condition from crops to forests across a wide geographical area. A widely used approach is to combine spectral information from multiple bands into a composite value known as a spectral vegetation index (Cohen and Goward 2004). The Normalised Difference Vegetation Index (NDVI) is widely used and provides an estimate of vegetation greenness or biomass per pixel (Goward *et al*. 1985). NDVI has also been related to leaf and plant area index (PAI) (Cunningham *et al*. 2007; Huemmrich and Goward 1997) and the fraction of absorbed photosynthetically active radiation (fPAR) (Veroustraete *et al*. 1996).

Vegetation indices have been used to model forest condition parameters such as basal area and stem density in forests as the spectral response of a forest is indirectly determined by these and other structural features (Ingram *et al*. 2005). Here the spectral response and forest structural factors are correlated so that broad-scale predictions of forest structure can be made from the remote sensing data (Lu *et al*. 2004). Vegetation indices such as NDVI have also been used in land cover change detection. This generally tends to be directed towards assessing dramatic changes in land cover caused by activities such as deforestation. The basic premise behind using remote sensing data for change detection is that land cover change results in changes in radiance values that are larger than radiance changes due to other factors such as differences in atmospheric conditions, soil moisture and sun angles (Mas 1999).

The simplest way to measure the response of vegetation to weir-pool raising would be to use a Before-After-Control-Impact (BACI) design. The difference in NDVI values recorded in the portion of the floodplain that was inundated both before and after weir-pool raising would be assessed. This difference would be compared with the change prior to, and after raising, in areas that remained dry; as would the difference between with the inundated area and the area that remained dry. Such a simple comparison would assume that a positive change in NDVI value is sufficient to demonstrate a positive response in the floodplain vegetation. In making this assessment all that is required is the delineation of inundated and dry areas and collection of data that could be used to assess NDVI. A range of satellites are able to provide this information at different costs, resolutions and frequencies of collection (e.g. Landsat, MODIS, SPOT, RapidEye). These data should be collected at the same time every year in order to reduce the effect of differences in sun angle and vegetation phenology (Singh 1989). Overcoming the problems of difference in atmospheric conditions, soil moisture and sun angles is addressed to some extent by comparison to the areas of floodplain that have not been inundated by the weir-pool raising. A pilot study is required to determine the most suitable methods of assessing inundation extent and collecting NDVI data.

The response of various floodplain vegetation communities can also be detected by spatially delineating them and separately assessing their response. Analysing data at this level of resolution provides more detailed information on the effects of weir-pool manipulation and also controls for the likely differences in spectral reflectance between the communities. Whilst there are a variety of communities present on the floodplain, some will be impacted more by weir-pool manipulation than others. Inundation modelling has estimated the area of the different vegetation types along the Lower River Murray that will be affected by weir-pool raising (Table 19; DEWNR 2012b). As black box woodland, river red gum woodland, lignum shrubland, samphire shrubland and terrestrial dry shrubland all cover over 100 ha of at least one weir-pool, the response of these vegetation associations should be monitored. Techniques using remote sensing that measure the stand condition of black box and river red gum are available (Cunningham *et al*. 2007, see below). However, methods for lignum, samphire and terrestrial dry shrubland need to be developed. Emergent sedgeland and flood dependent grassland are less affected by weir-pool raising but the large combined area from Lock 1 to the SA border may still warrant monitoring. Mallee shrubland, river coobah woodland and tea tree woodland are only minor components and are not recommended for monitoring.

In river red gum forests along the mid River Murray, Cunningham *et al*. (2007) used Landsat-derived NDVI values to predict stand condition, which was based on assessments of percentage live basal area, plant area index and crown vigour. This approach can be adopted to assess the effect of weir-pool manipulation on both river red gum and black box communities and will allow comparison with data collected at TLM icon sites.

Methods used to assess the shrub and sedgelands require development. Of the metrics used by Cunningham *et al*. (2007) to assess river red gum condition, the suitability of Plant Area Index (PAI) and vigour should be assessed over a suitably-sized plot and correlated to satellite derived NDVI data for the shrub and sedgeland vegetation types. Plant area index can be measured using a probe such as the Li-Cor LAI-2200 Plant canopy analyser (http://www.licor.com/env/products/leaf_area/LAI-2200/), with multiple readings taken across the quadrat to provide an estimate of PAI for each sample. More traditional methods such as estimating cover using a point quadrat approach may also be used. However, the sampling effort required in making these estimates using manual methods is likely to be prohibitive. For lignum, vigour could be measured across the entire plot, most likely by visual assessment.

Techniques for the emergent sedgeland, flood dependent grassland, samphire shrubland and terrestrial dry shrubland, which are all low growing, are thus likely to consist of the same measure of plant area index. As changes in the floristic composition of the sedgeland, grassland and terrestrial dry shrubland may occur due to inundation, an assessment of community composition should be made. For consistency with the rest of the program, multiple 15 m x 1 m quadrats should be surveyed at each site.

Table 19: Maximum estimated total area inundated (ha) of the different vegetation groups for each of the six weir reaches along the Lower River Murray in South Australia due to weir-pool raising. Data sourced from DEWNR (2012b)

	Total area inundated (ha)						
Vegetation group	Lock 1: 10 GL/day, 0.106 m rise	Lock 2: 10 GL/day, 0.7 m rise	Lock 3: 30 GL/day, 0.59 m rise	Lock 4: 10 GL/day, 0.114 m rise	Lock 5: 10 GL/day, 0.5 m rise	Lock 6: 10 GL/day, 0.5 m rise	Total area
Black box woodland	30.2	32.1	135	103.3	33.5	15	349.1
Emergent sedgeland	7.1	7.4	45.3	44.6	36.0	9.5	149.9
Flood dependent grassland	71.7	3.5	33.6	65.0	0.8	2.5	177.1
Lignum shrubland	125.1	39.2	683	258.2	53.2	66	1224.7
Mallee shrubland	0.1	0.1	0.0	0.9	0.0	0	1.1
River coobah woodland	0.7	1.4	1.8	0.0	0.3	18	22.2
River red gum woodland	358.9	83.0	582	501.7	389.7	244	2159.3
Samphire shrublands	47.1	71.4	633	407.2	354.2	50	1562.9
Tea tree woodland	0.0	0.0	0.3	16.9	0.1	1.7	19
Terrestrial dry shrublands	56.8	4.0	153	174.3	68.2	28	484.3
TOTAL VEGETATION AREA	698	242	2267	1572	936	434	6149

Nature of the vegetation response

The short-term response of the vegetation can be measured using the methods outlined above, with NDVI providing an indication of photosynthetic response. Time-series data obtained from multiple images will provide an understanding of the length of time that the initial response has lasted. On-ground monitoring of tree condition can pick up finer scale features that respond in the short-term, such as flowering, and delineate whether new growth in the tree crown is a result of epicormic or new tip growth. Long-term changes in vegetation structure may also occur with the expansion or contraction in the spatial extent and density of forest, woodland, shrubland and herbland areas. This can be calculated by using remotely sensed imagery to delineate the extent of the various communities, changes in which can be tracked over time.

Site selection for ground-truthing

A series of recommendations were made as a result of the monitoring carried out to assess the 2005 weir-pool surcharge. The selection of more responsive sites was recommended so that responses to a single surcharge can be better quantified. A range of criteria were developed for such sites, the main one being that they are disconnected from the river at pool level, but would be inundated during a surcharge.

The following attributes were assigned to potential sites:

- Not directly connected to the river at pool level, but likely to be periodically inundated by local rainfall or groundwater intrusions
- Connecting sill level (the height at which river and wetland become connected) that allows inundation during surcharge (identified using modelled inundation maps)
- A basin or depression capable of holding water as opposed to an open floodplain
- No significant salt intrusion
- Presence of a healthy flood-dependent or non-salt-affected community
- Easily accessible by vehicle or boat.

These sites should be monitored using the protocols outlined for wetland monitoring in this report. However, sampling frequency will need to be adjusted to meet the objectives of a particular weir-pool manipulation. There is merit in continuing with the previous littoral vegetation monitoring at the established sites along the Lock 5–7 reach so that long-term changes caused by weir manipulation can be established. The trees monitored at Chowilla used to report on the 2005 weir-pool manipulation (Souter and Walter unpublished) continue to be monitored as a part of the Chowilla Icon site and thus do not need to be monitored as a part of the RRP. It is highly unlikely that the lignum shrubs previously monitored would still be marked, given that the lignum were last surveyed in 2008 and the disturbance caused by overbank flows in 2011 and 2012. Further to this, lignum may be more suited to being monitored at the stand scale using remote sensing techniques.

The selection of ground-truthing sites should also be stratified so that there are an equal number of control sites and treatment sites. Initial oversampling is recommended so that representative sites can be chosen for continuous monitoring. This should be done to ensure that no anomalous sites are used for long-term monitoring and that the full gradient in vegetation cover across each of the vegetation association groups is sampled.

Similarly, as discussed in the *Stratification within wetlands* section for wetlands, the sampling design for weir-pool indicators will also rely on stratification of similar habitat for sub-sampling. The floodplain unit will be characterised not only by a range of wetland types, but also different vegetation associations. The interaction of these units and the extent of inundation will need to be considered when planning weir-pool manipulation monitoring.

How the weir-pools will be manipulated has yet to be established, but will determine the placement of sites within the affected area. The greatest effect is achieved by raising the weir-pools to their maximum height. If this is done routinely then sites within the area of maximum inundation can be randomly chosen. However, if the weir-pools are operated at a range of heights, sites should be stratified so that the area inundated by each of the operating levels is sampled.

The number of sites within each weir-pool can be reduced once pilot data are collected and analysed to ensure that adequate replication is achieved to calibrate NDVI data. Data from the Chowilla TLM site can be used to determine how many red gum and black box woodland sites are required, whilst sampling effort for the other vegetation types is unknown. The level of sampling intensity can also be determined by the proportion of the area affected by weir-pool raising. Using Lock 1 as an example, the bulk of the sampling effort would go towards river red gum woodlands and lignum shrublands, with less attention paid to the other shrublands and the black box woodland. However, until data points are collected and their costs and benefits assessed, such decisions cannot be made.

