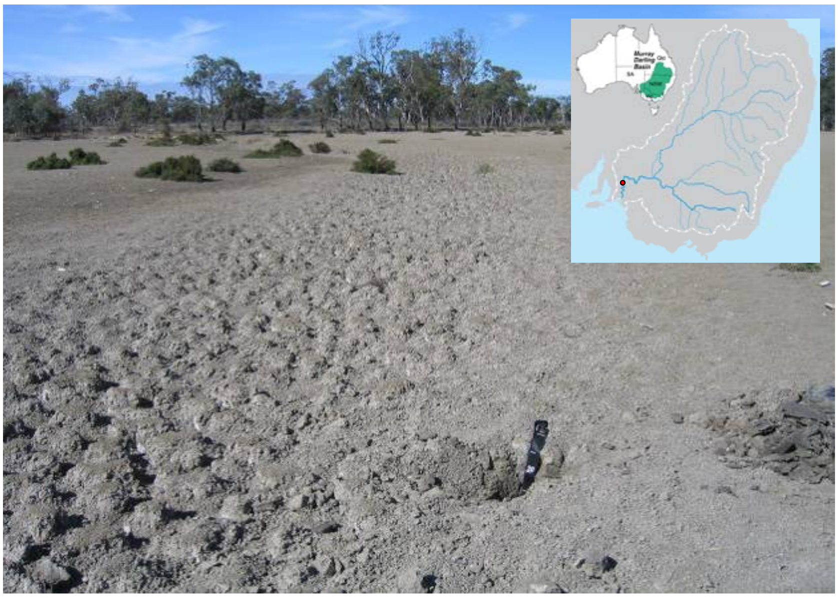

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Assessment of Acid Sulfate Soil Materials (Phase 2)
Swan Reach Complex wetland, South Australia

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Report to the Murray-Darling Basin Authority

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Cover Photograph:

Swan Reach Complex wetland
Photographer: Gerard Grealish
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EXECUTIVE SUMMARY

An initial Phase 1 acid sulfate soil investigation of the Swan Reach Complex wetland during August 2008 showed acid sulfate soils to be a priority concern within this wetland complex. Based on Phase 1 recommendations, a Phase 2 investigation was undertaken for the Swan Reach Complex wetland to determine the nature, severity and the specific risks associated with acid sulfate soil materials. The wetland was dry at the time of sampling.

The 24 hour **reactive metals** tests were undertaken to determine those metals and metalloids extractable with a moderately strong acid i.e. potentially available from binding sites on soil minerals such as iron (Fe), manganese (Mn) and aluminium (Al) oxides. Although comparisons can be made with soil and sediment quality guidelines, these are defined for total concentrations and not partial extractions. The results showed that concentrations were generally below the sediment quality guidelines (SQG) and soil ecological investigation levels (EIL) for those elements where guidelines are available. Although concentrations did not breach sediment quality guidelines and soil ecological investigation level trigger values, the concentrations of some elements were high enough that they may impact water quality if mobilised, particularly for aluminium (Al), manganese (Mn) and iron (Fe).

The **contaminant and metalloid dynamics** tests were undertaken to assess the release of metals during a water extraction, and to assess dynamics in response to saturation over time by incubating soil materials for periods of 1, 7, 14 and 35 days. The degree to which metal and metalloid concentrations exceed ANZECC/ARMCANZ environmental protection guideline values was used to characterise the degree of hazard. For Swan Reach Complex, aluminium (Al) and iron (Fe) were assigned a moderate hazard with concentrations exceeding ANZECC/ARMCANZ environmental protection guidelines by more than 10 times. A number of solutes also breached guideline values including arsenic (As), cobalt (Co), chromium (Cr), copper (Cu), manganese (Mn), nickel (Ni) and vanadium (V). However, many of these were only slightly above guideline values. Most increases were related to the reductive dissolution of iron (Fe) and/or manganese (Mn) oxides/oxyhydroxides in the soils of the wetland. This change, indicated by low Eh, was responsible for elevated concentrations of a number of metals and the metalloids arsenic (As) and vanadium (V).

The Swan Reach Complex wetland has been classified as medium conservation status by the SA Murray-Darling Basin Natural Resources Management Board (Miles *et al.* 2010). The main hazards considered in this study that may impact on wetland values are acidification, contaminant mobilisation and deoxygenation. The wetland has been allocated a **low** risk rating due to ***acidification*** and a **medium** ***contaminant*** risk rating for **soils**. For **surface waters**, the risk is largely dependent on surface and sub-surface hydrology and is thus scenario dependent. Taking into account the range of likely scenarios, from very low flows (highest risk) to very high flows (lowest risk), the risk to surface waters in the wetland has been allocated **low** risk rating for ***acidification*** and **low to medium** risk rating for ***contaminant mobilisation***. The risk associated with **deoxygenation** was determined to be **low** as there was no identified hazard associated with monosulfide formation and no evidence of monosulfides either in the wetland at the time of the Phase 1 field survey or forming during laboratory experiments.

In designing a management strategy for dealing with acid sulfate soils in Swan Reach Complex wetland, other values and uses of the wetland need to be taken into account to ensure that any intervention is compatible with other management plans and objectives for the wetland.

