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Australian & New Zealand GUIDELINES FOR FRESH & MARINE WATER QUALITY

Assessing and managing water quality in temporary waters

Technical Report

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Summary

A large proportion of Australia's inland waters are temporary in nature, yet little tailored guidance has been published on how to effectively assess water and/or sediment quality (in terms of chemical, physical and biological characteristics) in these types of ecosystems. To move towards redressing this deficiency, an expert workshop on water quality guidance for temporary waters was held on 17–18 September 2015 at the Brisbane Herbarium. The aim of the workshop was to bring together leading researchers and practitioners of temporary water science (e.g. ecology, water quality, hydrology) in Australia to provide guidance on the protection of water and/or sediment quality in temporary water systems. Many of the ideas and concepts in this guidance document arose from this workshop.

Empirical information about water quality in temporary waters lags behind that for permanent waters. Often, physical and chemical guideline values are not available for temporary waters, nor is there clear guidance on monitoring these ecosystem types. Instead, and until empirical data for multiple indicators for specific locations have been gathered, the advice in this guidance document relies on the development of suitable conceptual models within a water quality management framework, which are updated over time.

This document provides guidance for the application of the ANZG (2018) Water Quality Management Framework (WQMF) for assessing water and/or sediment quality of temporary waters. It assumes that the normal WQMF process is applied, but provides specific consideration for temporary waters. This document:

- defines temporary waters and explains their ubiquity in Australia (and overseas)
- explains why water quality assessment has been problematic in temporary waters and why this guidance document was required
- briefly describes the WQMF
- highlights some of the key benefits of the WQMF for assessing temporary waters, especially the use of conceptual models for defining spatial and temporal features of temporary waters for water quality assessment, and the use of multiple lines of evidence and how to select these according to the water regime and sampling intention
- summarises the challenges of monitoring water quality in temporary waters and how these might affect setting water quality objectives
- provides a list of potential water and/or sediment quality indicators for temporary waters, including brief description of their advantages and limitations.

The guidance provided in this document should be used in conjunction with the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018), although it does not replace or supersede government jurisdiction water quality legislation or other information pertaining to temporary waters unless otherwise advised by the relevant jurisdiction.

1 Introduction

1.1 What are temporary waters?

Temporary waters is a general term for all standing or flowing waterbodies that alternate between phases of inundation and lack of surface water. As such, this document refers to:

- *temporary streams*, which generally have a defined stream channel and alternate between flowing and either no-flow (i.e. disconnected waterholes) or dry stream bed states; and
- *temporary standing waters* (including pools, ponds, wetlands and lakes), which generally do not have through-flow of water or do not have a defined stream channel, and alternate between inundated and dry.

Also, t*emporary waters* include intermittent and ephemeral waters, which are defined as follows:

- *Intermittent waters*: subset of temporary waters that are predictably inundated each year (during a wet season), although the duration for which they retain water may be highly variable.
  - *Seasonal waters*: subset of intermittent waters, which are predictably inundated in one or more seasons per year.
- *Ephemeral waters*: subset of temporary waters that contain water only after irregular rainfall or flow events.
  - *Episodic waters*: more infrequently inundated subset of these waters for which inflow is a rare event (e.g. dune base wetlands in the Strzelecki Desert). Ephemeral and episodic waters may retain water for a long time and can be very productive (e.g. Lake Eyre).

Temporary streams cease flowing at some stage, but usually have remnant standing waterbodies along the stream for some time after cessation of flow (some of which may be perennial waterholes), resulting in an ecological overlap with temporary standing waters and, in some cases, perennial standing waters. Additionally, through-flow may occur in temporary standing waters during periods of heavy rainfall. Commonly, standing waters are fed and drained by temporary streams, but the absence of a defined stream channel is a useful distinguishing characteristic despite the overlapping ecological conditions of both water types. Supplementary to the above definitions, a recently published review of terminology for non-perennial rivers and streams (Busch et al. 2020) provided a useful groundwater-related distinction between intermittent and ephemeral streams. Examples of types of temporary waters are shown in Figure 1, Figure 2, Figure 3 and Figure 4.

Temporary waters typify the surface waters of arid and semi-arid regions of Australia (average of <250 mm and 250–350 mm rainfall per year, respectively) (Figure 5) but are not exclusive to these regions (Boulton et al. 2014). For example, temporary (intermittent) waters exist in northern (e.g. wet–dry tropics) and southern Australia (e.g. regions within Victoria and Tasmania), where average annual rainfall is typically much higher than in the arid and semi-arid regions. Sheldon et al. (2010) noted that as more than 75% of the nation's land surface is classified as arid or semi-arid and, as 95% of the nation's river channels are lowland rivers, most rivers are in drylands and are commonly temporary waters. Similarly, a large proportion of the nation's standing inland waters are temporary.

Figure 1 Example temporary water type, intermittent stream, southern Gulf of Carpentaria
Photo courtesy of Hydrobiology.

Figure 2 Example temporary water type, seasonal stream, central New South Wales
Photo courtesy of Hydrobiology.

Figure 3 Example temporary water type, ephemeral salt lake, Western Australia
Photo courtesy of Hydrobiology.

Figure 4 Example temporary water type, episodic wetland, Strzelecki Desert, South Australia
Photo courtesy of Hydrobiology.

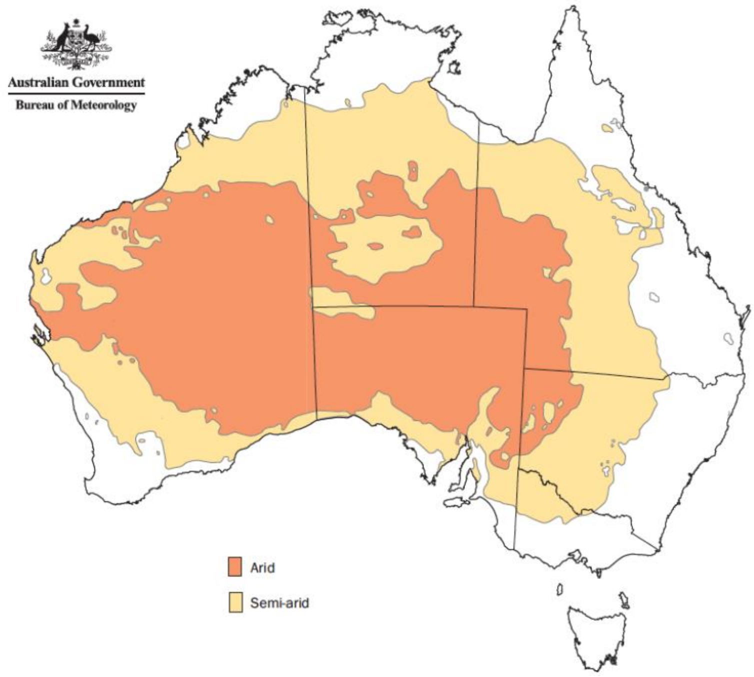

Figure 5 Arid and semi-arid zones of Australia

Source: Australian Bureau of Meteorology.

1.2 Temporary water types

With such prevalence of temporary waters across Australia, there are diverse typologies within each wetting–drying cycle category and sub-category defined in the Introduction. The characteristics of each typology are tightly inter-related with the wetting–drying cycle and the natural water quality in each waterbody. For the purposes of this guidance document, the development of a detailed typology of all temporary water types across Australia was out of scope. The Australian National Aquatic Ecosystem classification framework (ANAE 2012) provides a consistent national mechanism for users to classify waterbodies. Level 3 Aquatic classes, systems and habitats for inland waters include lacustrine, palustrine, riverine and floodplain systems (ANAE 2012), for which typical temporary water types (including salt lakes, upper catchment desert creeks, seasonal wet/dry tropics streams, dryland floodplain wetlands/desert swale wetlands, temporary waters in internally draining catchments, and vernal pools) may be nested. With the exception of some detail for the Lake Eyre Basin, desert salt lakes, and arid/semi-arid and wet–dry tropical inland waters, the advice provided in this document is general.

1.3 The water quality management framework

The centrepiece of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018) is the water quality management framework (WQMF). The WQMF provides a nationally consistent, 10 step approach for managing, assessing and monitoring water and/or sediment quality. In theory, the WQMF can be applied to any water quality issue in any type of waterbody, and it is supported by extensive guidance on how to assess and monitor water and/or sediment quality. However, and notwithstanding the prevalence of temporary waters in Australia, much of the guidance and guideline values provided by ANZG (2018) is more applicable to permanent waters than temporary waters. In fact, little guidance has previously been published on how to appropriately manage, assess and monitor water and/or sediment quality in temporary waters. This guidance document represents a move towards redressing this deficiency.

1.4 Scope and aims

An expert workshop on water quality guidance for temporary waters was held on 17–18 September 2015 at the Brisbane Herbarium. The aim of the workshop was to bring together leading researchers and practitioners of temporary water science (e.g. ecology, water quality, hydrology) in Australia to provide guidance on the protection of water and/or sediment quality in temporary water systems. The workshop discussions were to be used as the basis for a temporary waters guidance document associated with ANZG (2018). However, the provision of detailed guidance on how to manage, assess and monitor water and/or sediment quality for each or even a selection of the many different types of temporary waters that exist in Australia was beyond the scope of the expert workshop and this guidance document. Instead, this document focuses on the application of the various steps of the WQMF to temporary waters and, in doing so, draws on the concepts that were discussed at the workshop. Throughout ANZG (2018) and this document, the term “water quality” is not limited to just the chemical and physical characteristics of water, but also includes its biological characteristics.

Empirical information about water quality in temporary waters lags behind that for permanent waters. Often, physical and chemical guideline values are not available for temporary waters, nor is there clear guidance on monitoring these ecosystem types. Instead, and until empirical data for multiple (chemical, physical and biological) indicators for specific locations have been gathered, the advice in this document relies on the development of suitable conceptual models within a water quality management framework, which are updated over time.

Within this context, the aim of this document is to provide water managers, regulators, consultants and all others who have a role in protecting temporary waters with greater confidence in how to appropriately manage, assess and monitor temporary waters, and in a manner that is consistent with ANZG (2018). Although much information and guidance related to water quality monitoring for temporary waters is provided in this document, it is recognised that descriptions of example (or case study) monitoring programs for temporary waters could be a useful future addition. Finally, the guidance provided in this document should be used in conjunction with ANZG (2018), although it does not replace or supersede government jurisdiction water quality legislation or other information pertaining to temporary waters unless otherwise advised by the relevant jurisdiction.

2 Application of the water quality management framework to temporary waters

This section provides tailored guidance for how each of the steps of the WQMF can be applied to temporary waters, focusing on the unique characteristics, and associated considerations for water and/or sediment quality assessment, of temporary waters. Additional details for each of the steps of the WQMF can be obtained from ANZG (2018) (see Applying the framework)

2.1 Step 1 – examine current understanding

Step 1 involves developing the understanding of the waterway of interest and its associated water quality issues, and relies heavily on the use of conceptual models. This step forms the foundation for identifying the key community values and setting management goals (Step 2), selecting lines of evidence and subsequent water quality indicators (Step 3), determining appropriate water and/or sediment guideline values (Step 4) and developing water and/or sediment quality objectives (referred to herein as WQOs) (Step 5). Step 1 is used to determine the key pressures and stressors on the system and the predicted responses to them.

Human demands for water are intensifying, especially in semi-arid and arid zones where water is scarce and potentially threatened by increasing population pressure and climatic drying (Brooks 2009). Therefore, attention is now focussing on the impacts of water resource use on temporary waters (Acuna et al. 2014, Chiu et al. 2017, Datry et al. 2018) and sustainable provision of their ecosystem services (Boulton 2014, Datry et al. 2018). In the last decade or so, research on temporary waters in Australia and overseas has improved our understanding of the drivers of ecological condition and ecosystem health (e.g. reviews in Leigh et al. (2015) and Van den Broeck et al. (2015)), reiterating the major role played by the wetting–drying cycle in controlling water quality, composition and distribution of aquatic biota, and ecological processes in temporary waters. Consequently, much of the management focus in temporary waters has been on their susceptibility to alterations of the water regime (i.e. the quantity and spatial distribution of surface water and the duration, timing and extent of wetting and hydrological connectivity).

However, there has been less emphasis on the susceptibility of temporary water ecosystems to changes in water quality, although some studies describe potential effects from agriculture, urban land uses and mining on temporary waters (e.g. Botwe et al. 2015, Ramsay et al. 2012). As acknowledged in the previous guidelines (ANZECC/ARMCANZ 2000), there has also been much less research into the impacts of pulsed exposures of contaminants, although there has been some promising research into the area more recently (e.g. Ashauer et al. 2006, Angel et al. 2010, 2015). In temporary waters, the need to consider pulsed exposures arises from the pulsed nature of the wetting–drying cycle (Gómez et al. 2017). This is in contrast to perennial waters where the permanence of water leads to a focus on ecosystem sensitivity to continuous exposure to contaminants in the water column or sediments. This sensitivity is generally regarded as greater under continuous exposures, with effects occurring at lower toxicant concentrations (see section 3.1.7 of ANZECC/ARMCANZ (2000)).

Another knowledge gap is determining what WQOs are appropriate for temporary waters that have been converted to perennial or near-perennial waters by anthropogenic discharges (e.g. Luthy et al. 2015, Chiu et al. 2017). There are controls in Australia and overseas on water quality in discharges and/or receiving waters for perennial, near-perennial waters; however, there is only very limited guidance in any state or territory guidelines (e.g. Queensland Government 2016, Environmental Protection Authority 2018) on the combined impact of conversion from temporary to perennial or near-perennial status and alteration of water quality.

To provide the context for addressing these knowledge gaps and to summarise the recent advances in our scientific knowledge about temporary waters, Step 1 in the WQMF entails documenting the *current understanding* of the system (including its management strategies). This crucial step:

- provides a logical foundation for setting management goals
- illustrates the key drivers, pressures and stressors
- portrays relevant interactions among, and responses to, these drivers, pressures and stressors
- guides subsequent steps, such as selection of appropriate lines of evidence and relevant water quality indicators (Step 3) and development of WQOs (Step 5).

2.1.1 Conceptual models

Ideally, the process to explore and communicate current understanding of a system uses one of the most powerful tools—*conceptual models*. These can take many forms, from narratives and diagrams through to detailed numerical models and quantitative analytical tools. Conceptual models have particular advantages in environmental assessment (Lindenmayer & Likens 2010), including:

- specifying the scope and scales of the system
- illustrating the main components and interactions at a given scope and scale
- generating hypotheses to test particular interactions and outcomes
- integrating input from different experts into a formalised, shared understanding
- facilitating rapid communication of ecosystem components, interactions, responses and complexity among scientists, managers and the public
- revealing likely responses to one or more stressors so that potential management strategies to minimise impacts may be identified.

Comprehensive guidance on using conceptual models within the WQMF is provided in ANZG (2018) (see Conceptual models). A good conceptual model does not attempt to explain all possible relationships but instead tries to contain only the most relevant information. Too much information can conceal important aspects of the model, while too little information leads to a higher likelihood that the portrayal is not accurate and is contrary to what is observed. The model should only be as complex as is necessary; the level of complexity may be initially obvious or it may take several iterations of the model to realise.

The approach adopted for conceptual model development largely follows that of Gross (2003a, 2003b), which recognises *control* models (also known as *process* models) and *stressor* models (also referred to as *pressure-stressor-ecosystem receptor*, or *PSER*, models). Control models conceptualise the controls, feedback and interactions responsible for system dynamics. Stressor models articulate the relationships between the stressors and ecosystem components, effects to them, and indicators of those interactions. Negus et al. (2020) provide a detailed description of the use of stressor models (or, causative conceptual models, as they refer to them) for assessment of aquatic ecosystems in Queensland. The recommended steps to apply the conceptual modelling approach to the WQMF are:

1) identify the goals of the conceptual model(s)
2) identify the bounds of the system and important sub-systems
3) define the information requirements
4) compile the available information
5) develop control models of key systems and sub-system processes
6) identify natural and anthropogenic pressures and stressors
7) incorporate/identify the stressors (and indicators) in stressor models
8) articulate key questions or alternative approaches, and review assumptions
9) take model outputs to WQMF Step 2 and Step 3
10) review, revise and refine the models.

Various diagrammatic conceptual models have been developed for temporary waters (e.g. Figure 3.1 in Williams (2005) and Figure 2 in Bond and Cottingham (2008)), but they usually focus on factors affecting aquatic biota rather than the causes and effects of anthropogenic changes to water quality, and they seldom successfully capture the temporal dynamics of water quality variability. For temporary waters, an understanding of both the natural drivers of water quality in the system (through control models) and the effects of anthropogenic pressures and their stressors on water quality and aquatic biota (through stressor models), is especially important.

The expert workshop in September 2015 considered how to best derive conceptual models to illustrate the typical drivers of water quality in standing and flowing temporary waters in Australia, highlighting the major drivers and stressors and their interactions with each other and biological components such as microbes, algae, invertebrates and fishes (which can also influence water quality). The derived conceptual control model draws on the common themes from various conceptual models, both published and discussed in the expert workshop, and is presented in Figure 6.

