



**ECO-TOXICOLOGY ASSESSMENT:**

Analysis – mobilising contaminants at Reedy Lake

April 2013



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## Summary

Reedy Lake is part of the Lake Connewarre Complex, and an important wetland within the Port Phillip Bay (Western Shoreline) and Bellarine Peninsular Ramsar site. Prior to European settlement, Reedy Lake was most likely a variable system that experienced periods of drying and hypersalinity (Dr Jessica Reeves, pers. comm. March 2013). The hydrology and hydraulics of the system have been altered with several interventions since settlement, significantly changing the water regime of the wetland, and transitioning it to a permanent freshwater lake.

Future changes to the water regime have been proposed for Reedy Lake, to meet environmental objectives for the Lake Complex as defined by Lloyd *et al.* (2012). The proposed water regime includes more regular lake drying than has been experienced in the recent past. The new regime may include annual (summer) drying for the first 10 years for *Phragmites* control, followed by less regular (summer) drying (e.g. 1 in 4 years) for longer term ecological objectives. An important consideration for implementing a drying regime in coastal wetlands is the potential to expose acid sulfate soils, and mobilise other soil contaminants. Heavy metals are of particular concern in the Barwon River catchment that has experienced past mining, agriculture activities, and significant urbanisation and expansion of industry.

Based on geologic understanding of the region (Holocene sediments) and past recommendations (Fitzpatrick *et al.* 2007), there is a reasonable level of confidence that acid sulfate soils are present and that disturbance and/or exposure could pose a significant risk to the Reedy Lake ecosystem. The nature of this risk will vary as a function of the watering regime, concentrations of other contaminants, and other management interventions (e.g. *Phragmites* control, maintenance).

Analysis previously undertaken by RMIT and Ballarat University on a Reedy Lake sediment core revealed high levels of arsenic, chromium and nickel in many samples and more isolated high values of lead and mercury. Arsenic and chromium are particularly high compared to ANZECC sediment quality guidelines. No other data is currently available on contaminant concentrations in Reedy Lake.

As part of the screening level assessment completed for this investigation, potential contaminant concentrations in the water column were estimated using a partitioning coefficient approach. Results indicate that arsenic and chromium concentrations in the water are likely to be problematic at all lake levels associated with the past and proposed water regimes. While estimated concentrations are generally higher at lower lake levels, ANZECC guideline values are exceeded for nearly all scenarios assessed. At lower and dry lake levels (below 0.7m AHD) estimated concentrations of lead also begin to exceed guideline values.

Ecosystem risks posed by mobilizing contaminants were examined in this study through the calculation of hazard quotients. The hazard quotient calculation uses the estimated contaminant concentrations in the water (is not based on direct water samples). As such, this approach is also at the screening level of assessment only, intended to identify if there may be a problem. Results indicate that ecosystem risks associated with arsenic and chromium are likely to be high at all lake levels, and significantly increase for low/dry lake levels, raising concerns for ecosystem health associated with periodic drying of the lake. However, the amount of metal that is bioavailable (able to be taken up by biota) may be mitigated to some extent due by the presence of iron, sulfate and saline groundwater in Reedy Lake, which can bind up metals.

The past regulated water regime (predominantly high water levels) in the lake may have helped to limit the risk of acid sulfate soil exposure and mobilisation of contaminants to some extent, although there still may be ongoing problems of high arsenic and chromium concentrations. However, the high lake levels have negative implications for broader environmental objectives for the lake and the region (identified by Lloyd *et al.* 2012). *Phragmites* is continuing to colonise the lake, impacting on the mosaic of available habitat and vegetation types. Continuation of the past flow regime will not contribute to *Phragmites* control, and mechanical intervention would be required to remove *Phragmites*, which may increase the risk of disturbing the soil and triggering contaminant release.

The risk of both the annual and less regular drying have been assessed and are set out below:

**Annual drying:** An annual drying program for *Phragmites* control creates a high risk of acid sulfate soil exposure. Further, an annual drying program has the potential to create high contaminant concentrations in the lake at low lake levels (predominantly arsenic, chromium and lead). These risks are significant, and may outweigh the ecosystem benefits that the drying regime is intended to achieve (including *Phragmites* control). There may be factors that mitigate the bioavailability of the high levels of contaminants (e.g. saline groundwater intrusion, presence of iron and sulfate) and as a consequence the exact nature of the risk requires further investigation.

**Less regular drying:** In the case of less regular drying, there is a risk associated with contaminant release during drying years, however being less regular (1 in 4 years), there is opportunity, with appropriate monitoring in place, to detect and mitigate any contaminant issues before they become an issue that impacts on ecosystem health. Tidal flushing could be an option to reduce the risk of contaminant release associated with drying events.

It is important to emphasise that this investigation has been conducted at a screening level only, with no direct water quality samples available, and very limited sediment samples (one core only). While this investigation has revealed the potential for contaminant release to, and concentrations in, the water of Reedy Lake this does not mean that such release and concentrations will occur. Further investigations would be required to increase the confidence in the results of this screening level assessment.

This investigation has found that there is potential for contaminant release and high concentrations within the water of Reedy Lake associated with a drying program, and that as a consequence caution should be applied to the planning and implementation of any proposed drying program. Further investigations should be undertaken to inform and monitor any proposed drying program. A monitoring and evaluation program should include testing water quality (particularly pH), metal concentrations (soil and water), and metal bioavailability. Initial 'background' data should be collected and assessed prior to any interventions (drying or other intervention), and following drying events (if implemented). Such monitoring should extend into the future associated with the ongoing management of the lake. The monitoring and evaluation program should include (if and as possible) a control site that is not subject to the proposed interventions.

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# 1 Introduction

## 1.1 Project context

Reedy Lake is part of the Lake Connewarre Complex, within the Port Phillip Bay (Western Shoreline) and Bellarine Peninsular Ramsar site. More specifically, it is an important component of the Lower Barwon wetlands group on the Bellarine Peninsula.

In preparation for the establishment of an environmental entitlement for the Barwon River, Lloyd *et al.* (2012) reviewed flow/ecology relationships and scenarios for the Lower Barwon wetlands, and developed environmental flow recommendations to meet a suite of objectives for the Lake Complex. In developing these recommendations, Lloyd *et al.* (2012) noted that changes to the hydraulics of Reedy Lake since 1970 (including summer flooding) have resulted in conditions more favourable for *Phragmites australis*. The expansion of these reeds across Reedy Lake has reduced the extent of other aquatic plant communities and degraded fauna habitat, and is considered to threaten the habitat value of the lake as a whole (Lloyd *et al.* (2012)).

This threat forms the basis of an initial recommended watering regime by Lloyd *et al.* (2012) for Reedy Lake that includes annual summer drying for a ten year period. Once the reeds are controlled, Lloyd *et al.* (2012) recommend a watering regime for Reedy Lake that balances the water requirements of all flora and fauna, and includes summer drying one year in four.

An important consideration for implementing a drying regime in coastal wetlands is the potential to generate acid sulfate soils, and expose and mobilise other soil contaminants. Heavy metals are of particular concern in a catchment like the Barwon that has experienced past mining, agriculture activities, and significant urbanisation and expansion of industry. Preliminary analysis undertaken by RMIT and Ballarat University on a Reedy Lake bed sediment core revealed high levels of arsenic, chromium and nickel in many samples of the core, and more isolated high values of lead and mercury. The values of arsenic and nickel exceed the ANZECC guidelines (ANZECC/ARMCANZ 2000) for sediment found within the top 25 cm of the core, which is of particular concern.

Prior to European settlement, Reedy Lake was most likely a variable system that received minor inflows from a small local catchment, major flows from high flow in the Barwon River, and experienced periods of drying and hypersalinity (Dr Jessica Reeves, pers. comm. March 2013). Water depths were also likely to have been quite variable, with the lake occasionally drying out and at other times being full. The hydraulics of the system have been altered with several interventions since European settlement, significantly changing the water regime of the wetland, and transitioning it to a permanently freshwater lake, with the exception of occasional drying events for Carp control (approximately every 7 – 10 years). While metals remain in the sediment and inundated, they are relatively inert. However, alteration of the redox state and pH from exposure after drying may result in the metals either changing states and/or becoming mobile.

Before implementing a program of regular drying of Reedy Lake for both *Phragmites* control (initial 10 year period) and for long term ecological objectives, it is important to understand the risks of mobilising contaminants, and implications for Reedy Lake.

## 1.2 Project purpose

The purpose of this investigation was to gain an understanding of the potential risks associated with contaminants in Reedy Lake. The scope included to:

1. Detail the risk of mobilising contaminants and/or causing acid sulfate soils in the sediment of Reedy Lake if the following watering regimes were implemented:
  - a. Historical management of Reedy Lake
  - b. Annual drying of Reedy Lake over summer
  - c. Less regular drying of Reedy Lake over summer.
2. Identify the risks posed by mobilising the contaminant in the sediment or causing acid sulfate soils, which may include ecological, human health, economic and social risks.
3. Identify strategies that could be used to minimise risks under each of the proposed management regimes, and ongoing monitoring recommendations associated with key risks.

The scope of this investigation was limited to a desktop assessment using existing literature and available knowledge of the system.

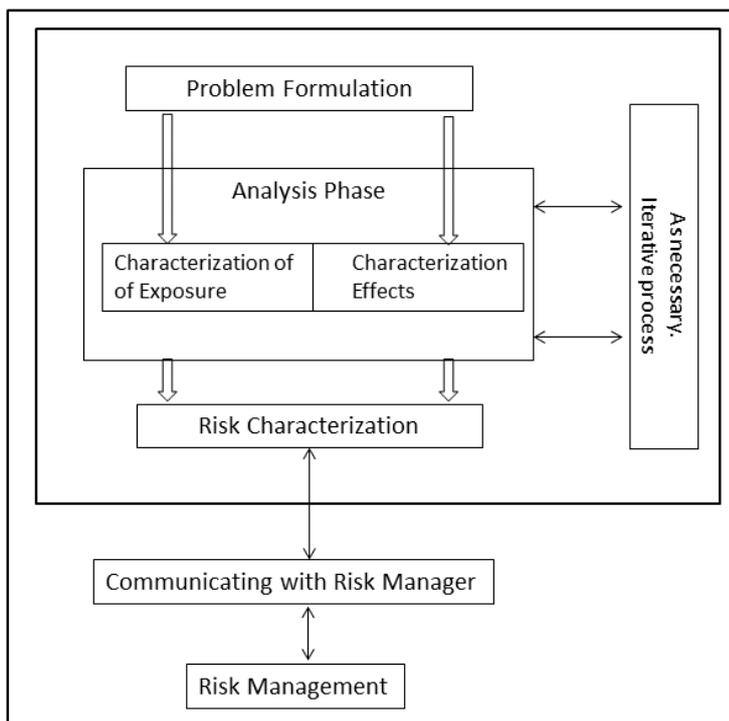
It is understood that the outcomes from this investigation will be used to inform future management decisions. In addition the investigation will inform the direction and outcomes of a concurrent investigation into alternative *Phragmites australis* control options for Reedy Lake (Alluvium 2013).

### 1.3 Project approach

#### Ecological risk assessment (ERA) framework

The ecological risk assessment (ERA) framework provides a consistent approach to structuring eco-toxicology investigations of this nature. The framework forms the basis for how information is structured within this report. The main steps of an ERA framework (USEPA 1998c) are (Figure 1):

1. **Problem formulation:** a preliminary characterization of exposure and effects is conducted by gathering data on the compounds of concern and of the ecosystem potentially at risk through literature research and by defining the objectives of the ERA. During the problem formulation phase, the assessor must develop a conceptual model containing working hypotheses regarding the stressor and its effect. The conceptual model is the main input to the analysis phase in which the model is evaluated and refined with further data.
2. **Analysis phase:** aims to characterise exposure and effect. Exposure is characterized through prediction or measurement of the spatial and temporal distribution of the stressor. The process of effects characterization identifies and quantifies the effect on the considered ecosystem component and is used to extrapolate a cause-effect relationship.
3. **Risk characterisation:** aims to determine the likelihood that exposure to the stressor results in adverse ecological effect. During risk characterisation, results from the model are compared and summarised.