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Appendix 1 – Souter and Watts (unpublished)

Title: A current condition-future trend model to inform conservation management

Keywords: Tree condition; Current condition-future trend; Trend; Murray-Darling Basin; Floodplain; *Eucalyptus camaldulensis*

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Summary

Conservation managers frequently make decisions on the basis of limited information. This information is often obtained from only a single observation, which may provide little indication of future trend or the likely effects of management. When data is limited, measuring parameters that measure both current condition and future trend can improve decision making. As an example, a conceptual model in which visually assessed parameters measured from the Australian floodplain eucalypt, river red gum (*Eucalyptus camaldulensis*) are partitioned into current condition and future trend indices. Current condition is the product of tree crown extent and density and describes the current status of a tree. Future trend indicators aim to predict changes in condition and are either responsive to short-term environmental changes or directly affect tree health. Future trend parameters are either positive (new-tip growth, reproduction and epicormic growth) or negative (leaf die-off, insect herbivory causing leaf damage, mistletoe infestation and bark condition). These indicators are weighted and combined with a ranging formula to indicate the direction and strength of the trees' likely future change in condition. For a particular stand the frequency of trees in each condition class is graphically presented along with the mean future trend score for all trees of that condition. A comparison between river red gum on two floodplains on the Lower River Murray in South Australia demonstrates how the method differentiates between stands with different management histories. The current condition-future trend model allows managers to compare the likely response of competing tree stands when allocating environmental water. The current condition-future trend method is applicable to a wide range of situations where variables that can express both current condition and predict future trend can be measured, providing an aid to management where data is limited.

Keywords Tree condition; Current condition-future trend; Trend; Murray-Darling Basin; Floodplain; *Eucalyptus camaldulensis*

Introduction

Conservation managers often have to make prompt decisions on the basis of limited data (Brook *et al*. 2002). As management decisions have future consequences, managers need techniques that can make reliable forecasts, which improves decision making (Clark *et al*. 2001). Thus there is a need for techniques that, with limited data, can quickly provide useful predictions. A range of models and approaches have been used for ecological forecasting. For example, population viability analysis (PVA) provides accurate predictions that assist in endangered species management (Brook *et al*. 2000). Whilst PVA provides rigorous methodologies it can require data that may not be available (Akçakaya and Sjögren-Gulve 2000). For example, PVA state transition occupancy models require data from two or more yearly inventories (Akçakaya and Sjögren-Gulve 2000). However, even with limited data, the use of techniques such as PVA can still contribute to decision making (Brook *et al*. 2002). Other methods such as probit analysis can require extensive time series data (e.g. Albertson *et al*. 2009). Simulation models may be used to help predict the consequences of management actions (e.g. Glasscock *et al*. 2005), but to be effective they require sufficient data and an understanding of system dynamics with which to construct the models. Where data is abundant, data assimilation tools are being developed to undertake quantitative ecological forecasting (Luo *et al*. 2011).

Whilst the aforementioned techniques have proven their worth, they generally require extensive data sets. These data sets often need to have been collected on multiple occasions. For example, when managing the condition of long lived species such as trees, long-term datasets can be used to make predictions (e.g. Dobbertin and Brang 2001). However, suitable long-term datasets are rare and many management decisions need to be made in their absence. Thus techniques that can use data collected from one observation to predict future condition would be of considerable value to conservation managers. Added to this, managers often have limited resources and must trade off which environmental assets receive attention, and those which remain unmanaged. A system that assesses current condition and indicates trend from a single assessment has the potential to improve management decisions in the absence of long-term data.

The rapid decline in floodplain forest condition along the Lower River Murray in south-eastern Australia presented a case where rapid decision making needed to be made with very limited data and where suitable techniques were lacking. In addressing this problem we have developed the current condition-future trend approach to conservation decision making.

On the floodplain of the Lower River Murray in South Australia, large numbers of river red gum (*Eucalyptus camaldulensis*) are suffering the combined effects of over allocation of water for human use, drought and soil salinisation (Jolly 1996). Reduced flood frequency and water availability has caused widespread tree mortality and loss of tree condition (Jolly *et al*. 1993; Overton *et al*. 2006). A range of visually assessed tree crown parameters are routinely used to monitor forest condition. These parameters are usually presented and analysed separately (e.g. Lorenz *et al*. 2008) or combined into a single tree condition metric (e.g. Busotti *et al*. 2002). Both approaches require a temporal data series before trends in condition can be determined. These approaches are unsuited to situations where temporal data series are unavailable and immediate management decisions need to be made.

The amount of water available for environmental flows in the Lower River Murray is limited and sound decisions on its application are required. For example in 2006–2007 only 1% of the total flow (16.9 Gl out of 1470 Gl) crossing the South Australian border was available for environmental use. At this time the scarce environmental water was used to fill wetlands and floodplain deflation basins fringed by water stressed trees. For many of these sites long-term tree condition data sets were lacking. However, with many more potential watering sites than water available to fill them, choices were needed as to where environmental water would generate the best outcome. Added to this, the time between receiving notice that environmental water was available and its delivery was often short. Thus decisions on where to deliver water had to be made quickly once an allocation had been received. Here the need for a suitable decision support tool became evident. For these floodplain trees we developed a measure of current tree condition and an index that provides a prediction of future changes in condition. As an illustration, we use this method to assess stands of trees on two contrasting floodplains.

Methods

Current condition-future trend model

The basis of the current condition-future trend model is the partitioning of a suite of parameters into those that measure current condition and those that indicate a likely change in condition: future trend. Whilst condition is a longer term measure, future trend parameters signal either a likely improvement or decline in condition. The first step in producing a current condition-future trend model is the development of a conceptual model which partitions parameters into those that measure current condition and those that predict future trend. The basis of the river red gum model is that tree condition – the amount of foliage in the tree crown (Souter *et al*. 2010) – will change according to environmental conditions. However, observing a change in condition requires multiple site visits. The current condition-future trend model presumes that prior to a change in condition being observed a range of indicators will reveal the direction of a trees likely future change in condition (Figure A1.1). These indicators of future trend either measure the growth (new tip growth, epicormic growth) or loss of foliage (leaf die-off, insect herbivory) or reveal the capacity of the tree to produce new foliage (reproductive capacity, mistletoe load, bark condition). Our choice of indicators and evidence of their suitability is detailed below.

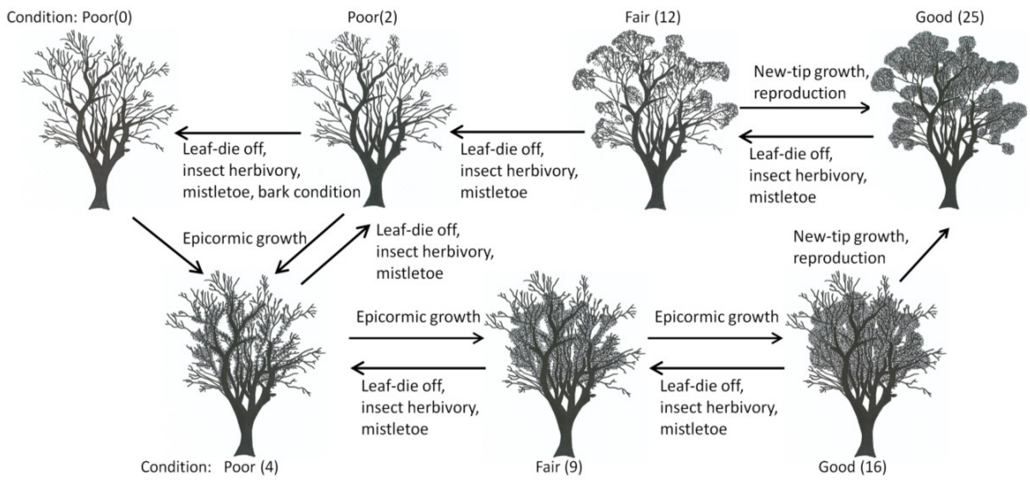

Figure A1.1: Conceptual model of the relationship between current condition and future trend parameters for river red gum. Condition class is presented in brackets after the condition rating

Current condition. Red gum condition is assessed according to two visually assessed measures of crown foliation: crown extent and density. Crown extent is the amount of foliage on the outer crown edge and density is the crown foliage density (Souter *et al*. 2010). Both parameters are compared against the tree if it possessed a fully foliated assessable crown (UN/ECE 2006). Crown extent and density are assessed as one of five classes (*Minimal* (1-10%), *Sparse* (11-25%), *Medium* (26-75%), *Major* (76-90%), *Maximum* (91-100%)) according to a conceptual model of crown decline and recovery (Souter *et al.* 2010).

Future trend. Seven parameters are used to assess future trend, three of which indicate a likely improvement in condition, whilst four indicate a likely decline. We based the selection of parameters on information from the literature and extensive field observations.

Three attributes indicate a likely future improvement in condition: new-tip growth; reproduction (buds, flowers and seeds) and epicormic growth. In response to favourable conditions (i.e. sufficient water in the right season) a tree in good condition will grow new shoots from the peripheral tips of the tree branches. Conversely, a tree in poor condition seldom produces such growth.

A tree in good condition will typically – in the right season and given favorable environmental conditions – produce buds, flowers and seeds. As tree condition declines so does its ability to reproduce. Stressed river red gums have fewer stunted buds and a slower rate of bud development (George 2004). They also show reduced flowering, both in relative volume and number of trees, and produce fewer seeds (George 2004). Such a tree is unlikely to have the energy to direct towards foliage growth. Measuring reproductive status is confounded by seasonality and the cyclical nature of bud crop development. In the Lower River Murray river red gum form buds in summer (January – February) and flower in the following spring – early summer (September – December) (George 2004; Jensen *et al*. 2007). Mature fruit may be retained for up to 24 months before being shed (Jensen *et al*. 2007). However, as the majority of river red gum in the Lower River Murray display a biennial cycle of flowering (Jensen *et al*. 2007) the lack of, or reduced flowering, may be due to this biennial cycle rather than stress. To mitigate these confounding influences, reproductive behaviour is recorded as the combined relative abundance of buds, flowers and/or fruit.

Epicormic growth is the sprouting of new shoots from the main trunk or primary (and less commonly secondary) branches of the tree. Whilst epicormic growth is produced by a tree under physiological stress it has been included as an indicator of improving condition. The growth of new leaves is a positive response to favourable environmental conditions (Cunningham *et al*. 2007), leading to an improvement in condition.