This generic conceptual control model can be used as a guide for users to develop their own specific conceptual control model (linked to or combined with a stressor model) when a water quality management plan is being developed for a particular temporary waterbody; that is, identifying the likely anthropogenic and non-anthropogenic drivers (Step 1), guiding selection of water quality indicators (Step 3), determining the appropriate guideline values (Step 4) and developing WQOs (Step 5). Many of the drivers (e.g. climate, physical geography, geology, land use and hydrological connectivity) operate at least in part at the regional scale (landscape or catchment; Figure 6), whereas others such as bathymetry, fringing vegetation and substrata operate at the local scale (basin) within a given pool or stream reach (Figure 6). At a fine (i.e. high resolution) spatial and temporal scale—not depicted here—internal limnological factors and processes operate (e.g. depth of surface mixing and vertical stratification, wind and wave action, reaeration rates and temperature variations). These may also act as drivers of ambient water quality variations in lentic waters. Drawing up a conceptual model prompts discussion about the interactions among different drivers, the likely features of a specific temporary water that potentially affect water quality at different times, and the indicators (Step 3) that would reflect salient water quality stressors at a given location.

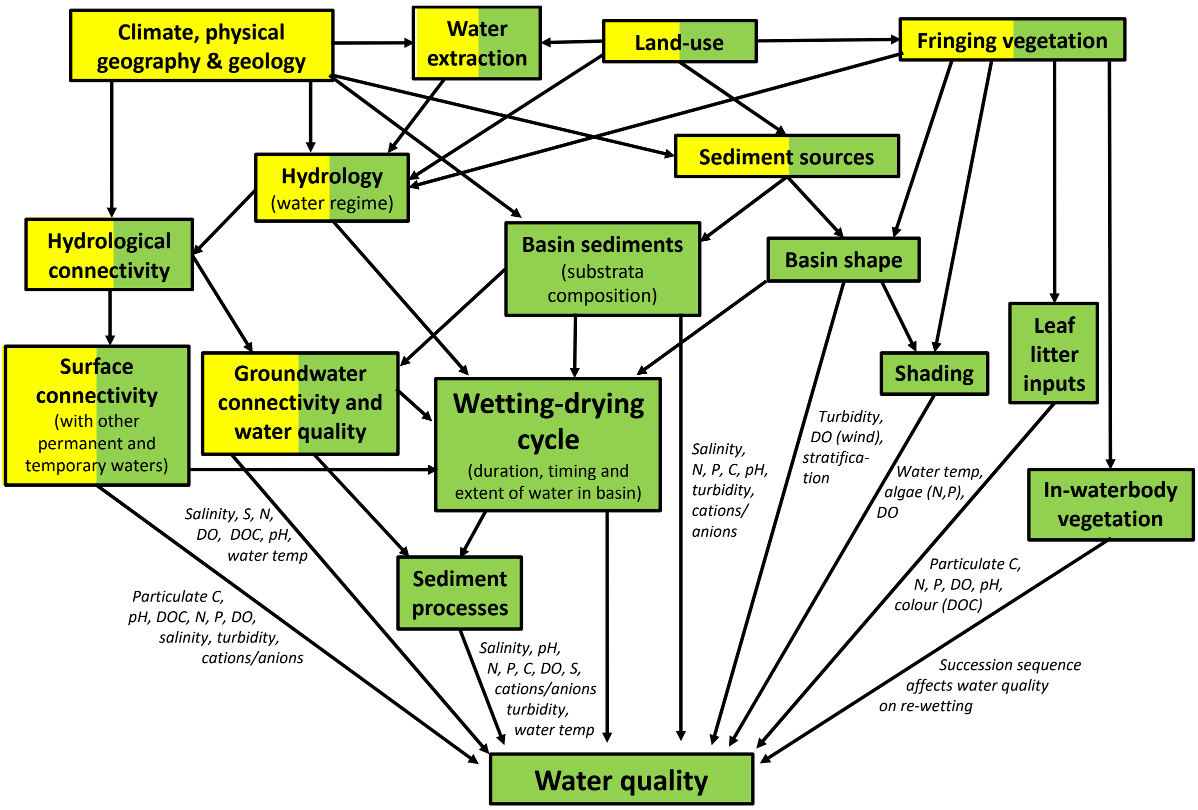

Figure 6 Illustrative conceptual control model, key drivers of water quality in temporary pool

Yellow shading: regional scale.
Green shading: local scale.
C – carbon; DO – dissolved oxygen; DOC – dissolved organic carbon; N – nitrogen; P – phosphorus; S - sulphur.
Note: Italicised text indicates examples of specific water quality parameters affected by each driver. For clarity, not all arrows of interaction are shown (e.g. effect of land use on groundwater connectivity and water quality; effect of basin shape and sediments on fringing vegetation).
Note: While drivers of water quality predicate uni-direction of pathway arrows, a number of the interactions are bi-directional; for example, there are potentially very strong two-way interactions between water quality, sediment processes, aquatic vegetation and physical limnology.

2.1.2 Using conceptual models to develop understanding of temporary waters

One of the most effective ways to use a conceptual model is to generate questions or hypotheses to help guide logical investigation and development of current understanding (see ANZG 2018) for particular temporary waters. These questions can be grouped by their main drivers. Example questions and associated implications for a conceptual model are provided in Table 1.

Table 1 Example conceptual model questions and implications

Questions	Implications for conceptual model	Control or stressor model
Climate, landscape setting and geology		
• Is the climate arid, semi-arid, temperate, Mediterranean or wet–dry tropical? • Is rainfall seasonal and/or predictable within or among years? • When does evaporation exceed precipitation? Is this predictable within or among years?	These questions influence considerations of the wetting–drying cycle for the waterbody. They will also influence subsequent selection of indicators and design of sampling programs. Many geochemical processes will be influenced by the durations of the wet–dry cycles and the durations of inflows to and outflows from the waterbody.	Control
• Is the temporary water within a defined drainage line (e.g. a waterhole within a river channel)? • Is it a depression in undulating land and not part of any coordinated drainage? • Is it on a floodplain or near other temporary or permanent waters? • Is it near saline waterbodies that could flood into it and alter its water quality? • Conversely, if the temporary water is saline, is there nearby freshwater that might flood into the basin and reduce the salinity significantly?	Connectivity between the waterbody and other surface water will strongly influence the initial water quality when inundation occurs, particularly where salinity will be strongly controlled by inflows. If part of a larger watercourse, initial conditions are likely to be largely shared with other waterbodies along the watercourse, but local interactions may subsequently override these initial conditions, leading to divergence in water characteristics between waterbodies over time once flow ceases to connect them.	Control / stressor
• What are the regional geology and soil-influencing properties of the catchment? • How might these affect catchment water quality through weathering and erosion? • How might local geology affect surface and sub-surface water inputs and outputs (e.g. sediment permeability and porosity, faults and cracks)? • Is the basin on bedrock or over alluvial sediments with high porosity and permeability or clays that bake hard when dry and might allow the temporary water to perch above the water table? • Is there a local mineral outcropping within or near the waterbody that might affect its water quality?	Regional geology will influence the typical initial water characteristics during infilling, including the salt mix dissolved in the water, the initial water pH and composition and concentration of suspended particulate matter (SPM). Local geology can modify the water characteristics through exchange with the substrata. This influence can be expected to increase through the drying phase once surface flow connection to other waterbodies has ceased. Connection to an aquifer through permeable substrata will control the longevity of the presence of surface water if inflows from the aquifer can substantially offset evaporation once flow connectivity is lost. Such a connection can also afford access for hyporheic fauna and influence the development of the faunal assemblage and provide refuge opportunities for fauna not available in waterbodies isolated by a clay or bedrock layer.	Control
Land use, vegetation cover and sediment processes		

• What is the predominant land use in the catchment and are there established land and water management plans in place?
• What land uses are in the upstream sub-catchment?
• What is the likely water quality of any runoff?
• Are there intensive agriculture, grazing, mining or urban inputs?
• Are there identified point and diffuse sources of pollution that enter the basin?
• For specific land uses:
    o Grazing: Are the riparian zones and recharge areas of key groundwater aquifers and spring heads of waterways fenced to control livestock access? Are there off-stream watering points? Are feral animals being managed?
    o Intensive agriculture: What management plans and regulatory mechanisms are in place? What kinds of irrigation are employed (source, application method, frequency, timing)? What are the main crops and what is the cropping cycle? What specific agrochemicals are used (quantities and types of fertilisers and pesticides) and are there potential sources of biological oxygen demand?
    o Mining: There are analogous questions to those posed for ‘intensive agriculture’. For mining, persistent toxicants (metals, metalloids) and process chemicals (e.g. cyanide, ammonia) are also among the potential stressors.
	Consideration of the local land use will help identify likely sources of stressors to the waterbody, such as nutrients (e.g. from stock faeces), turbidity and SPM (due to gullying and soil erosion), pesticides (e.g. from cropping or residential areas), and industrial or urban discharges. For example, in temporary water catchments in grazing lands, there is a tendency in the dry phase for stock to congregate in riparian zones for shade and access to residual water, resulting in a net flux of nutrients closer to, and within, the watercourse. Bank erosion and accumulation of excreta along the riparian zone in an extended dry period can be substantial and can result in initial high suspended sediment and nutrient loads in the first overland flows.	Control

• What is the dominant vegetation type around the waterbody? Is it natural? • Is there fringing vegetation around the temporary water? If so, is it intact? • Does it shade the waterbody? • Is it a likely source of leaf litter or fallen fruit?	Vegetation types in the catchment of the waterbody control the nature of organic matter that enters the waterbody. Shading can strongly influence diel temperature cycles and also diel dissolved oxygen cycling. Decomposing plant material in the waterbody also affects the limnology, including turbidity, SPM and oxygen environment.	Control
• Do fine sediments enter the waterbody? Is their passage and/or composition altered by land use or fringing vegetation cover? • Are the fine sediments likely to be contaminated with adsorbed materials (e.g. metals, pollutants)? • Is the basin shallow (<1 m deep), steep-sided or wind-exposed? • Is the water always turbid? • Are the bed sediments fine and what is their composition (i.e. clay, sand, gravel or bedrock)? • Are they rich in organic matter (rotting leaf litter and ‘ooze’)? • Is the water tea-coloured (rich in dissolved organic matter)?	Fine sediments will usually be carried into a waterbody to some extent, but the flux of them will be influenced by land use, specific sources of fine sediments (e.g. roads, construction activities, mining and other landscape alteration) and natural (e.g. drought) and anthropogenic (e.g. overstocking) alteration of vegetation. The chemical characteristics of these sediments will have strong influence on water characteristics, and their interactions with other substances can be a control of bioavailability of other stressors. These processes need to be taken into account in the conceptual model.	Control / stressor

Questions	Implications for conceptual model	Control or stressor model
Hydrology, water budget and hydrological connectivity		
• How often does the basin become inundated each year? • Is it decadal, is the frequency and timing (e.g. seasonal) of inundation predictable? • For how long is it inundated? • What is the approximate volume of the basin (noting that the rates of drawdown by evaporation will often be faster in smaller volumes)?	These questions relate to the predictability of timing and duration of inundation of the waterbody. These influence both the development and predictability of biogeochemical processes and the likely drivers of aquatic ecosystem biological composition. More predictable inundation regimes will typically lead to more predictable chemical and biological processes. Highly unpredictable inundation and short wetted periods will lead to less predictability in water quality and ecosystem composition, which will need to be accommodated in the conceptual model and in the selection of indicators and development of monitoring designs. Such unpredictability would increase the potential utility of selecting multiple indicators and the use of a flexible weight of evidence approach. Potential contamination risks vary considerably over a hydrograph. In many arid systems, contaminant (including sediment) transport occurs during episodic storm events, and these may be difficult to detect and monitor. Limited dilution may also exacerbate potential impacts. These considerations are a prerequisite for determining when, where and what to monitor to detect these episodic impacts.	Control
• When the basin fills, does it overflow to fill other basins? • What are the sources of water to the basin? • Is there input of groundwater and what is the water quality of the groundwater (especially the salinity)? • What is the composition of dissolved salts in these inflows? • Is the basin filled by water flowing along a channel? Or is early filling primarily by overland flow from the catchment? • Is the water entering the basin tea-coloured (rich in dissolved organic matter) and/or nutrient-rich? • What are the outputs of water from the basin? • Is there substantial loss to the groundwater or through evaporation or transpiration?	These questions relate to the likely initial water quality characteristics of the waterbody and also how similar it is likely to be to other waterbodies in the catchment. Initial inflow water characteristics set the starting point for subsequent development of water quality during the wet phase, and in turn can constrain the initial biological colonisation after inundation. Connection to groundwater will affect these initial conditions and the path of development of water characteristics, and potentially the duration of the wet phase in each cycle. Inflows of waters high in dissolved organic carbon can strongly influence the bioavailability of some toxicants under the initial conditions, while inputs of nutrients can be a stressor for the aquatic ecosystem.	Control / stressor
Biophysical components potentially affecting water quality		
• What are the main sources of primary productivity in the basin? • Are algal blooms evident (and likely to affect dissolved nutrient concentrations and daily cycles of dissolved oxygen and pH)? • Is leaf litter abundant in the basin and likely to support detritivores? • Has the waterbody been colonised by amphibious or terrestrial vegetation during the drying phase (on re-wetting, amphibious or terrestrial vegetation can directly affect water quality)? • Are there aquatic or semi-aquatic weeds such as water hyacinth, salvinia, para grass or hymenachne present?	Primary production is the driver of development of all other biological components of the wet phase ecosystem in the waterbody. Understanding the main sources of it will help identify key risks to overall ecosystem productivity, such as via increased turbidity and fluctuating dissolved oxygen if primary production is naturally predominantly *in situ*. Phytoplankton and macrophytes seasonally-senescing in the waterbody also affect the limnology, including turbidity, SPM and oxygen environment. Leaf litter and other particulate organic matter are often important sources of food for aquatic fauna, and are sources of dissolved organic substances that can influence the bioavailability of some toxicants. However, native aquatic organisms cannot make direct use of some types of leaf litter, including exotic plant leaves containing natural toxins and many types of grasses that do not enter the food chain except via microbial decomposition. Knowing these constraints on productivity and sources of organic matter can be useful in the conceptual model to assess impact and amelioration pathways and key potential ecosystem sensitivities to water quality changes. This knowledge, in turn, can aid the selection of indicators and WQOs.	Control / stressor
• Once flow ceases, does the waterbody stratify; that is, do any key water quality parameters (e.g. temperature, dissolved oxygen, pH, salinity and/or turbidity) vary significantly with depth? If so, what is the duration and frequency of stratification episodes? • Is the waterbody frequently exposed to winds that are strong enough to create wave action? • Does the wave action promote surface mixing and, therefore, reaeration (i.e. exchange of oxygen and carbon dioxide between the air and the water)? • How often and over what proportion of the waterbody does the depth of surface mixing exceed the standing water depth? • When and where does the euphotic depth exceed the standing water depth? That is, when and where does the benthos receive sufficient light to support primary production? • How much do temperature, dissolved oxygen and pH vary over the course of each day?	Diurnal stratification is a very common characteristic of most lentic waters (for much of the year in the tropics and during summer months in cooler climates). Accordingly, water samples and meter readings collected near the water surface during daylight hours may not be representative of conditions at the benthos or of the conditions that will develop overnight. Deep and/or sheltered waterbodies may remain thermally stratified for months at a time, in which case the differences between the surface and benthos can become dramatically different. In such cases, the warm, well-mixed surface water layer (the epilimnion) is typically normoxic with relatively low concentrations of dissolved contaminants, while the cool bottom waters (hypolimnion) are typically characterised by low oxygen levels and high dissolved contaminant concentrations. The distribution of particulate contaminants through the lentic water column is largely independent of water stratification and is dictated by settling rates (a function of the size and density of the suspended particles) and resuspension processes (a function of wave action and disturbances by animals). Typically, concentrations increase with depth, and in waterbodies with clear waters near the surface there is often a distinct nepheloid layer just above the benthos (i.e. a layer of water with very high turbidity). In such cases, the benthos may not receive sufficient light to sustain benthic algal/plant communities, and productivity is mainly confined to phytoplankton and/or emergent macrophytes.	Control / stressor
• Do fine bottom sediments smell of 'rotten eggs' when stirred up (indicating likely anoxia and hydrogen sulfide, with potential effects on pH) and/or are sediments and lower surfaces or rocks blackened (indicating past anoxia)? • Are dried algal mats present as indicators of previous extensive algal growth? • Is salinity likely to affect the composition of aquatic biota (most freshwater groups tolerate <3 g/L, many decline or disappear when salinity is >10 g/L, very few tolerate >50 g/L, but, depending on the composition of the salts, these thresholds for change can be lower (Dunlop et al. 2008, 2011, Nielsen & Brock 2009, Prasad et al. 2012, Beyer-Robson 2015).	Strong anoxia, particularly leading to generation of hydrogen sulfide, is a biogeochemical process that can preserve some toxins (preventing decomposition of some organic toxins) or greatly reduce bioavailability of others (most metal sulfides are poorly soluble), but with wetting–drying cycles can also lead to periods of generation of acid from oxidation of sulfides to sulfates (Gómez *et al.* 2017). Knowing whether or not these processes are likely to be prevalent or controlling factors in the waterbody will be a key input to the conceptual model. Similarly, indications of strong salinity cycles or pulses and algal blooms will be useful to characterise the likely effects of these on the aquatic ecosystem.	Stressor

Questions	Implications for conceptual model	Control or stressor model
• Are terrestrial animals likely to affect water quality (e.g. nutrient inputs through excretion)? • Is there evidence of pugging and trampling, disturbing bed sediments? • Are there large congregations of waterfowl? • Are there any major bird or bat colonies in the immediate vicinity? • Are carp or other exotic or native fishes present? Are they likely to affect water quality by excretion or feeding activities? • Does access by these animal groups differ between or within wetting–drying cycles.	Stock and feral animal influences on the physico-chemical characteristics of the waterbody can be strong in areas of high stocking rates and/or feral animal densities or when extensive dry periods limit the opportunities for stock to access waterbodies (Steward et al. 2018). Similarly, exotic aquatic fauna and flora can also influence abiotic characteristics of the waterbody as well as potentially provide competition or predation pressure on native aquatic biota. These influences need to be accounted for in the conceptual model and may constrain potential indicators suitable for assessing other water quality characteristics. For example, in a waterbody that regularly experiences high rates of carp feeding on macroinvertebrates, the latter group might be an insensitive indicator for toxins. Feral animals can have significant localised impacts; care must be taken when selecting the locations of reference sites or test sites intended to detect broader scale anthropogenic impacts rather than localised wildlife impacts.	Control / stressor

2.1.3 Water quality temporal changes during the wetting–drying cycle

Figure 6 does not capture how water quality variables in a temporary waterbody change over time, especially during different phases of the wetting–drying cycle. Therefore, a second conceptual model is needed to portray likely trends in parameters such as water temperature, dissolved oxygen, conductivity (salinity), nutrients and pH (Figure 7). When the basin first fills from surface runoff, water temperatures may be moderate, decline during the filled phase, and rise as the basin dries (usually during warm dry seasons). Dissolved oxygen saturation may be quite high on first filling, especially if there has been physical turbulence during the process, but often declines quickly as it is consumed by microbes consuming carbon that accumulated as litter or terrestrial vegetation during the dry phase (Hladyz et al. 2011). Dissolved oxygen may then be generally high during the filled phase notwithstanding typical diurnal variation, before declining during the drying phase (Figure 7) as water temperature rises and organic matter in the basin breaks down. There are often wide diurnal variations of dissolved oxygen and pH in response to alternating peaks in photosynthesis (day) and respiration (night), especially as water volumes decline with drying.