**Figure 1.** Ecological risk assessment framework (modified from USEPA 1992)

#### Screening level ERA

A full ERA requires detailed information on field characteristics, geochemistry, mineralogy as well as consideration of water transport, flow regimes, lateral transport of acids and solutes and the role of groundwater. Only limited data is available on soil properties in Reedy Lake, and no data is currently available on water quality, therefore a screening level ERA was the most appropriate option for this investigation.

The problem formulation phase for this investigation involved a review of all relevant background information for Reedy Lake, including past landuse changes and lake modifications (Section 2 of this report). The analysis

phase included consideration of the sediment ecosystem, range of contaminants and effects of exposure (Section 3 and 4).

The screening level ERA completed for this investigation has three components to the risk characterisation phase (Section 5):

1. **Contaminants in the sediment:** Review of existing information on contaminants in the sediment (based on the sediment core analysis by RMIT and Ballarat University) and comparison to sediment quality guidelines.
2. **Contaminants in the water:** Estimation of the potential level of contaminants in the water (using RMIT and Ballarat University sediment data and partitioning coefficient ( $K_d$ ) values from the literature) and comparison to water quality guidelines.
3. **Mobilising contaminants:** Estimation of the risk to the ecosystem associated with mobilising contaminants, using the hazard quotient (Figure 2) (based on the potential water contaminant values calculated in 2., and ANZECC guidelines).

### Water and sediment quality guidelines

Numerous methodologies have been developed and have been used within ERAs to characterise the impairment of an environment. Where data is limited, water and sediment quality guideline values can be used within the risk assessment to identify concerns. Guidelines for contaminant concentrations in sediment are shown in Table 1, and for water in Table 2.

**Table 1. ANZECC/ARMCANZ sediment quality guidelines for selected metals and metalloids (ANZECC/ARMCANZ 2000)**

Contaminant	ISQG-low (mg/kg)	ISQG-high (mg/kg)
Chromium	80	370
Lead	50	220
Mercury	0.15	1
Nickel	21	52
Zinc	200	410
Arsenic	20	70

**Table 2. ANZECC/ARMCANZ (ANZECC/ARMCANZ 2000) water quality guidelines – the 95% protection of species values are used for the analysis in this report (green column)**

	Level of protection ( $\mu\text{g/l}$ ) (% species)			
	99%	95%	90%	80%
<b>As(III)</b>	1	24	94	360
<b>As(V)</b>	0.8	13	42	140
<b>Cr(VI)</b>	0.01	1	6	40
<b>Pb</b>	1	3.4	5.6	9.4
<b>Hg</b>	0.06	0.6	1.9	5.4
<b>Ni</b>	8	11	13	17

### Hazard quotient

At screening level (Suter 1993), the method most widely used to estimate risk is the quotient method (Landis and Yu 2004; USEPA 1992; USEPA 1998a), in which the hazard quotient (HQ) is calculated as shown in Figure 2.

$$HQ = \frac{PEC}{PNEC}$$

Where:

HQ = Hazard quotient

PEC = Predicted Environmental Concentration at the site, i.e. how much contaminant is in the soil, sediment, or water (e.g. mg contaminant/kg soil).

PNEC = Predicted No-Effect Concentration at the site, i.e. generally a no-adverse effects level concentration. If the contamination concentration is below this level, the contaminant is not likely to cause adverse effects. To quantify the risk for the water overlying the sediment, the PNEC value should be sourced where possible from evidence based reference data. However in the absence of this, estimates can be obtained from the ANZECC/ARMCANZ (2000) water quality guidelines values that protect 95% of species with 50% confidence (Table 2).

If the quotient is less than some threshold level, the concentration in the environment is considered safe, otherwise it is unsafe. Usually (USEPA 1998a):

HQ > 1: Harmful effects are likely due to the contaminant in question

HQ = 1: Contaminant alone is not likely to cause ecological risk

HQ < 1: Harmful effects are not likely.

The occurrence of multiple contaminants can be modelled as additive at the screening level (Suter *et al.* 2000):

$$HQ = \sum_i \frac{PEC_i}{PNEC_i}$$

The assumption of additive toxicity is accurate for chemicals with the same mode of action, and is conservative in most cases for mixed mechanisms of action (Suter *et al.* 2000).

**Figure 2.** Hazard quotient formula

Where significant concerns are identified at this screening level of assessment, further detailed investigation is recommended, along with other management recommendations (Section 6).

## 2 Background

### 2.1 Site description

Lloyd *et al* (2012) provide an extensive description of Reedy Lake from European settlement to present time, as well as an overview of the ecosystem dynamics including geomorphological processes, vegetation, waterbirds and fish. Exerts of background information relevant to this investigation are summarised here and in the following sub-sections (from Lloyd *et al* (2012) unless otherwise indicated).

Reedy Lake is a shallow freshwater sub-circular basin of 550 ha bordered by a higher terrestrial landscape on the west, north and eastern sides. It is located to the north-east of the lower Barwon River and is connected to Lake Connewarre on the southern side. The lake has a small local catchment of 27 km<sup>2</sup> flooded almost entirely from the Barwon River (Figure 3). The Barwon River channel flows from north-west to south-east on the southern margins of Reedy Lake, and discharges to the tidal, open water environment of Lake Connewarre. The river is separated from Reedy Lake by a natural levee.



Figure 3. Reedy Lake in the Lower Barwon wetland region (courtesy of CCMA)

### 2.2 Landuse change

The site has seen a series of agricultural and industrial developments since the early 1800s, when the area was opened up to sheep grazing and associated industry (wool scour, tannery). During the 1850's gold rush vast tracts of land upstream were cleared for mining, and tailings from the mining operations flowed through the catchment and into the Barwon River and its lower wetlands. These tailings have been found to be a source of toxic, heavy-metal contaminants, as discussed below. During the first half of the 20<sup>th</sup> century recreational shooting became common in the area (Dahlhaus *et al.* 2007). The Connewarre complex more generally has been slowly infilling over time, due to ongoing inputs of sediment from the catchment (Victorian Saltmarsh Study 2011). These sediments have brought with them a chronic nutrient load, resulting in nutrient enrichment and eutrophication of the wetland complex (Victorian Saltmarsh Study 2011).

A detailed history of landuse changes for the Lower Barwon region is documented by Lloyd *et al* (2012) based on work by Dr Erica Nathan (in Dahlhaus *et al.* 2007), and key periods are noted for reference in Table 3.

**Table 3. Periods of landuse change in the Lower Barwon region (after Lloyd *et al.* 2012)**

Period	Events
Pre settlement	Aboriginal land management – light harvesting of the lakes natural resources and mosaic burning of the surrounding country.
1830 – mid 1850s	Early pastoral runs and first breakwater established.
Mid 1850s - 1890	Agricultural subdivision, gold mining waste begins, extensive riverfront industries, two major flood events, dredging, commercial shooting of wildfowl.
1890 - 1950	Agricultural decline, tourism, harbour trust management (lower breakwater), gold mining waste ends, Geelong sewerage schemes, river stabilisation works.
1950 - 1980	Growth of Geelong and Bellarine Peninsula, State Rivers and Water Supply Commission lower breakwater works, early conservation management, cement quarry discharge, erosion of lower Barwon, recreational lobby, high rainfall of early 1950s.
1980 - present	Ramsar wetland listing, Parks Victoria management, Hospital Swamps project, cessation of grazing and irrigation licences, estuarine studies, residential pressure.

High concentrations of chromium have been found in the Reedy Lake sediments (by the RMIT and Ballarat University sediment core pilot study). This element is most likely to have been a by-product from a local tannery. High levels of arsenic and mercury are also present, these are likely to have been a result of gold mining and refining in the catchment during the gold rush. Lead has also been found in the sediments, this is most likely associated with lead-shot from the game reserve and from use of lead at the wool scour (Grundell *et al.* 2012). The nickel found in the sediments is likely to have originated from the local volcanic rocks (Fabris *et al.* 2006).

### 2.3 Lake and water management

Prior to European settlement, Reedy Lake was most likely a variable system that received minor inflows from a small local catchment, major flows from high flow in the Barwon River, and experienced periods of drying and hypersalinity (Dr Jessica Reeves, pers. comm. March 2013). The river was originally estuarine upstream of the lake, and the lake would have received a combination of saline and freshwater inflows, depending on the contribution of high tides and river levels.

Prior to modifications, the spill of water from the river to Reedy Lake was controlled by the natural levee of the river bank. Water would have spilled onto the wetland several times a year in response to flood events and winter and spring freshes. On the basis of seasonality in regional climatic patterns, it is believed that the wetland flooded most often in winter and spring, and then dried out over summer and autumn. Inflows regularly exceeded the lake's capacity and discharged to Lake Connewarre to the south. The sill between the wetland and Lake Connewarre allowed saline estuarine water to enter the wetland during high tides.

The hydrology and hydraulics of the system has been altered with several interventions since settlement, significantly changing the water regime of the wetland, which is now a freshwater system. Lloyd *et al.* (2012) document a detailed history of lake and flow regime modifications, a summary of which is provided in Table 4.

**Table 4. Reedy Lake past modifications (after Lloyd *et al.* 2012)**

Period	Events
1898	A weir (the lower breakwater) was constructed where the Barwon River discharges to Lake Connewarre to raise the river level upstream and prevent the incursion of saline estuary water. The weir raised the level adjacent to Reedy Lake and presumably promoted inflow events.
1906	Reedy lake was described as drying out only in very prolonged dry weather. The water level in the Barwon River was 0.3 m higher than the lake, so that water could easily be diverted into the lake.
1950s	The lower breakwater was replaced (by SRWSC) with a floating gate structure at a lower level to manage the higher rainfall that was occurring at the time and the high flows that were expected to result from the Corangamite Drainage Scheme. The new structure held the Barwon River at 0.3 m below the lake.