Four attributes indicate a likely future decline in condition: leaf die-off, insect herbivory causing leaf damage, mistletoe infestation and bark condition. In response to dry conditions, river red gum shed leaves to reduce leaf area and hence water demand and heat load (Gibson *et al*. 1994; Roberts 2001). Before leaves are shed they brown off and die. The leaf die-off parameter refers to the initial browning of these leaves and transfers to a loss in condition as extent and density decline when the dead leaves fall from the tree.

Insect herbivory is a cause of dieback in eucalypts (Lowman and Heatwole 1992) and forty-nine phytophagous insects have been identified from river red gum canopies at Gulpa Island State Forest (Stone and Bacon 1994). Insects such as *Uraba lugens* (gumleaf skeletoniser) (Dalton 1990) and *Doratifera* spp. (cup moths) (CSIRO 2004) have been known to cause considerable defoliation in eucalypts. Eucalypts may either suffer insect attack because they are already in poor condition (Lowman and Heatwole 1992), whilst the loss of leaf biomass (Stone and Bacon 1995) will reduce condition via defoliation.

Across Australia, river red gum are host to thirteen species of mistletoe (Downey 1998). Infestations are often localised and trees already stressed by drought or insect attack may be more susceptible (CSIRO 2004). Severe mistletoe infestation can cause tree death (Dalton 1990).

Highly water stressed red gum have vertical cracks in the bark of the trunk and major support branches which exposes the heartwood. River red gum bark cracks under severe water stress and occurs when trees have either very few or no leaves.

We measured future trend parameters on a three category scale after UN/ECE (2006) as either: *Absent or Scarce*, effect is not seen in a cursory examination; *Common*, effect is clearly visible or *Abundant*, effect dominates the appearance of the tree. Cracked bark was assessed as either present or absent.

Assessment

We gave the five crown extent and density classes scores of 0-5 such that both *Minimal* extent and density score 1 and *Maximal* extent and density score 5 (Souter *et al.* 2010), trees with no leaves score 0. All the future trend parameters trees assessed as *Absent or Scarce* received a score of 0. For positive future trend parameters trees assessed as *Common* received a score of 1, and *Abundant* 2. Negative trend parameters, except bark condition, when *Common* scored -1, and *Abundant* -2. As very few trees with cracked bark seem to recover upon watering, trees with cracked bark scored -2, otherwise they scored 0.

We measured condition as the product of the crown extent and density scores (Range 0-25). Future trend was calculated by summing the future trend parameters and then using the ranging formula: $y_i' = (y_i - y_{min})/(y_{max} - y_{min})$. Here y_i' is a value between 0 and 1, and $y_{min} = -8$, which was the maximum negative future trend score a tree can receive if it had no positive future trend parameters and all negative parameters were *Abundant*. The term $y_{max} = 14$ was the maximum positive future trend score a tree would have received had it no negative future trend parameters and all positive future trend parameters were *Abundant*.

In order to see the likely future direction of change 0.57, which occurs when $y_i' = 0$, was subtracted from the value of trend for each condition class, in which case neither positive nor negative trend parameters were dominant.

Condition and future trend are depicted together (Figure A1.2). Condition is presented as the frequency of trees in each of the 25 condition classes and describes the stands' current status. Trees in condition classes 0-5 are considered 'poor'; those in classes 6-15 'fair'; and 16-25 'good'. Future trend provides an indication of the likely future condition of trees in each condition class and is calculated as the mean of the future trend scores for each tree in that class.

A Practical Example

Thirty six river red gum were surveyed on Rillie's (34° 23'S, 140° 35'E) and thirty on Clark's (34° 21'S, 140° 34'E) floodplains downstream of Lock 4 on the Lower River Murray in South Australia on 1 November 2007. Trees on Rillie's floodplain have never received environmental water. On Clark's floodplain thirteen trees were either inundated or located within 15 m of a deflation basin filled by an environmental water allocation in winter 2005 and spring 2006. Seventeen trees were located beyond the influence (>15 m *cf.* Bacon *et al*. 1993) of this water.

In order to compare between floodplains, three current condition-future trend plots are presented (Figure A1.2). Each plot depicts the frequency of trees in each condition class (0-16) present on each floodplain. The future trend indicators are depicted adjacent to each condition bar to show the direction and strength of these trees' likely future trend.

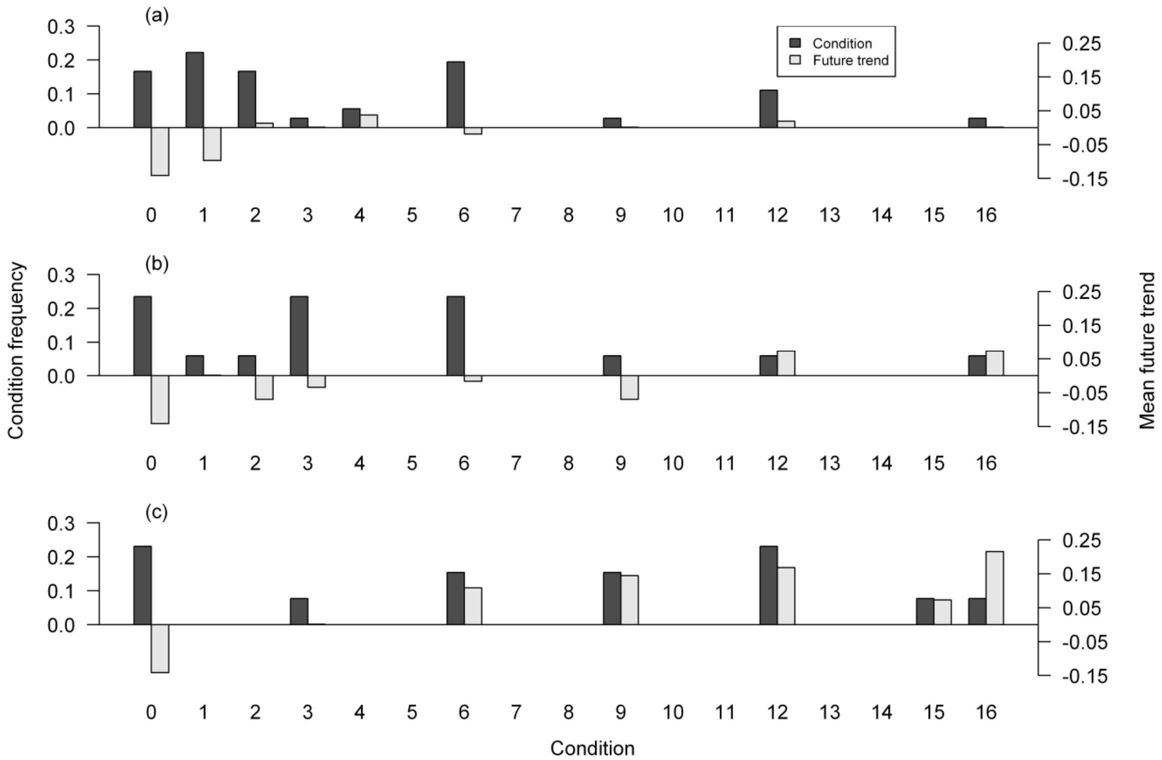

Figure A1.2: River red gum condition and future trend at (a) Rillie's and (b) Clark's – unwatered and (c) Clark's – watered floodplains

Results

On Rillie's floodplain, river red gums were observed in nine condition classes (Figure A1.2), with the majority of trees in poor (<5) condition. The future trend of the trees in differing condition varied. Those in better condition showed a neutral future trend and would be expected not to change condition in the near future if environmental conditions remained the same.

The majority of unwatered trees on Clark's floodplain were in poor or fair condition (Figure A1.2). With the exception of trees in condition class 1, which were only slightly positive, all trees of condition class 9 and below had negative future trend, largely due to insect damage. Trees in the two highest condition classes (12 and 16) had positive mean future trend due to reproductive activity. This contrasts with watered trees, most of which were in fair to good condition and showed positive future trend due to epicormic growth and reproduction (Figure A1.2). Poor trees in class 3 showed minimal positive future trend. Over time it would be expected that if conditions remained favourable watered trees showing positive trend would shift to higher classes as epicormic growth filled out the crown. The lack of response for defoliated trees with strong negative trend suggests they were dead at the time of sampling.

Discussion

The current condition-future trend model assists managers in making informed decisions with limited data. The strength of this method is that it indicates trend from a single survey. The floodplain tree example shows how a range of visually assessed parameters can summarise current condition and provide an assessment of future trend in a stand of trees. Presented graphically, this information is easily interpreted and provides an aid to management. By using information from the literature and extensive field experience, our example shows how the model can be constructed with very limited data. In situations where the literature is limited, expert opinion alone can be used. Even with very limited knowledge, the process of model building is beneficial as it: clarifies assumptions, integrates knowledge and provides a structure for explicit and rigorous reasoning (Akçakaya and Sjögren-Gulve 2000). This alone should assist in decision making.

Whilst the condition-future trend approach arose out of the need to make decisions from limited data, it is also applicable where data is more plentiful. Even though the river red gum model has been developed from the literature and extensive field experience, its application to Lower River Murray floodplain trees shows how the current condition-future trend method can be used to differentiate between sites and guide management decisions. However, in any application of the method validation that the positive future trend indicators lead to improved condition (and negative indicators a decline) needs to be verified as further data is collected. This process is helped as the structure of the model makes it amenable to hypothesis testing. Examples from the floodplain tree example are: "Did the presence of flowers predict an improvement in either crown extent or density?" and if so "Was there any difference in the response depending upon the level of flowering?" The results of these tests can then be used to refine the model. Further improvements can also be made through optimising the weights applied to each of the future trend parameters, as some may be better predictors of future condition than others.

In its current form the river red gum model future trend estimate provides a prognosis of future conditions according to Luo *et al*.'s (2011) definition. They define prognosis as a more subjective approach to forecasting based on the scientist's view of the state of the system. This is an apt description as the model has been developed from the literature and expert opinion. As monitoring data is collected the model can be modified to provide a quantitative result and become more predictive, or under a Bayesian framework provide forecasts by making probabilistic statements on future condition.

Our comparison of three stands of river red gum demonstrates the utility of the approach. The frequency of trees in the different condition classes was similar between the three stands which ranged from poor to good condition. The juxtaposition of predicted trend next to these condition frequencies differentiates between stands, with the high positive predicted trend of the watered trees differentiating them from the two stands that remained dry. This positive response was expected due to the watering and suggests that their condition will improve as foliage growth fills the tree crown, increasing both extent and density. These graphs show that whilst floodplains can have similar distributions of tree condition, some can be improving whilst others decline or remain neutral. When allocating water, the declining stands of trees can be prioritised for attention over stands in better current condition that also show likely improvements in future condition.