Initially, salinity (often measured as electrical conductivity) is often high, reflecting the high concentrations of dissolved salts and organic acids in surface runoff, but it tends to decline through subsequent dilution before steadily increasing during the drying phase (Figure 7) due to evapoconcentration. Newly-wetted sediments often release large amounts of bioavailable nitrogen (e.g. Baldwin et al. 2005) and, more occasionally, phosphorus (e.g. Schönbrunner et al. 2012), before these are taken up by primary producers such as algae. During drying, dissolved nutrient concentrations may increase again because of evapoconcentration (e.g. Scholz et al. 2002), but these trends are not consistent and depend on other variables such as shading, turbidity and algal biomass, as well as biotic and abiotic sediment processes (Baldwin & Mitchell 2000, von Schiller et al. 2017). Temporal trends in pH also vary; it is often low initially because of humic acids leaching from newly-wetted leaf litter, rises during the filled phase, and then declines as the basin dries (Figure 7) and organic matter accumulates, leaching humic acids and other dissolved organic compounds (Boulton & Lake 1990). These general trends are often overlaid by diel cycles in temperature, dissolved oxygen and pH, and are influenced by the extent to which stratification of the water develops and breaks down.

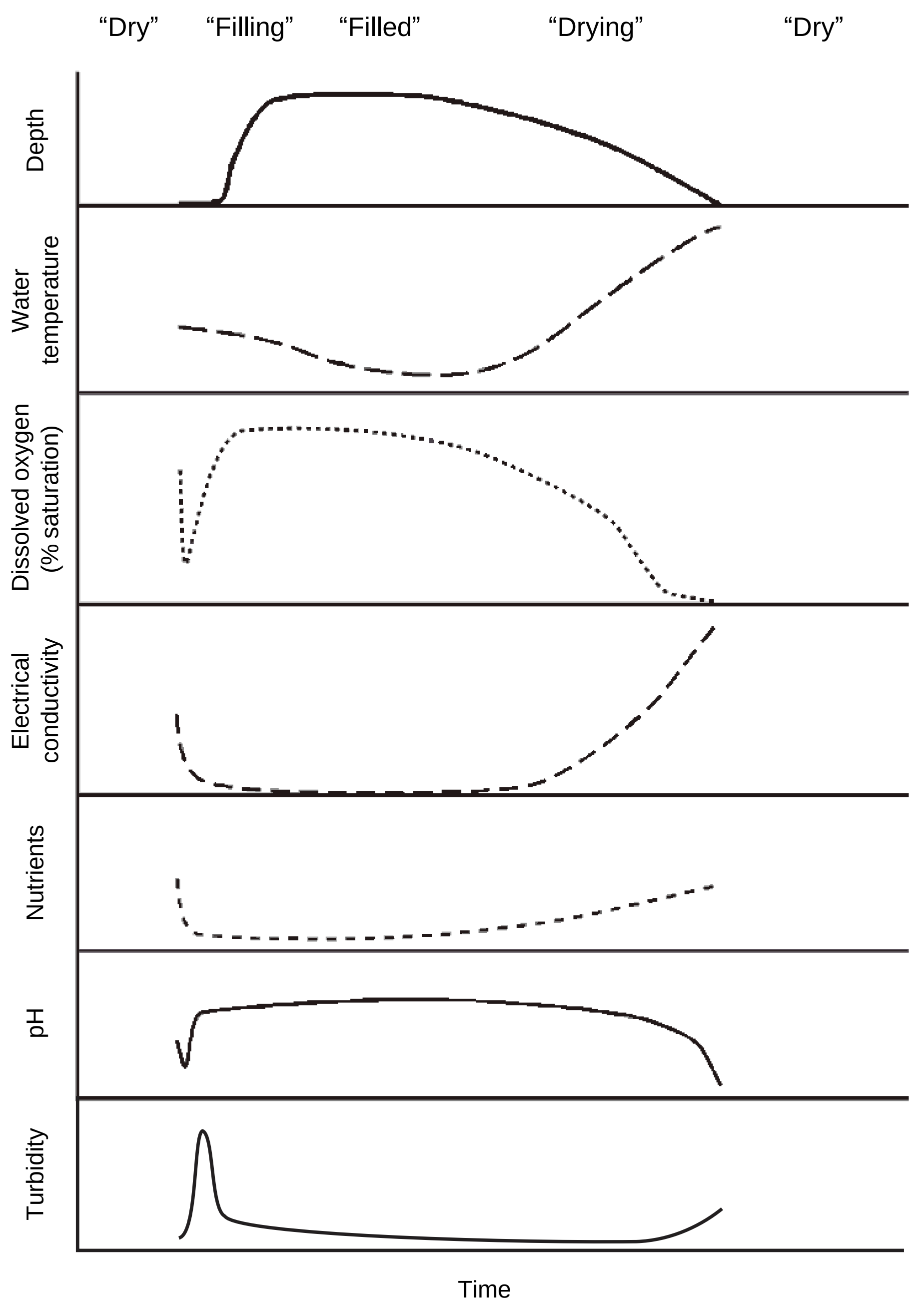

Figure 7 Illustrative conceptual model, likely changes in water quality in temporary pool during wetting–drying cycle consecutive phases

Adapted from Boulton et al. (2014).

Note: this model assumes that there is no connection with groundwater and rapid dissolution of salts on filling.

Examples of the use of conceptual models

The following examples are not meant to be templates for the use of conceptual models for temporary waters; rather, they are examples of the utility of conceptual models. Each temporary waterbody is unique, and the circumstances of the application of the WQMF will be unique in each instance. This is why site-specific conceptual models are useful.

Turbidity in the Lake Eyre–Bulloo

The rivers and streams of this biogeographic province are part of a vast catchment and have the unusual characteristics of extremely variable flows and, when flows are high, connection to extensive temporary lakes and wetlands. The Queensland Government developed conceptual models for the catchment (Figure 8).

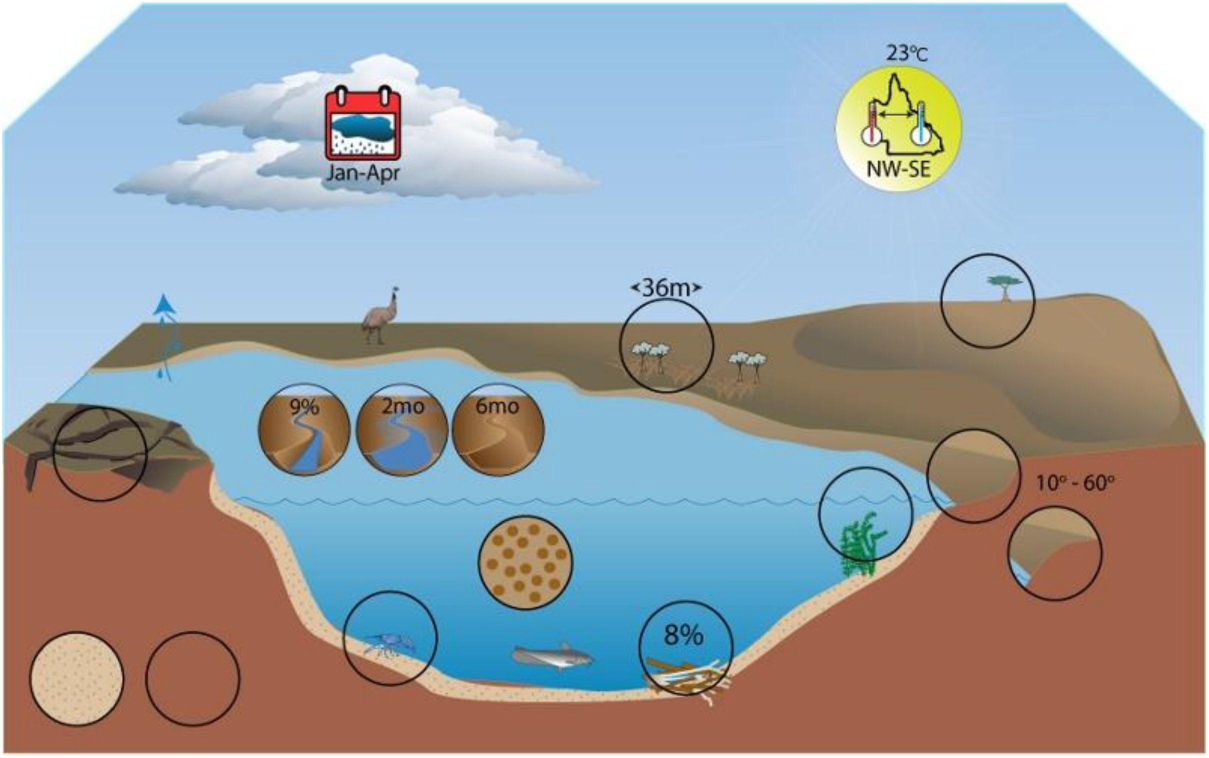

Figure 8 Conceptual model, Lake Eyre–Bulloo systems

Source: Queensland Government.

Turbidity tends to be high in these systems, averaging 359±31 NTU, but can be lower in upland areas. Hence, the input of turbid discharges (e.g. runoff from grazing land) will be into waters that are already turbid. The Wetland Info website depicts a control model of the main controls and features of the Lake Eyre–Bulloo systems and provides text describing the stressor models. The text describes the influence of turbidity on photic depth, and how photic depth affects primary production. The primary production of these systems is mostly from littoral zone algae, which, despite the high turbidity and limited area of the littoral zone within the photic depth, can make the systems net carbon producers. Therefore, further increases in turbidity caused by the turbid discharge (if above the background turbidity) would reduce the photic depth, and so reduce the amount of primary production. This information could be depicted in a simple manner, as shown in Figure 9.

↑Turbidity → ↓Photic Depth → ↓Primary Production

Figure 9 Stressor model, turbidity effects in Lake Eyre–Bulloo systems

Thus, on the basis of these *control* and *stressor* models, a key potential impact of increased turbidity in the receiving system would be reduced primary production, assuming primary production sustained at natural levels is an important management goal for the water. This conceptual understanding could be used to develop water quality guideline values, which could, for example, include a maximum increase in the average turbidity, a maximum reduction in the photic depth, and a maximum reduction in the average primary production rates. Hence, potential indicators would be direct measurement of average turbidity, but also indirect measurement of turbidity, such as the average photic depth, and/or measurement of net primary production in the littoral zone or measurement of the littoral zone algal assemblage status (biomass, species composition).

The use of control and process conceptual models facilitates understanding a key potential impacting process, and highlights indicators that could be used to provide multiple lines of evidence to assess water quality. In systems with highly variable flows, it can be problematic to take direct measurements of physical properties (such as turbidity) due to maintenance issues with automated samplers being unpredictably exposed and inundated, and difficulty in accessing the watercourse and monitoring sites during discharge. Hence, the use of multiple lines of evidence provides additional reassurance that an assessment can be made when such problems arise, particularly a biological measure that can integrate ecosystem response over a rainfall/discharge event when direct measurements may fail.

Desert salt lakes

Desert salt lakes mostly lie within very long, flat palaeovalleys in areas of very high annual evaporation and low rainfall. Salt lakes represent areas of shallow water table within the paleochannel where there is discharge of hypersaline groundwater to the surface through percolation and evaporation. Salt lakes may remain dry for several years at a time. Large flood events resulting in surface flow in the paleochannel are very rare, but the accumulation of small amounts of free surface water after moderate rainfall events, as a result of direct precipitation on the lake bed and its immediate catchment, are not unusual. Salinity usually varies from hyposaline to saline when free water first appears and then becomes hypersaline and reaches saturation as the lake dries. The ecology of desert salt lakes revolves around the pattern of flooding and salinity, with species adapted to a short hydroperiod, hatching in the relatively low salinities of the initial inundation and then senescing as salinity becomes too high or the water dries (Boulton et al. 2014), as shown in Figure 10. Discharge of hypersaline groundwater into a salt lake, which may occur when an adjacent below-water table mine pit is dewatered to provide access to the mineral resource (e.g. as occurs in Western Australia; Gregory et al. 2009), may not substantially increase salt loads but can alter the natural temporal dynamics of salinity by raising the base salinity prior to flooding, and this can lead to impaired ecological responses (Figure 11) (Gregory et al. 2009, Outback Ecology 2009). Another issue for desert salt lakes in Western Australia is that of potash mining, where the groundwater and hyporheos are dewatered in order to extract potassium salts. This activity can also potentially impact the water quality and ecology of salt lakes due to various processes, including increased turbidity due to sediment mobilisation, acidification and mobilisation of metals due to oxidation of shallow sediments, and increases in salinity due to saline discharges and increased infiltration of runoff (Environmental Protection Authority 2019).

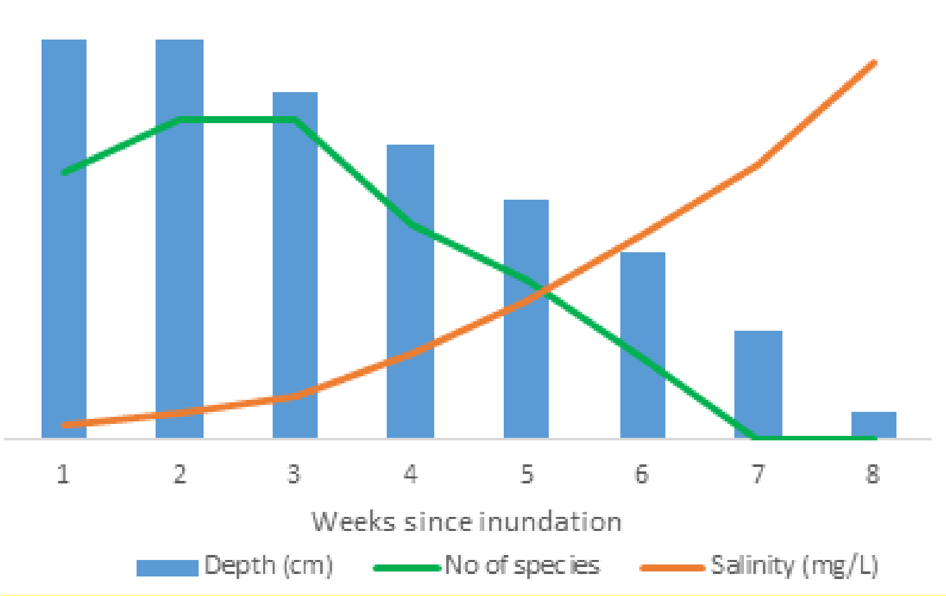

Figure 10 Conceptual model, main interactions in desert salt lakes

↑Hypersaline discharge → ↑Initial salinity → ↓Biological diversity

Figure 11 Stressor model, effects of hypersaline discharge into salt lakes

Understanding the nature of these human activities in Step 1 of the WQMF is critical as it will inform discussions and decisions at subsequent steps of the framework about whether or not such activities should occur at all and, if so, what should be protected, what regulatory actions are required and how the monitoring can be appropriately designed for compliance assessment. Considering just the specific case of dewatering discharges to salt lakes, a framework for assessing and managing potential impacts has been prepared for the Western Australia government (Outback Ecology 2009).

On the basis of these simple *control* and *stressor* models (Figures 10 and 11), the potential impact of increased salinity in a salt lake receiving hypersaline groundwater discharge would be reduced biodiversity. While this conceptual understanding could be used to develop objectives for the nature of the discharge and the expected changes in salinity around the discharge site, in practice the large size of desert salt lakes and their complex patterns of flooding and wind-driven water movement can present a challenge for assessing the extent of compliance.