Period	Events
1953	A channel was cut between the Barwon River regulator and Reedy Lake to aid inflows. Water entering the lake rapidly drained to the estuary.
1960s	<p>Competing interests emerged during the dry years of the 1960s, particularly 1960, 1961 and the drought period of 1967-68. Irrigators required a high Reedy lake water level to provide water for irrigation, while graziers sought a low summer level to allow dairy cattle grazing. Field and Game sought to retain water in Reedy Lake in summer over a minimum of two thirds of the wetland area to provide habitat for waterfowl. The inlet structure was sabotaged on several occasions including in 1967 when the lake was dry and farmers sought to fill the lake. The typical water regime in the 1960s has been described as flooding in the central lake 3 to 4 times per year, with water retained between floods and mostly dry by February (prior to duck hunting season).</p> <p>Management objectives for Reedy Lake changes after 1961 when the Wildlife reserves Investigation Committee recommended the area as one of the new State Wildlife Reserves. There was concern that waterbird numbers has been declining over the previous 50 years and that the lowering of the river level by the new lower breakwater had adversely impacted Reedy Lake. The lake came under the management of the Division of Fisheries and Wildlife, however much of the on-ground management work (fencing, revegetation) was undertaken by the Field and Game Association.</p>
1967-68	In order to maintain water levels in the 1967-68 drought, the bank at the outlet was raised to 0.7 – 0.8 m AHD. These arrangements increased the wetlands volume, depth, area and permanence.
1970	The outlet was modified by construction of an embankment across the natural outlet drain with a spillway height of around 0.7m AHD. The effect of the embankment was however to prevent tidal water egress.
1973	A report was prepared on the effects of grazing in Reedy lake.
1974 (circa)	A new larger capacity inlet and channel was constructed (the current structure).
1983	All grazing and irrigation licences were terminated by 1983.
1995	Until the 1990s, Barwon River flows were supplemented by brackish water discharged to the Moorabool River from a dewatered quarry. This maintained the weir pool upstream of the lower breakwater an increased the frequency of spill events form the river to Reedy lake. In 1995 Field and Game drained the lake by cutting a new outlet.
1996	<p>In 1995/96 the lake was drained to eradicate the carp population that had developed and facilitate works on the drainage flow path. The flow path from the lake to the Barwon River below the Lower Breakwater was excavated to a depth of approximately 0.1 m AHD, allowing the wetland to be drained when lake levels permit. The drop board regulator on the outlet channel was constructed at this time.</p> <p>A major management objective has since been to dry Reedy Lake every 6 to 7 years for Carp control.</p>
1997	The regulator from the Barwon River was upgraded by closing two of the box culverts and replacing the flap gate on the third with a penstock regulator and fish screen.
2005-2006	<p>The lake was dried out and the channel from the inlet structure to the lake bed was excavated to 0.1 m AHD and cleared of vegetation. Accumulated sediment and vegetation was also cleared form the outlet channel, to a depth of 0.1 m AHD.</p> <p>The inlet fish screen was replaced with a sloping screen to minimise clogging, single penstock removed and all culverts re-opened. Groundwater monitoring bores were also installed.</p>

## 2.4 Groundwater

Groundwater salinity varies considerably across the lower Barwon River and estuary complex. In the Reedy lake sediments, groundwater EC varies from 15 – 25 mS/cm, whereas at the northern and western edges of the lake it is consistently a little higher. Salinities of groundwater in the shallow geological units surrounding Reedy Lake and Hospital Swamps (Newer Volcanics, Fyansford Formation and Quaternary sediments) generally range from 20 to 60 mS/cm (Dr Matthew Currell, pers. comm., March 2013). By way of comparison, seawater has an EC of about 52 mS/cm, so the groundwater underneath the wetlands complex is roughly between one-quarter and one-half seawater salinity. The source of salt in the groundwater may be partly oceanic, but it is also likely that evapotranspiration is a major control on groundwater salinity (Currell *et al.* 2012).

The salt and water environment in Reedy Lake is understood to have been largely stable since the 1970's when the last modifications to the hydraulics of the system were completed. However, this has not been confirmed with monitoring data as groundwater monitoring in the region did not commence till 2006 (Dr Matthew Currell, pers. comm., March 2013).

## 2.5 Geology and sediment

The geology of the sediments underlying the Lower Barwon is of the Holocene epoch. The wetland sediments are fine-grained, being dominantly clay-, silt- and fine sand-sized. Significant sediment deposition is episodic, occurring only when turbid river water enters the wetlands during high flow events. The bed sediments can be re-suspended, causing turbid water conditions, episodically in association with periods of high wind velocity. The events also generate waves, which can erode exposed shorelines. Bed sediments are also subject to the ongoing process of mixing by organisms (bioturbation by animal burrowing). In its current configuration, Reedy Lake receives only a small percentage of the Barwon River sediment load, with most of it being transferred to Lake Connewarre (Lloyd *et al.* 2012).

## 2.6 Ecosystem

Reedy Lake is now a freshwater wetland containing diverse aquatic habitats and supporting 16 native fish species. These habitats include open water, reed beds, and submerged aquatic plant beds. Reedy Lake has been recognised as an area of significant wetland vegetation since settlement. Lake salinities declined after the installation of an embankment on the outlet around 1970, which excluded saline estuary water from the lake. Submerged aquatic plants increased in response so that by 1979 the lake supported approximately 25% open water, 25% submerged aquatics and 50% reeds. Despite the arrival of carp around 1979, the lake vegetation retained this general structure until 1990 (Lloyd *et al.* 2012).

Extensive changes in vegetation in the seasonally waterlogged periphery of Reedy Lake occurred between 1983 and 2010 corresponding with the removal of stock. The extent of *Phragmites australis* and *Typha orientalis* in Reedy Lake has fluctuated dramatically (Lloyd *et al.* 2012) between 1983 and 2010, with carp strongly indicated as a dominant influence (along with removal of stock). Carp were introduced in 1979 and when in high abundance, reduce the variety of plant communities present in the lake and change the physical and chemical character of the water column (Ecological Associates 2012). Carp are benthivorous fish that feed in and on the sediments to a depth of about 12 cm into the soil. They resuspend sediments and in doing so likely reduce visibility for visual-feeding fish. Carp can destroy existing vegetation and, in particular, soft bodied and young, recolonising plants. Carp also alter prey availability for non-benthic predators (Driver *et al.* 2005).

Modifications to Reedy Lake have contributed to an increase in the depth and permanence of water and reduced salinity, making conditions more favourable for *Phragmites australis* growth than existed in pre-European times. *Phragmites* grows well in areas that are subject to spring and summer inundation to a depth of 0.2 to 1.0 m, and an increase in wetland depth and permanence would have expanded the available habitat. *Typha* reeds are also a dominant component of the wetland vegetation mosaic.

The diversity and populations of birds supported by habitats within Reedy Lake have generally declined, relative to other wetlands in the complex. The foraging conditions in Reedy Lake that in the past attracted large populations of waterbirds, particularly waders, have over the last forty years occurred less frequently. Habitat destruction by Carp is considered to be one of the primary factors in this decline, as significant recovery in bird numbers has been observed after Carp control (drying) events (Ian McLachlan, Field and Game Association, pers. comm. March 2013).

Fish diversity is high with Reedy Lake having 16 native fish species. The vast majority of the fish species prefer either open water or areas of dense submerged aquatic vegetation. The fish are important for their conservation value, value in fisheries (eels), and as food for waterbirds. Many of the fish species require, or utilise, aquatic vegetation beds as critical habitat. Fish require a water regime that supports aquatic vegetation and open water habitats at key stages of their biology such as feeding and growth, spawning of eggs and recruitment of juveniles. The lake has had a semi-permanent water regime in the past with long periods of inundation which has proved ideal conditions for the species present.

## 2.7 Watering regimes

The water regime of the most recent past as described by Lloyd *et al.* (2012), has seen the lake at a high level in winter/spring (0.7m AHD), drying to average 0.4m by late summer, and refilling in response to autumn breaks, usually April/May (Table 5).

**Table 5. Reedy Lake Water Regime in the recent past (Lloyd *et al.* 2012)**

Season	Typical hydrological environment
Winter/spring	High ( <b>0.7 m</b> AHD)
Late summer	Drying to average <b>0.4 m</b>
Autumn	Refilling in response to autumn breaks, usually April/May

To support a trajectory of change towards the vegetation types recorded in 1983, Lloyd (2012) proposed the water regimes summarised in Table 6 and Table 7. The lake levels (in m AHD) identified for the flow regimes (past and proposed) are used in the risk characterisation undertaken in Section 5 of this report.

**Table 6. Water regime recommended for Reedy Lake (modified from (Lloyd *et al.* 2012))**

Season	Typical hydrological environment	Hydrological objective	Frequency
Early Winter (June to July)	Low flows in the Barwon with the possibility of minor freshes.	Allow wetland to fill with moderate and high flows in Barwon River.	9 years in 10
Winter-spring High Flow Period (August to October)	High flows in the Barwon River. Overbank flows occur intermittently.	Fill Reedy Lake and allow overbank flows to surcharge and flush the wetland.	9 years in 10 High flows to extend through to November 4 years in 10 years
Late Spring – Early Summer Drawdown Period (November to December)	Moderate flows in the Barwon. Overbank flows less frequent.	Drain Reedy Lake to a level of less than 0.3 m AHD before the end of December.	Every year; No more than one year in 10 without drawdown (in very wet years)
Late Summer – Autumn (January to May)	Low flows in the Barwon. Overbank flows rare.	Maintain water level at or below 0.3 m AHD. Drain overbank flows at normal rate	8 years in 10; no more than 5 years between events.
Late Summer – Autumn (January to May)	Low flows in the Barwon. Overbank flows rare.	Allow water level to drop below 0.3 m AHD and dry. Drain overbank flows as rapidly as possible.	2 years in 10; Wetland dry - March to May.

**Table 7. Future water regime for Reedy Lake (modified from (Lloyd *et al.* 2012))**

Scenario	Typical hydrological environment	Hydrological objective	Frequency
Wettest 25% of years	High flows in the Barwon with multiple overbank flow events in winter, spring and summer	Maintain high lake level (at or near <b>0.8 m AHD</b> ) throughout the year.	1 year in 4
Typical (interquartile) years	Moderate flows in the Barwon with frequent freshes and overbank flow events in winter and spring	Allow wetland to fill in winter and spring to <b>0.8 m AHD</b> . Gradually reduce water levels to <b>0.3 m</b> at an approximate rate of 7 cm per week, starting December 1. Restart drawdown following overbank flows in summer, if any.	2 years in 4
Driest 25% of years	Low flows in the Barwon with infrequent freshes	Allow wetland to fill in winter Gradually reduce water levels to <b>0.0 m</b> at an approximate rate of 7 cm per week, starting November 1. Restart drawdown following overbank flows in summer, if any.	1 year in 4

## 2.8 Stressor identification

A stressor is any physical, chemical or biological entity that induces an adverse response. Stressors of particular concern for Reedy Lake include acid sulfate soils and heavy metal contaminants.

### Acid sulfate soil

Acid sulfate soil is the common name given to soils and sediments containing iron sulphides. When exposed to air due to drainage or disturbance, these soils produce sulfuric acid, often releasing toxic quantities of iron, aluminium and heavy metals. Acid sulfate soils commonly occur on coastal wetlands as layers of Holocene marine muds and sands deposited in protected low-energy environments such as barrier estuaries and coastal lakes. In similar environments they are still being formed. Formation occurs when seawater or sulfate rich water mixes with land sediments containing iron oxides and organic matter in a waterlogged situation, in the absence of oxygen.

Acid sulfate soils can also form under modern freshwater conditions in inland settings, especially in the higher rainfall (> 500 mm per annum) mediterranean environments of Australia where locally extreme changes in hydrology and geochemistry have occurred. These inland acid sulfate soils develop as a result of contemporary land clearing, excess discharge of saline groundwater and erosion to form unsightly discharge areas, with eroded "iron ochre scalds" (Fitzpatrick 2003).

Reedy Lake's geology has been identified as of the Holocene era (Lloyd *et al.* 2012), and is a typical environment for acid sulfate soils (Sammut *et al.* 1995).

### Heavy metals

Preliminary analysis undertaken by RMIT and Ballarat University on the Reedy Lake core samples revealed high levels of arsenic, chromium and nickel in many samples and more isolated high values of lead and mercury, exceeding the ANZECC guidelines for sediment (ANZECC/ARMCANZ 2000).

The degree to which heavy metals impact on the ecosystem depends on the toxicity and uptake. Metal toxicity is related to the internal exposure or dose, while metal uptake from water or sediment is related to the metal bioavailability. Metal bioavailability is dependent on the chemical form of the metal (speciation) that relates to the water quality characteristics such as pH, hardness, alkalinity, salinity and dissolved organic matter. In general, a decrease in pH increases the amount of potentially toxic "free" metal ion - this can be counteracted by the increase in protons ( $H^+$ ) which compete with metal binding on biota, thereby reducing toxicity. Metals in dissolved form are generally more bioavailable, at least via the water exposure route, than metals bound to particulates. A metal, once inside the organism can exert toxic effects if it binds to active sites (eg. fish gills) (Luoma 1989; Tessier and Campbell 1987).