The wetland soils studied were largely dry at the time of sampling, therefore management options considered should relate to controlling or treating acidification and the protection of connected or adjacent wetlands. Due to the medium risks to the wetland values associated with contaminant mobilisation in Swan Reach Complex wetland, a monitoring program is recommended during any disturbance to the soils.

1. INTRODUCTION

At its March 2008 meeting, the Murray–Darling Basin Ministerial Council discussed the emerging issue of inland acid sulfate soils and the associated risks to Murray–Darling Basin waterways and agreed that the extent of the threat posed by this issue required assessment. The purpose of the Murray–Darling Basin Acid Sulfate Soils Risk Assessment Project was to determine the spatial occurrence of, and risk posed by, acid sulfate soils at priority wetlands in the River Murray system, wetlands listed under the Ramsar Convention on Wetlands of International Importance and other key environmental sites in the Murray–Darling Basin. The project involved the selection of wetlands of environmental significance, as well as those that may pose a risk to surrounding waters. These wetlands were then subjected to a tiered assessment program, whereby wetlands were screened through a desktop assessment stage, followed by a rapid on-ground appraisal, and then detailed on-ground assessment if results of previous stages indicated an increased likelihood of occurrence of acid sulfate soils.

Detailed assessments of acid sulfate soils within the Murray-Darling Basin (MDB) are conducted as a two-phase process under the MDB Acid Sulfate Soils Risk Assessment Project (ASSRAP). Phase 1 investigations are initially undertaken to determine whether or not acid sulfate soil materials are present in the study area, and provide characterisation of the properties and types of acid sulfate soils. Phase 2 investigations are only conducted if the acid sulfate soil materials from Phase 1 are determined to be a priority concern for the study area and, based on Phase 1 recommendations, selected samples undergo further investigations to determine the nature, severity and the specific risks associated with the acid sulfate soil materials. Phase 2 activities include: (i) soil laboratory analysis to confirm and refine the hazards associated with contaminant mobilisation and/or deoxygenation, (ii) a risk assessment, and (iii) interpretation and reporting, including discussion on broad acid sulfate soil management options.

Detailed Phase 1 acid sulfate soil assessments were undertaken at almost 200 wetlands and river channels throughout the Murray-Darling Basin. In South Australia, 56 wetlands along the River Murray between Lock 1 and Lock 5 were investigated by CSIRO Land and Water (Grealish *et al.* 2010). From these Phase 1 investigations, 13 wetlands were selected for further investigation. Nearly all of the wetlands along the River Murray between Wellington and Blanchetown (Lock 1) in South Australia also received detailed Phase 1 acid sulfate soil assessments (Grealish *et al.* 2011) and of these 23 wetlands were selected for further investigation in Phase 2. This included some wetlands below Lock 1 from earlier studies (Fitzpatrick *et al.* 2008; Fitzpatrick *et al.* 2010).

Following the Swan Reach Complex wetland Phase 1 assessment (Grealish *et al.* 2011) and the priority ranking criteria adopted by the Scientific Reference Panel of the MDB ASSRAP (see Table 1-1), Swan Reach Complex wetland was selected for Phase 2 detailed assessment. The Phase 1 assessment sampled from 8 sites (Figure 1-1) with each site comprising 2-4 soil layers. The Phase 1 assessment identified no high priority sites based on the presence of sulfuric materials, no high priority sites based on the presence of hypersulfidic materials, no high priority sites based on hyposulfidic materials with SCR ≥ 0.10% and 5 moderate priority sites based on the presence of hyposulfidic materials with SCR < 0.10%. Phase 2 investigations were carried out on selected surface soil samples from three sites (SRE 1, SRE 2 and SRE 8) identified in the Phase 1 assessment (Grealish *et al*. 2011).

The soils were all dry cracking clays with a surface clayey to sandy loam at two sites (Grealish *et al*. 2011). Net acidities were generally low, varying overall from 3 to 55 mol H^+/tonne.

Table 1-1 Priority ranking criteria adopted by the Scientific Reference Panel of the Murray-Darling Basin Acid Sulfate Soils Risk Assessment Project, from MDBA (2010).

Priority	Soil material
High Priority	All sulfuric materials. All hypersulfidic materials (as recognised by either 1) incubation of sulfidic materials or 2) a positive net acidity result with a Fineness Factor of 1.5 being used). All hyposulfidic materials with S_{CR} contents $\geq 0.10\%$ S. All surface soil materials (i.e. within 0-20 cm) with water soluble sulfate (1:5 soil:water) contents ≥ 100 mg kg^{-1} SO_4. All monosulfidic materials.
Moderate Priority	All hyposulfidic materials with S_{CR} contents $< 0.10\%$ S.
No Further Assessment	Other acidic soil materials. All other soil materials.