Examining potential impacts on desert salt lakes in the context of their natural hydrological patterns and biological responses provides a framework for selecting potential indicators to assess whether there has been meaningful changes to water quality. However, in large desert salt lakes that do not regularly flood, it is difficult to determine the impact of localised changes as a result of saline discharge (or other factors) on the ecology of the lake as a whole. The selection of appropriate sites for monitoring may also be difficult unless the pattern of water movement within the lake is well understood (e.g. via hydrodynamic modelling and monitoring prior to any discharge). Notwithstanding these difficulties, potential indicators of the effects of discharge salinity include salinity during the first few weeks of inundation and numbers of species present.

Natural and anthropogenic turbidity variations of arid and semi-arid tropical inland waters
Many of the river systems of northern Australia are episodic, flowing briefly with intense rain events arising from tropical lows. While the water quality of the stormwater generated by such events reflects overall catchment condition, most of the time the river systems fragment into a mosaic of disconnected and often hydrogeomorphologically distinctive waterbodies. The water quality conditions within each waterbody each time flows cease can vary substantially, depending on the size and duration of the flow event and antecedent catchment conditions (i.e. each flow event potentially resets the system to a different point). During prolonged periods between flow events, subsequent water quality variations are governed by localised influences that are independent of the rest of the catchment (Preite & Pearson 2017). These characteristics present unique challenges for water quality assessments, most notably the difficulty obtaining meaningful reference data.

The conventional approach of establishing reference and test sites upstream and downstream of an impact source, respectively, is rarely valid, especially once the system becomes fragmented and each site takes on a trajectory governed by its inherent natural stresses (e.g. evapoconcentration rates increase exponentially with decreasing water depth, so even minor between-site differences in bathymetry can lead to substantial differences in water quality). Moreover, due to the flashy nature of these streams, the event hydrograph often comprises a series of brief peaks, each of which generates a pulse of stormwater contaminants of highly variable composition. The time taken for each pulse to travel from an upstream reference site to a downstream test site can vary by orders of magnitude over the course of each event, making it difficult to make valid comparisons between reference and test data unless sampling has been quite intensive. Correspondence between reference and test sites can also be lost during large scale rain events if groundwater levels at a test site rise to the point where a losing stream (which carries only surface runoff) is temporarily transformed into a gaining stream (which carries a significant proportion of groundwater-derived baseflow).

Since stormwaters are typically very turbid, whereas baseflows and hyporheic (below stream bed) waters are comparatively clear, the spatio-temporal variability in water quality is visually evident in ground observations and remote sensing imagery (e.g. Lymberner et al 2007). Available evidence (e.g. Bartley et al. 2018) indicates that the quantities of fine suspended sediment currently being exported to the coast from arid and semi-arid tropical rangelands (and by extension the turbidity concentrations contained in stormwaters) are three to eight times greater than pre-European levels. This is a potential issue for coastal environments and some off-channel/floodplain wetlands where siltation effects may be evident, but the majority of the fine (clay and colloid) sediment load passes rapidly through the main river channels without settling (as evidenced by the coarse-grained texture of basal sands). Residual effects on ambient water quality in the rivers are not always evident and often vary among sites and between flow events, depending on the intensity and duration of the flow event and the amount of groundwater-driven baseflow generated at each site. At one extreme, there are rivers that never experience significant baseflow and, therefore, always retain turbid stormwater; at the other extreme, there are spring-fed systems where stormwaters are rapidly displaced by clear baseflows on the receding limb of the storm hydrograph.

However, many sites fall between these two extremes, potentially receiving enough baseflow to run clear only if there is sufficient rainfall to increase groundwater levels above a certain critical point. Figure 12 and its figure notes illustrate how these *natural* hydrological processes can influence the turbidity of the water retained in the river system after flood flows have ceased, while Figure 13 and its figure notes illustrate how anthropogenic activities can alter these relationships. The information provided in Figure 12 and Figure 13 is derived from studies conducted in the free range grazing areas of inland river catchments in the arid and semi-arid (drought-prone) tropics of northern and north-western Queensland. Note that the suspensoids responsible for the turbidity are typically too small to settle unless they flocculate. Accordingly, the turbidity present at cease-to-flow persists for very long periods unless electrical conductivity or pH levels change to the point where flocculation occurs.

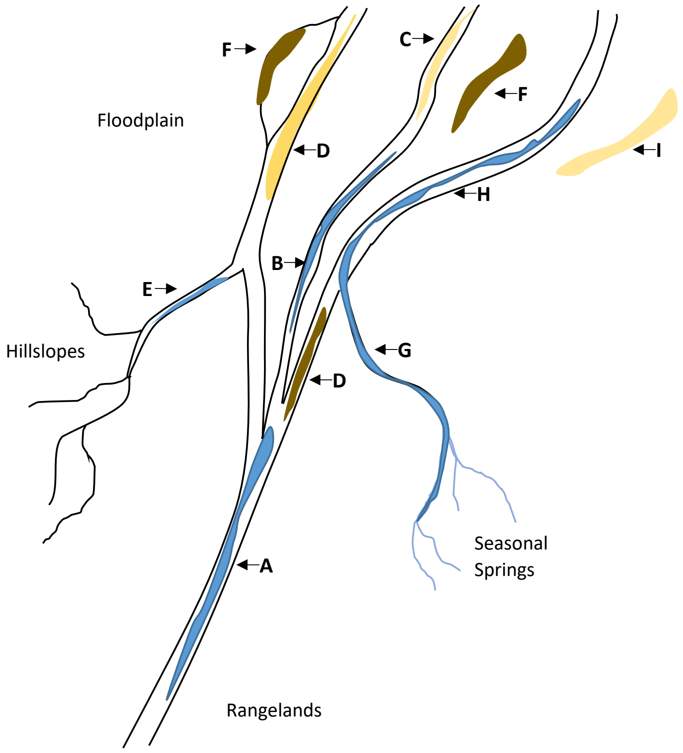

Figure 12 Process model, suspended particulate matter and turbidity, pre-European, dry season, post-flood

Note: Shaded sections in figure correspond to low (blue), moderate (yellow) and high (brown) turbidity.

A Water is clear because groundwater-driven baseflows have been sufficient to wash turbid flood water downstream. Systems that carry little baseflow may always remain turbid (as in **D**).

B Water beginning to clear and will become clear if baseflow persists.

C Similar to **B**, but baseflow needs to be sustained for a longer period to remove turbidity.

D Overflow channels only receiving water from **A**, when close to bank full. These reaches become isolated early on the falling limb of the flood hydrograph when water still turbid.

E Water in this temporary/seasonal waterhole runs clear before flow ceases, indicating that the sub-catchment has been effectively flushed (sediment supplies exhausted) and/or that there has been sufficient baseflow to wash away turbid stormwater.

F Off-channel wetlands (oxbow lakes, palustrine wetlands, billabongs) only connect to the river on the peak of the flood hydrograph; therefore, they often retain highly turbid water depending on macrophyte extent, persistence and timing of establishment.

G Due to floodwaters recharging local springs, this tributary rapidly runs clear on the falling limb of the hydrograph and sustains strong baseflow for much of the year.

H This braid of the main river also runs clear due to baseflow contribution from **G**. In very wet years, baseflow may persist for most of the year.

I Similar to **F**, but the water is not as turbid because the wetland is on the eastern side of the floodplain where the flood plume is less turbid due to lower sediment inputs from the sub-catchment of tributary **G**.

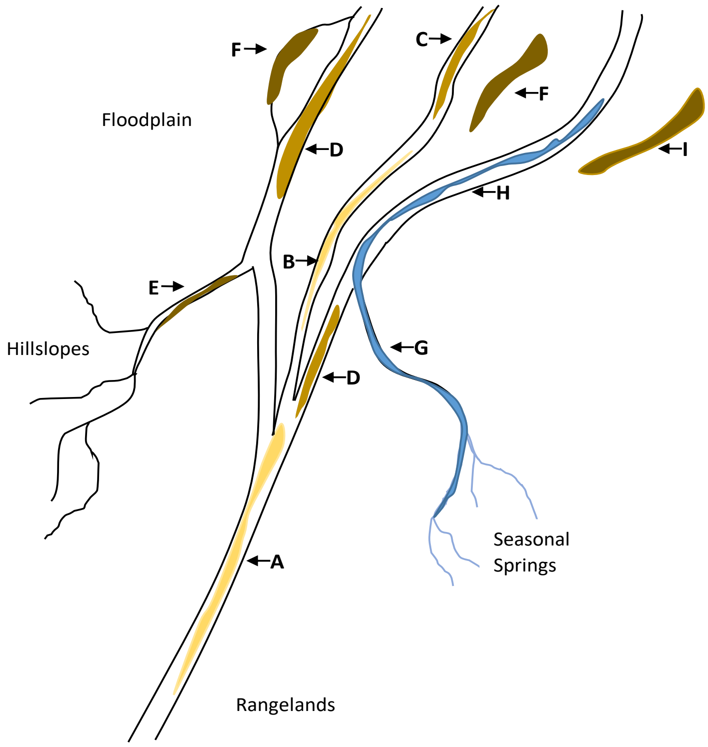

Figure 13 Stressor model, suspended particulate matter and turbidity, post-European, dry season, post-flood

Note: Shaded sections in figure correspond to low (blue), moderate (yellow) and high (light brown) and very high (dark brown) turbidity.

A Disturbance pressure from livestock and feral animals throughout the dry season may prevent groundwater fed sites from becoming clear. Systems that carry little baseflow will likely be more turbid (as in **D**) and *vice versa* (as in **G** and **H**).

B Similar to **A**, but may become more turbid than **A** if baseflow is not sustained, and as isolated pools become areas of congregation for livestock and feral animals.

C Similar to **B**, but may become more turbid than **B** if baseflow is not sustained, and as isolated pools become areas of congregation for livestock and feral animals.

D Overflow channels only receiving water from **A**, when close to bank full. These reaches become isolated early on in the dry season when water still turbid, and will become more turbid throughout the dry season as disturbance pressure grows.

E Temporary/seasonal waterhole that is clear at the beginning of the dry season and will become turbid during the dry season due to increasing disturbance pressure.

F Off-channel wetlands (oxbow lakes, palustrine wetlands, billabongs) that are generally highly turbid due to retention of turbid water during peak flood flows may become progressively more turbid throughout the dry season as disturbance pressure grows.

G Due to the available dilution, dispersion and flocculation capacity provided by sustained baseflow, groundwater fed sites generally remain clear during the dry season.

H Similar to **G**.

I Moderately turbid off-channel wetland at the beginning of the dry season that will become progressively more turbid throughout the dry season as disturbance pressure grows.

Numerous anthropogenic pressures (including sand extraction, mining, recreational uses, road crossings and irrigation runoff) can impact on turbidity levels, but impacts from feral and domesticated livestock are the most widespread across the landscape (Steward et al. 2018). Fine-grained sediments rarely consolidate when they settle to the bottom (sometimes forming thick floc layers near the bottom) and are readily resuspended if disturbed. Therefore, isolated lentic waterbodies are particularly susceptible to physical disturbances from animals, especially species that wallow such as feral pigs, horses and water buffalo. The potential effects of this on turbidity are depicted in Figure 13. The severity of impacts is inversely proportional to the size and volume of each waterbody. However, factors such as accessibility to livestock, stream bed morphology and aquatic community types also have a bearing. For example, certain wetland plants attract feral pigs.

From an ecosystem receptor perspective, the sensitivity of aquatic communities to turbidity varies depending on the typical state of the waterbody. Typically, clear waterbodies develop significant autotroph communities that are particularly sensitive to changes in water transparency, and even moderate increases in turbidity can impair these communities with flow-on effects for rest of the ecosystem. Lentic sites that support high autotrophic biomass are most susceptible because they often rely on photosynthetic production to sustain oxygen levels. Such sites can experience severe oxygen sags if turbidity levels become high enough to prevent photosynthesis. In contrast to these sensitive waterbodies and communities, naturally turbid waterbodies support smaller autotrophic communities due to the lower light availability beyond the first few centimetres below the surface. Accordingly, such ecosystems are comparatively insensitive to transparency variations caused by turbidity fluctuations. However, because such ecosystems are net consumers of oxygen, they are also potentially vulnerable to severe oxygen sags due to the biological oxygen demand associated with turbidity.

The pressures and associated effects on these systems are substantially exacerbated during droughts. Groundwater levels decline, springs run dry, wet season baseflows are inadequate to displace turbid stormwater, and so clear waters become rare. Waterbodies are smaller and fewer, instream biomass is concentrated, respiration (oxygen consumption) rates are high, water temperatures are elevated, livestock is thirstier and green feed is largely confined to riparian zones.

Practical ramifications for monitoring and assessment

The complexity of turbidity variations in inland waters needs to be considered when undertaking monitoring and assessment. Key considerations include:

- Ambient turbidity levels are determined by localised hydrological conditions and are not a reliable indicator of the quantity of sediment exported to downstream environments during the peak of the hydrograph.
- Turbidity variations play a fundamental role in determining the metabolism, function and community structure of these aquatic ecosystems. Accordingly, when selecting reference sites for biological assessments, turbidity regime and related factors such as oxygen and temperature status must be factored into site selections.
- The random variability component of natural water quality fluctuations is so high that the prospects of successfully employing referential methods for quantitative water quality assessment are very poor. Such methods would be unsuitable for regulatory compliance assessment applications.
- The ANZECC/ARMCANZ (2000) recommendation to employ the most recent two years of reference data for assessing current conditions is not valid in highly dynamic drought-prone landscapes. Alternative approaches need to be considered for monitoring and assessment.
  - The revised Guidelines (ANZG 2018) recommend modelling approaches for monitoring and assessment that consider stressor relationships to hydrological conditions, such as discharge, more meaningful.

  - In the absence of these modelling approaches, it would be more defensible to use the historical reference data that best represent contemporary hydrographic and climatic conditions.
  - An alternative approach is to use an extended network of reference and/or control sites and to use spatial variability as a surrogate for temporal variability at the site. Noting the high chance that such a network of sites introduces isolated waterbodies, each of which is subject to independent localised natural influences, the biological characteristics of these and additional sites of putative impact could be modelled with physical and chemical stressors to derive a regional-specific, biological effects-based guideline value (see ANZG 2018; Deriving guideline values using field effects data).

- During droughts, clear waters are regionally rare and often largely confined to highly saline or spring-fed systems. Spring-fed systems are potentially vital regional drought refugia for a number of taxa (Davis et al. 2013) and should be a focal point for management measures, such as installation of fencing to exclude livestock and provision of off-stream watering points.
- Sites that are naturally sufficiently turbid to be light-limited are relatively insensitive to further turbidity increases, but they may be vulnerable to impacts from associated contaminants such as organic matter, biological oxygen demand and nutrients. At such sites, turbidity alone is not a reliable indicator of impacts. Conversely, sites that are sufficiently clear to be autotrophic are sensitive to the light attenuation effects caused by even moderate turbidity increases—turbidity is a relatively sensitive indicator of potential impacts at such sites.
- The most significant turbidity-related impacts are often episodic and severe (potentially resulting in acute mortality events such as fish kills). Such incidents are not reliably detected in random sampling programs, especially if sampling frequencies are low. Similarly, conventional statistical measures of central tendency, variance and trend may not be effective tools for assessing annual outcomes for the ecosystem in situations where a site maintains good conditions for most of the year but suffers a single catastrophic event prior to the next flow event. However, these considerations need to be weighed against the fact that ecosystem response is related to the timing in the wetting–drying cycle and the duration of the high turbidity exposures, not just the maximum levels reached.
- The turbidity of the water contained in each waterbody at the beginning of each dry season varies across years and is dependent upon frequency, intensity and duration of wet season rainfall, existing groundwater levels, presence or absence of aquatic plant species, antecedent catchment conditions, and the volume, duration and quality of baseflows.

Other sources of conceptual models relevant to temporary waters

Among other sources of conceptual models relevant to temporary waters, those developed under the Bioregional Assessment Program, which has examined the impacts of coal seam gas (CSG) and large coal mining developments on water resources and water-dependent assets over six bioregions in Australia, may provide further data, lists and descriptions of assets and conceptual models that are relevant beyond their main (CSG/coal development) purpose. The conceptual models depict causal pathways of how the bioregion or subregion works and how it might respond to development. They are a collection of evidence-base narratives, diagrams, graphics and hypotheses represented as a set of nested conceptual models focusing on certain parts of the bioregion or subregion, or portraying alternative conceptualisations or hypotheses about the systems.

2.2 Step 2 – define community values and management goals

Concurrently with Step 1 (Section 2.1), stakeholders set the management goals for water quality management of the waterbodies of interest. These goals relate to the appropriate community values to be protected, which broadly include aquatic ecosystems, primary industries, human health, and cultural and spiritual values. Large parts of arid and semi-arid Australia are under native title or Indigenous tenure, making cultural and spiritual values often of key importance for stakeholders.

Given the dynamic nature and high temporal variability intrinsic to temporary waters, users need to allow for inter-wetting and intra-wetting and drying cycle variability in the community values and management goals to be protected, and recognise that emphases may shift among these values (e.g. Datry et al. 2018) and associated goals over those cycles. Being fundamental to water quality management, the management goals are used to focus efforts at all subsequent WQMF steps. This step is usually conducted concurrently or iteratively with Step 1 because the management goals determine which values are to be considered and the associated level of effort required for developing the *current understanding*.