Alteration of the redox state and pH, could arise from the planned wetting and drying regimes, and may result in the metals either changing states and/or becoming mobile and bioavailable (Du Laing *et al.* 2009).

## 2.9 Assessment endpoints

In considering the background physio-chemical environment and ecosystem values of Reedy Lake, the assessment endpoints associated with this eco-toxicology assessment are to:

1. Maintain a physico-chemical environment (pH, salinity, turbidity) that supports a healthy aquatic ecosystem
2. Maintain habitat and key species.

These assessment endpoints cannot be specifically measured in a screening ecological risk assessment (due to data limitations). However, water and sediment quality guidelines e.g. ANZECC/ARMCANZ (2000) can be used as reference data to identify significant concerns for the relevant stressors (as outlined in Section 1).

### 3 The sediment ecosystem

This section provides a detailed literature review on dynamics of the sediment ecosystem, providing background understanding to the lake bed environment and ecosystem dynamics.

#### 3.1 Sediments

The sediment environment is an integral component of the aquatic ecosystem. Sediments are semi-solid media comprised of minerals, organic materials, interstitial water and biological components (Hakanson 1992). The sediments are deeply influenced by the lithological characteristics of the watershed, by the coastal and or riparian vegetation and by the activity and biomass of the aquatic community (Connell 2007). The biotic and the abiotic factors of sediments are intimately linked; changes in the physical environment will influence the biological community and vice versa. The sediment community typically comprises bacteria, algae, macrophytes, microfauna, meiofauna and macroinvertebrates.

#### 3.2 Sediment-water interface

Chemical partitioning to sediment and colloidal particles via adsorption is controlled to some extent by sediment temperature, interstitial spacing and contact between particles and dissolved organic and inorganic compounds in the interstitial or pore water (Mudroch et al. 1997). The surface layer of sediment is normally oxidised over a depth of one to several millimetres, acting as a barrier to the transport of dissolved and reduced inorganic complexes.

The sediment-water interface is a site of high biological activity. Particulate organic matter that reaches the sediment is subject to extensive degradation by the benthos prior to burial (Carlton and Klug 1990). Remineralisation is significant in the top 10 cm of sediment. Most of the particulate organic matter is remineralized after death of organisms or through excretion. Particulate matter can bind to various chemicals and remain suspended in the water column or deposited into the sediment. The continual flux of inorganic and organic compounds through the sediment-water interface can be accelerated by biological activity (Burton 1992). Deeper sediments are passive and more permanently in place. The deeper layers may be reintroduced into the active portion of the ecosystem by dredging activities, bioturbation, severe storms and hydrological events (Burton 1992). Burrowing biota (e.g. Carp) may also disturb deeper sediments.

#### 3.3 Benthic communities

Benthic communities are a critical component of aquatic ecosystems. They influence surface productivity, alter the physical and chemical condition of the sediment and sediment-water interface, and transfer energy to higher trophic levels.

Microbially colonised particles represent the main food source of burrowing organisms (Krantzberg 1985). Billions of bacteria per gram of sediment occupy critical niches, metabolizing and cycling organic and inorganic compounds, acting as mediators in the cycling of many elements and compounds, contributing to benthic respiration of oxygen and to metabolism of many compounds (Burgess and Scott 1992). The metabolic by-products of a group of bacteria are energy sources for other organisms (Burton and MacPherson 1995).

Structural changes to the sediment's substrate (e.g. vegetation and polychaete tubes) also modify the hydrodynamics near the sediment-water interface, and consequently influence the settlement and habitat of residing benthos, and the deposition of detrital and sedimentary material. The activity of mobile and sedimentary benthos (feeding, burrowing, tube construction and irrigation) move sedimentary particles and fluids affecting geochemical processes near the sediment-water interface. Bioturbation involves particle transport and exchange of sediment pore water and overlying water (Kennish 1986): benthic macroinvertebrates burrow into the sediment's anoxic zones and create oxidised conditions within their burrows, altering their exposure to ionic species of contaminants and resulting in transport of contaminants to and from sediments (Burton and MacPherson 1995). Physical and chemical changes in sediment parameters resulting from bioturbation may enhance microbial activity, which in turn result in some of the chemical changes observed as a consequence of bioturbation (Krantzberg 1985).

Trophic-group distribution related to foraging activities appears to be an essential element in the modification of the benthic environment (Kennish 1986). Sediments constitute a rich substrate for different types of organisms of all trophic levels; however, heterotrophs are the main group. The metabolic by-products of a group of bacteria are energy sources for other organisms (Burton and MacPherson 1995). If sediments are reached by sunlight, they will host photosynthetic activity. The epiphytic algae use the nutrients stored in the sediments, and actively participate to the cycle of numerous compounds. The mixture of benthic and planktonic forms may be particularly pronounced in shallow lakes where planktonic algae are in continuous contact with the sediment surface (Hankanson and Jansson 1983).

Trophic relationships exist between marine benthic organisms and fish (Xu and Wang 2002) and direct transfer of metals from invertebrates to fish have been demonstrated (Mathews et al. 2008). Although fish do not strictly live in the sediment, many are bottom-feeding and represent the top of the food chain (*Clearwater et al.* 2005).

### **3.4 Aquatic plants**

Macrophytes can be free floating or permanently attached to the sediments, they are able to create a special environment sheltered from wave turbulence and with a rich variety of environmental microzones with high biological activity due to dense benthic populations and intense bacterial metabolism (Burton 1992). A direct link between plants and sediments occurs via the root system, which is the main way of nutrition. Nutrition represents a considerable path of transport of essential elements from the sediment to the plant biomass and to the lake waters, after the death of the plants. Emergent and submergent aquatic plants in estuaries, bays, and wetlands provide shelter for many fish and invertebrate species and are important also as food resources and in energy exchange. The presence of phytotoxicants and phytostimulants in water and sediment can alter plant condition and community composition and result in excessive plant growth and reductions in diversity and density (Lewis et al. 2001). Macroalgae support a wide variety of invertebrates, including molluscs, annelids and crustaceans; higher trophic organisms such as fish as supported by algae (herbivores) as well as by invertebrates (carnivores).

## 4 Contaminants and associated risks

### 4.1 Contaminants in sediments

Contaminants can alter the structure and dynamics of freshwater, estuarine and marine environments. Changes can arise when contaminants impair feeding, growth, development energetics, and recruitment of populations (USEPA 1998b), or cause death. Sources of contaminants comprise wastewater discharges, chemical industry wastes, oil spills and leakages, dredged spoil, mining, agriculture and aquaculture wastes. Contaminants incorporated in the food chain at lower trophic levels may result in harmful effects at higher trophic levels (Kennish 1998).

Contaminated sediments can have an effect on aquatic life by making areas uninhabitable for benthic organisms, and they can affect fish and wildlife by contributing to the bioaccumulation and biomagnification of contaminants in the food chain. Documented adverse ecological effects from contaminated sediments include fin rot, increased tumour frequency, and reproductive toxicity in fish as well as decreased biodiversity in aquatic ecosystems (USEPA 1998b). In addition to effects on benthic organisms, some sediment-associated contaminants may pose a direct risk to wildlife and human health through direct consumption of contaminated benthic organisms such as clams and lobsters, or an indirect risk through the trophic transfer of contaminants up the food chain into edible fish (Brown 2005; Luoma et al. 2005).

### 4.2 Metals

#### Uptake and accumulation

Metals have been recognised as different from the organic contaminants (Chapman and Wang 2000), presenting unique assessment issues (Fairbrother et al. 2007). Metals can be present in both the sediment (particulate phase) or water (dissolved phase), and have various transport pathways including groundwater inputs, surface water runoff, and disturbance of bed sediments.

Trace metals can be taken up from solution and from food, both of which are influenced by metals in sediment. In solution, salinity and chelating agents can affect bioavailability. In sediments, metals partition not only among geochemical forms but also between inorganic and organic material. Bioavailability differs among metals and for each metal it probably differs widely among food sources as it will depend on how food is internally processed (Luoma et al. 2005). Sulfides influence metal distribution between particles and water and hence the metal assimilation via diet. The trophic transfer can be influenced by the biogeochemical conditions which affect the form of metal in particles and hence the quantity that is taken up via diet. Metals taken up from diet or water can have different disposition within the body of fish and invertebrates. Diet-borne metal exposure can result in whole body or tissue-specific metal burdens well in excess of those resulting from waterborne exposure, causing adverse effects (Mount 2005).

Trace metals accumulate at very variable concentrations in aquatic invertebrates (Rainbow 2002; Wang and Rainbow 2008). Essential metals (copper, zinc) provide essential function in the organisms and are required as micronutrients, whereas non-essential metals (cadmium, lead) are not required by the organisms and may induce toxic effects even at low doses (Weis 1991). Essential and non-essential metals are known to behave differently (Rainbow 2002). Toxic effects ensue if an excess of metal is present internally in a form that is metabolically available (Luoma and Rainbow 2005). Most aquatic organisms have specific mechanisms for uptake, internal transport, sequestration and depuration of metals (Wang and Rainbow 2008). In some cases, metals are actively regulated, in part due to the organism's physiological requirements for those metals that are necessary for normal metabolic functions. The same mechanisms will apply to those metals that mimic essential elements. Detoxification of essential and non-essential metals can involve the formation of metallothioneins.

Six biological processes have been indicated as important in the cycling of contaminants in benthic systems (Burgess and Scott 1992): bioaccumulation, trophic transfer, migration, biodegradation, biodeposition, and bioturbation. Environmental processes affect the concentration of metal ingested by an organism, the form of the metal that is ingested and an animal's access to diet-borne contaminant. The exposure of benthic

organisms to contaminants, and thus to metals, is affected by a multitude of factors, associated with the variability of the organisms, of the sediments, and of the contaminants (Luoma and Rainbow 2008).

### **Metals in lakes**

Metals in contaminated lakes have been shown to exert direct and indirect effects on biota. In freshwater environments, dietary exposure to metals poses a threat of reduced survival and growth to fish, especially in early life stages, when feeding almost exclusively on small invertebrates (Maret *et al.* 2003). In estuarine environments, salinity is an important controlling factor for contaminant partitioning between the sediment and the overlying water and thus for bioavailability.

For metals, an increase in salinity leads to an increase of the partitioning coefficient between sediment and water, and thus to the removal of the metals from the water column. The partitioning of the metal in either the sediment or the water phase will lead to higher uptake by deposit feeders or by filter feeders and by dermal uptake respectively (Chapman and Wang 2001). However in some cases, greater salinity may lead to greater competition for metal binding sites, and actually increase metal content in water (via de-sorption) (Dr Suzanne Reichman, pers. comm. March 2013).

Exposure of wetland plants to contaminants may not be easily attributed to a single medium (Suter *et al.* 2000), as the water/sediment equilibrium will strongly affect the exposure. In metal contaminated marshes, live and dead plant tissue can become a sink for metal pollutants (van der Welle *et al.* 2007; Windham *et al.* 2004).

Burrowing and feeding by benthic organisms resuspends contaminants, increasing their biological availability both to the benthic fauna and flora, and to the higher order organisms which feed on them (Taylor and Maher 2010). However, the presence of high concentration of total metal in sediments is not per se indicative of ecosystem impairment. Experiments aimed at linking exposure to heavy metals in the sediment to bioaccumulation in the tissue and toxicity effects (Taylor and Maher 2012), showed that although final tissue concentrations were considerably lower than the sediment exposure concentrations, elevated lead and cadmium concentrations in sediments have the potential to increase biologically active lead burdens and impair the antioxidant reduction capacity, leading to a series of associated effects from lipid peroxidation to cellular perturbation and genotoxic damage.