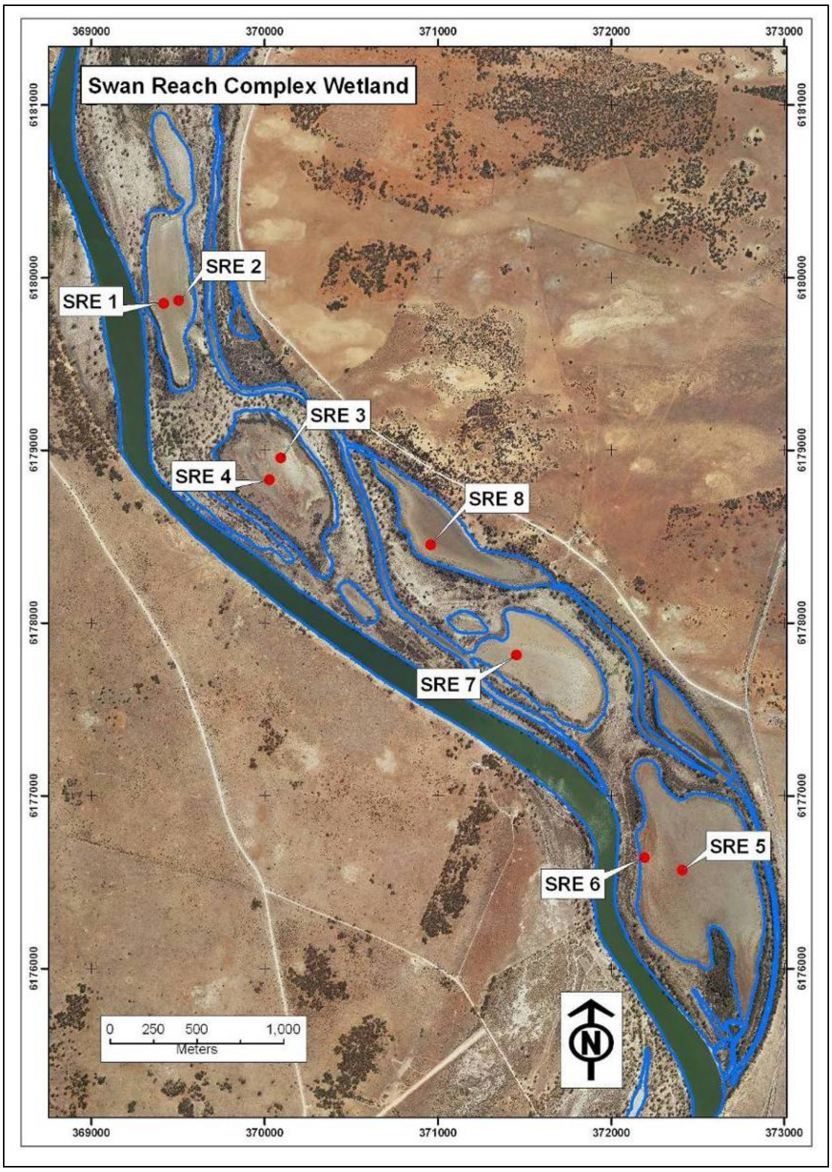

Figure 1-1 Swan Reach Complex wetland aerial photograph with Phase 1 sampling sites identified.

A summary of the soil laboratory analyses undertaken as part of the Phase 2 assessment and the sample selection criteria for each analysis is given in Table 1-2. Soil samples identified to undergo Phase 2 laboratory analysis are primarily from the surface and near-surface layers, as these are the soils most likely to have initial contact with water. A list of the samples selected for Phase 2 analysis for the Swan Reach Complex wetland is presented in Table 1-3.

Table 1-2 Rationale for Phase 2 sample selection, from MDBA (2010).

Parameter	Samples selected
Reactive metals	Conducted on selected upper two surface samples.
Contaminant and metalloid dynamics	Conducted on selected upper two surface samples.
Monosulfide formation potential	Conducted on surface samples of dry sites that meet the water extractable sulfate criteria for monosulfides.
Mineral identification by X-ray diffraction (XRD)	Conducted on a limited number of selected crystals and minerals (if present). Most likely to be associated with sulfuric layers to confirm acid mineral presences.
Acid base accounting data	Conducted only on samples from wetlands below Lock 1 and Burnt Creek/Loddon River if not previously analysed and pH_{KCl}<4.5.

Table 1-3 Summary of Swan Reach Complex wetland samples analysed for Phase 2 assessment.

Soil Laboratory Test	Swan Reach Complex wetland samples	Depth of sample (cm)	Number of samples analysed
Reactive metals	SRE1.1	0-3	4
	SRE1.2	3-25	
	SRE2.1	0-5	
	SRE 8.1	0-10	
Contaminant and metalloid dynamics	SRE1.1	0-3	4
	SRE1.2	3-25	
	SRE2.1	0-5	
	SRE 8.1	0-10	
Monosulfide formation potential	SRE 2.1	0-5	1
Mineral identification by X-ray diffraction (XRD)	N/A	-	0

2. LABORATORY METHODS

2.1. Laboratory analysis methods

2.1.1. Summary of laboratory methods

A list of the method objectives for the Phase 2 assessment are summarised below in Table 2-1. All soil samples analysed in this Phase 2 assessment were collected and subsequently stored as part of the Phase 1 field assessment.

Table 2-1 Phase 2 data requirements - list of parameters and objective for conducting the test, from MDBA (2010).

Parameter	Objective
Reactive metals	Assists with determining impacts on water quality by determining weakly to moderately strongly bound metals.
Contaminant and metalloid dynamics	Assists with determining impacts on water quality by simulating longer time frames that create anaerobic conditions. Identifies metal release concentrations that may occur over a 5 week time frame.
Monosulfide formation potential	Determine relative propensity for monosulfides to form following inundation.
Mineral identification by X-ray diffraction (XRD)	Characterisation and confirmation of minerals present.