Whilst we have demonstrated the use of the current condition-future trend method in the context of managing tree condition, its principles are transferrable to other systems. It is readily transferrable to the TLM method of assessing tree condition which uses similar parameters (Souter *et al.* 2010). For other systems, all that is required is that a range of variables are measured separately and that some can be used to define condition with others able to predict future changes in condition. Such an approach may be useful when using state-and-transition models (Suding *et al*. 2004), which are commonly employed for vegetation communities (Stringham *et al*. 2003; Spooner and Alcock 2006). An assessment of the current state (condition) could be determined using measures of community composition, whilst the likelihood of a future change in state (predicted trend) could be predicted by the appearance or decline of certain species associated with each state.

The use of the current condition-future trend model enables sites to be visited once, their condition established and their likely trend forecast. This provides an aid to adaptive management, which in our floodplain tree example is deciding where to allocate scarce environmental water. The method described can conveniently be modified for use in any environmental assessment protocol where parameters that predict future changes in condition can be identified.

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Appendix 2 – Critical effect size determination

Critical effect sizes have been determined from baseline data. This is the only data available for this analysis and was collected during a drought period. Consequently, it is recommended that calculations of critical effect sizes are repeated as monitoring data becomes available. It also needs to be noted that different methods may have been has across sites and times providing a potential, but unavoidable, source of error. Revisiting calculations of critical effect size once appropriate data becomes available will also address this issue.

Native fish baseline data analysis for critical effect sizes

Data availability and processing

Twenty four sites were repeat sampled in spring 2003 and summer 2004 baseline surveys, with a further 13 sites sampled during autumn and spring 2005. Full data analysis is available in SKM (2006a) and Smith and Fleer (2007) respectively. Data on native fish species were selected from these survey databases for analysis.

Data were sorted into paired responses and estimates for overall, site and seasonal mean values and data spread were calculated. The distribution of mean differences was calculated (Cumming 2012) and used to determine the range of likely variations at sites. A critical effect size for management intervention is defined as the difference between pre and post values at a managed site; or between a managed site and controls greater than that predicted by baseline data to occur with no intervention at 95% confidence.

Native species richness

Overall mean native species richness was 6.08 [95% CI: 5.46-6.70] species. A small degree of seasonal variation was implied by the data, with slightly higher median richness observed in autumn (Figure A2.1), although this difference was not statistically significant (Kruskal-Wallis chi squared = 3.8; $p = 0.15$). Sampling effort was not even across seasons (autumn = 13; spring = 37; summer = 24) and hence no conclusions regarding optimal sampling season can be drawn from this analysis.

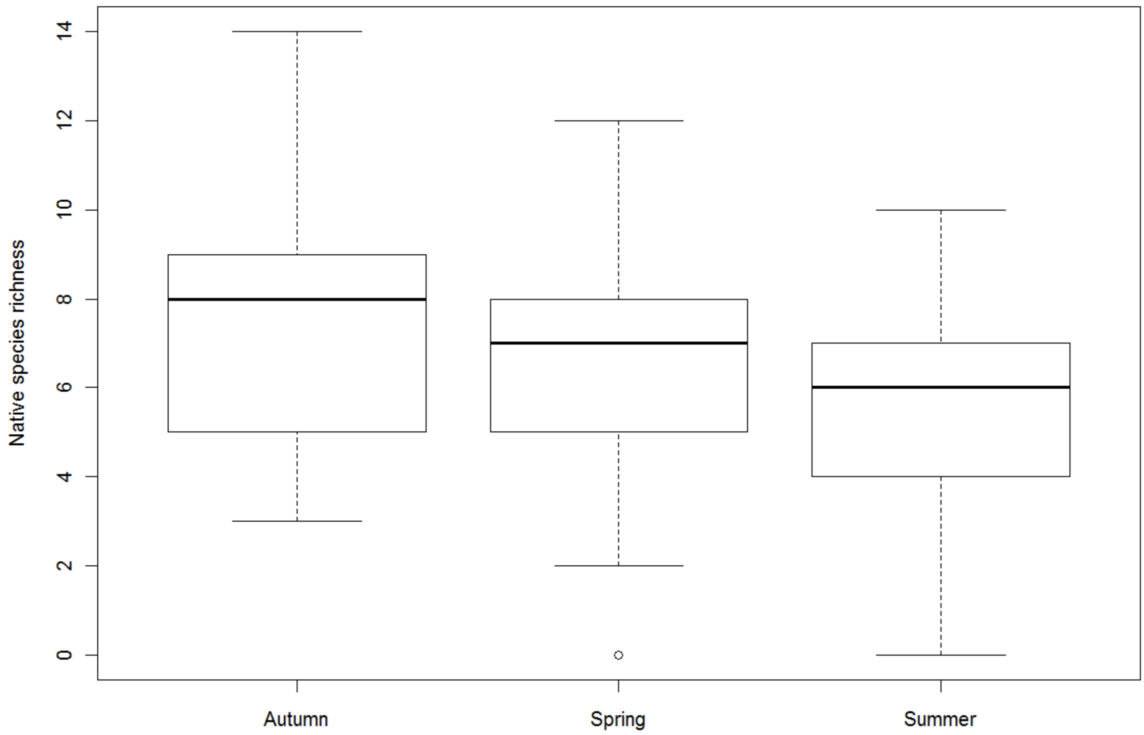

Figure A2.1: Box and whisker plots for native fish species richness as a function of season sampled (n = 74 samples at 37 sites). Data are from baseline surveys all gear types combined. The top and bottom of the box represent 25th and 75th percentiles, the middle band is the median and dots represent outliers

Season was disregarded in effect size analysis, and the maximum (6.8 [95% CI: 5.8-7.5]) and minimum (5.5 [95% CI: 4.6–6.4]) species richness recorded at each site was used. The mean difference in the 37 repeat surveys (Figure A2.2) was 1.2 [95% CI: 0.85-1.53] species.

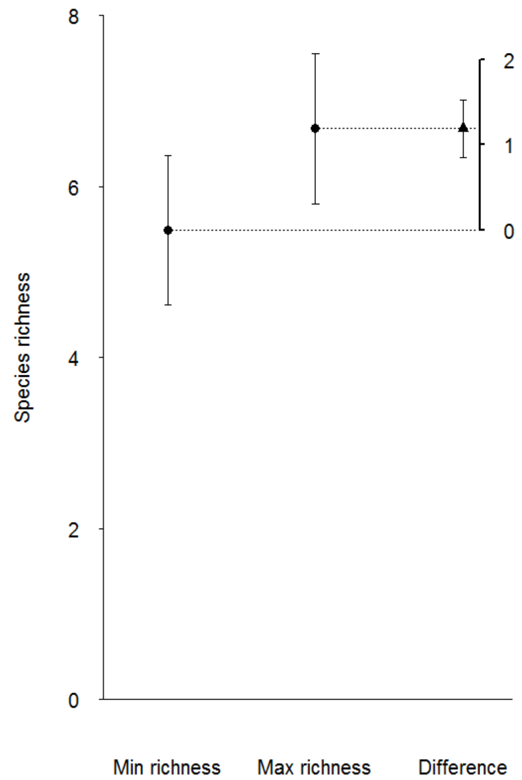

Figure A2.2: Mean maximum, minimum and difference in native fish richness [±95%CI] from baseline surveys at 37 wetlands in the Lower River Murray. After Cumming (2012)

These data suggest that at 95% confidence, the mean native fish species richness at Lower River Murray wetlands varies by less than 2 species. If RRP monitoring data consistently indicate native fish richness exceeding controls by 2 or more species, this would suggest an increase has occurred. A critical effect size suggested for data analysis of native fish richness is therefore 2 species.

The standard deviation in the native species richness baseline data is 2.75, meaning a difference of 2 species represents a change of 0.75 standard deviation units. Correlation in the paired baseline data is 0.92. A sample size of 6 should provide adequate replication to detect a difference of this size in pre and post comparisons of managed sites with baseline data at a power of 0.953.

Native to alien ratio

The overall native to alien ratio was 1.96 [95% CI: 1.78–2.14] species. No evidence of seasonal variation was suggested by the data (Figure A2.3). Although sampling effort was not even across seasons (autumn = 12; spring = 35; summer = 22) these data suggest that any seasonal variation is minimal.

Season was disregarded in effect size calculation, and the maximum (2.24 [95% CI: 1.97-2.50]) and minimum (1.68 [95% CI: 1.46–1.90]) ratio recorded at each site was used. The mean difference in the native to exotic ratio (Figure A2.4) for the 35 repeat surveys was 0.56 [95% CI: 0.38-0.73].

These data suggest that at 95% confidence, the mean native to exotic ratio in fish populations of Lower River Murray wetlands varies by less than 0.8. A difference of at least this value between managed sites and controls (or at managed sites compared to baseline data) indicates a change in excess of that expected in the absence of management. A critical effect size suggested for data analysis of the native to alien fish ratio is 0.8.

In standard deviation units this represents a CES of 1.6. At 0.92 correlation, a sample size of 4 should enable a difference this large to be detected. To attain a margin of error equivalent to half the standard deviation with assurance, 8 samples are needed to detect a difference of this size in pre and post comparisons of managed sites with baseline data.