At Reedy Lake, concerns have been raised regarding the release of arsenic, chromium, nickel, lead and mercury, likely originating from past mining, industrial and recreational activities, and local geology (nickel, arsenic) (refer Table 3).

### **Arsenic**

Arsenic (As) is found in the environment as a result of natural weathering processes, volcanic activity and anthropogenic inputs, and is usually associated with sulphide minerals (Merian 2004), the most common of which are arsenopyrite (FeAsS) and As-rich pyrite. When acid sulfate soils are exposed to aerobic conditions, the oxidation of As-rich pyrite can release significant amounts of arsenic into the environment.

Several forms of arsenic occur in natural waters, depending on the redox potential and pH; however As (III) and As (V) are the most common. Once in the aquatic environment, arsenic may be adsorbed to metal oxides in sediments, in particular, iron, aluminium and manganese. Clays also adsorb arsenic as a result of the oxide-like character of their edges. Similar to soils, under acidic, anaerobic conditions, arsenic release from sediments tends to increase as iron is reduced and arsenic is redissolved. As (III) is the more toxic form, it is removed (sorption) by sulphides and As (V) by clays (and clays can adsorb both redox states of arsenic). Acute toxicity of arsenic (III) to freshwater invertebrates occurs at concentration as low as 812 µg/l. Adult freshwater fish are usually less sensitive to arsenic: acute toxic response was observed at arsenic concentration above 13300µg/l (ANZECC/ARMCANZ 2000).

### **Chromium**

In the natural environment, chromium (Cr) occurs as either trivalent Cr(III) or hexavalent Cr(VI). The two differ greatly in terms of their physical and chemical properties, as well as their bioavailability and toxicity to aquatic biota. Cr(III) is considered to be an essential trace element for most living organisms. In animal cells, it plays an

important role in the metabolism of glucose and fatty acids. In contrast, Cr(VI) is highly toxic aquatic organisms and able to freely diffuse across cell membranes and oxidise intracellular compounds. Once inside the cell, the reduction of Cr(VI) to Cr(III) also forms free radicals, and the Cr(III) that is formed can exert additional toxicity by inhibiting some metallo-enzyme systems (Merian 2004).

In natural waters, the ratio and presence of Cr(III) and Cr(VI) is controlled by pH, oxygen concentration and the presence of reducing agents, oxidation mediators and complexing agents. In anoxic to suboxic conditions, Cr(III) should be the only form present. In oxygenated waters, the strong oxidant Cr(VI) reacts readily with reducing agents such as sulfur dioxide to form Cr(III), while the oxidation of Cr(III) to Cr(VI) is slow, but increases with temperature and the presence of manganese oxides. In soils and sediments, chromium can undergo a number of transformations including oxidation, reduction, sorption, precipitation and dissolution. A large portion of Cr(III) is readily adsorbed to clay compounds. In oxic sediments, a portion of Cr(III) can be oxidised to Cr(VI) by MnO<sub>2</sub> and, to a lesser extent, dissolved oxygen. The lowest freshwater acute toxicity reported for Cr(III) was 3300 µg/l for fish and 1200 µg/l for *Daphnia magna*, while for Cr(VI) was 220 µg/l for fish and 5.3 µg/l for *Ceriodaphnia magna* (ANZECC/ARMCANZ 2000).

### **Lead**

Lead is usually present in very low concentrations in natural waters; in freshwater the main species is PbCaCO<sub>3</sub> and lead-organic complexes. Lead is absorbed by suspended clay, humic substances and other suspended material. Lead is a non-essential metal with no nutritional role, however, its detoxification and storage have been documented. Lead will bind to metallothionein and probably has a higher affinity for other metabolic ligands, as it is often associated with deposited inorganic granules with high concentrations of calcium (McGeer *et al.* 2004).

Lead can bioaccumulate in aquatic organisms if present at sufficient concentrations. Lead toxicity is hardness dependant; lead uptake and toxicity in freshwater organisms decreases with increasing water hardness and alkalinity. At hardness of 50mg/l (as CaCO<sub>3</sub>) the acute toxicity of 10 freshwater species ranged from 143 µg/l to 236 µg/l (ANZECC/ARMCANZ 2000). In fish, lead does not exert acute toxicity but it accumulates in internal organs and leads to the disruption of the central nervous system, with the most likely primary effect to be a reduced reproductive success (ANZECC/ARMCANZ 2000).

### **Nickel**

Nickel can originate from weathering of mineral and rocks and from anthropogenic sources. It is an essential element for aquatic organisms at low concentration, but can be toxic at higher concentrations. Nickel toxicity decreases with increased hardness and with decrease in pH in water. Nickel acute toxicity to freshwater organisms ranges from 510 µg/l for cladoceran to 43000 µg/l for fish.

### **Mercury**

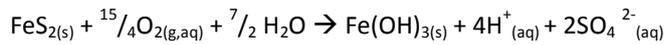
In aquatic systems mercury exists mainly as complexes of mercury (II) and as organomercurial. Mercury toxicity is hardness dependent; organisms' uptake rate increases with decreasing water hardness and pH. Inorganic forms of mercury may be converted by bacteria into organomercury complexes that are more toxic and tend to bioaccumulate. Bioconcentration factors for fish for mercury are very high. Acute toxicity of mercury (II) to freshwater organisms ranged from 2.2 µg/l for *Daphnia magna*, to 2000 µg/l for a mayfly (ANZECC/ARMCANZ 2000).

### 4.3 Acid sulfate soils

#### Soil acidity

Acid sulfate soils are saline soils or sediments containing pyrite  $\text{FeS}_2$ , which once drained (as part of land management or development measures), become acidic and release large amounts of acidity and other contaminants to the environment.

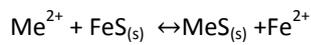
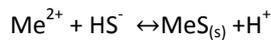
The production of sulfuric acid from pyrite brings the soil pH below 4, sometimes even below 3. Each mole of pyrite yields 4 moles of acidity and the overall equation for the oxidation of pyrite is:



The acidity in soil and sediment can then cause:

- Minerals in soils to dissolve and liberate soluble and colloidal aluminium and iron, which may leak into drains, streams and floodwaters killing vegetation and aquatic life (acidic waters rich in metals, deoxygenated waters, fish kills, fish disease and localised losses of fish and aquaculture production)
- Acidification
- Arsenic contamination of groundwater aquifers
- Damage to infrastructure such as roads, concrete and steel pipes, buildings, housing estates, tourism assets, bridges and culverts constructed on acid sulfate soils.

In anaerobic sediments with high acid generating potential, Acid Volatile Sulfide (AVS) is the main binding phase for metals (Me). AVS is composed by solid phase iron-monosulfides (FeS) (Casas and Crecelius 1994):



### Submerged / exposed soil conceptual model

Typically, submerged sediments in inland aquatic ecosystems have very little oxygen below the first few millimetres and can therefore be sites of sulfate reduction (Figure 4). Under wet conditions, sulfur can be integrated in the organic matter through the  $\text{SO}_4^{2-}$  reduction, occurring when algae and bacteria use sulfate to produce sulfur-protein within their cells. Volatile Organic Sulfur gases (VOSC) will be released by wetland water and sediments when wet.

When wetlands dry, they will gradually oxidise the sulphides stored in the sediments, producing both sulfuric acid ( $\text{H}_2\text{S}$ ) and sulfur dioxide ( $\text{SO}_2$ ). The acidity produced during the oxidation of acid sulfate soils can cause metals and metalloids to be dissolved and released (Environment Protection and Heritage Council and the Natural Resource Management Ministerial Council 2011; Hicks and Lamontagne 2006). These processes are illustrated by the conceptual models (Figure 4) proposed by Hicks and Lamontagne (2006).

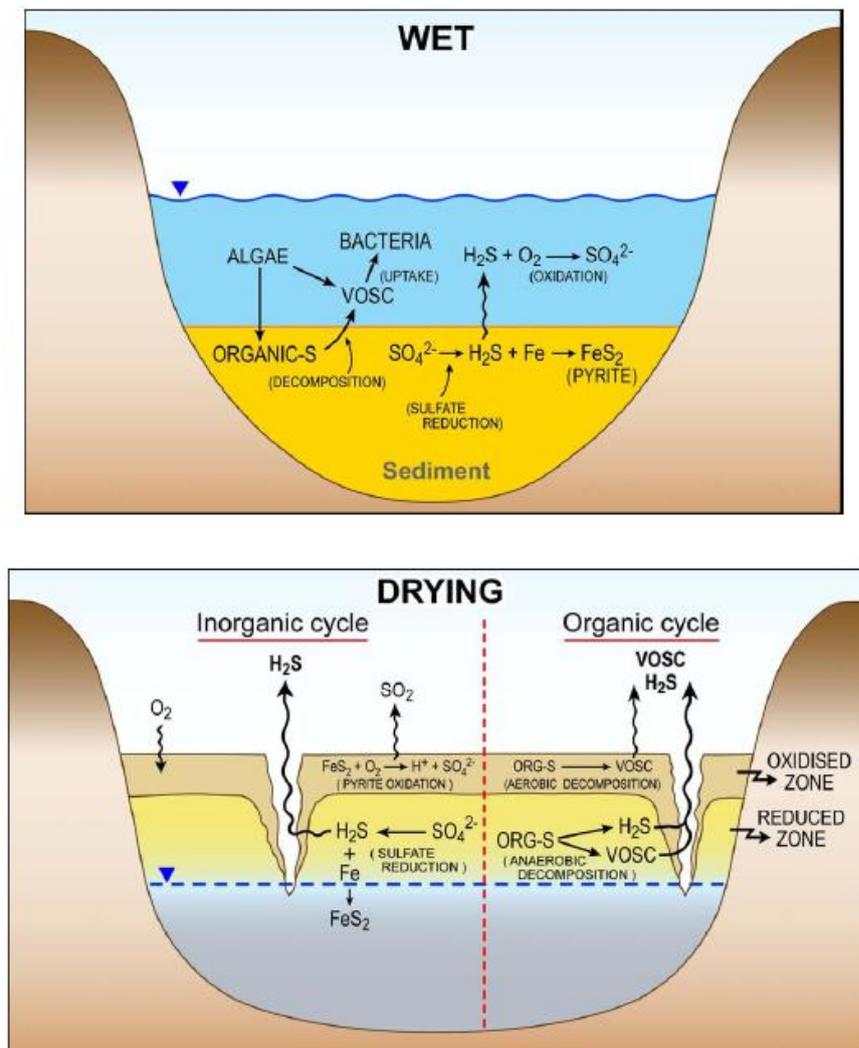


Figure 4. Conceptual model for acid sulfate soils in Australia (Hicks and Lamontagne 2006)

### Mobilising contaminants

In the case of drying-rewetting of acid sulfate soils, heavy metals may be mobilised. The conceptual model for mobilisation assumes that sediments re-wetted by water will result in the release of substances to the associated waters. However, there is some uncertainty as to how much metal will be released as the production of iron hydroxide immobilises metals. The mixing of the acidic metal-rich waters is likely to result in some removal of the trace metals through co-precipitation and adsorption to freshly forming Al, Fe and Mn oxyhydroxide phases (Simpson and Batley 2003). However the flocculation of aluminium may cause toxicity to

a range of aquatic fish, and the trace metal-enriched precipitates may be toxic to a variety of filter feeding or benthic organisms.