Guidelines on the approaches that were followed as part of this Phase 2 assessment are presented in full in the detailed assessment protocols (MDBA 2010).

2.1.2. Reactive metals method

The guidelines for the reactive metals method are outlined as an addendum to the detailed assessment protocols (MDBA 2010). In this method, samples were prepared by disaggregation (not grinding) using a jaw crusher, and then sieved to include only the <2 mm fine earth fraction. A total of 2.5 g soil was added to 40 ml of 0.1 M HCl, gently mixed for 1 hour and filtered through a pre-acid washed 0.45 μm nitro-cellulose filter. The metals examined comprised silver (Ag), aluminium (Al), arsenic (As), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), manganese (Mn), nickel (Ni), lead (Pb), antimony (Sb), selenium (Se), vanadium (V) and zinc (Zn).

2.1.3. Contaminant and metalloid dynamics method

The guidelines for the contaminant and metalloid dynamics method are outlined in Appendix 7 of the detailed assessment protocols (MDBA 2010). The contaminant and metalloid dynamics method was designed to determine the release of metals and metalloids in soils after 24 hours. The data represent the availability of metals and metalloids from a weak extraction (water, and thus easily bioavailable) of saturated soils, and for dry wetland soils, those easily mobilised from mineral surfaces and readily soluble mineral phases (such as salts). The exercise was repeated in a batch process for longer time periods (7 days, 14 days, 35 days). The latter approach was aimed at understanding changes in concentrations over time. This is particularly important for dried soils which have been in contact with the atmosphere. The soil materials and the release/uptake of metals/metalloids are expected to change as the chemical environment changes from oxidising to reducing. The data can be compared to existing water quality guidelines, although care should be taken when extrapolating to surface waters without knowledge of hydrological conditions and natural chemical barriers. The impact on surface waters will be governed by the upward chemical flux which is a function of soil type, water flow, diffusion and the chemistry of the soils near the sediment-water interface.

Redox potential (Eh) and pH were determined using calibrated electrodes linked to a TPS WP-80 meter; Eh measurements were undertaken in an anaerobic chamber to minimise the rapid changes encountered due to contact with the atmosphere, and are presented relative to the standard hydrogen electrode (SHE). Specific electrical conductance (SEC) was determined using a calibrated electrode linked to a TPS WP-81 meter. All parameters were measured on filtered (0.45 µm) water samples.

2.1.4. Monosulfide formation potential method

The guidelines for the monosulfide formation potential method are outlined in Appendix 8 of the detailed assessment protocols (MDBA 2010). In this study 3.6 g/L sucrose was used as an organic substrate instead of the 7.2 g/L outlined in the protocols. In addition to sampling after seven weeks, water samples were collected and analysed immediately after inundating the soils (i.e. Day 0). The pore-water pH and Eh were determined at Day 0.

The reactive iron (Fe) fraction in field moist sediments was extracted using 1.0 M HCl (Claff *et al.* 2010). The ferrous iron (Fe^{2+}) and total iron (Fe^{2+} + Fe^{3+}) fractions were immediately fixed following extraction. The ferrous iron trap was made up from a phenanthroline solution with an ammonium acetate buffer (APHA 2005), and the total iron trap also included a hydroxylamine solution (APHA 2005). The iron species were quantified colorimetrically using a Hach DR 2800 spectrophotometer.

Redox potential and pH were determined using calibrated electrodes linked to a TPS WP-80 meter; Eh measurements are presented versus the standard hydrogen electrode. In this study the solid phase elemental sulfur fraction was extracted using toluene as a solvent and quantified by high-performance liquid chromatography (HPLC) (McGuire and Hamers 2000). Pore-water sulfide was preserved in zinc acetate prior to determination by the spectrophotometric method of Cline (1969).

2.1.5. Mineral identification by x-ray diffraction

The guidelines for mineral identification by x-ray diffraction are outlined in the detailed assessment protocols (MDBA 2010).

2.2. Quality assurance and quality control

For all tests and analyses, the quality assurance and quality control procedures were equivalent to those endorsed by NATA (National Association of Testing Authorities). The standard procedures included the monitoring of blanks, duplicate analysis of at least 1 in 10 samples, and the inclusion of standards in each batch.

Reagent blanks and method blanks were prepared and analysed for each method. All blanks examined here were either at, or very close to, the limits of detection. On average, the frequencies of quality control samples processed were: 10% blanks, 10% laboratory duplicates, and 10% laboratory controls. The analytical precision was ±10% for all analyses. In addition, for all samples, reactive metals and contaminant and metalloid dynamics tests were duplicated. For the reactive metals, two International Standards (Reference Stream Sediment STSD-2 and STSD-3 Canadian Certified Reference Materials) were processed in an identical manner to the samples. Precision was excellent with the coefficient of variation (standard deviation/mean*100) typically being in the range < 1 to 2 %.