Normally at this step, the ecosystem condition and associated levels of protection are selected for the aquatic ecosystem community value. Some temporary waters may be assigned, *a priori*, high conservation value (e.g. particular mound springs, streams under wild rivers protection, and other systems in national parks), with an associated high level of protection. Otherwise, the appropriate level of protection for temporary waters may not be so straightforward to set, as these systems commonly have cosmopolitan taxa with high taxonomic turnover and stochastic determination (Smith & Pearson 1987, Bunn & Hughes 1997, Bunn & Davies 2000, Davies & Bunn 2003, Sheldon 2005, Sheldon et al. 2010), with overall biodiversity, which may be high, reliant on connectivity between a larger-scale network of waterbodies. Nor will the level of protection necessarily be set in a uniform manner across the waterbodies of interest because of that reliance on the network of waterbodies, including key refugia, to maintain overall biodiversity.

Refugia of aquatic (and terrestrial) ecosystem biota have an overarching importance for maintenance of biodiversity for the whole system and may require special consideration (e.g. perennial waterholes within a temporary stream network). While they may not be typically regarded as high conservation value waterbodies, within the context of the temporary water catchment or complex that they are part of, they may have a critical role to play in maintaining biodiversity, and so may warrant consideration for a higher level of protection than non-refugial waterbodies in the system (Davis et al. 2013). Key sites of reproduction and dispersal may also warrant special consideration. In short, the combination of spatial and temporal variability in inundation may impose spatial and temporal requirements on the setting of management goals and the allocation of levels of protection to specific waters.

2.3 Step 3 – define relevant lines of evidence and associated indicators

At this step, measures of the pressures, stressors and ecosystem receptors (indicative of the community values) of the system are selected. ANZG (2018) emphasises the selection of multiple lines of evidence to increase the credibility, rigour and reliability of water quality assessments. This approach is relevant to temporary waters where most indicators are highly variable in space and time, necessitating as complete a dataset (i.e. multiple lines of evidence) as possible to optimise interpretation. Furthermore, data collection opportunities (particularly for ephemeral and episodic waters) for water quality are inherently limited and often opportunistic, prompting exploration of other lines of evidence that do not rely on sampling the aquatic phase, and also maximising the information gained by collecting multiple lines of evidence when water is present. Note that the development of conceptual models in Step 1 (Section 2.1) will highlight the ways in which the multiple lines of evidence are linked and/or reflect different aspects of water quality, providing additional support for the selection of multiple lines of evidence.

Step 3 of the WQMF is also a common (but not sole) step for the commencement of monitoring to acquire data to derive water quality guideline values and assemble suitable chemical and biological baselines. The fundamental elements of water quality monitoring are discussed in ANZG (2018), but there are three aspects that need particular attention in temporary waters. The first relates to the temporal variation in most water quality indicators, from diel to seasonal scales (Figure 7).

- The diel nature of temperature, dissolved oxygen and pH can be substantial in many lentic waters (diel ranges greater than 10˚C, 150% oxygen saturation and 2 pH units are quite common), and diurnal stratification is very common (i.e. waters stratify during the day and mix at night). Accordingly, there are many lentic sites where meaningful data for these physical and chemical stressors cannot be obtained from spot samples and meter readings.
- The extent of variation during the wetting–drying cycle differs among indicators; some such as electrical conductivity show predictable trends during drying and evapoconcentration of salts, whereas others such as pH in poorly buffered waters might fluctuate in a highly variable manner and over a much broader spatio-temporal scale (from metres to kilometres and minutes to months). The frequency of sampling may also need to be tailored to the expected temporal variation in different indicators; for example, sampling may need to be more frequent (if possible) during the initial pulse of re-wetting for indicators such as electrical conductivity or dissolved oxygen (Figure 7).

The second relates to spatial variation. Not only will different water quality indicators interact with each other (e.g. electrical conductivity, dissolved organic carbon and pH), they can vary *within* a temporary waterbody, especially in stratified standing waters (Boulton et al. 2014). Where stratification or other processes (e.g. flocculation of suspended sediments) cause significant spatial variability, samples either need to be collected from different strata or collected as composite or integrated samples that combine sub-samples from different strata. It is wise to do a preliminary study that samples water quality from different parts of a temporary waterbody at several different stages of the wetting–drying cycle to establish the extent of spatial variation within the waterbody. This information can then be used to decide whether to collect composite/integrated samples or to sample separate strata for a given indicator or baseline dataset. Other factors to consider, or that may assist in determining where to collect samples from, include the following.

- The choice of biological indicators to accurately identify correlations between water quality and biological condition. For example, if invertebrate sampling is being conducted in edge habitats (which normally occupy the mixed surface layer or epilimnion), a surface sample would be required, but if pool bottom habitats are being sampled, a sample from near the bottom would be required. In both cases, an integrated sample could be misleading (e.g. surface dissolved oxygen levels could be cycling from 50% to 150% while the bottom levels remain consistently close to zero).
- Depth integrated samples are preferable for evaluating contaminant loads. For depth integrated sampling, it is important to ensure that samples integrate the entire depth of the water column to account for potential chemoclines within the metalimnion (i.e. partially mixed layers of water between the epilimnion and hypolimnion within which there is a contaminant concentration gradient).

Spatial variation in water quality also occurs among temporary waters, even those close to each other. Differences in waterbody bathymetry, fringing vegetation, extent of groundwater connectivity, features of the wetting–drying cycle and other drivers shown in Figure 6 often cause substantial inter-waterbody differences in water quality. The hydrogeochemistry of the reference sites, in particular, should be a valid match for the test sites. This is especially challenging in temporary waterbodies because they become isolated and are potentially subject to distinctive and sometimes unique local hydrogeological influences. Accordingly, astute reference site selection and the development of meaningful conceptual models requires a detailed knowledge of the local hydrogeology. For example, a spring-fed waterhole will have a geochemical signature characteristic of the source aquifer and could never be meaningfully compared to any other waters in the region unless they are fed by the same or a very similar aquifer. Also, the water quality of a site downstream of a sub-catchment with deep, highly dispersible clay soils or a natural mineralogical anomaly will always differ from that of a site in a rocky catchment with thin weathered soils or upstream of the mineral anomaly. Thus, before using only one or several temporary waters to represent a regional baseline of local water quality, a preliminary study of inter-waterbody differences in water quality, and at different stages of the wetting–drying cycle, is needed.

Thirdly, many temporary waters are remote and may be challenging to access when inundated. Such accessibility problems and the very high spatial and temporal variation in water quality over the wetting–drying cycle require suitable approaches for collection and measurement of water physico-chemistry. Some approaches to addressing these issues include:

- automatic samplers (refrigerated if necessary) triggered by events (or via telemetry)
- continuous or integrated monitoring
  - of stressors (loggers and telemetry) with potential to extend to direct measurement of some toxicants
  - of water level, particularly where waterbody bathymetry and/or invert (inflow/outflow) levels are known, or logging of inflows and outflows
  - use of passive samplers that integrate chemical concentrations over time (e.g. gel diffusion samplers, peepers, chelex-resin columns, and polar and non-polar organic molecule samplers)
- remote sensing and hyperspectral and other imagery (e.g. salts, turbidity, chlorophyll and (with verification of the approach) the frequency of presence of water)
- sediment chemistry, including pore water chemistry, as an archive of recent and past water quality, including the use of diatom stratigraphy in cores and radiometric markers and sediment physical properties (e.g. particle size distribution) and sediment erosion and deposition rates as indicators of alteration of sediment transport regime.

Further discussion of these approaches can be found in Moran et al. (2008) and ANZG (2018).

Similarly, where access during the wet phase is particularly challenging, and also for the less predictably inundated waterbodies, the use of surrogate/proxy datasets that can be obtained during the dry phase is also likely to be beneficial. Examples include:

- Direct Toxicity Assessment of potential discharges coupled with hydrological and/or geochemical modelling to provide a prediction of safe whole-effluent dilutions and probability of exceedance of them in the receiving environment

- assessment of sediment chemistry as a direct measure of sediment quality, as an archive of recent and past water quality, and a proxy of potential water quality during the wet phase and potential water quality detriment footprint (as informed by leaching tests)
- use of terrestrial phase (e.g. dry riverbed) assessments as surrogates for aquatic phase water/sediment quality assessments, such as terrestrial invertebrate health indices, riparian vegetation condition indices, and in pastoral areas, measures of stock access/trampling and defecation rates (e.g. Steward et al. 2018)
- assessment of hyporheic communities, which contain varying proportions of obligate hyporheic fauna and refuging surface water fauna that are a major source of recruitment associated with wetting–drying cycles (e.g. Stanford & Ward 1988, Boulton 2000)
- assessment of propagule (eggs, spores, resting stages) bank status as a proxy for *in situ* recruitment potential (e.g. Angeler & García 2005, Stubbington & Datry 2013)
- assessment of permanent refuge water, sediment and ecological status as an indicator of probable wet phase ecosystem health
- remote sensing and hyperspectral and other imagery to detect deposited salts.

Although some of these methods (e.g. use of terrestrial-phase invertebrate fauna for assessing dry riverbed health (Steward et al. 2012, 2018)) were not originally intended to be, and have not yet been tested as, direct surrogates of water quality in temporary waters, there has been some limited assessment of the effects of water quality on a few of the other potential surrogate measures. One of these is the invertebrate propagule bank in the sediments of temporary waters. For example, Skinner et al. (2001) experimentally demonstrated that increased salinity reduced the diversity of invertebrates emerging from propagules in re-wetted sediments of two River Murray wetlands. However, it should be noted that the propagules of many organisms respond only to select salinity ranges, and so the propagules that do respond may reflect the salinity used to extract responses in addition to antecedent salinity conditions. Similarly, the diversity of rotifer assemblages emerging from re-wetted sediments from Spanish temporary waters was reduced by high salinities as well as elevated nitrogen concentrations (Angeler et al. 2010). The hypoxic conditions associated with 'blackwater events' (i.e. high dissolved organic carbon and low dissolved oxygen) were experimentally shown to suppress the emergence of zooplankton from the re-wetted sediments of two floodplain temporary waters of the southern River Murray (Ning et al. 2014), although high dissolved organic carbon concentrations alone did not have a consistent effect.

There is still much to be learned about the interacting influences of water quality, variations in aspects of the wetting–drying cycle (e.g. duration of wetting, timing and predictability) and the effects of sediment characteristics (e.g. moisture retention) before the composition of the invertebrate or plant assemblages emerging from propagules in re-wetted sediments can be used as a reliable proxy for water quality in temporary waters (Chiu et al. 2017). However, this approach shows promise as a useful line of evidence to supplement other indirect measures of water quality sampled during the dry phase.

Given the inherent high variability of physical and chemical stressors in temporary waters, and the effects of first flushes, spates and evapoconcentration on them, ecological lines of evidence that integrate this variability in abiotic conditions through time (e.g. biological indicators such as invertebrate assemblage composition (Van den Broeck et al. 2015)) are useful inclusions for water quality assessment. However, the following factors can strongly influence the variability in the development of assemblages of organisms between wetting–drying cycles and, geographically, within temporary water networks:

- stochastic recruitment effects on assemblage development (e.g. Vanschoenwinkel et al. 2010)
- in-built genetic variability in timing and triggers for ending aestivation within populations ('spreading the risk') and among different species (e.g. Simovich & Hathaway 1997)
- physical and chemical constraints on assemblage successions and variability among years will require different benchmarking between inundation events. For example, the initial assemblage composition (and hence the process of ecosystem successional development) in salt lakes is contingent on the amount of inflow in the initial re-wetting of the ecosystem, with different taxa favoured by different salinities (e.g. Suter et al. 1995, Halse et al. 1998, Cale et al. 2004)
- changes in the relative input of surface water and groundwater flows (particularly to pools/refugia) at different phases of the wetting–drying cycle, with implications for water persistence and water quality
- the extent of connectivity between refugia and newly-inundated habitats, geographically and temporally, strongly affecting recruitment opportunities and sequences and hence the resultant biological interactions (Sheldon et al. 2003, 2010).

All of these factors will affect the selection of ecological lines of evidence and their value for detecting water quality changes in temporary waters.

Process-related indicators can be very useful in predicting the potential sensitivity of the waterbody to a potential stressor. Three key issues to consider are:

- how the bioavailability of a chemical stressor could be moderated by the system; for example, sulfate reduction/sulfide oxidation reactions, which modify metal bioavailability
- mobilisation and concentration processes within the system
- the sensitivity of the system biota to the stressor.

The salinity of the system and the patterns of salinity change will strongly influence these processes, as indicated in the conceptual models presented in Step 1 (Section 2.1). Dissolved oxygen cycling over diel and wetting–drying cycle timescales can also be a strong controlling process (Gómez et al. 2017). Rapid declines in dissolved oxygen, or oxygen sags, (e.g. Figure 14) often occur in isolated waterbodies and strongly affect biogeochemical and ecological processes and, hence, sensitivity to other stressors. Some stressors will be inter-related in terms of potential for detrimental impact. For example, many dryland rivers in Australia are highly turbid, conditions under which light limitation may control the level of ecosystem response to nutrient loading.

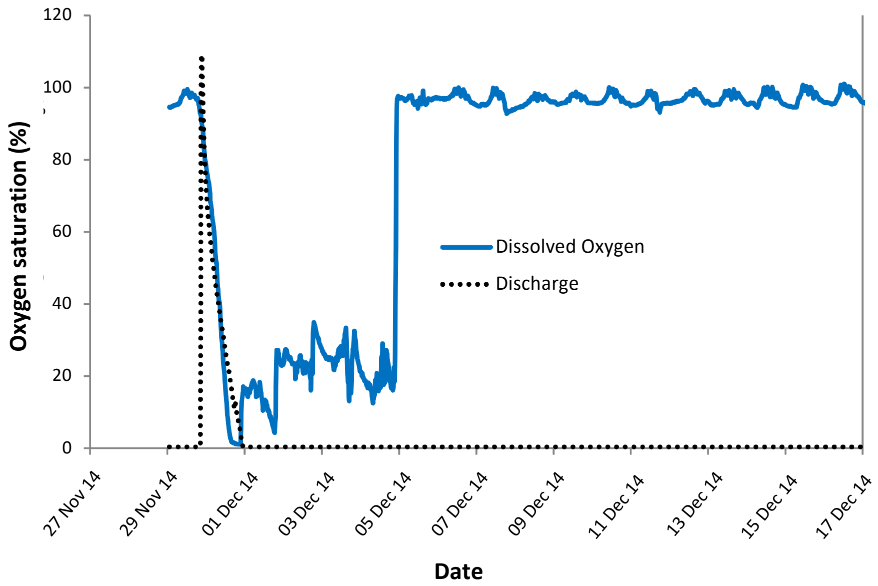

Figure 14 Example of oxygen sag in response to initial inflow to a Flinders River tributary, Queensland

Source: unpublished data from B. Butler, TropWATER at James Cook University.

It is important that successional changes in monitoring organisms over the wetting–drying cycle and of assemblage 'similarity' among adjacent populations—such as between reference and (potentially) 'exposed' locations—are understood when undertaking monitoring to detect change. Such conceptual understanding is provided in Figure 15 for macroinvertebrate communities in temporary stream sites of the wet–dry tropics. The patterns are explained in Townsend et al. (2012), but it is the relevance of the changing 'between-site' similarity over the annual hydrograph that is important. Aquatic health assessments typically compare community structures at test sites with those from a reference site. Annual monitoring must be standardised to the same or similar hydrological conditions, otherwise natural shifts in community structure among sites may be incorrectly attributed (e.g. to human-related disturbance), confounding monitoring results (see Sheldon 2005). Should standardisation not be possible, or monitoring during different seasons be required, conceptual understandings such as that depicted in Figure 15 are essential so that hydrological (flow, water level) modelling of biological responses may be undertaken to account for natural temporal variability.

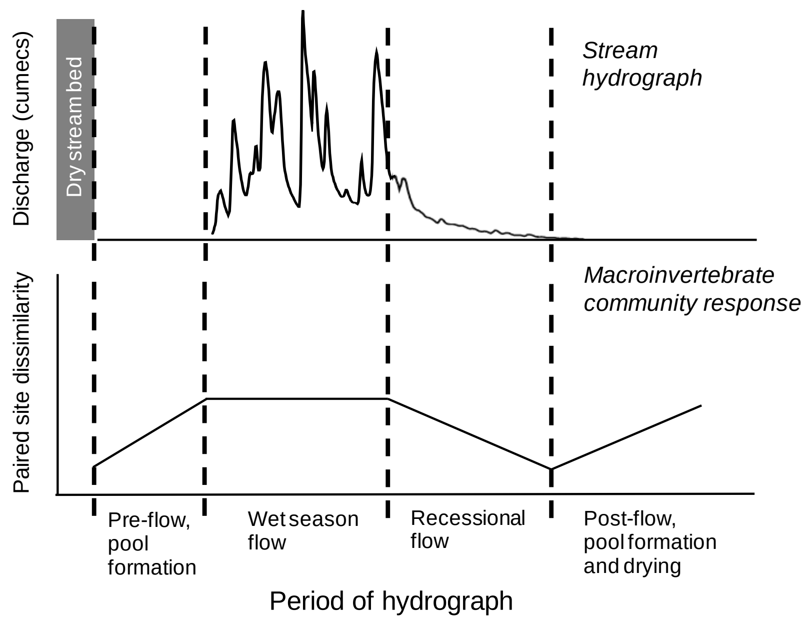

Figure 15 Behaviour of paired-site (upstream–downstream) Bray–Curtis dissimilarity measures for different phases of the hydrograph in northern seasonally flowing streams that cease to flow during the latter part of the dry season

Source: Humphrey and Douglas (unpublished data) [from Townsend et al. 2012].