Given the presence of the contaminants in the sediment, receptors which may be impacted by contaminant release include fish, aquatic invertebrates and aquatic plants. Wildlife exposure may also eventuate as direct effects due to ingestion of water and food. Lower trophic levels will be affected by the contamination directly, while the upper trophic levels may be affected indirectly, e.g. reduction in food (Suter et al. 2000).

### **Acid sulfate soils in Reedy Lake**

Investigation into the Corangamite region's soil (Cox *et al.* 2005; Fitzpatrick *et al.* 2007) has revealed the presence of acid sulfate soil in Reedy Lake. Acid sulfate soils pose a number of significant risks such as (Environment Protection and Heritage Council and the Natural Resource Management Ministerial Council 2011):

- *Acidification*: When exposed to oxygen, acid sulfate soils undergo a complex series of oxidation reactions that ultimately produce acid. If the amount of acidity produced by this oxidation process is greater than the system's ability to absorb that acidity (the acid neutralising capacity) the pH of the system falls.
- *Deoxygenation*: the oxidation process caused by exposure to oxygen, consumes oxygen in the sediment itself, and in extreme cases can remove all of the oxygen from the water column, resulting in the death of aquatic organisms. This is most likely to occur when highly reactive forms of sulfide, such as those found in monosulfidic black oozes, are physically disturbed and distributed throughout a water column.
- *Release of metals and metalloids*: Oxidation of sulfidic materials may also lead to heavy metals present in Reedy Lake (such as chromium and lead) and metalloids (such as arsenic) becoming more available in the environment. Many heavy metals and metalloids form sulfidic minerals. If those metal sulfides are oxidised, the heavy metals or metalloids are released into the pore water or into the overlying water column, where they may be incorporated into animal or plant tissue and potentially into the food chain.

Effects of acid sulfate soils on Australian ecosystems have been documented and include fish kills and the outbreak of red spot disease, while laboratory toxicity testing has demonstrated a delay in the hatching of fertilised Australian bass eggs and a decrease in the development of oyster embryos (Hyne and Wilson 1997; Sammut *et al.* 1996; Wilson and Hyne 1997). A fish kill event reportedly occurred in Reedy Lake in 2009/10, attributed to a 'blackwater' event (Ian McLachlan, Field and Game Association, pers. comm., March 2013). In the absence of monitoring data it is not possible to determine if this event was linked to previous exposure of acid sulphate soils, however it is certainly a possibility.

In addition to direct effects on the environment, acid sulfate soils may lead to other social and economic problems for the Reedy Lake area. Offensive odour problems have been encountered in areas rich in sulfidic materials. They occur when sediments extremely enriched in sulfides are exposed to the atmosphere. Hydrogen sulfide production (H<sub>2</sub>S – the rotten egg smell) from drying sulfidic materials is thought to be a significant cause of offensive odour. Drying sulfidic materials also produces sulfur dioxide (SO<sub>2</sub>). H<sub>2</sub>S and SO<sub>2</sub> are also of concern for human health at high concentrations e.g. in confined spaces such as excavations. With increasing residential development in the vicinity of Reedy Lake, the release of offensive odour from the lake bed sediments would detract from the amenity and social values of the region, impacting on adjacent development and recreational use of the lake.

## 5 Risk characterisation

### 5.1 Acid sulfate soils

The risk of acid sulfate soil being present in Reedy Lake has previously been ranked as moderate (Fitzpatrick *et al.* 2007) with a subsequent management recommendation of 'do not disturb'. There is currently no data available on water quality, in particular pH values, which would provide more confidence on the degree of soil acidity and likelihood and extent of acid sulfate soils in Reedy Lake.

However, based on geologic understanding of the region (Holocene sediments) and past recommendations (Fitzpatrick *et al.* 2007), there is a reasonable level of confidence that acid sulfate soils are present and that disturbance and/or exposure could pose a significant risk to the Reedy Lake ecosystem (as discussed in Section 4.3). The nature of this risk will vary in association with the watering regime, concentrations of other contaminants, and other management actions including those associated with *Phragmites australis* control (summarised in Section 6).

### 5.2 Metal concentrations

#### Metals in sediments

As part of an investigation into anthropogenic influences on the Barwon estuary system, RMIT and Ballarat University undertook a pilot study on heavy metal concentrations in the lake bed sediments. One sediment core was taken from the lake bed in 2011 (from the 'big hole' area), and samples from the core profile were taken every 5 cm (5 – 50 cm) (Dr Suzanne Reichman, pers. comm. March 2013, Grundell *et al.* 2012).

Chromium levels in all but the lowermost samples were above the ANZECC guidelines (a likely by-product from local tannery), and arsenic levels were also above ANZECC guideline values from 5 – 35 cm (associated with gold rush runoff and wool scour), reaching above ecological health values at 18 – 24 cm. The chromium and arsenic values are also quite high when compared to similar environments (Stauber *et al.* 2008).

Lead was found to be above ANZECC guideline values from 25 – 18 cm (related to lead-shot accumulation), and nickel was also found in significant quantities (likely associated with local volcanic rocks). Concentrations of all other metals were within ANZECC guidelines.

The RMIT and Ballarat University investigation indicates that concentrations of chromium, arsenic and lead in the sediments are a significant concern for Reedy Lake. However, given the limited data, this does not discount concern over the concentrations of other metals, which may also be in high concentrations in areas that were not sampled.

Soil samples undertaken previously by Fitzpatrick *et al.* (2007) in Reedy Lake found similar levels of heavy metals to the RMIT and Ballarat University pilot study. Fitzpatrick *et al.* (2007) also identified high levels of arsenic in surrounding areas, indicating that background levels may be naturally high and associated with basalt geology (similar to the high levels of nickel). Fitzpatrick *et al.* (2007) concluded that elevated levels of heavy metals (Cr, Ni) and metalloids (As) in soils and sediments may in fact (or at least in part) be associated with natural background levels, however their ecotoxicity and mobility under acid conditions still needs to be established.

## Metals in water

No direct water quality samples are currently available for Reedy Lake. Therefore, to estimate the potential concentration of metals in the water of Reedy Lake ( $C_{\text{water}}$ ), the following formula was used (McCall *et al.* 1983):

$$\frac{C_{\text{sed}}}{K_d} = C_{\text{water}}$$

Where  $K_d$  is the metal partition coefficient,  $C_{\text{sed}}$  is the sorbed metal concentration (metal bound to solid), and  $C_{\text{water}}$  is the (in this case unknown) dissolved metal concentration.

$K_d$  (partitioning coefficient) values for arsenic (As), chromium (Cr(VI)), Lead (Pb), Nickel (Ni), and mercury (Hg) were sourced from relevant literature (Caruso 2004; USEPA 2004). The  $K_d$  of a particular metal depends on the nature of suspended solids or sediment, and key geochemical parameters of the water. Geochemical parameters that have the greatest influence on the magnitude of  $K_d$  include the pH of the system and the nature and concentration of sorbents associated with the soil or surface water. Given the many variables involved, maximum and minimum  $K_d$  values were identified from the range of values in the literature (Table 8).

**Table 8.  $K_d$  values for selected metals and metalloids (after Caruso 2004; USEPA 2004)**

	<b><math>K_{d\text{min}}</math> (ml/g) (low)</b>	<b><math>K_{d\text{max}}</math> (ml/g) (high)</b>
<b>Cr(VI)</b>	1	35
<b>Pb</b>	710	23270
<b>As</b>	0.12	6.46
<b>Hg</b>	631	3981071
<b>Ni</b>	10000	N/A

$C_{\text{sed}}$  (concentration in sediment) values were estimated from the RMIT and Ballarat University data (**Error! eference source not found.**), and best (lowest) and worst case (highest) sediment concentration values were also identified from the range of samples.

The potential concentration of metals in the water of Reedy Lake ( $C_{\text{water}}$ ) was estimated for the past water regime levels (Table 9) as well as for the levels proposed by Lloyd *et al.* (2012) (Table 10). The volume and area of the lake at the various AHD levels were used in the estimations (Ian McLachlan *pers. comm.*). Potential concentrations were estimated for scenarios of high and low  $K_d$  values, and further split by the best and worst case scenarios of sediment concentrations. The resulting range of potential concentrations was compared to the ANZECC guideline values (Table 9, Table 10).

**Table 9. Estimated potential levels of metals in the water of Reedy Lake (past water regime levels) – red (very high) indicates estimated concentrations exceed ANZECC guideline values**

	C <sub>water</sub>				ANZECC guidelines (µg/l) (95% species protection)
	K <sub>d</sub> <sub>min</sub>		K <sub>d</sub> <sub>max</sub>		
	Best case scenario (C <sub>sed</sub> min) (µg/l)	Worst case scenario (C <sub>sed</sub> max) (µg/l)	Best case scenario (C <sub>sed</sub> min) (µg/l)	Worst case scenario (C <sub>sed</sub> min) (µg/l)	
<b>0.4m AHD</b>					
Arsenic	>24	>>24	>24	>24	24
Chromium(VI)	>>1	>>1	>1	>>1	1c
Lead	<3.4	>3.4	<3.4	<3.4	3.4
Mercury	<<0.6	<0.6	<<0.6	<<0.6	0.6
Nickel	<<11	<11	N/A	N/A	11
<b>0.7m AHD</b>					
Arsenic	>24	>>24	<24	>24	24
Chromium(VI)	>1	>>1	>1	>1	1c
Lead	<3.4	<3.4	<3.4	<3.4	3.4
Mercury	<0.6	<0.6	<0.6	<0.6	0.6
Nickel	<11	<11	N/A	N/A	11

**Table 10. Estimated potential levels of metals in the water of Reedy Lake (proposed water regime levels) – red (very high) indicates estimated concentrations exceed ANZECC guideline values**

	C <sub>water</sub>				ANZECC guidelines (µg/l) (95% species protection)
	K <sub>d</sub> <sub>min</sub>		K <sub>d</sub> <sub>max</sub>		
	Best case scenario (C <sub>sed</sub> min) (µg/l)	Worst case scenario (C <sub>sed</sub> max) (µg/l)	Best case scenario (C <sub>sed</sub> min) (µg/l)	Worst case scenario (C <sub>sed</sub> max) (µg/l)	
<b>0 m AHD</b>					
Arsenic	>>24	>>24	>24	>>24	24
Chromium(VI)	>>1	>>1	>1	>1	1c
Lead	<3.4	>3.4	<<3.4	<3.4	3.4
Mercury	<0.6	<0.6	<<0.6	<<0.6	0.6
Nickel	<11	>11	N/A	N/A	11
<b>0.3 m AHD</b>					
Arsenic	>>24	>>24	>24	>>24	24
Chromium(VI)	>>1	>>1	>1	>1	1c
Lead	<3.4	>3.4	<<3.4	<3.4	3.4
Mercury	<<0.6	<0.6	<<0.6	<<0.6	0.6
Nickel	<11	<11	N/A	N/A	11
<b>0.8m AHD</b>					
Arsenic	>24	>>24	<24	>24	24
Chromium(VI)	>1	>>1	>1	>1	1c
Lead	<3.4	<3.4	<<3.4	<<3.4	3.4
Mercury	<<0.6	<0.6	<<0.6	<<0.6	0.6
Nickel	<11	<11	N/A	N/A	11
<b>0.9m AHD</b>					
Arsenic	>24	>>24	<24	>24	24
Chromium(VI)	>1	>>1	>1	>1	1c
Lead	<3.4	<3.4	<<3.4	<3.4	3.4
Mercury	<<0.6	<0.6	<<0.6	<<0.6	0.6
Nickel	<11	<11	N/A	N/A	11

For the past water regime levels (Table 9), when lake levels are low (0.4 m AHD), estimated arsenic and chromium concentrations in the water significantly exceed ANZECC guidelines for all scenarios (min, max K<sub>d</sub>, best and worst case C<sub>sed</sub>). Estimated lead concentrations also exceed guideline values where K<sub>d</sub> is a minimum and C<sub>sed</sub> is high (worst case). At the higher lake level (0.7 m), estimated arsenic and chromium concentrations still exceed guideline values, with the exception of where K<sub>d</sub> is high and C<sub>sed</sub> is low (best case). Estimates of other metals are within guideline values.