3. RESULTS AND DISCUSSION

3.1. Summary of soil laboratory results

3.1.1. Reactive metals data

The data are presented on a dry weight basis (mg kg^{-1}) and shown in Table 3-1. The 24 hour reactive metals studies provide an indication of those metals and metalloids which are more strongly bound to minerals (or weakly soluble with an acid extraction) than would be soluble with a water extraction, and thus have the potential to be released. The use of a moderately strong acid (0.1 M HCl) should provide an indication of "stored metals" and metalloids associated with iron (Fe) and manganese (Mn) oxides and organic materials as well as acid soluble minerals. It is commonly found that the concentrations of metals and metalloids released using extractions are much higher than those found in solution (Gooddy *et al.* 1995). Although guideline values exist for soils and sediments, these are generally for total soil concentrations, and therefore, are not directly appropriate for the data from metal mobilisation studies. Nevertheless, they provide a basis for comparison; and concentrations close to or above guideline values indicate an elevated hazard.

The concentrations of metals and metalloids were below sediment quality guideline (SQG) values and soil ecological investigation levels (EIL) for those elements where guidelines are available (Table 3-1). The concentrations for most reactive metals and metalloids are relatively low, being highest for aluminium (Al), iron (Fe) and manganese (Mn).

Table 3-1 Swan Reach Complex wetland reactive metals data.

Sample	Ag*	Al	As	Cd*	Co	Cr*	Cu	Fe	Mn	Ni	Pb	Sb*	Se*	V	Zn
SRE 1.1	11	311	0.79	51	2.5	101	4.4	321	119	3.4	3.4	< 1.7	31	9.5	3.3
SRE 1.2	9.7	270	0.67	49	0.52	36	4.0	326	80	3.2	3.1	< 1.8	32	8.5	1.8
SRE 2.1	10	313	0.81	54	2.7	94	4.0	566	216	3.6	3.7	< 1.7	39	9.8	3.6
SRE 8.1	7.8	341	1.1	49	4.4	154	4.0	439	176	3.6	3.6	< 2.4	39	7.5	5.6
[1]SQG	1000	-	20	1500	-	80000	65	-	-	21	50	2000	-	-	200
[2]Soil EIL	-	-	20	3000	-	-	100	-	500	60	600	-	-	50	200

* Units are in µg kg^{-1}
< value is below detection limit
[1]SQG: Sediment Quality Guideline Value (Australian and New Zealand Guidelines for Fresh and Marine Water Quality 2000)
[2]Soil EIL: Soil – Ecological Investigation Level (NEPC 1999)

3.1.2. Contaminant and metalloid dynamics data

The contaminant and metalloid dynamics data for the four Swan Reach Complex wetland soil materials examined are presented in Appendix 2, summarised in Table 3-2 and plotted against time in Figure 3-1 to Figure 3-3. Table 3-2 also compares the pore-water metal contents to the relevant national water quality guideline for environmental protection (ANZECC/ARMCANZ 2000).

Table 3-2 Summary of contaminant and metalloid dynamics data

Parameter	units	ANZECC Guidelines	Swan Reach Complex		
			Min.	Median	Max.
pH		6.5-8.0	🟥 **4.2**	🟨 **6.3**	6.7
EC*	µS cm^{-1}	2200	95	193	350
Eh	mV	-	4.0	310	422
Ag	µg l^{-1}	0.05	<0.01	<0.01	<0.04
Al[A]	mg l^{-1}	0.055	<0.05	🟨 **0.06**	🟧 **0.69**
As[B]	µg l^{-1}	13	<0.20	9.7	🟨 **30**
Cd	µg l^{-1}	0.2	<0.01	<0.06	<0.3
Co	µg l^{-1}	2.8	0.05	🟨 **4.4**	🟨 **12**
Cr[C]	µg l^{-1}	1	<0.10	<0.50	🟨 **1.5**
Cu[H]	µg l^{-1}	1.4	<0.20	1.2	🟨 **11**
Fe[I]	mg l^{-1}	0.3	<0.10	🟨 **0.54**	🟧 **11**
Mn	µg l^{-1}	1700	5.2	392	🟨 **1944**
Ni[H]	µg l^{-1}	11	0.49	5.3	🟨 **14**
Pb[H]	µg l^{-1}	3.4	<0.06	<1.0	<3.0
Sb	µg l^{-1}	9	<0.60	<4.0	<10.0
Se	µg l^{-1}	11	0.06	0.34	0.65
V	µg l^{-1}	6	0.80	3.3	🟨 **13**
Zn[H]	µg l^{-1}	8	<0.7	0.95	4.0

🟨 ***Exceeded ANZECC Guideline (x1)***

🟧 ***Exceeded ANZECC Guideline (x10)***

🟥 ***Exceeded ANZECC Guideline (x100)***

Notes.
The ANZECC guideline values for toxicants refer to the Ecosystem Protection – Freshwater Guideline for protection of 95% of biota in 'slightly-moderately disturbed' systems, as outlined in the Australian Water Quality Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ 2000).
** ANZECC water quality upper guideline (125-2200 µS cm^{-1}) for freshwater lowland rivers in South-east Australia is provided for salinity (there are currently no trigger values defined for 'Wetlands'.)*

[A] *Guideline is for Aluminium in freshwater where pH > 6.5.*
[B] *Guideline assumes As in solution as Arsenic (AsV).*
[C] *Guideline for Chromium is applicable to Chromium (CrVI) only.*
[H] *Hardness affected (refer to Guidelines).*
[I] *Fe Guideline for recreational purposes*

The pH of all soil materials was slightly acidic after the day 1 contaminant and metalloid dynamics tests, and showed a slight increase over time (Figure 3-1). One of the samples SRE 1.2 showed a decrease on day 7, but subsequently increased. By day 35, three of the four samples had a pH above the lower ANZECC/ARMCANZ environmental protection guideline value (Figure 3-1).