Lists of potentially useful abiotic and biotic indicators along with their strengths and weaknesses are provided in Table A 1 and Table A 2, respectively, in Appendix A: Potential indicators for temporary waters. These lists are not meant to be comprehensive or to indicate any preferences or recommendations but to provide guidance for selection of indicators by the user. The actual set of indicators used should be guided by the development of the current understanding in Step 1 (Section 2.1) and the setting of management goals in Step 2 (Section 2.2).

2.4 Step 4 – determine water/sediment quality guideline values

Step 4 determines the water/sediment quality guideline values for each of the selected physical, chemical and biological indicators/lines of evidence associated with the desired level of protection for the primary management goals for the waterbody. There are specific recommended approaches for deriving guideline values for water and sediment physico-chemistry (e.g. ANZECC/ARMCANZ 2000, DERM 2009, ANZG 2018). Guideline values for toxicants in waters and sediment, such as metals, pesticides and other anthropogenic compounds, are typically provided through national defaults (global effects-based). Very few sites in Australia have derived site-specific toxicant guideline values (although they are preferred if available) unless values based on a reference condition are deemed suitable (e.g. for sites of high conservation value). Biological effects-based guideline values for physical and chemical stressors in waters, such as pH, dissolved oxygen, ammonia, oxides of nitrogen (NOx), total nitrogen (TN), total phosphorus (TP), filterable reactive phosphorus (FRP), turbidity, chlorophyll-a and electrical conductivity/salinity, are not commonly derived but are preferred. Most commonly, guideline values for these stressors are derived from a reference condition (e.g. for a slightly-to-moderately disturbed system, the guideline value is typically the 80^{th} percentile of the reference data; see ANZG 2018), with regional default guideline values provided. This includes (eco)regional default values for some temporary waters, particularly for the seasonal subtropics/tropics). However, it is recommended that site-specific (referentially-based) values be derived if effects-based values cannot be obtained (notwithstanding the issues associated with identifying suitable reference sites—as discussed in Section 2.1.3).

Many of the national and regional default guideline values may be unrepresentative for temporary waters. Consequently, it will usually be necessary to initiate or continue/refine existing monitoring to establish more locally relevant guideline values. In particular, little has been done to develop toxicant guideline values specifically for temporary waters, with those developed so far based on chronic toxicity of continuous exposures of the toxicants. Smith et al. (2004) noted that some phases of the hydrograph for temporary waters were characterised by highly variable flows or water level fluctuations, invoking pulsed exposure of resident organisms to potential contaminants. The duration of exposure within the wetting–drying cycle will affect the potential for significant ecological impact of a toxicant. For example, where pulsed exposures are identified, sensitivity is likely to be less than typical for continuous exposures (e.g. Hogan et al. 2013), though provision for adequate dilution must also be considered under scenarios of highly fluctuating flow and water levels. Assemblage successions and associated natural water quality may also need to be considered when setting toxicant guideline values. To this end, Smith et al. (2004) also noted that the presence of sensitive life history stages coincident with natural declines in pH could adversely affect organism tolerances to toxicants.

The default toxicant guideline values in ANZG (2018) are derived from suites of species used in ecotoxicity testing that, where possible, were selected to be representative of a broad range of taxonomic and trophic groups. For many temporary waters, the taxonomic composition of the aquatic ecosystem may be relatively restricted. This can be particularly true for the more isolated and episodic systems where lack of connectivity will restrict the opportunity for colonisation by many groups. Furthermore, physical (e.g. temperature) and chemical (e.g. salinity) extremes may also restrict the ability of taxonomic groups to use the temporary water (Stubbington et al. 2017), and this may change over the wetting–drying cycle. For example, the taxonomic composition of the aquatic ecosystem of a salt lake can vary widely within and between wetting–drying cycles, from a freshwater ecosystem composition to a highly specialist salt lake taxonomic grouping able to tolerate hypersaline conditions (e.g. Timms 1993, 1998, Halse et al. 1998, Cale et al. 2004). Therefore, the taxonomic and trophic representativeness of the default guideline values may be questionable. In addition, it may be necessary to consider several sets of guideline values for different phases of the wetting–drying cycle to be protective of naturally-changing suites of biota between phases, or to focus attention on the phase of the wetting–drying cycle that is likely to be most sensitive depending on the timing and nature of the stressor and likely sensitivity of the biotic assemblages at different phases.

While ANZG (2018) promotes the development of site-specific guideline values, this may be difficult to achieve in practice. Where the taxonomic peculiarities of the ecosystem result from the exclusion of particular taxonomic groups (e.g. fishes), a more locally-relevant guideline value may be able to be developed by deleting those groups from the dataset used to calculate the default guideline value, at least as a first step. Various approaches for developing site-specific guideline values, some of which may be suitable for temporary water ecosystems, are detailed in van Dam et al. (2019).

The development of guideline values based on reference conditions can also be more challenging in temporary waters than for perennial waters. Smith et al. (2004) noted the following challenges for mining areas:

- there are relatively few opportunities to collect reference water quality samples for ephemeral/episodic systems
- rainfall that inundates potential impact sites does not necessarily inundate potential reference sites and *vice versa*
- in the geologically old, flat landscapes of much of arid and semi-arid Australia, the process of mineralisation (particularly via intrusion) and differential erosion have often elevated the ore bodies; hence, the watercourses of interest are above the surrounding landscape, making the location of (upstream) reference sites difficult. Alternatively, the ore body, as a mineralogical anomaly, may be on a physiographic divide, so the natural water quality expectations of sites upstream and downstream of that divide are rarely similar.

In addition to mining-related challenges, much of semi-arid and arid Australia is rangelands with broadscale coverage by pastoral operations, and so locating true reference conditions may be impossible or impractical. Accepting some pastoral signature (e.g. enhanced nutrients) among best-available reference sites may be the only pragmatic solution if a reference condition benchmarked against good biological condition is not possible. In the event that the best-available reference site is a modified ecosystem, using the 80th percentile value of an analyte as the site-specific guideline value may not be sufficiently protective of water quality (and therefore, aquatic ecosystems), and a lower percentile may be more appropriate (ANZG 2018, Queensland Government 2018). For example, Queensland Government (2018) suggests using the 40th percentile from a best-available reference site as a starting point for stakeholder discussions on the WQO for a moderately disturbed site, an approach that aims to both protect the aquatic ecosystem and guide improvements in water quality.

Temporary waters are typically more variable in initial water quality conditions following inundation and may naturally vary widely in size, connectivity and amount of rainfall received compared to permanent waters. A further issue is that surface water may rarely be present at sites long enough to collect more than one or two samples per year. This variability and ephemerality are also found within relatively small spatial areas (e.g. sub-catchment) and a wetting–drying cycle. Therefore, adequately determining the range of reference conditions for representativeness requires a more extensive set of reference sites and a longer period of reference data collection than would be needed for a comparable perennial waterbody. Pre-development baseline data (e.g. three years (ANZG 2018)) is essential for best characterising the reference condition. So, not only can location and access of reference sites be relatively more difficult for temporary water sites, more reference sites may be needed to achieve accurate and precise assessment of water quality.

For unique sites, such as key perennial refugia within a temporary water catchment, or in situations where the hydrogeochemistry of all potential reference sites is inherently different to the test sites, suitable reference sites may not be available, and effects-based guideline values and/or additional lines of evidence may need to be developed to provide appropriate levels of protection. For example, other lines of evidence might focus on measuring the loads of a contaminant rather than concentrations, which may not be reliable indicators (e.g. nutrients) because the stressor is rapidly assimilated by wetland plants.

The default method of setting referentially-based guideline values using (typically) monthly monitoring of reference sites is not broadly applicable to temporary waters without modification. To date, most legislatures have dealt with this by allowing for some level of seasonality in the reference dataset—indeed this was a recommendation in ANZECC/ARMCANZ (2000) and remains so in ANZG (2018)—at least to account for the period of no water. Options that have been recommended, used and/or warrant further investigation include the following.

- Using multiple reference sites in lieu of longer-term datasets from single (or few) reference sites (i.e. emphasis on spatial replication).
- Developing guideline values separately for each phase of the wetting–drying cycle (while accounting for the influence of antecedent conditions and potentially using the multiple reference sites approach). For example, for a dryland river, this would comprise the no-flow phase, the within channel flow phase and the overbank/floodplain flow phase.
- Developing guideline values separately for a subset of the wetting–drying cycle phases, such as only for the eurheic and oligorheic phases of stream flow (*sensu* Gallart et al. 2012).
- A variant of temporal partitioning of guideline values is use of stressor-hydrological (regression) modelling to derive a ‘continuous’ guideline dataset across the entire hydrograph (or relevant parts thereof). Continuous water quality data from loggers are well suited to population of such relationships, examples of which are provided in Figure 16. In adapting this approach to temporary waters, any number of hydrological proxies may be used as the independent variable (e.g. water level or extent of flooding). Upper and/or lower confidence bands about derived regression relationships can serve as ‘continuous’ guideline values.
- For systems for which conductivity falls within distinct temporal or spatial ranges, such as waterbodies with connection to saline groundwater, or for which there are distinct biological assemblages associated with conductivity ranges, such as in salt lakes, referential guideline values can be developed separately for these conductivity ranges by examining relationships comparable to those for discharge shown in Figure 16.
- Setting broader guideline value ranges rather than single values to account for spatial and temporal variability.
- Where true reference site conditions cannot be found, setting guideline values based on ‘least disturbed’ or ‘best attainable condition’ *sensu* Sánchez-Montoya et al. (2012) provides more reference site opportunities. Alternatively, where a clear disturbance gradient is evident, guideline values can be developed relative to the level of disturbance.

As with any referentially-based assessments, it is essential that the reference sites are valid for comparison with the assessment site(s). In temporary water catchments where there can be substantial differences between waterbodies in the wetting–drying cycle, connectivity and substrata, this can limit the number of relevant reference sites that can be identified. This selection of suitable reference sites should be guided by the control conceptual model developed for the assessment.

Due to the inherent unavailability of water for some periods of time in temporary waters, it must be anticipated that the time required to collect sufficient data to be able to derive reference-site-based guideline values will be longer than for a perennial waterbody. This will be especially so the more episodic the reference waterbodies are. The use of multiple reference sites can partly alleviate this problem, but for the more ephemeral and episodic systems, there will always be a greater time requirement to characterise inundation-phase and inter-inundation variability in reference condition than is required for comparable perennial systems.

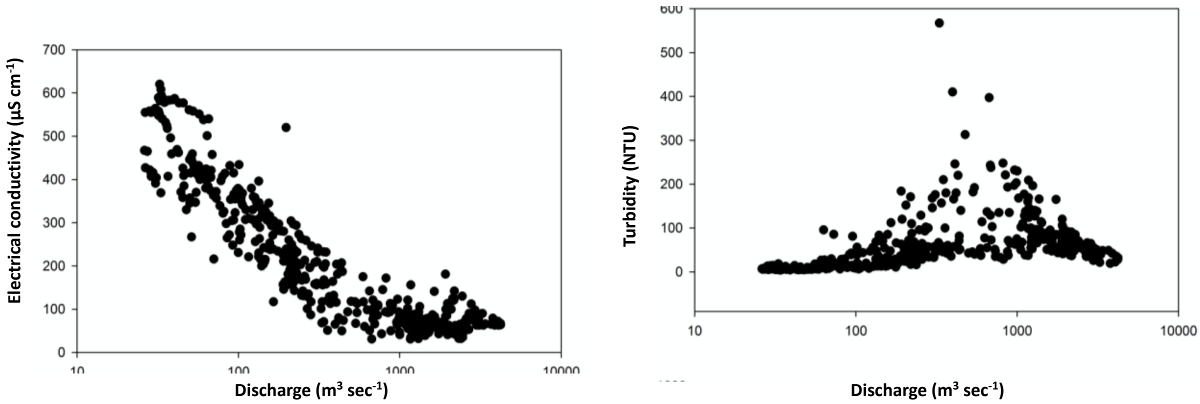

Figure 16 Relationship between physical and chemical stressors (conductivity and turbidity) and discharge in Daly River, Northern Territory

Source: S. Townsend, Northern Territory Department of Environment and Natural Resources, unpublished data.

The ANZG (2018) default sediment quality guideline values have their origin in United States databases of sediment quality and associated biological effects across a wide range of marine, estuarine (mostly port) and freshwater locations. They included both field and laboratory assessments of sediments. However, it is unlikely that any of the sediment information in the databases was derived from temporary waters. The numbers used as guideline values are percentiles of parameter concentrations associated with biological effects, and or a consensus-derived value from that approach and additional laboratory-based approaches. Thus, they are not based on single parameter causal testing or toxicity identification and evaluation (TIE) studies, just co-occurrence of measured concentrations of parameters with biological detriment. They are not comparable to the default water quality guideline values, which are based on concentration-response data for individual toxicants. Their relevance to temporary waters is uncertain as the databases contained few, if any, sediments subjected to cycles of wetting and drying; therefore, comparison with contaminant concentrations and biological detriment to temporary waters is highly questionable. They remain the best default guideline values to use, but developing site-specific guideline values for temporary water sites is strongly recommended if it was determined in Step 1 that sediment contamination and potential effects on benthic communities is a key issue.

Therefore, setting physico-chemical guideline values for temporary waters can generally be expected to be substantially more difficult than for perennial waterbodies. Care is needed when making decisions regarding the guideline values to use, and multiple lines of evidence and an associated weight of evidence assessment process should be used. The more informed the current understanding at Step 1 (Section 2.1), the better the ability to make sound decisions at Step 4 will be.

2.5 Step 5 – define water/sediment quality objectives

Step 5 defines the WQOs for the assigned community values using the relevant water/sediment quality guideline values (determined in Step 4 (Section 2.4)) appropriate for the management goals (Step 2 (Section 2.2)). These WQOs, which are usually the lowest of the guideline values for each indicator across the community values as agreed to by the stakeholders, become the measures used to assess management performance via comparison with the relevant indicator measurements.

2.6 Step 6 – assess if water/sediment quality objectives are met

At this step, monitoring data for the selected indicators across the selected lines of evidence are compared with the WQOs, preferably using a weight of evidence process. The assessment determines whether the WQOs have been clearly met or not met, or whether there was uncertainty in the assessment. In the latter case, the use of additional lines of evidence can be considered.

The use of multiple lines of evidence using a weight of evidence process, as recommended in ANZG (2018), is advantageous for temporary waters. This is because there are often high levels of variability in most parameters within and between wetting–drying cycles, and the variability among reference site measurements commonly makes assessment using a single line of evidence equivocal. An important step for the weight of evidence assessment is the evaluation assessment (i.e. Step 6). For temporary waters, the evaluation may also differ between and within wetting–drying cycles. For example, the initial and final stages of inundation are often associated with stochastically determined biological assemblages or assemblages that are constrained by extreme natural physical and/or chemical water quality. Biological assessments of water quality in these stages may have limited efficacy, and the requirement for sampling to occur some time (e.g. days to months) after the first flush or most recent spate is common in many programs (e.g. Conrick & Cockayne 2001, Humphrey et al. 2006). The period of 'no sampling' after such events is location-specific and dependent on climate and proximity to refugial recruitment sources, among other factors. For example, Smith (1982) found that artificial pools (created within the stream bed from artificial structures filled with water and natural sediments) within a temporary stream in north Queensland developed a fully functional macroinvertebrate assemblage (i.e. comparable to well-established natural pools in the same stream system) within one or two weeks of creation. Therefore, even pulsed impacts that result in complete denudation of macroinvertebrates from pools within some temporary streams may be indistinguishable from reference conditions a fortnight later.

2.7 Step 7 – consider additional indicators or refine water/sediment quality objectives

In temporary waters, as the current understanding is developed and as better understanding of natural variability in physical, chemical and biological indicators across wetting–drying cycles is gained, the lines of evidence can be refined. Initially and in the early cycles through the WQMF, it would be beneficial to include as many relevant lines of evidence as possible, reviewing and optimising these as knowledge accrues. An equivocal assessment at Step 6 (Section 2.6) may require additional lines of evidence/indicators. For temporary waters, the assessment may require collection of more monitoring data over more wetting–drying cycles to improve the power to detect change compared with the variability observed at (multiple) reference sites. This approach was used for setting baseline water quality and nominating water quality guideline values for the South of Embley Project (Rio Tinto Alcan 2011). The collection of more monitoring data may take some time to achieve, particularly for waterbodies that are infrequently inundated; therefore, this needs to be considered when assessing the sufficiency of the selected lines of evidence. As more data are acquired, the power of the assessment at Step 6 can be expected to improve, whereupon it may be possible to maintain the overall quality of evidence while reducing the number of lines of evidence used.

2.8 Step 8 – consider alternative management strategies

This step is normally taken if the assessment at Step 6 (Section 2.6) indicates a failure to achieve the WQOs or if the assessment is equivocal and management options to improve the certainty of attaining the WQOs are identified. In principle, the application of this step to temporary waters is no different from application to perennial waters, but there may be some management options that are specifically applicable to the waterbodies of interest because of their temporary nature. For example, in the arid and semi-arid regions of Australia, evaporation will always be an important consideration.

2.9 Step 9 – assess if water/sediment quality objectives are achievable

If the chosen management strategies (Step 8 in Section 2.8) are considered likely to ensure the WQOs are met, go to Step 10 (Section 2.10). If not (e.g. because the costs/impacts associated with the necessary improvements to meet the WQOs are not viable or modelling and/or monitoring data have demonstrated that the available management strategies will not meet the WQOs in practice), repeat the WQMF steps, typically bypassing Step 10. At Step 1, improvements to the current understanding can be made based on knowledge gained from monitoring and the assessments made at Step 7 and Step 8. At Step 2, the primary management goals may be reconsidered, and the stakeholders may need to reconsider the levels of protection that are achievable and acceptable. In some cases, the pressure under consideration may be assessed to have unacceptable water quality consequences, and closure and/or remediation strategies may be implemented.