For water levels associated with the proposed water regime (Table 10), estimated arsenic and chromium concentrations exceed ANZECC guideline values for all scenarios, with the exception of high K<sub>d</sub> and low C<sub>sed</sub> (best case) for the higher lake levels (0.8 AHD, 0.9 AHD). Estimated lead concentrations exceed guideline values for the dry/lower lake levels (0m and 0.3m AHD) under the scenario of low K<sub>d</sub> and high C<sub>sed</sub> (worst case). Estimated nickel concentrations exceed guidelines when the lake is dry (0m AHD) under the scenario of low K<sub>d</sub> and high C<sub>sed</sub> (worst case), however is likely associated with natural geology of the region so is not of particular concern. Estimates of other metals are within guideline values.

Overall, the screening level results indicate that arsenic and chromium concentrations in the water of Reedy Lake may be problematic across all water levels of the past and proposed water regimes. Concentrations are likely to be highest at lower lake levels. At lower and dry lake levels (below 0.7m AHD) concentrations of lead may also become problematic.

**Risk associated with mobilising contaminants**

To assist with determining the level of risk associated with mobilising contaminants, the hazard quotient (as shown below and detailed previously in Section 1) was estimated for the range of flow levels associated with the historical and proposed flow regimes.

$$HQ = \frac{PEC}{PNEC}$$

Where:

HQ = Hazard quotient

PEC = Predicted environmental concentration at the site, i.e. how much contaminant is in the water.

PNEC = Predicted no-effect concentration at the site, i.e. generally a no-adverse effects level concentration.

The C<sub>water</sub> (concentration of metal in the water) values previously estimated were used for the PEC values in the hazard quotient formula. The C<sub>water</sub> values for the range of scenarios - K<sub>d</sub> max and min, C<sub>sed</sub> low (worst case) and high (best case), were used to calculate a corresponding range of hazard quotients.

The PNEC values were drawn from the ANZECC/ARMCANZ (2000) freshwater trigger values for chronic effects, converted to acute guideline by a reverse application of an acute/chronic ratio of 10, to comply with a water hardness higher than 30 mg CaCO<sub>3</sub>/L and pH 6.5-9 (Stauber *et al.* 2008) (Table 11).

**Table 11. PNEC (predicted no-effect concentration) values used in the Risk Analysis**

	Acute Trigger Value (µg/)
<b>Arsenic (V)</b>	130
<b>Chromium(VI)</b>	10
<b>Lead</b>	34
<b>Mercury</b>	6
<b>Nickel</b>	110

The hazard quotient was estimated for lake levels associated with the past (Table 12) and proposed (Table 13) water regimes for Reedy Lake. Hazard quotients were estimated for scenarios of high and low K<sub>d</sub> values, and further split by the best and worst case scenarios of sediment concentrations. Where hazard quotients are greater than one, there is likely to be significant concern for adverse environmental impacts associated with mobilising the contaminant.

**Table 12. Estimated hazard quotients for selected metals and metalloids in the water of Reedy Lake (past water regime) – red indicates estimated hazard quotient > 1**

	0.4 AHD		0.7 AHD	
	HQ Csed max	HQ Csed min	HQ Csed max	HQ Csed min
	(worst case)	(best case)	(worst case)	(best case)
<b>Kd<sub>min</sub></b>				
As	>>1	>1	>>1	>1
Cr	>>>1	>1	>>1	>1
Pb	<1	<<1	<1	<<1
Ni	<1	<<1	<1	<<1
Hg	<1	<<1	<1	<<1
<b>Kd<sub>max</sub></b>				
As	>1	<1	>1	<1
Cr	>1	>1	>1	<1
Pb	<<1	<<1	<<1	<<1
Ni	<<1	<<1	<<1	<<1
Hg	<<1	<<1	<<1	<<1

**Table 13. Estimated hazard quotients for selected metals and metalloids in the water of Reedy Lake (proposed water regime) – red indicates estimated hazard quotient > 1**

	0 AHD		0.3 AHD		0.8 AHD		0.9 AHD	
	HQ Csed max	HQ Csed min						
	(worst case)	(best case)						
<b>Kd min</b>								
As	>>1	>1	>>1	>1	>>1	>1	>>1	>1
Cr	>>1	>>1	>>1	>1	>>1	>1	>>1	>1
Pb	<1	<<1	<1	<<1	<1	<<1	<1	<<1
Ni	<1	<1	<1	<<1	<1	<<1	<1	<<1
Hg	<<1	<<1	<<1	<<1	<<1	<<1	<1	<<1
<b>Kd max</b>								
As	>1	<1	>1	<1	>1	<1	>1	<1
Cr	>1	>1	>1	>1	>1	<1	>1	<1
Pb	<1	<<1	<1	<<1	<<1	<<1	<<1	<<1
Ni	<<1	<<1	<<1	<<1	<<1	<<1	<<1	<<1
Hg	<<1	<<1	<<1	<<1	<<1	<<1	<<1	<<1

The hazard quotient analysis indicates concern for the impact of chromium and arsenic in the water of Reedy Lake. Estimated values exceed the acceptable ratio value (of 1) by several orders of magnitude for the dry and lower lake values. At higher lake levels, under the scenario where  $K_d$  is high and  $C_{sed}$  is low (best case), the estimated hazard quotient for arsenic and chromium is below one (acceptable), however for the comparable worst case scenario the hazard quotients are significantly higher than one and therefore these potential concentrations are still a concern.

Estimated hazard quotients for other contaminants are below one (acceptable). However, caution should be taken in interpreting these results as many assumptions have been made (e.g. water pH) due to limited data. Other contaminants may also prove to be of concern with improved data on sediment concentrations and water pH.

Although problematic at all lake levels, estimated hazard quotients for arsenic and chromium are orders of magnitude higher (by an approximate factor of 10) for low/dry lake levels, raising significant concerns for ecosystem health associated with periodic drying of the lake.

It is important to emphasize that this hazard quotient analysis is based on only on theoretical estimations of potential contaminant concentrations in the water column. No adverse effects have been reported to date on flora and fauna in Reedy Lake. This may be associated with a lack of data / investigations into this issue, or it may indicate limited bioavailability of the metals, or that metals are not actually in the water column as predicted. Detailed toxicity or bioaccumulation tests are required to confirm actual contaminant concentrations in the water column, the degree of bioavailability of metals in Reedy Lake, and the subsequent level of risk to species. Some consideration of factors affecting bioavailability of metals is discussed below.

#### **Consideration of bioavailability**

As detailed previously (in Section 4), when metals are in anoxic sediment they can form metal sulfides that are relatively stable. When exposed to the air (e.g. drying of the lake bed) oxygenation of the sediment occurs, which may lead to the metals in the first layers of sediment becoming more mobile and thus bioavailable.

Two factors that may reduce the bioavailability of metals are the presence of iron and sulfate in the sediment. Under anoxic conditions, the presence of iron ( $Fe^{2+}$ ) can reduce the bioavailability of metals. Since FeS (iron sulphide), FeS<sub>2</sub> (iron disulfide, or iron pyrite) and MnS have higher solubility products than all the metal-sulfides, metals will displace Fe and Mn to form more stable metal-sulfides. The oxidation of sulphide can result in the release of the associated metals; however, only a small portion is released (Chapman et al. 1998). Solubility values for a variety of metal sulfides indicate that, many metal or metalloid ions, including Hg and As, can form metal or metalloid sulfides less soluble than iron and manganese monosulfides. The toxicity of these metals in anoxic sulfidic sediments is also likely to be controlled by sulfide (Wang and Chapman 1999).

In addition, all sediments where the overlying water is oxic will have an oxidised layer containing iron hydroxides. These Fe-hydroxides sequester metals, and in this case drying would promote formation of oxides and prevent release of metals (Simpson and Batley 2003).

In the case of Reedy Lake, the concentrations of  $Fe^{2+}$  and sulfate in the sediments of concern are unknown, and so it is uncertain how much of a mitigating effect there currently is on metal bioavailability. In addition there is currently no data available to directly determine bioavailability. Reedy Lake was originally an estuarine wetland, and so it is possible that the concentration of sulfate is high. Hydraulic interventions have altered the salinity of the lake (Lloyd *et al.* 2012), to become a freshwater body, and so ferrous oxides in the sediments may have also increased. It is also important to note that regardless of the ability of  $Fe^{2+}$  to reduce bioavailability, any metals in the soils are still toxic.

Another factor in the bioavailability of metals in Reedy Lake is the presence of *Phragmites* rhizomes (horizontal root network). In transporting oxygen to and through the root structure, *Phragmites* plants can generate an oxygenated zone around the rhizome network, which may have some buffering (mitigating) effect on metal bioavailability. However, conversely, the presence of Carp disturbing sediments and *Phragmites* root network may be acting to increase the bioavailability of metals.

### 5.3 Uncertainties

The analysis undertaken in this investigation is at the screening level of assessment only, and based on limited data on the soil and water properties of Reedy Lake. A range of assumptions have been made for the calculations, and predicted environmental concentrations for the range of release/dilution scenarios may over or underestimate the real concentrations present.

A high level of confidence can be had in the identification of major stressors (chromium, arsenic) and the variations with water levels that are of an order of magnitude (or several). However, only low levels of confidence can be had in the actual values calculated during the assessment, due the range of other uncertainties that could not be accounted for (data not available). These uncertainties include:

- Limited sediment data. Only one recent sediment core sample was taken from the lake bed in 2011 for the RMIT and Ballarat University pilot study, and no replicates. The lack of spatial distribution of sediment samples, and lack of replicates, may mean data is over-representing or under-representing contaminant concentrations.
- Dissolved organic matter concentrations that may influence the distribution of the contaminants between the dissolved and particulate phase.
- Partitioning coefficient values ( $K_d$ ). Although data was collected from the literature, more accurate estimation is necessary as rates can vary widely with water chemistry and sediment characteristics (Gandhi *et al.* 2011; Hassan *et al.* 1996).
- Background contaminant concentrations in the water and sediment of Reedy Lake. No data was available for use in calculations / comparisons with current values (relied solely on ANZECC guidelines).
- pH and hardness of the water, a significant factor modifying solubility of metals during rewetting. No data was available for use in calculations and so assumptions were made based on the literature.

To reduce the uncertainties *ad hoc* experiments could be conducted to provide information on the metal release from the sediments considering the soil pH and different soil types during wetting and drying regimes.

It is important to note that in some ways the screening level risk characterisation undertaken may be overly conservative by not considering:

- The buffering capacity of the water
- Removal processes such as neutralisation or precipitation.

Conversely, the screening level risk characterisation may also be under-conservative as it does not take into account:

- Interactions of metals
- Interactions of metals and pH
- Effects of precipitates on benthic biota (by using only dissolved metals in guideline comparisons).

More detailed investigations that take into account uncertainties associated with estimating sediment concentrations in Reedy Lake (sediment and water) are recommended once data availability increases, presumably in partnership with the establishment of a monitoring and evaluation program.

## 6 Management implications

### 6.1 Concerns and mitigation

#### Past water regime

The past water regime (of the recent past) has typically involved keeping the lake relatively full over winter/spring (0.7 m AHD), allowing to reduce over summer (0.4 m AHD), and periodic drying for carp control (every 6 – 7 years).