The Eh showed a significant decrease over the 35 day period in all samples, following a slight increase for some samples on day 7 (Figure 3-1). Final Eh varied from slightly (SRE 1.2) to quite strongly reducing (SRE 2.1 and SRE 8.1). The SEC was variable, with the samples from SRE 1 being significantly lower than SRE 2 and 8.

Manganese (Mn) concentrations increased over time in the three surface soils, remaining low in the sub-soil sample SRE 1.2 (Figure 3-2). The ANZECC/ARMCANZ environmental protection guideline value was breached in only one sample, but there was a trend of increasing manganese with time. Iron (Fe) concentrations displayed a similar trend to manganese (Mn), reaching high concentrations, up to 11.25 mg l^{-1}. The data are in agreement with the change to more reducing conditions, where manganese (Mn) and then iron (Fe) become soluble. The sample (SRE 1.2) with lowest concentrations of iron and manganese had the highest Eh.

Aluminium (Al) concentrations were relatively high in some samples, but showed no clear trend (Figure 3-1). At the pH values observed it is likely that the aluminium is present in colloidal form. This may be related to the dispersion of clays due to dilution, but not possible to validate with existing data. Arsenic (As) concentrations increased in the three most reducing samples and were similar to iron (Fe) and manganese (Mn), suggesting a strong redox control. The increase was sufficient to allow significant exceedance of the ANZECC/ARMCANZ environmental protection guideline.

Cobalt (Co) and nickel (Ni) both displayed similar trends to iron (Fe) and manganese (Mn) initially, increasing in concentration ((Figure 3-2 and Figure 3-3), however, both decreased between days 14 and 35. Although both breached the ANZECC/ARMCANZ environmental protection guideline, by day 35 the concentrations in many samples had decreased below the guideline value (Figure 3-2 and Figure 3-3). Chromium (Cr) showed a slight increase with time, breaching the ANZECC/ARMCANZ environmental protection guideline slightly. Copper (Cu) showed a sharp increase up to day 14 in one sample (SRE 1.1), but as with cobalt (Co) and nickel (Ni), subsequently decreased by day 35. The highest vanadium (V) concentration was found in the most reducing sample (SRE 8.1), with most samples showing an increase over time.

The magnitude of metal mobilisation is affected by many factors that include but are not exclusive to: 1) the abundance and form of metal and metalloid contaminants; 2) the abundance and lability of organic matter; 3) the abundance and reactivity of iron minerals; 4) availability of sulfate; 5) acid/alkalinity buffering capacity; 6) pH; 7) EC; 8) clay content; 9) microbial activity; 10) temperature; and 11) porosity (MDBA 2010). The relationship with pH for metals and metalloids showed no consistent behaviour, although the most acidic sample generally contained the lowest concentrations of metals and metalloids. The patterns for most metals and metalloids were similar to iron (Fe) and manganese (Mn) suggesting a geochemical control, the exception being copper (Cu), which may be related more to organic matter in the soils.