2.10 Step 10 – implement agreed management strategy

This step is reached because acceptable water/sediment quality has or can be achieved. However, there may be implications for the monitoring program that have arisen from the considerations in Step 6, Step 7 and Step 8 that need to be included in the revision of ongoing monitoring programs, or alternative management actions from Step 8 that need to be incorporated into the management systems. Continual improvement of both aspects (i.e. monitoring and management) should be considered at Step 10 regardless of the route within the WQMF by which this step was reached. Ongoing monitoring and management lead to continued cycling through the framework, via a potentially rapid reconsideration of the current understanding at Step 1.

3 Conclusion

Temporary waters abound across Australia, especially in arid and semi-arid regions. In these temporary waters, the variations in duration, timing, frequency and rates of drying and re-wetting largely drive the substantial variation in water quality parameters (e.g. temperature, electrical conductivity, pH, dissolved oxygen, turbidity, nutrients) at multiple spatial scales within and among waterbodies. These variations occur against a diverse backdrop of variable climatic, geological and landscape settings in catchments that can vary widely in land use, vegetation cover, sediment processes and runoff features. Activities of aquatic and terrestrial biota (including humans) can further alter water quality in different temporary waters, and all these processes interact with each other as portrayed conceptually in Figure 6 (see also Table 1).

These interactions and the vast spatial and temporal variability in water quality created by the wetting–drying cycle must be considered when designing water quality monitoring programs to assess and manage temporary waters. Consequently, application of the WQMF steps (Section 2) to temporary waters must explicitly recognise the inherently high variability in water quality within and across these types of waters, and modify the methods and indicators (Table A 1 and Table A 2) used to assess water quality accordingly. These are all affected by the water regime produced by the wetting–drying cycle in each temporary water. For example, when temporary waters begin to dry up, there is often an elevation in solute concentrations that may increase the vulnerability of aquatic biota and ecological processes to direct and indirect human impacts. Later, inundation or flooding of previously dry lakebeds and river channels can lead to pulses of high concentrations of solutes or to the export of solute-rich water downstream and onto floodplains.

Extremes and fluctuations in the physico-chemistry of temporary waters define their ecological conditions, govern the composition of their biota, and may influence community values, including cultural and spiritual ones. Therefore, appropriately defining water quality objectives and choosing suitable indicators (usually several in a multiple lines of evidence approach) are essential to properly assessing and wisely managing the water quality of temporary waters across Australia. Currently, empirical information about water quality in temporary waters lags far behind that about permanent waters, necessitating the use of conceptual models and extrapolations from other, perhaps unsuitable, examples. Where possible, water resource managers should collect empirical data of multiple indicators and refine their conceptual models to improve management within a suitably modified WQMF.

Appendix A: Potential indicators for temporary waters

Table A 1 Potential abiotic indicators for assessing water quality in temporary waters

Potential indicator	Indicative of: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Electrical conductivity (salinity)	• Salinisation (including saline groundwater inputs) • Inputs of freshwater, especially into temporary salt lakes (e.g. co-produced water)	Yes	• Easily measured (spot or automated) • Inexpensive • Conservative indicator (and samples do not require refrigeration) • Well-known effects on most aquatic biotic groups • Wide public recognition • Sediment salinity can be measured in the field on 1:5 (sediment:distilled water) slurries • Can be used as an indicator for the presence of sulfidic sediments	• The relationship between electrical conductivity and total dissolved solids concentrations is dependent on salt composition • Because of this, electrical conductivity is not a good direct measure of the potential toxicity of the salt mixture without knowledge of and, if necessary, adjustment for the salt mixture present	• Electrical conductivity is routinely measured at many gauging stations • Potential for stratification, necessitating sampling from different depths
Macronutrient concentrations (total and dissolved nitrogen and phosphorus in water)	• Eutrophication • Effluent spills • Agricultural activities	Potentially	• Standard methods for measurement (spot or automated depending on location and limits of detection required) • Well-known effects on algae and aquatic plants (and subsequent effects on most aquatic biota) • Wide public recognition	• Largely restricted to wet phase (but can be estimated from nitrogen and/or phosphorus release from dry sediments re-wetted in lab) • Some complexity in spatial and temporal variations (von Schiller et al. 2017), especially in bioavailable forms (may require preliminary sampling to assess) • Appropriate sample curation needed for accuracy (challenging if sample delivery times to the laboratory exceed recommended hold times)	• Potential for nitrogen and phosphorus measures to supplement bioassays (see Table A 2) to assess potential macronutrient limitation
Macronutrient concentrations (sediment-associated nitrogen and phosphorus)	• Eutrophication • Sewage effluent spills or intentional releases • Some agricultural activities (e.g. poor fertiliser use)	Yes	• Standard methods for measurement • Well-known effects on algae and aquatic plants (and subsequent effects on most aquatic biota) • Can be measured in wet and dry phases	• Complex extraction techniques may be required for different sediments • Techniques (e.g. 31P nmr) can be used for speciation but they are very expensive	• Alternative techniques such as adsorption isotherms can be used to measure the sediment's affinity for nutrients, but these are labour-intensive
Dissolved organic carbon (concentration)	• 'Blackwater events', especially in lowland temporary waters • By-product of particulate organic matter inputs • Control on bioavailability of some toxicants such as some metals (e.g. copper, nickel, zinc)	No	• Standard methods for measurement (high concentrations evident in field samples by 'tea' colour) • Inexpensive	–	• High levels of dissolved organic carbon can lead to hypoxia and/or anoxia • At high concentrations, may impair photosynthesis by reducing transparency • An important control on toxicant bioavailability that is under-utilised • Can contribute to pH and electrical conductivity (e.g. humic acids)
Particulate organic matter (concentration, standing crop)	• Poorly managed fringing vegetation (e.g. slash from forestry activities entering basin or channel)	Yes	• Easily measured as 'grab samples' • Inexpensive	• Often high spatial and temporal variability	• Many aquatic food webs rely heavily on particulate organic matter • In some instances, the likelihood of hypoxia occurring can be estimated from standing stocks of litter on the dry bed of the waterbody

Potential indicator	Indicative of: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Sulfidic sediments	• Exposure of acid sulfate soils • Control of bioavailability of some toxicants, including many metals • Some mining activities (e.g. coal mining), and eutrophication/reduction of sulfate with sub-oxic conditions	Yes	• Standard methods for measurement include rapid assessment techniques • Well-known effects on most aquatic biota if associated with low pH • Wide public recognition	• Can be both a source and sink for toxicants so needs to be considered within the context of the biogeochemical and wetting–drying cycles of the waterbody	• Acid volatile solids associated with sulfidic sediments can alter the bioavailability of heavy metals, but oxidation of sulfidic sediments can lead to release of these and other toxicants
Ionic composition (inorganic ions)	• Water releases (e.g. co-produced water) • Alteration of connectivity to groundwater and surface water • Water hardness and related processes (e.g. bioavailability of some toxins, suitability for some community uses, flocculation)	Potentially	• Standard methods for measurement • Inexpensive • Usually very highly correlated with electrical conductivity	• Needs discrete samples and laboratory analysis • Relationship with electrical conductivity can differ depending on salt composition, even within catchments	• Do not assume the same relationship with electrical conductivity between water samples from different sites and/or times
Extremes of pH	• Acidification (e.g. exposure of acid sulfate soils, acid mine drainage) • Some types of industrial and urban runoff	Potentially	• Easily measured (spot or automated) • Inexpensive • Well-known effects on most aquatic biotic groups • Wide public recognition	• Largely restricted to wet phase (but can be estimated from pH release from dry sediments re-wetted in laboratory or field) • Highly temporally variable in poorly buffered temporary waters • Can be strongly influenced by bioconcentration and increased rates of photosynthesis and respiration late in the wet phase	• Indirect effects of pH include increased bioavailability of many metals • pH is routinely measured at some flow gauging stations • Maintaining and checking calibration in temporary waters for pH loggers can be particularly challenging, especially where the logger has to cope with dry periods
Suspended sediments (turbidity)	• Poorly managed land clearance • Sediment discharges (e.g. mining, agriculture, urban catchments) • Excessive erosion and/or bank slumping (altered flows or surface drainage)	No	• Easily measured (spot for both or automated for turbidity) • Inexpensive • Conservative indicator—well-known effects on many aquatic biotic groups (although acute/chronic thresholds often poorly determined) • Wide public recognition	• Restricted to wet phase • Turbidity needs to be measured in the field • Relationship between turbidity and suspended sediments concentrations can change within catchments and through the flow cycle	• Turbidity is routinely measured at many gauging stations • Suspended solids concentrations are easy and inexpensive to measure but less commonly used due to the time delay in obtaining results • Biological responses to turbidity and suspended solids are not always equivalent, can be highly variable in nature between taxa, highly non-linear in relation to changes in suspended solids loads, and are often substantially temporally and/or spatially displaced from the sediment impact (see Dunlop et al. 2008)
Altered sediment composition (especially siltation and colmation)	• Poorly managed land clearance • Sediment inputs (e.g. mining, agriculture, urban catchment discharges) • Excessive erosion and/or bank slumping (altered flows or surface drainage, altered vegetation) • Disturbance or fining of sediments by livestock or vehicles	Yes	• Easily measured during wet and dry phases • Inexpensive (sieving) • Conservative indicator (and samples do not require refrigeration) • Potential value as 'archive' of past events when sediments collected from different depths • Well-known effects on most aquatic biotic groups (although acute/chronic thresholds often poorly determined) • Wide public recognition	• Often high spatial and temporal variability (especially in temporary flowing waters)	• Colmation refers to where fine sediments clog stream beds, impairing hyporheic exchange

Potential indicator	Indicative of: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Sediment redox potential	• Sediment inputs (e.g. mining, agriculture, urban catchment, wastewater treatment plant discharges) • Acidification (e.g. exposure of acid sulfate soils, acid mine drainage) • Eutrophication • Some types of industrial and urban runoff	Yes	• Easily measured (spot or automated) during wet and dry phases • Well-known effects on many biogeochemical processes in sediments and overlying waters • Some public recognition	• Often high spatial (especially with depth in sediments) and temporal variability	• Although a widely used parameter, its calculation by *in situ* probes is prone to artefacts
Extremes of water temperature	• Thermal pollution (e.g. cooling tower effluent, deep-release from stratified dams) • Reduction in water volume (thermal mass) due to water abstraction • Temperature changes related to turbidity alteration	No	• Easily measured (spot or automated) • Inexpensive • Well-known effects on most aquatic biotic groups and biogeochemical processes in sediments and overlying waters • Wide public recognition	• Must be measured in field • Often high spatial and temporal (diel and longer-term) variability (especially in standing temporary waters)	• Water temperature is routinely measured at many gauging stations • Potential for thermal stratification, necessitating sampling from different depths
Dissolved oxygen	• Hypoxic or anoxic conditions (e.g. from disturbance of sulfidic sediments, eutrophication, blackwater events, organic discharges, excessive growth of aquatic weeds)	No	• Easily measured (spot or automated) • Inexpensive • Well-known effects on most aquatic biotic groups and biogeochemical processes in sediments and overlying waters • Wide public recognition	• Often high spatial and temporal variability (especially in standing temporary waters) • Can be very strong diurnal cycling, especially late in the wet phase due to bioconcentration of aquatic organisms, requiring careful standardisation of time of measurement	• Potential for stratification, necessitating sampling from different depths • Plots of temporal variation (e.g. diel cycles) in a waterbody can reveal production/respiration ratio, oxygen 'sags' and periods of supersaturation (see Figure 14)
Chemical oxygen demand Biological oxygen demand	• Organic pollution (e.g. wastewater effluent, agricultural runoff)	Potentially	• Standard methods for measurement • Well-known effects on most aquatic biota • Some public recognition	• Largely restricted to wet phase (but response to first flush indicated by chemical/biological oxygen demand of dry sediments re-wetted in laboratory can be measured) • Often high spatial and temporal variability (especially in standing temporary waters)	• Indirect measure of the amount of organic matter that can be broken down (i.e. oxidised biologically or chemically) by aerobic organisms in the water column or sediments
Toxicants	• Where particular toxicants are known to be potential stressors, direct measurement of them is recommended	Potentially	• Standard methods for measurement • Direct measure of stressor of concern • Well-known effects on aquatic biota for many toxicants • Public recognition	• Sensitivity of temporary water ecosystems may differ from perennial ecosystems and may differ over time through the wetting–drying cycle, making default guideline values potentially over-protective or under-protective	• Often mobilised during 'first flush' events and/or evapoconcentrated during drying • Many metals become mobilised by altered pH • Many potential indicators above are important modifiers of bioavailability of toxicants and should be assessed parallel with the measurement of toxicant concentrations

a Applicability of measuring the indicator during the dry phase of the wetting-drying cycle, indicated as follows: green highlight = applicable; orange highlight = potentially applicable; red highlight = not applicable.

Table A 2 Potential biotic indicators for assessing water quality in temporary waters

Potential indicator	Impacted by: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Frog abundance/density	• Agricultural pesticides (e.g. chlordane, dieldrin), herbicides and fungicides • Stormwater runoff • Salinisation • Some heavy metals (e.g. mercury, copper, zinc)	Potentially	• Calls and adults of most temporary water species are easily identified • Call detection is inexpensive and non-invasive • Sensitivities established for a few toxicants for a few species • Wide public recognition • Adults may be detected when water is absent (e.g. burrowing frogs in moist sediments) • Improved keys available to identify tadpoles, making aquatic sampling of them a feasible option when sampling for adults is difficult	• Direct search is more difficult, requires permits and ethics clearance • Success in detecting frogs by calls/direct search is often weather-dependent • Some training needed to identify calls/adults • Detection success often limited to certain times of the year and phases of the wetting–drying cycle • Only a subset of frogs live in temporary waters (mostly standing ones) • Absence is not necessarily due to water quality issues • Relatively few toxicity endpoints available compared with other taxonomic groups	• Seldom used to assess water quality in temporary waters because of the many disadvantages
Tadpole abundance/density	• Agricultural pesticides, herbicides, pharmaceuticals and fungicides • Salinisation • Mining • Stormwater runoff • Acidification	No	• Sampling is easy (e.g. dip-net, electrofisher), inexpensive and can be non-invasive (returning identified specimens to the water) • Sensitivities established for a few toxicants for a few species • Among the more sensitive of taxonomic groups to endocrine-disrupting substances • Wide public recognition	• Identification of some species is challenging (requiring microscopic assessment) • Sampling requires permits and ethics clearance • Restricted to wet phase • Only a subset of species live in temporary waters (mostly standing ones) • Absence is not necessarily due to water quality issues • Relatively few toxicity endpoints available compared with other taxonomic groups	• Seldom used to assess water quality in temporary waters because of the many disadvantages
Native fish abundance/density (adults and larvae) and fish assemblage composition (Kerezsy et al. 2017)	• Organic pollution (leading to hypoxia) • Salinisation • Acidification • Agricultural, industrial and urban runoff • Mining discharges	No	• Adults and most juveniles/larvae of temporary water species are easily identified • Standard methods for sampling fish in standing and flowing waters • Non-destructive, non-invasive ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Well-known effects of water quality impairment on most species • Can respond by avoidance as well as direct toxicity effects, meaning distribution patterns can be informative • Wide public recognition and community values	• Sampling requires permits and ethics clearance • Some methods (e.g. electrofishing, larval sampling) require specialised equipment and training • Restricted to wet phase • Only a subset of Australian native fish live in temporary waters (mostly flowing ones) • Absence is not necessarily due to water quality issues (e.g. effects of refuge availability, hydrological connectivity) • Dominant species living in temporary water are likely to be tolerant of poor water quality	• Although the many disadvantages mean that fish are seldom used to assess water quality in temporary waters, some Australian native fish species may warrant consideration as water quality indicators in some temporary streams (Bond & Cottingham 2008) • As more information becomes available, functional traits (e.g. reproduction cycles, feeding) of temporary water fish may be viable indicators of specific water quality parameters