The screening level risk characterisation process undertaken for this investigation has indicated that arsenic and chromium levels may be high for all lake levels, beyond the ANZECC guideline values for both the sediment and water, and may pose a significant risk to ecosystem health. However, keeping the lake levels high may avoid the very high contaminant concentrations predicted for lower lake levels. In this way, maintaining the past water regime may have helped to limit the mobilisation of contaminants to some extent.

Exposure of the lake bed sediment has still occurred periodically for carp control and maintenance, and the risk of exposing acid sulfate soils is high when this occurs. However, this occasional drying is likely to carry a lower risk of long term ecosystem effects compared to annual or more regular drying of the lake (due to more limited occurrence soil exposure and low lake levels).

While keeping lake levels high may assist with reducing the risk of acid sulfate soil exposure and contaminant mobilisation, the high lake level has other negative implications for the suite of other environmental objectives for the lake and the region (identified by Lloyd *et al.* 2012). In addition, continuing the past flow regime will not promote reduction in *Phragmites* extent, and mechanical intervention would be required to remove *Phragmites*. Such mechanical removal may increase the risk of disturbing the soil and triggering contaminant release. Minimising the disturbance to the soil associated with *Phragmites* control and other management interventions in the lake (e.g. inlet / outlet maintenance) will help reduce the risk of increasing contaminant concentrations in the water and long term ecosystem impacts.

#### Annual drying

Annual drying of the lake has been proposed as an option for an initial 10 year period, primarily for *Phragmites* control. This will subject the lake to frequent exposure of the sediments over this time, and significant periods of time at low lake levels. Simply due to the frequent nature of the drying, this regime poses a relatively high risk for the exposure and effects of acid sulfate soils. There is also a risk associated with potentially high contaminant concentrations estimated for the water at low lake levels (predominantly arsenic, chromium and lead), and the associated significant risk of ecosystem impacts. These risks are significant, and may outweigh the ecosystem benefits that the drying regime is intended to achieve (including *Phragmites* control).

However, there may be factors that mitigate the bioavailability of the high levels of contaminants, and associated ecosystem impacts. Drying of the lake is expected to promote the intrusion of saline groundwater, which would assist with *Phragmites* control, and would help naturally mitigate the bioavailability of metals. The presence of iron and sulfate in the soils may also limit bioavailability. However with only limited data available for this investigation, it is uncertain how significant the impact on bioavailability would be, and therefore it is still assumed that annual drying poses a high risk of increased contaminant mobilisation.

If successful as a *Phragmites* control technique, annual drying would reduce the need for any additional physical intervention techniques, and therefore minimising the risk associated with disturbing the soil.

There are also management options that may assist with mitigating the effects of acid sulfate soils and the release of metal contaminants, including:

- Liming or other localised treatments for acid sulfate soils
- Periodic flooding of the lake with seawater (tidal flush). Flooding with seawater theoretically has benefits of neutralising acidity, maintaining sufficient water levels to inhibit oxidation and adding sulfate which locks up iron.

Detailed monitoring of the first few drying events would assist with determining risk to ecosystem health, however based on this initial screening level assessment, frequent drying is likely to have a greater risk than less regular or no drying.

### Less regular drying

Following an initial period of annual drying (10 years) for *Phragmites* control, Lloyd (2012) proposed a less regular drying regime as a component of the longer term management of Reedy Lake, aimed at achieving a suite of environmental objectives (Lloyd *et al.* 2012).

The concerns associated with annual drying listed above (acid sulfate soil exposure and contaminant mobilisation) also hold for less regular drying, although being less regular (more time at higher lake levels) the longer term ecosystem impacts may be more moderate. Even if this is not the case, less regular drying also provides opportunity, with appropriate monitoring in place, to detect and mitigate any contaminant issues before they become a significant concern.

Potential high concentrations of contaminants predicted in the water at low lake levels are still a concern, however as mentioned previously, bioavailability of metals may be mitigated by the presence of iron, sulfate, and the intrusion of saline groundwater.

As noted previously, minimising disturbance to the soil during periodic mechanical *Phragmites* control (if undertaken) will be important to limit contaminant mobilization, particularly avoiding any major disturbances that expose deeper sediment to the air.

An appropriate monitoring and evaluation plan will be an essential component of any proposed drying regime for the lake, and should include testing water quality, metal concentrations (sediment and water), and bioavailability, outlined in Section 6.3.

## 6.2 Summary

A summary of the key points discussed above for each water regime is provided in Table 14.

**Table 14. Summary table of likely concerns for various water regimes**

Concerns	Past water regime	Annual summer drying (10 years)	Less regular drying
<b>Potential to expose acid sulfate soils</b>	Low : only during rare drying events for carp control	High: exposure with each annual drying event	Moderate – high: less regular exposure
<b>Potential to mobilise heavy metals</b>	Low: only during rare drying events for carp control	High: annual exposure of soils and re-wetting is high risk for mobilising contaminants into water column	Moderate – high: less regular exposure and re-wetting creating potential for mobilising contaminants – opportunity to mitigate
<b>Physical <i>Phragmites</i> control considerations</b>	To control reeds, mechanical intervention will be required (cutting) that may cause some disturbance to soil.	May require less physical intervention (less chance of soil disturbance) if drying is effective in reed reduction.	Periodic mechanical intervention (cutting) likely required to control reeds.
<b>Mitigation options</b>	Boat based cutting mainly in deep water areas to minimise bed disturbance.	Potential for using a tidal flush to mitigate risk of contaminant mobilisation from drying events. Monitoring of initial drying events to better assess contaminant risks.	Potential for using a tidal flush to mitigate risk of contaminant mobilisation from drying events. Monitoring of initial drying events to better assess contaminant risks.
<b>Potential problems of arsenic and chromium in the water column</b>	High: current levels may already be above guideline values	High: due to frequent exposure and potential mobilisation, and more time spent at lower lake levels in first 10 years.	High: some increase compared to current levels expected. Will depend on any mitigation.

## 7 Further investigations, monitoring and evaluation

### Monitoring and evaluation

It is recommended that a detailed monitoring and evaluation plan for contaminants be developed and implemented prior to the commencement of a drying regime for the lake. This includes any future drying for Carp control. As such, it is a priority that the monitoring plan be design and implemented as soon as possible within the next year.

The monitoring plan should include further investigations to fill background data gaps, and then ongoing monitoring and evaluation based on the background data. Recommended inclusions for the monitoring plan are provided in Table 15, however each area will need to be developed in more detail by the relevant specialists in each field as part of a further investigation.

The monitoring program should include:

- **Background (current) water quality (including pH), sediment and water contaminant concentrations** for all metals. It is important that this data be collected prior to further intervention in the water regime, to enable a comparison between present and future contaminant concentrations in the lake. The spatial and temporal distribution of sampling should be appropriate for obtaining a statistically significant data sample.
- **Background ecosystem data.** Where possible, data should be obtained on current ecosystem impacts, including contaminant levels in fish and other biota. This will assist in determining the long term ecosystem risk of current contaminant levels, and potential impact of future changes.
- **Background contaminant bioavailability data.** There are many factors that may be effectively limiting the bioavailability of metals, including the presence of iron, sulfate and saline groundwater, however no data available to test these hypotheses. Further testing would assist in determining the current bioavailability of the metals for uptake in the ecosystem.
- **Periodic water quality (including pH), sediment and water contaminant concentrations** (preferably all metals, but particularly hexavalent chromium, arsenic and lead). Samples should be collected at the range of lake levels included in the new water regime for the lake. This will be particularly important during trial periods of a new watering regime.
- **Species specific monitoring.** Monitoring programs for key species – fish, macroinvertebrates, birds and vegetation, would assist with more accurate determination of long term impacts associated with water regime changes and potential contaminant release. Rather than having to infer likely ecosystem impacts from other information, clear documentation of species changes over time will facilitate improved and targeted management of the lake, and greater confidence in decision making.

In the absence of any background data, any measurements that can be taken initially would be of benefit for improving confidence in the risk assessment (water quality, ecosystem, bioavailability). A summary of recommended components for inclusion in the monitoring program is provided in Table 15, along with an indication of the highest priority components.

**Table 15. Recommended inclusions for contaminants monitoring program as background and ongoing monitoring data – red indicates highest priority**

Types of data	Test for	Spatial locations	Sampling frequency
<b>Sediment samples</b>	<ul style="list-style-type: none"> <li>- <b>Metal concentrations (chromium, arsenic and lead as a priority)</b></li> <li>- Soil acidity</li> <li>- Iron</li> <li>- Sulfate</li> </ul>	<ul style="list-style-type: none"> <li>- <b>Lake margin (particularly in areas where reed control may be undertaken)</b></li> <li>- Mid-depth regions of lake</li> <li>- Deep areas (the 'big hole' where previous sediment core was taken in 2011)</li> </ul>	<p>As a minimum, samples should be taken:</p> <ul style="list-style-type: none"> <li>- <b>As soon as possible to determine current conditions</b></li> <li>- After each drying event, or other management interventions (tidal flush, reed control) into the future.</li> </ul>
<b>Water quality samples</b>	<ul style="list-style-type: none"> <li>- <b>pH</b></li> <li>- Temperature</li> <li>- Salinity</li> <li>- Iron</li> <li>- Sulfate</li> <li>- <b>Metal concentrations (chromium, arsenic and lead as a priority)</b></li> </ul>	<ul style="list-style-type: none"> <li>- <b>Lake margin</b></li> <li>- <b>Mid-depth regions of lake</b></li> <li>- <b>Deep areas of lake</b></li> <li>- <b>Different levels of the water column (surface, mid and near bed)</b></li> </ul>	<p>As above, plus:</p> <ul style="list-style-type: none"> <li>- At a range of different lake levels</li> <li>- After floods</li> <li>- <b>At low lake levels (where screening assessment indicated high levels of contaminants may be present).</b></li> </ul>
<b>Groundwater samples</b>	<ul style="list-style-type: none"> <li>- pH</li> <li>- Temperature</li> <li>- Salinity</li> <li>- Iron</li> <li>- Sulfate</li> <li>- Metal concentrations (chromium, arsenic and lead as a priority)</li> </ul>	<ul style="list-style-type: none"> <li>- Lake margin</li> <li>- Bore holes</li> </ul>	<p>As a minimum, samples should be taken:</p> <ul style="list-style-type: none"> <li>- As soon as possible to determine current conditions</li> <li>- Annually to monitor any changes in groundwater conditions, in particular salinity and metal concentrations.</li> </ul>
<b>Flora and fauna testing</b>	<ul style="list-style-type: none"> <li>- <b>Metal concentrations (chromium, arsenic and lead as a priority).</b></li> </ul> <ul style="list-style-type: none"> <li>- fish</li> <li>- birds</li> <li>- reeds/ rhizomes</li> </ul>	<ul style="list-style-type: none"> <li>- Various parts of the lake / where feasible</li> </ul>	<p>As a minimum, samples should be taken:</p> <ul style="list-style-type: none"> <li>- As soon as possible to determine current conditions</li> <li>- Annually to monitor any changes in metal concentrations in flora and fauna, particularly in the years following drying events.</li> </ul>

### Long term strategy

In the longer term, there will also need to be some consideration of the implication of reduced *Phragmites* (and potentially reduced *Typha*) extent. Current *Phragmites* stands are likely have a significant role in the stabilisation of soils, contaminants, and uptake of trace metals from the water column in Reedy Lake. It is uncertain from this screening level assessment how contaminant concentrations will respond to varied water regimes combined with *Phragmites* removal / reduction. However a reduction in *Phragmites* extent may have implications for increased contaminant release in the longer term. Re-planting strategies with appropriate vegetation may assist with mitigating this problem, as well as achieving the desired vegetation mosaic for the wetland.

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