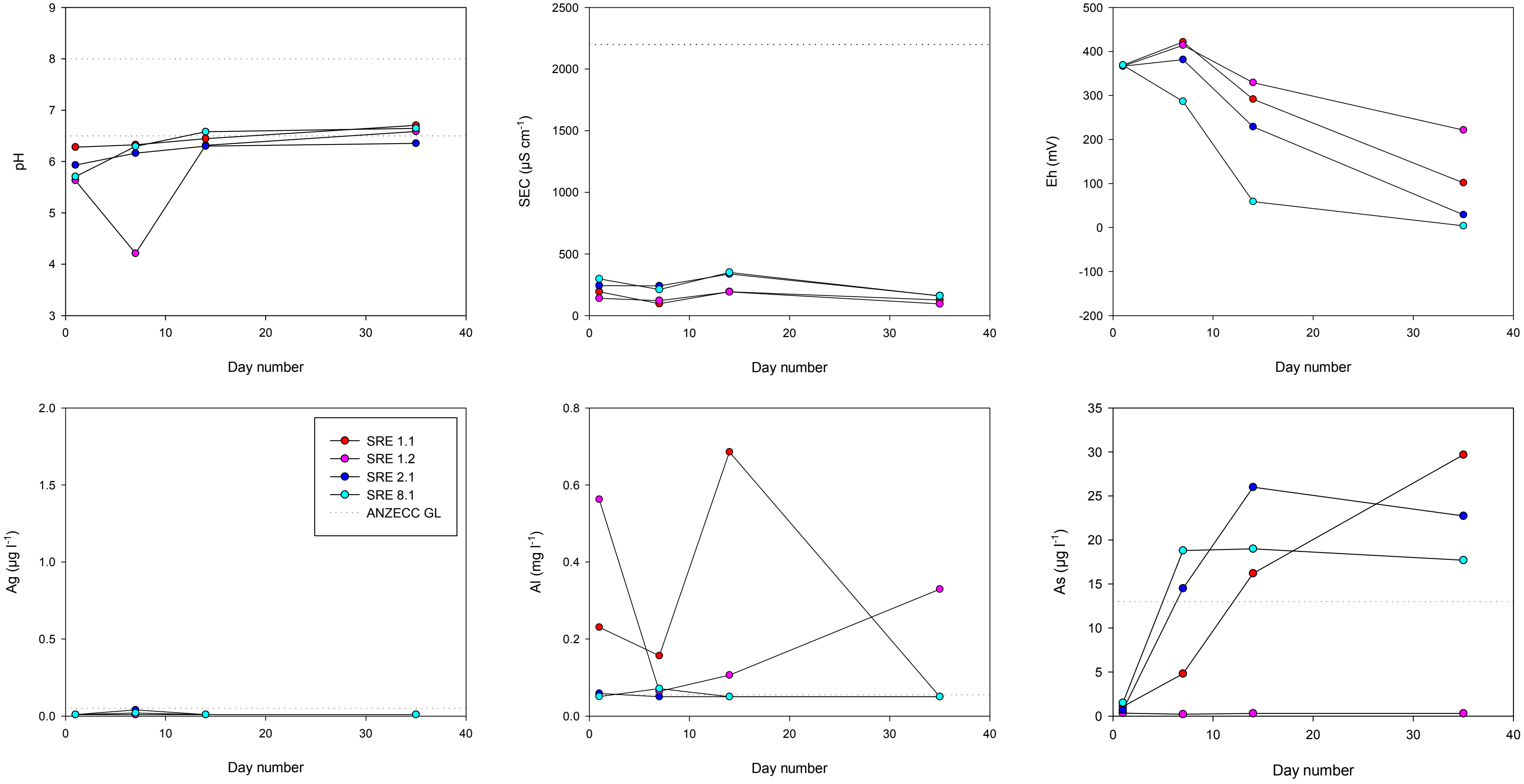

Figure 3-1 Contaminant and metalloid dynamics results for Swan Reach Complex soil materials for pH, SEC, Eh, silver (Ag), aluminium (Al) and arsenic (As).

Note: silver (Ag) was all < detection limit, data represent detection limits which vary according to required dilutions.