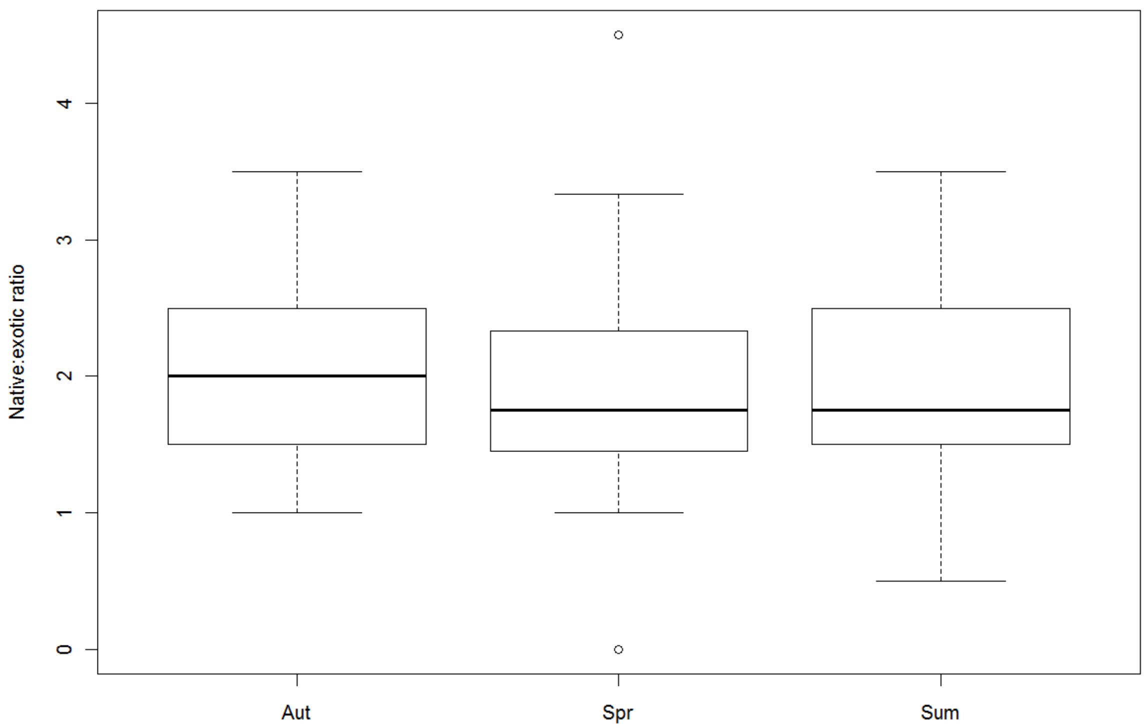

Figure A2.3: Ratio of native to alien fish in Lower River Murray wetlands as a function of sampling season. Data are from baseline surveys representing 70 samples from 35 sites during 2003 – 2005 (Aut n = 13; Spr n = 35; Sum n = 22). The top and bottom of the box represent 25th and 75th percentiles and the middle band is the median

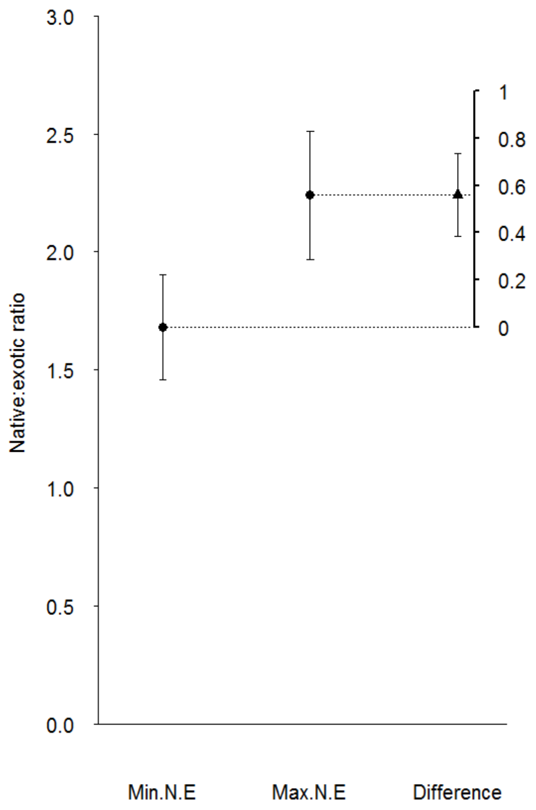

Figure A2.4: Mean maximum, minimum and difference in native to exotic ratio [±95%CI] in native fish species in wetlands (n = 35). After Cumming (2012)

Frog data analysis from baseline surveys

A total of 79 samples collected from 20 sites[8] within the Riverine Recovery Program area during 2005 were analysed. Replication at individual sites ranged from 3-5, with sampling stratified by season (notionally 2 visits each site in autumn and spring).

Ten species were recorded across all samples in baseline 2, and all species except *Pseudophryne bibroni* (the most rarely recorded species) were more frequently detected in spring (Figure A2.5). Almost 80% of samples detected no frogs during autumn surveys, and only *Crinia signifera* was detected in more than 20% of surveys during this season.

[8] Surveyed wetlands: Boggy Flat, Little Toolunka Flat, Loveday Bay, Murtho Park, Murrundi, Overland Corner, Paisley Creek, Paringa Island, Pelican Lagoon, Point Sturt, Poltalloch, Reedy Creek, Rocky Gully, Spectacle Lakes, Sweeney's, Ukee Boat Club, Weila, Wellington East, Yatco Lagoon, Younghusband

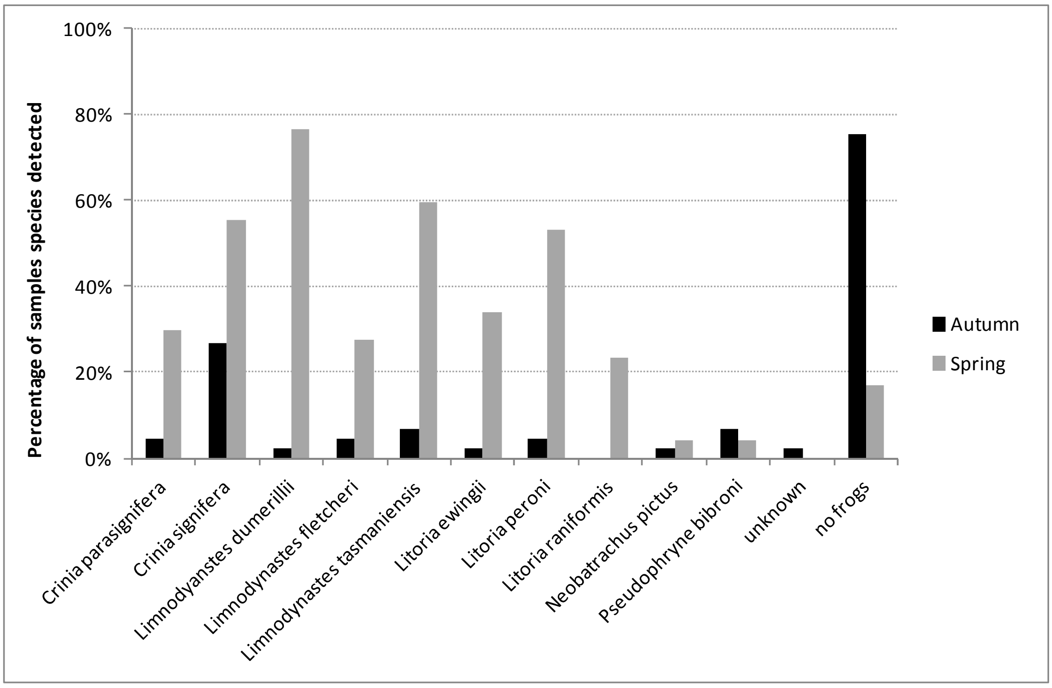

Figure A2.5: Detection percentage for all frog species recorded by seasonality. Data from 79 surveys across 24 sites in 2005: autumn n = 39 (includes 6 June surveys); spring n = 40 (includes 4 August surveys)

The mean species richness of 2.17 for all samples was low. Seasonal variation in detection probability was evident with mean spring richness of 3.7 species (median 4) and winter 0.6 species (median 0). Maximum richness in any one sample was six recorded at around 25% of sites.

The spread of data was comparable between most sites (Figure A2.6), with Weila (WEI) the least variable and highest median richness. Little Toolunka Ck (LIT) and Paisley Creek (PAI) consistently recorded low, or zero, richness.

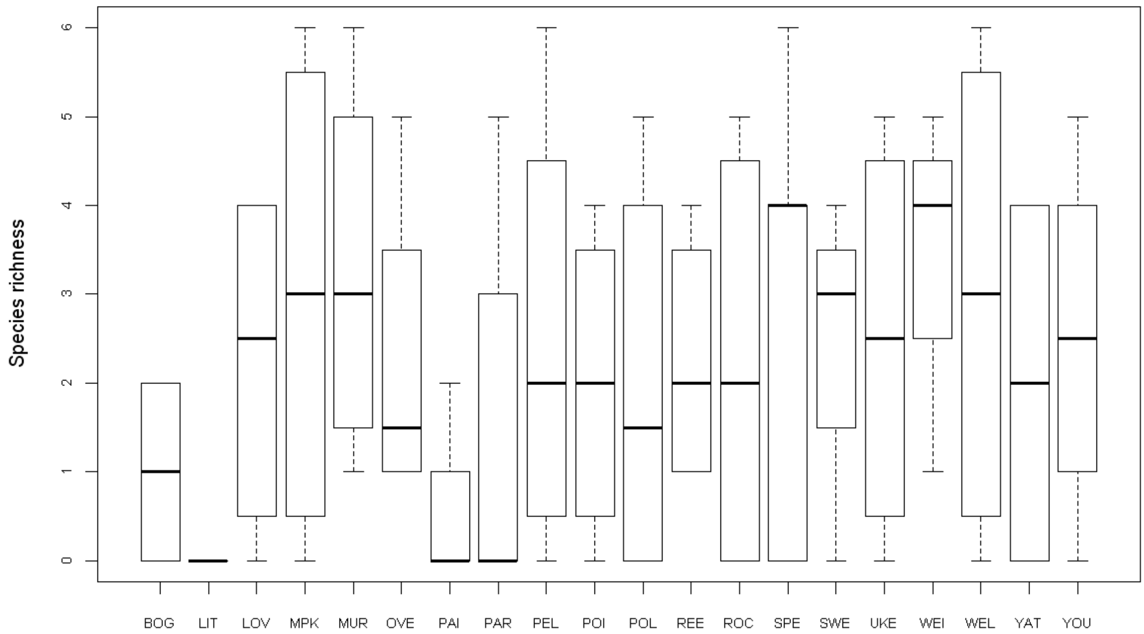

Figure A2.6: Frog species richness for all baseline survey sites with data available. The top and bottom of the box represent 25th and 75th percentiles and the middle band is the median

Geomorphic reach was not a significant influence on richness (Kruskal Wallis p = 0.54) (Figure A2.7).