Potential indicator	Impacted by: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Tissue accumulation of toxicants in fishes and larger invertebrates (e.g. crayfish, prawns)	• Pesticides, metals	Potentially	• Line of evidence for exposure to toxicants • Often a simple addition to a program of sampling biota diversity and abundance • Established methods for analysis • Established generally expected levels for a number of toxicants for some taxonomic groups that can be used for initial screening assessment (ANZFA 2001) • Same dataset can be used for risk screening for human consumption risk assessment • Some groups can be collected in the dry phase (e.g. crayfish from burrows) • Wide public recognition	• Measurement of exposure to toxicants rather than effects • Tissue and taxa used need to be selected to accumulate rather than regulate the toxicant(s) of concern	• A common addition to monitoring programs to address stakeholder concerns • Interpretation can require specialised training • Often a useful addition to a multiple lines of evidence approach
Decapod crustacean diversity/abundance	• Organic pollution (leading to hypoxia) • Salinisation • Acidification • Agricultural, industrial and urban runoff • Mining discharges	Potentially	• Advantages as for *native fish abundance/density (adults and larvae) and fish assemblage composition* • Commonly collected along with fishes using similar sampling techniques, so addition to program requires little additional effort • Burrowing taxa can potentially be sampled in the dry phase and may be present during seasons of low water inflow that prevent colonisation by fishes	• Disadvantages as for *native fish abundance/density (adults and larvae) and fish assemblage composition*, except ethics clearance is generally not needed • Some taxa have been the focus of ecotoxicity testing, providing established toxicant sensitivities for those taxa and potentially for congeners	• An important taxonomic group in many temporary waters that is poorly sampled by more commonly used general macroinvertebrate sampling methods, but can be included in fish sampling programs and can be specifically targeted
Aquatic macroinvertebrates Assemblage composition (using ordinal scoring systems e.g. AusRivAs scores, SIGNAL scores, EPT index, and/or via quantitative or qualitative multivariate and/or univariate comparisons with reference to temporary waters)	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff • Thermal pollution • Mining	No	• Most temporary water macroinvertebrate species are easily identified to family level (often used for assessing assemblage composition) • Standard methods for sampling macroinvertebrates in standing and flowing waters • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Reasonably well-known effects of water quality impairment on assemblages (but not always specific) • Reasonable database, especially in Australian rivers (although fewer temporary rivers than perennial rivers) • Macroinvertebrates occur in almost all temporary waters and play diverse ecological roles (e.g. food web dynamics) • Wide public recognition	• Some methods (e.g. AusRivAs) require specialised training • Largely restricted to wet phase • Variation in assemblage composition is not necessarily due to water quality issues (e.g. low SIGNAL scores may arise from natural drying, (Boulton et al. 2000)) • Many common families in temporary waters are likely to be tolerant of poor water quality • Relatively few toxicity endpoints available compared with other taxonomic groups, but the database is growing, particularly for salinity	• Growing database on 'expected' assemblage composition in different river types for AusRivAs may assist water quality assessments, but it is still challenging to identify clear associations with specific water quality parameters • Flow stage specific sampling designs have been shown to provide robust monitoring data (e.g. Faith et al. 1995, Humphrey & Pidgeon 2001, Humphrey et al. 2006) • Very widely used approach that is well regarded by regulators

Potential indicator	Impacted by: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Aquatic macroinvertebrates Diversity within selected orders or families (e.g. Coleoptera diversity); abundance of particular taxa (e.g. freshwater crayfish)	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff • Thermal pollution	No	• Less work involved in processing samples when focused on specific order, family or taxonomic group compared to full assemblage/community • Standard methods for sampling different groups of macroinvertebrates in standing and flowing waters • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Some data on water quality impairment on a few selected groups or species • Macroinvertebrates occur in almost all temporary waters and play diverse ecological roles (e.g. food web dynamics) • Wide public recognition, especially of freshwater crayfish that show behavioural responses to poor water quality (e.g. to 'blackwater events' (King et al. 2012))	• Species-level taxonomy requires specialised training • Largely restricted to wet phase • Many common orders (Coleoptera, Hemiptera) and families (Chironomidae) in temporary waters are likely to be tolerant of poor water quality, although some groups still prove useful (e.g. Coleoptera diversity for salinity and turbidity in temporary pools (Gutiérrez-Estrada & Bilton 2010)) • Major gaps remain in basic understanding of ecology of many Australian temporary water species, limiting their use as water quality indicators	• This approach is more widely used for water quality assessment in European temporary waters (e.g. references in Van den Broeck et al. (2015)), perhaps because there is more life history data from those regions
Aquatic macro-invertebrates Temporary water taxa possessing particular functional traits associated with poor water quality	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff • Thermal pollution • Specific toxicants	No	• Less work involved in processing samples when focused on specific order, family or taxonomic group compared to full assemblage/community • Standard methods for sampling different groups of macroinvertebrates in standing and flowing waters • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Increasing evidence that macroinvertebrates can be grouped by functional traits for some aspects of water quality (e.g. tolerance of hypoxia, high water temperature, salinity (Horrigan et al. 2005, Chessman 2015)) • Macroinvertebrates occur in almost all temporary waters and play diverse ecological roles (e.g. food web dynamics) • Wide public recognition	• Species-level taxonomy (usually needed for detailed functional trait analysis) requires specialised training • Largely restricted to wet phase • Many macroinvertebrate functional traits that respond to water quality also respond to natural wetting–drying cycles in temporary waters (Boulton et al. 2000); it is challenging to separate these interacting drivers • Major gaps remain in basic understanding of functional trait ecology of many Australian temporary water species, limiting their use as water quality indicators	• Although a promising approach for assessing macroinvertebrate responses to altered water quality and quantity in temporary flowing waters (Stitz et al. 2014, Chessman 2015), a dearth of basic biological information for most taxa constrains this method; only a few correlations (few very strong) have been shown so far for macroinvertebrates and water quality in Australian temporary waters

Potential indicator	Impacted by: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Zooplankton/Phytoplankton Assemblage composition (e.g. multivariate comparisons with reference temporary waters)	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff • Specific toxicants	Potentially	• Most temporary water zooplankton/phytoplankton can be identified to genus or species level • Standard methods for sampling zooplankton (especially in standing waters) • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Reasonably well-known effects of some water quality parameters (e.g. salinity, metals, pH) on some assemblages • Reasonable database for some Australian temporary standing waters • Wide range of ecotoxicity testing data for specific toxicant sensitivity for a number of Australian microinvertebrate/microalgal taxa, including within the toxicity database developed for ANZG (2018) • Zooplankton/phytoplankton occur in almost all temporary waters and play diverse ecological roles (e.g. food web dynamics) • Many temporary water zooplankton/phytoplankton have desiccation-resistant stages, and their assemblage composition may be potentially estimated (at least in part) from the egg/propagule bank of dry sediments	• Taxonomic resolution requires specialised training and access to microscopes, but commercial facilities to process samples are becoming more available • Although assemblage composition might be estimated from the egg/propagule bank sampled during the dry phase (Stubbington *et al.* 2017), this may give incomplete data • Variation in assemblage composition is not necessarily due to water quality issues (assemblage composition is strongly influenced by aspects of the wetting–drying cycle and predator interactions) • Many zooplankton/phytoplankton taxa in temporary water are likely to be tolerant of poor water quality • Zooplankton/phytoplankton may be rare in temporary flowing waters while they are flowing • Often high spatial and temporal variability • Limited public recognition	• Considerable strength of interpretation potentially to be gained from combining field collection data with toxicity database sensitivity distribution data for individual toxicants

Potential indicator	**Impacted by:** **(example stressors and pressures)**	**Applicable to dry phase?** [a]	**Advantages**	**Disadvantages**	**Comments**
Zooplankton/Phytoplankton Diversity within selected groups (e.g. Branchiopoda diversity); abundance of particular taxa	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff	Potentially	• Less work involved in processing samples when focused on specific group compared to full assemblage/community • Standard methods for sampling zooplankton/phytoplankton (especially in standing waters) • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Well-known effects of some water quality parameters (including ecotoxicological testing of specific chemicals) on some groups (e.g. Daphniidae, Chlorophyta) • Reasonable database for some Australian temporary standing waters • Wide range of ecotoxicity testing data for specific toxicant sensitivity for a number of Australian microinvertebrate/microalgal taxa (particularly Daphniidae and Chlorophyta), including within the ANZG (2018) guidelines toxicity database • Zooplankton/phytoplankton occur in almost all temporary waters and play diverse ecological roles (e.g. food web dynamics) • Selected groups of temporary water zooplankton/phytoplankton have desiccation-resistant stages and their assemblage composition may be estimated (at least in part) from the egg/propagule bank of dry sediments	• Taxonomic resolution requires specialised training and access to microscopes • Although assemblage composition might be estimated from the egg/propagule bank sampled during the dry phase, this may give incomplete data • Variation in assemblage composition is not necessarily due to water quality issues (assemblage composition strongly influenced by aspects of the wetting–drying cycle) • Many zooplankton/phytoplankton taxa in temporary waters are likely to be tolerant of poor water quality • Zooplankton/phytoplankton may be rare in temporary flowing waters while they are flowing • Often high spatial and temporal variability • Limited public recognition • Major gaps remain in basic understanding of ecology of many Australian temporary water zooplankton/phytoplankton, limiting their use as water quality indicators, but more study will readily fill these information gaps and many taxa are cosmopolitan and well studied	• Considerable strength of interpretation potentially to be gained from combining field collection data with toxicity database sensitivity distribution data for individual toxicants
Zooplankton/Phytoplankton Egg bank dynamics (e.g. hatching success, diversity of unhatched eggs)	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff	Yes	• Easy, cost-efficient sampling of dry sediments • Widely used (almost standard methods) (e.g. Boulton & Lloyd 1992, Angeler et al. 2010, Ning et al. 2014) • Australian data on effects of water quality parameters (e.g. salinity, hypoxia) on hatching success • Deeper egg-bearing layers in sediments 'archive' past events • Reasonable data base for some Australian temporary standing waters and floodplains • Zooplankton occur in almost all temporary waters and play diverse ecological roles (e.g. food web dynamics)	• Requires equipment and laboratory space for aquaria for emergence trials • Little evidence for diverse egg banks in sediments of many temporary streams • Taxonomic resolution requires specialised training and access to microscopes • Australian 'egg taxonomy' in infancy • Variation in egg bank composition is not solely due to water quality issues (strongly influenced by aspects of the wetting–drying cycle (Jenkins & Boulton 2007)) • Many zooplankton taxa in egg banks are likely to be tolerant of poor water quality • Often high spatial and temporal variability • Limited public recognition • Major gaps remain in basic understanding of ecology of many Australian temporary water zooplankton, limiting their use as water quality indicators	• Despite the disadvantages, this is a promising approach because of its tractability for sampling temporary waters during the dry phase and potential for experimental assessment of hatching success to particular water quality variables (e.g. Skinner et al. 2001, Ning et al. 2014)

Potential indicator	Impacted by: (example stressors and pressures)	Applicable to dry phase? [a]	Advantages	Disadvantages	Comments
Hyporheic faunal communities Diversity within selected groups; abundance of particular taxa	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff	Yes	• Quantitative sampling during dry and wet phases, using inexpensive equipment (endorheic pots or direct sampling) • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Can be related to toxicant concentrations in sediment/pore water • Provide an indication of groundwater and surface water related contamination • Hyporheic communities can be spatially extensive and less variable than temporary surface waters	• May be less abundant in temporary waters of high temperature desert environments and in streams with bedrock or unstable sandy substrata • Taxonomic resolution requires specialised training and access to microscopes • Variation in assemblage composition is not necessarily due to water quality issues • Many taxa in temporary waters are likely to be tolerant of poor water quality • Potential high spatial and temporal variability • Limited public recognition • Major gaps remain in basic understanding of ecology of Australian temporary water hyporheic communities, limiting their use as water quality indicators, but more study will readily fill these information gaps	–
Diatoms and other benthic algae	• Salinisation or 'freshening' • Eutrophication • Acidification • Organic pollution • Agricultural, industrial and urban runoff	Potentially	• Easy, cost-effective sampling of temporary waters • Established methods available for Australian waters (e.g. John 2000a, 2000b) • Ecogenomic approaches (e.g. eDNA metabarcoding) for analysing samples are becoming more refined and validated • Reasonable database of sensitivities and affinities for particular water quality characteristics for these cosmopolitan taxa • Sensitivities of some taxa to specific toxicants available from ecotoxicity databases	• Taxonomic resolution requires specialised training and access to microscopes, but commercial laboratory services are developing • Assessment of assemblage sensitivities and affinities require knowledge of the literature and specialist knowledge	• A well-established water quality assessment tool globally, but is not commonly used for water quality assessment in Australia
Aquatic macrophytes	• Nutrients • Agricultural, mining and urban discharges • Herbicides • Turbidity/suspended solids	Potentially	• Readily surveyed with standard methods • Seed/propagule (rhizome, tuber) bank potentially able to be surveyed in dry phase but responses to water quality not well-established	• Responsive to successional development of vegetation, including increased shading by riparian vegetation during wet phase • Timeliness of responses to water quality changes by be slow • May be subject to other pressures such as grazing that are un-related to water quality changes and override responses to water quality	• Despite disadvantages, is an under-used group for water quality assessment under appropriate circumstances
Ecological processes (e.g. production/respiration ratio, leaf-pack mass loss, cotton-strip tensile strength loss, algal bioassays)	• Nutrients • Pesticides/herbicides • Metals • Agricultural, mining, industrial and urban discharges	No	• Existing methods (e.g. Davies & Bunn 2003, Bunn et al. 2010) • Assess key ecosystem functions rather than taxonomic composition, so potentially less influenced by stochastic recruitment processes	• Not widely used, and variable sensitivities and results reported for applications to date • Requires specialist knowledge, equipment (for some measures) and training to implement	• These methods have been used with some success (e.g. Burrows et al.2017) but are not widely adopted, and there have been some difficulties with successful adoption when trialled

Note: The table excludes waterbirds, semi-aquatic vertebrates and terrestrial invertebrates because of the dearth of information on how these groups might respond to water quality in temporary waters.

a Applicability of measuring the indicator during the dry phase of the wet-dry cycle, indicated as follows: green highlight = applicable; orange highlight = potentially applicable; red highlight = not applicable.

Glossary

Term	Definition
antecedent	Preceding in time or order; previous or pre-existing.
AusRivAs	Australian River Assessment System.
community value	A particular value or use of the environment that is important for a healthy ecosystem or for public benefit, welfare, safety or health and that requires protection from the effects of stressors. Also known as 'environmental values', 'beneficial uses' or 'national values'.
conceptual model	An abstraction (e.g. narrative, table, matrices of factor, pictorial, box-and-arrow diagram) that describes the components and processes that are deemed important in a system and the relationships between them. It documents assumptions about how components and processes are related and identifies gaps in knowledge. It is a working hypothesis about system form and function.
direct toxicity assessment	The use of toxicity tests to determine the acute and/or chronic toxicity of wastewater discharges or total pollutant loads in receiving waters.
ecosystem condition	The current or desired state of health of an ecosystem, relative to the degree of human disturbance.
ecosystem receptor	Any living organism or natural habitat that could be exposed to a stressor.
ephemeral waters	Subset of temporary waters that contain water only after irregular rainfall or flow events.
episodic waters	More infrequently inundated subset of ephemeral waters for which inflow is a rare event (e.g. dune base wetlands in the Strzelecki Desert).
EPT index	Total number of distinct taxa within the groups Ephemeroptera, Plecoptera and Trichoptera.
FRP	Filterable reactive phosphorus.
guideline value	A measurable quantity (e.g. concentration) or condition of an indicator for a specific community value below which (or above which, in the case of stressors such as pH, dissolved oxygen and many biodiversity responses) there is considered to be a low risk of unacceptable effects occurring to that community value. Guideline values for more than one indicator should be used simultaneously in a multiple lines of evidence approach.
hydroperiod	The number of days per year that an area of land is wet or the length of time that there is standing water at a location.
hyporheic	Of the region of sediment and porous space beneath and alongside a stream bed, where there is mixing of shallow groundwater and surface water (i.e. referred to as the hyporheic zone).
indicator	A parameter that can be used to provide a measure of a pressure, stressor and/or ecosystem condition response.
intermittent waters	Subset of temporary waters that are predictably inundated each year (during a wet season), although the duration for which they retain water may be highly variable.
lentic	Non-flowing.
level of protection	A degree of protection afforded to a waterbody based on its ecosystem condition.
littoral zone	The zone of a waterbody that is close to the shore. It is often characterised by alternating intervals of submergence and exposure, with the presence of submerged and emergent aquatic plants.

Term	Definition
management goal	A measure or statement used to assess whether community values are being attained or maintained. It should reflect the desired level of protection and provide precise and detailed descriptions of which aspects of the community values are to be protected.
management strategy	Documentation of actions and approaches to achieve the water quality objectives and management goals and, hence, protect the community values.
multiple lines of evidence	Two or more lines of evidence that can be combined to monitor, assess or manage water or sediment quality.
NOx	Oxides of nitrogen.
NH_3	Ammonia.
perennial waters	Waterbodies that contain surface water for most or all of the time.
photic depth	The depth of a waterbody to which sunlight penetrates.
pressure	Any human activity or biophysical change that has the potential to have an impact on the natural environment.
reference site	A site that is similar to the site being assessed (e.g. similar climate, relief and geology) but is as minimally impacted as possible, with limited exposure to anthropogenic drivers, and has sufficient historical data to characterise water quality condition and variability.
refugia	Waterbodies in which populations or communities of organisms can survive through a period of unfavourable conditions.
seasonal waters	Subset of intermittent waters, which are predictably inundated in one or more seasons per year.
SIGNAL	Stream Invertebrate Grade Number Average Level.
stressor	Any physical, chemical or biological substance or process arising from a pressure that has the potential to induce an adverse environmental response to a community value.
temporary standing waters	Waterbodies that generally do not have through-flow of water or do not have a defined stream channel (e.g. pools, ponds, wetlands and lakes).
temporary streams	Watercourses that have flowing water for at least part of the wetting–drying cycle, generally resulting in a defined stream channel.
temporary waters	All waterbodies, be they standing or flowing waters, which alternate between phases of inundation and lack of surface water.
TN	Total nitrogen.
TP	Total phosphorus.
typology	A classification of items based on distinct types of categories.
water quality	the physical, chemical and biological characteristics of water and the measure of its condition relative to the requirements for one or more biotic species and/or to any human need or purpose.
water quality objective	The guideline value and/or narrative statement for each selected indicator for water quality that should ensure the protection of all identified community values. For the purpose of this document, this also includes sediment quality objectives.
weight of evidence	A qualitative, semi-quantitative or quantitative combination of multiple lines of evidence to make an overall assessment of water or sediment quality and/or associated management. This assessment incorporates judgments about the quality, quantity, relevance and congruence of the data in the different lines of evidence.

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