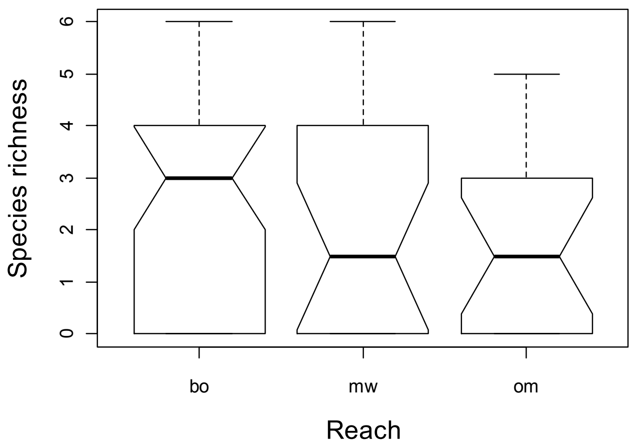

Figure A2.7: Frog survey species richness by River reach: bo = Border to Overland Corner; MW = Morgan-Wellington; om = Overland Corner to Mannum. The top and bottom of the box represent 25th and 75th percentiles and the middle band is the median. Notches indicate approximate 95% confidence intervals

Paired sample analysis

A critical effect size for biological communities can be established based on the observed natural range of variation (Munkittrick *et al*. 2009). The difference in richness between consecutive samples at individual sites in spring 2005 (n = 19) was used to provide a basis for determining the range of natural variability in unmanaged sites.

The estimated difference in the mean and its 95% confidence interval suggest that the difference between richness at a given site under pre-intervention water regimes in spring varies by less than 2 species (mean 1.87; Figure A2.8). If the difference in frog species richness at managed wetlands was to exceed that at control sites by three or more species, this would be outside the natural range of variation and provide evidence that wetland management had been beneficial for the frog community.

Adopting a critical effect size of three species gives a δ of 1.79 (standard deviation is 1.68). Correlation between the paired measures is 0.71. A sample size of five should provide a power[9] of 0.97 to detect a difference of this size in pre and post comparisons of managed sites with baseline data.

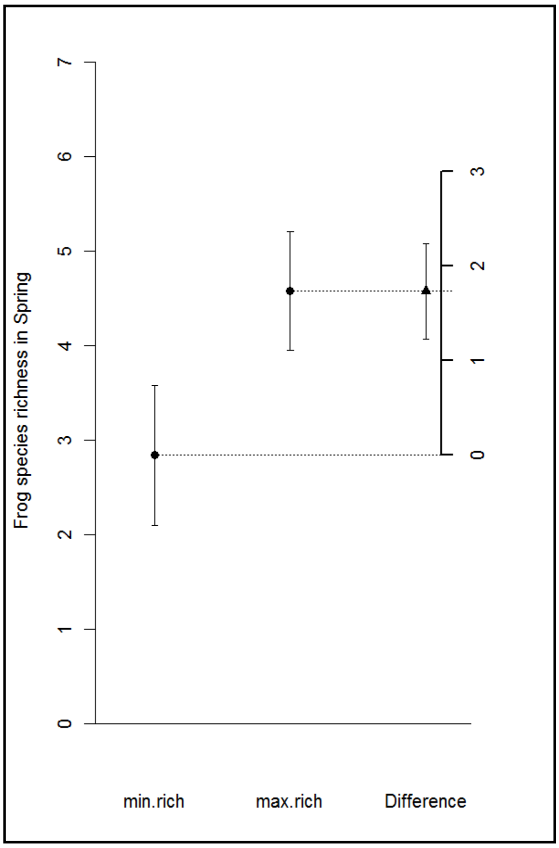

Figure A2.8: Mean maximum, minimum and difference in the two (±95%CI) in frog species richness (After Cumming 2012). Data are repeated wetland baseline surveys conducted at 19 sites during spring 2005

[9] Calculations performed with the 'Power paired' worksheet of the ESCI software (Version 4 Jul 2011) for Microsoft Excel (Cumming 2012).

Water-bird baseline survey data analysis

A total of 70 water-bird species were recorded from repeat surveys collected during autumn (including some February surveys in 2004) and spring (including some December surveys in 2003) at 57 wetlands. Richness over all surveys varied between 2 and 31 species. Autumn surveys were generally less diverse, but richness did not differ significantly between years (Figure A2.9; Kruskal Wallis test chi sq = 4.3; p = 0.114) or seasons (Figure A2.10; Wilcoxon Rank Sum W = 1344; p = 0.112).

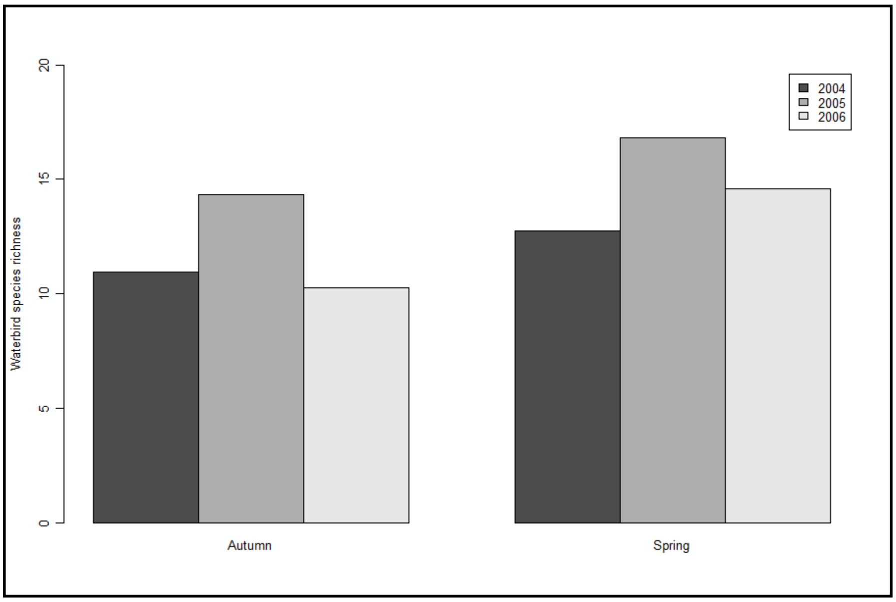

Figure A2.9: Mean water-bird species richness by year and season

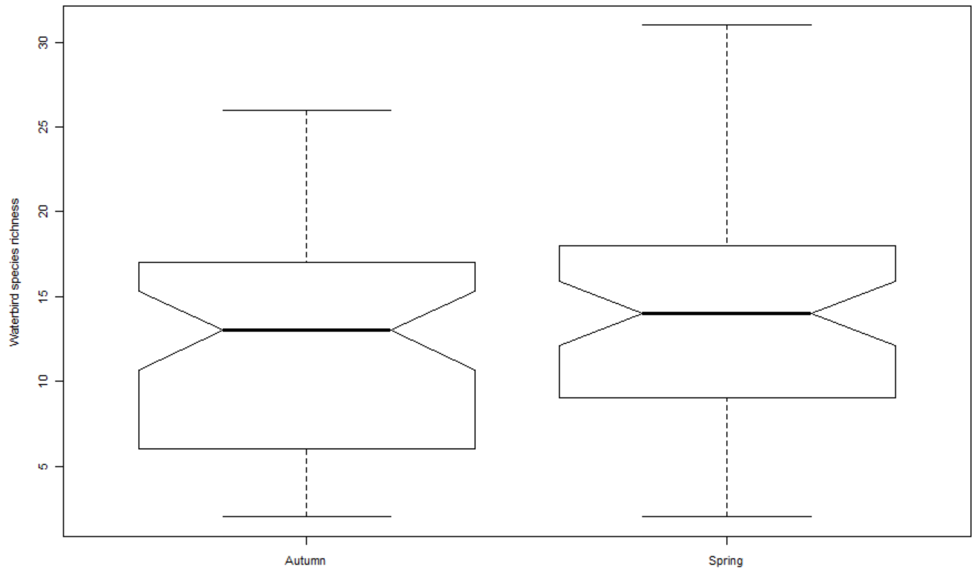

Figure A2.10: Distribution of water-bird species richness samples by season. The top and bottom of the box represent 25th and 75th percentiles and the middle band is the median. Notches indicate approximate 95% CI on median

Critical effect size

Differences in richness between repeat surveys at single sites were analysed to determine the range of natural variation at unmanaged sites. This enables an empirical estimate of a critical effect size. Season and year were not taken into account, and the comparison is between the minimum and maximum recorded from the two surveys at each site.

The maximum observed mean difference at a given site for the 57 repeat surveys conducted between 2003 and 2006 was 5 species (mean 4.96 [95%CI: 4.1 – 5.9]; Figure A2.11), a critical effect size for water-bird species richness during spring surveys is suggested as being 6 species. If the difference between water-bird richness from surveys at managed and unmanaged wetlands consistently exceeded 6 species, then this suggests that management was producing favourable conditions.

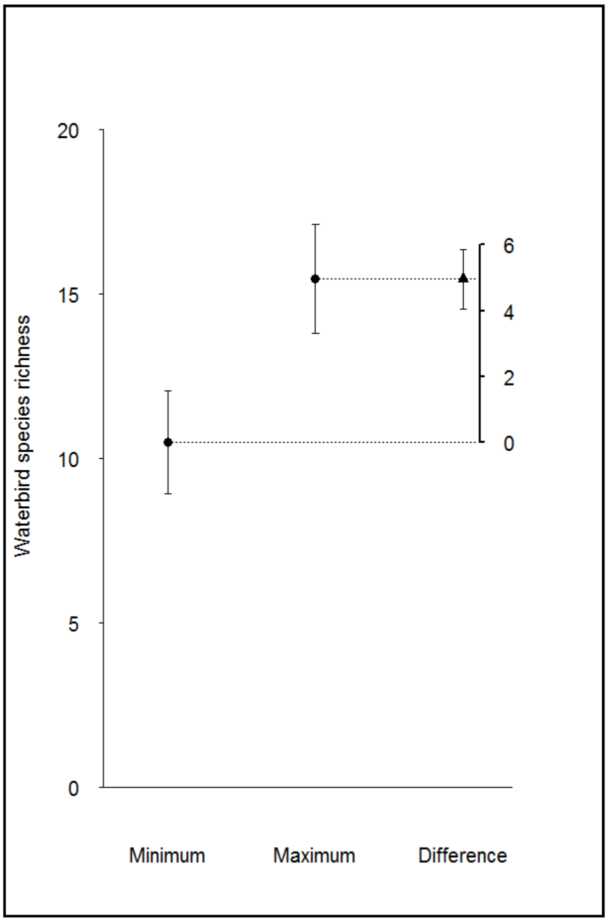

Figure A2.11: Mean maximum, minimum and difference [±95%CI] in water-bird species richness for repeat samples at 57 wetlands between 2003 and 2006 (Figure design after Cumming 2012)

This threshold could also be used to determine any improvement in habitat over baseline data. The standard deviation for the difference in repeat surveys is 3.42, meaning the CES in SD units is 1.75. Correlation between the paired measures is 0.84. A sample size of six should provide a power of 0.969 to detect a difference of this size in pre and post comparisons of managed sites with baseline data.

Water-quality baseline survey data analysis

Critical effect size for turbidity

Baseline survey data gave a total dataset of 36 repeat survey observations that were used to estimate the range in estimated turbidity at unmanaged sites. The mean of the paired difference measures is 57.8 NTU, with a standard deviation of 66.2 NTU (Figure A2.12). The margin of error for 95% confidence is 22.4, giving a maximum upper threshold of change under natural conditions of 80.2 NTU at this level of confidence.

A difference exceeding this at a site would be strong evidence that management had improved turbidity.

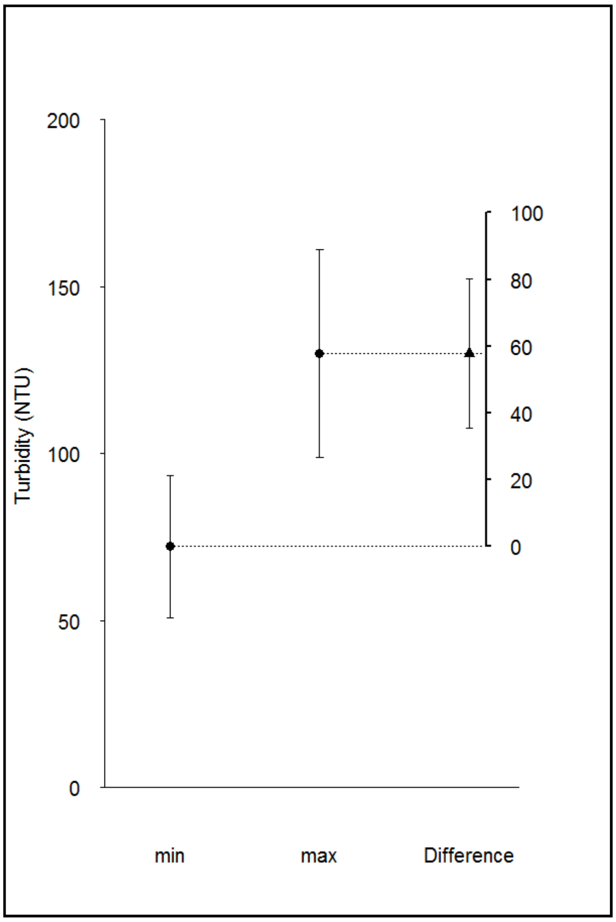

Figure A2.12: Maximum and minimum mean turbidity estimates and the difference (±95% CI). Data are estimates based on 3-5 replicate samples taken at two different times within a calendar at 36 wetlands during baseline surveys 2003-2005

Critical effect size for pH

Mean pH was calculated for baseline surveys, with maximum and minimum mean measurements for each wetland used in a repeat survey analysis (n = 20). The mean difference back-transformed to pH units was 1.1 [95% CI: 0.87-1.35; Figure A2.13].

A change in pH value of more than 1.4 pH units exceeds the natural variation and would be evidence that something has affected wetland pH. The value of 1.4 is suggested as a critical effect size likely to be attributable to an external driver. If the value observed was below any prior minimum in baseline data at the site by the CES, this would suggest additional investigation is warranted as there may be a potential acid sulfate soil problem.

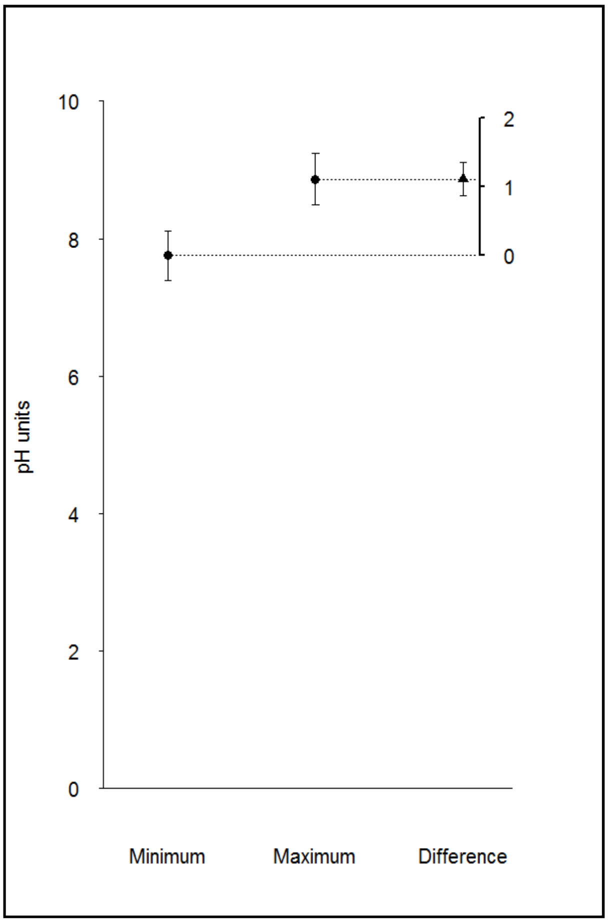

Figure A2.13: Maximum and minimum mean and difference in the mean and 95% confidence intervals for baseline survey repeat measures pH

Measurement precision for electrical conductivity

A review of the baseline data indicates that the error in the estimate of mean electrical conductivity increases linearly with the mean (Figure A2.14). The collection of four replicate electrical conductivity samples would result in wide confidence intervals for the electrical conductivity values likely to be encountered during a wetland drawdown (Figure A2.15).

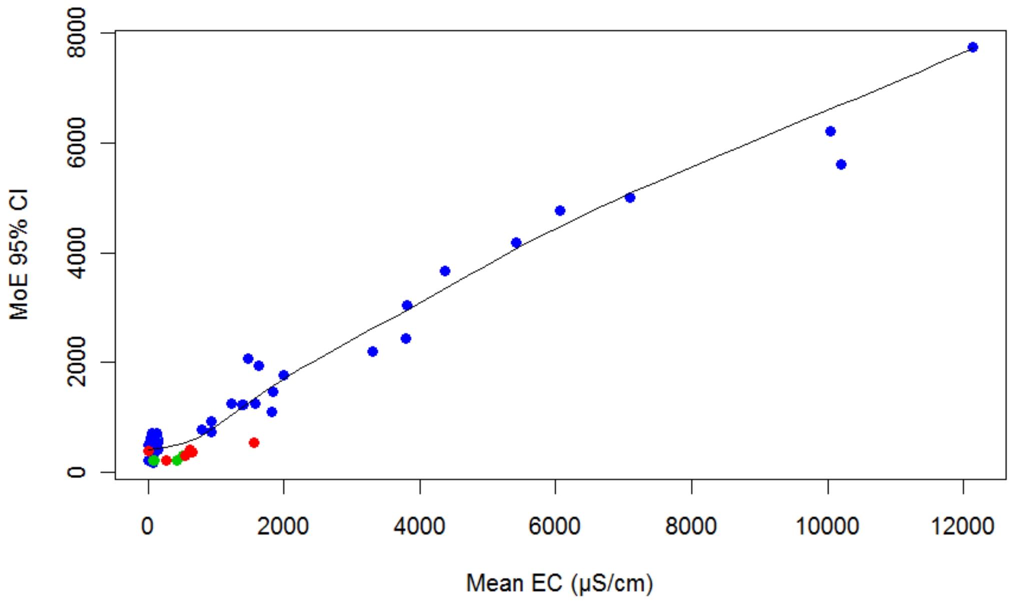

Figure A2.14: Margin of error in mean estimates of electrical conductivity for baseline surveys as a function of observed EC. Colour indicates number of replicates: red = 3,blue = 4,green = 5, light blue = 6

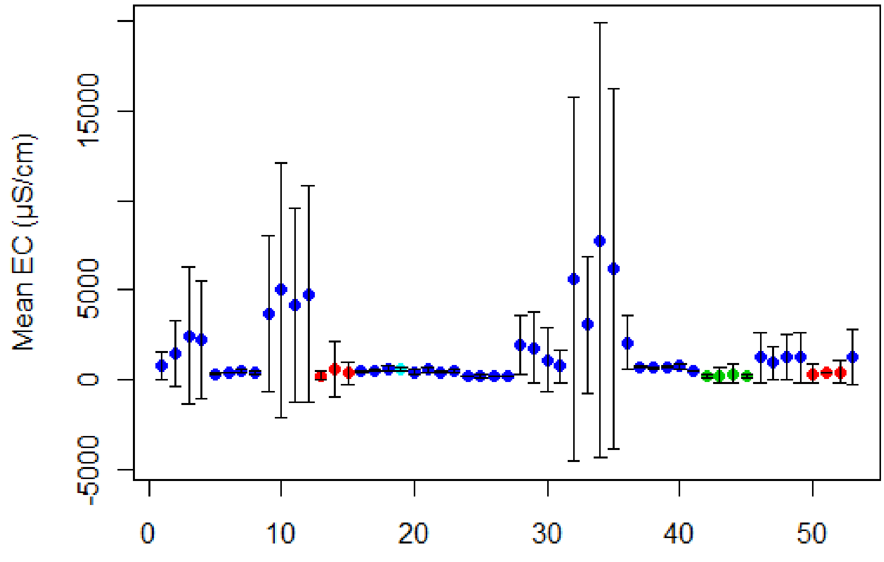

Figure A2.15: Mean electrical conductivity (±95%CI) for baseline surveys. Colour indicates number of replicates: red = 3,blue = 4,green = 5, light blue = 6

Where the maximum deviation from mean sample electrical conductivity varies by more than 34%, a step change in data is observed (Figure A2.16). Samples that deviated less than 40% in magnitude between the mean were taken at sites where mean conductivity was below 450 µS/cm. Clearly the level of replication employed in baseline surveys is inadequate to estimate margins of error that will be meaningful to wetland managers in the 500 – 2500 µS/cm range and above.

Figure A2.16: Ranked ascending maximum absolute deviation from the mean in electrical conductivity samples from baseline surveys (n = 52). Samples that deviate more than 40% of the mean show a step change

An acceptable precision is likely to be obtained with recommended replication (n = 8), provided the largest deviation from the mean in the sample is less than 40% of the mean. Where the maximum absolute deviation for a single measurement within a sample exceeds 40% of the mean, additional samples are required to improve precision.

Oversampling is required to determine how many more measurements will be required to attain adequate sample mean precision. Where time permits, replication in water-quality measures during a pilot study should take up to 20 samples in total in order to provide the data to develop more precise replication guidelines.

Appendix 3 – Estimated replication for vegetation surveys

As baseline survey vegetation data were not collected in a consistent manner, and no repeat survey data were available, a critical effect size could not be calculated. Thus a critical effect size will need to be determined from the initial program implementation and a value of 0.3 in proportional cover for the amphibious plant functional group is suggested as a starting point.

In order to establish the level of replication required to produce adequate precision for a sampling unit, a permutational study was undertaken on data collected from sites in the Lower River Murray using the adopted measurement protocols. Inverse Simpson diversity was used as the state variable and the study aim was to determine the levels of within site variability that can be expected in the region. From this the level of sampling necessary to reduce variability in community level estimates to an optimal level of precision was determined.

Data from single surveys of 28 Lower River Murray sites were analysed, with species recorded as counts (0-15) from each quadrat. Replication within sites ranged from 9 to 17 quadrats. Data was firstly ranged to take values from 0-1, and the Inverse Simpson index (Simpson 1949) was calculated for each quadrat. The distribution of values for each site differed greatly (Figure A3.1), but did not appear to vary as a function of replication.

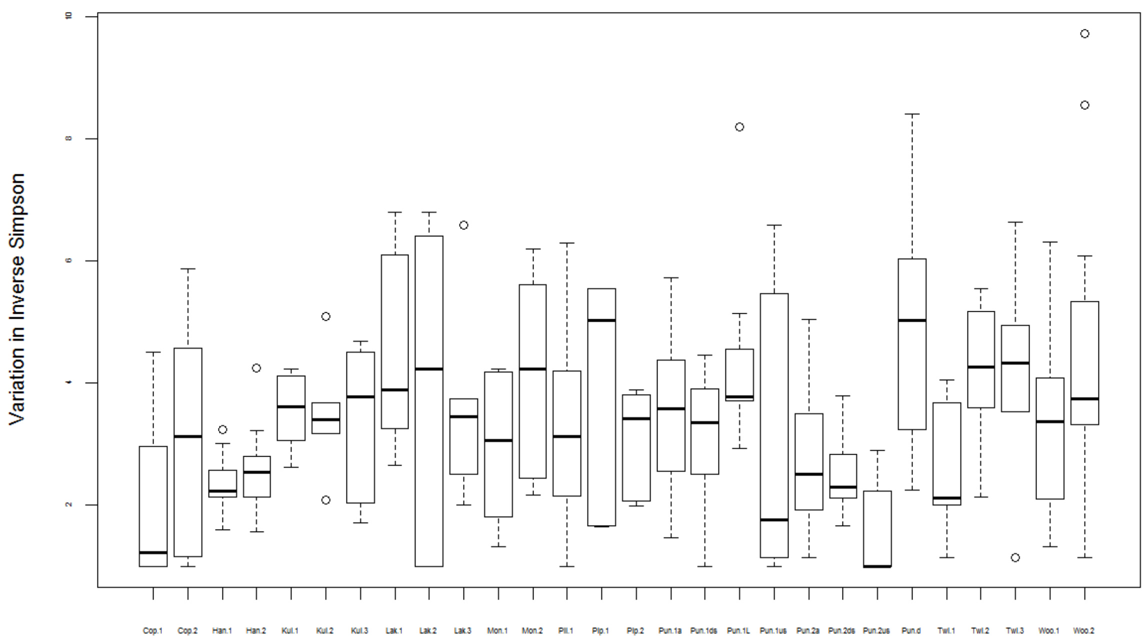

Figure A3.1: Distribution of Inverse Simpson indices for individual quadrats using measurement protocols. Replication ranges from 9 -17 quadrats per site

The site exhibiting the greatest variation (Lake Littra) consisted of 18 quadrats, and these data were selected for simulation. Species data was permuted individually and 1000 random draws were taken after each permutation and an Inverse Simpson index was calculated on each permutation. For each level of replication, the mean and 95% confidence intervals around the estimated Simpson Index were calculated (Figure A3.2).

Lower levels of replication underestimated the mean value compared with the value calculated on the full data set (red triangle in Figure A3.2). The margin of error decreased with increasing sample size, asymptoting to a similar value as the overall dataset by around 10-12 draws (Figure A3.3).

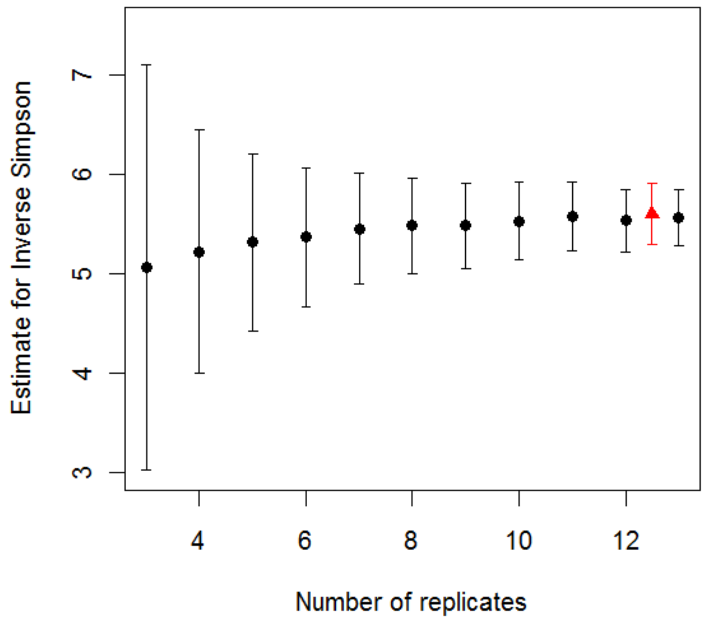

Figure A3.2: Estimation accuracy and precision in Inverse Simpson indices as a function of replication

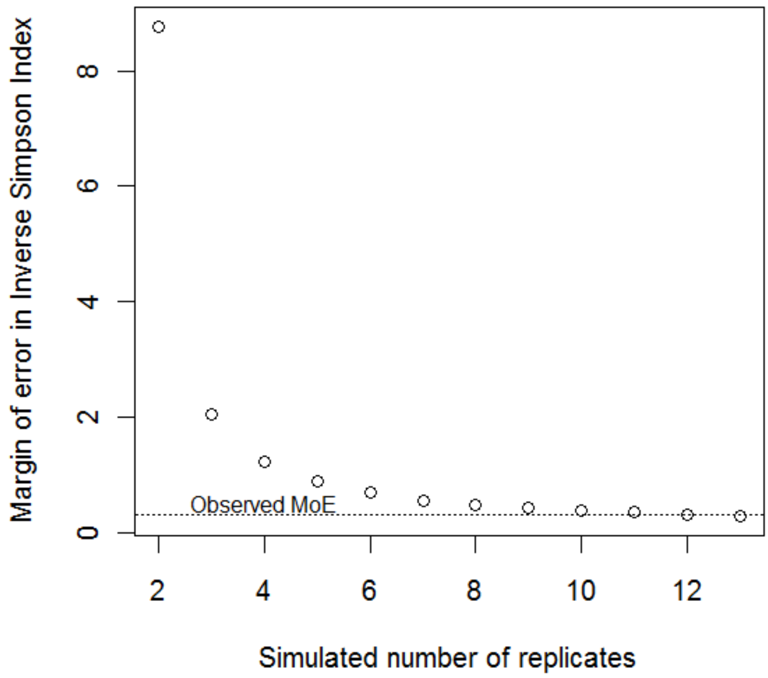

Figure A3.3: Margin of error in simulated estimates of Inverse Simpson index as a function of replication

Recommendation

Until pilot study data can be analysed to determine levels of variability in plant functional group and community diversity measures, sampling at least 12 quadrats is recommended.

Appendix 4 – Field equipment

Table A4.1: Field monitoring equipment list

Monitoring category	Equipment list
General	4WD vehicle and recovery equipment Boat and safety equipment Communications equipment to meet OHS Remote or Isolated working conditions as appropriate Personal Protective Equipment (e.g. long sleeve shirt and trousers, sunglasses, hat) Other OHS requirements: First Aid Kit, Snake bite kit, fire blanket, fire extinguisher etc. Adequate drinking water for field crew Waders Laptop computer Hand-held Personal Digital Assistance (PDA) with integrated GPS (loaded with site imagery) OR GPS unit and datasheets Waterproof digital camera plus blackboard/whiteboard and markers for photopoint designation Clipboard with field sheet, notebook and pencil Spare batteries and/or battery charger for all electronic equipment (water-quality meter, GPS, laptop, PDA, camera)
Additional equipment required for site establishment:	Surveying equipment (dumpy level and staff) At least 10 x 1.5 m star pickets and high visibility end caps to mark vegetation transects Aerial photographs of sites indicating broad vegetation associations used in stratifying sampling design
Surface water monitoring	Sampling rod and laboratory sampling bottles (if taking samples for verification of salinity readings or for other analytes e.g. Chlorophyll a; Total suspended solids) Multi-parameter waterproof hand-held meter plus any transducers and calibration solutions (Calibrate probe prior to field investigation)

Monitoring category	Equipment list
	Demineralised water (clean probe after every sample)
Vegetation zonation and composition (NOTE TLM floodplain canopy methods have a separate gear list – see Souter *et al*. 2010)	Compass 100 m measuring tape Range pole (2 m) Plant press and vouchering equipment Identification handbooks
Water-birds	Binoculars Spotting scope, case, tripod Identification handbooks
Fish	Fyke Nets Bait Traps Seine Net Scales and measuring equipment Identification handbooks
Frogs	Spotlight Digital recorder with external microphone MP3 player (or equivalent) and external speakers with pre-recorded frog calls Batteries
Groundwater (where existing observation well network is to be maintained)	Water level probe (e.g. Solinst 101) Bailer Multi-parameter water hand-held water-quality meter as for surface water monitoring

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