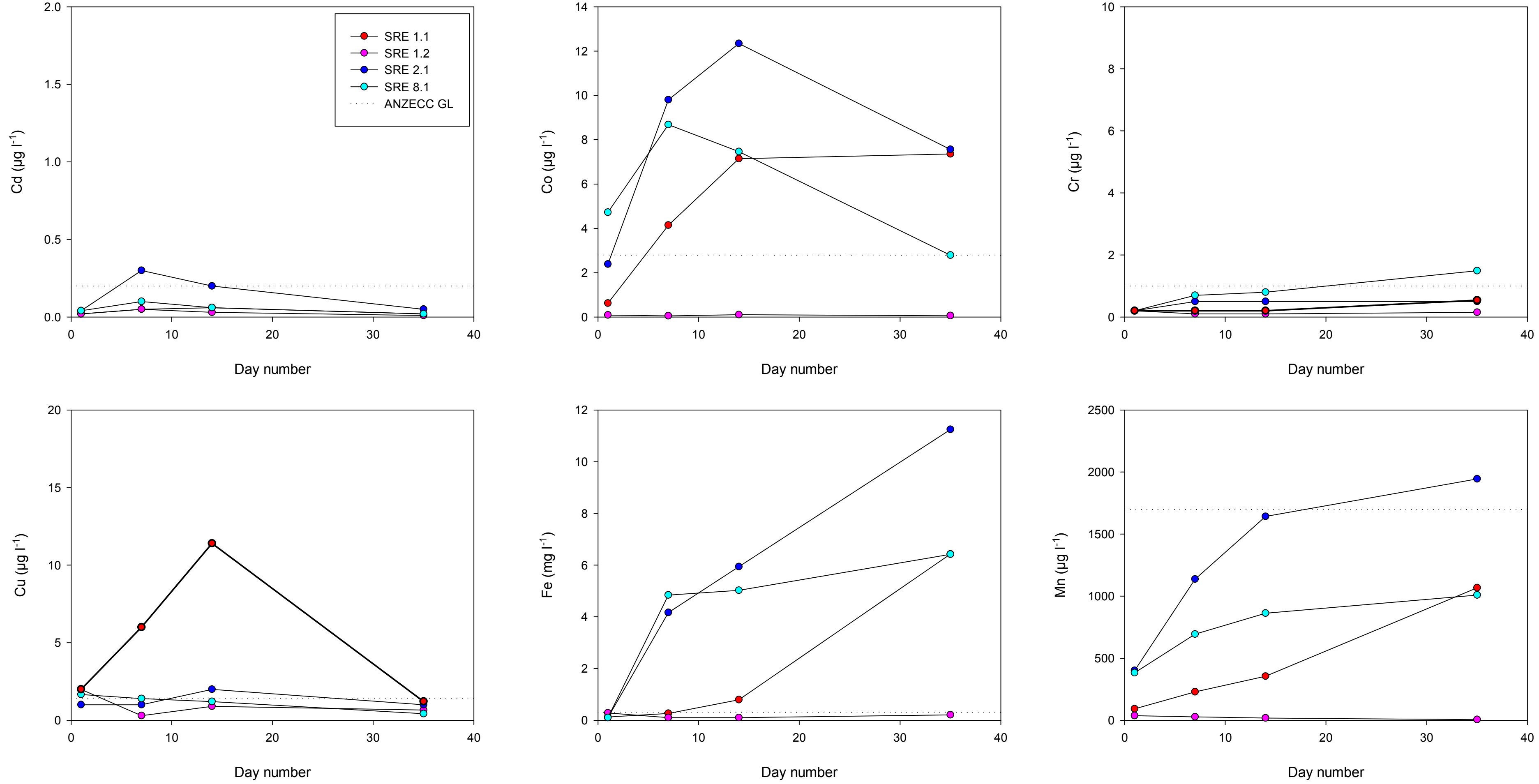

Figure 3-2 Contaminant and metalloid dynamics results for Swan Reach Complex wetland soil materials for cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe) and manganese (Mn).

Note: cadmium (Cd) was all < detection limit, data represent detection limits which vary according to required dilutions.

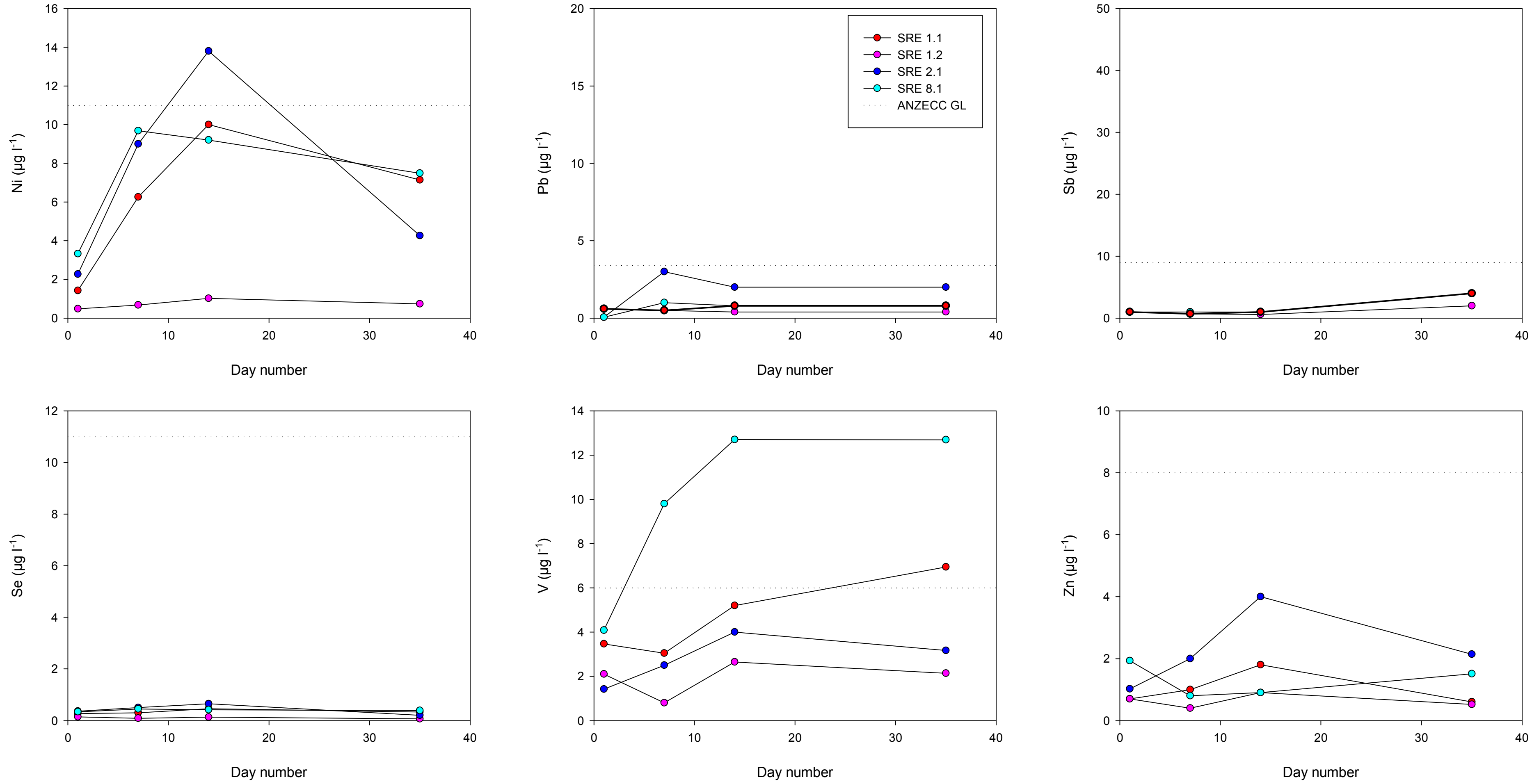

Figure 3-3 Contaminant and metalloid dynamics results for Swan Reach Complex wetland soil materials for nickel (Ni), lead (Pb), antimony (Sb), selenium (Se), vanadium (V) and zinc (Zn).

Note: lead (Pb) and antimony (Sb) were all < detection limit, data represent detection limits which vary according to required dilutions.

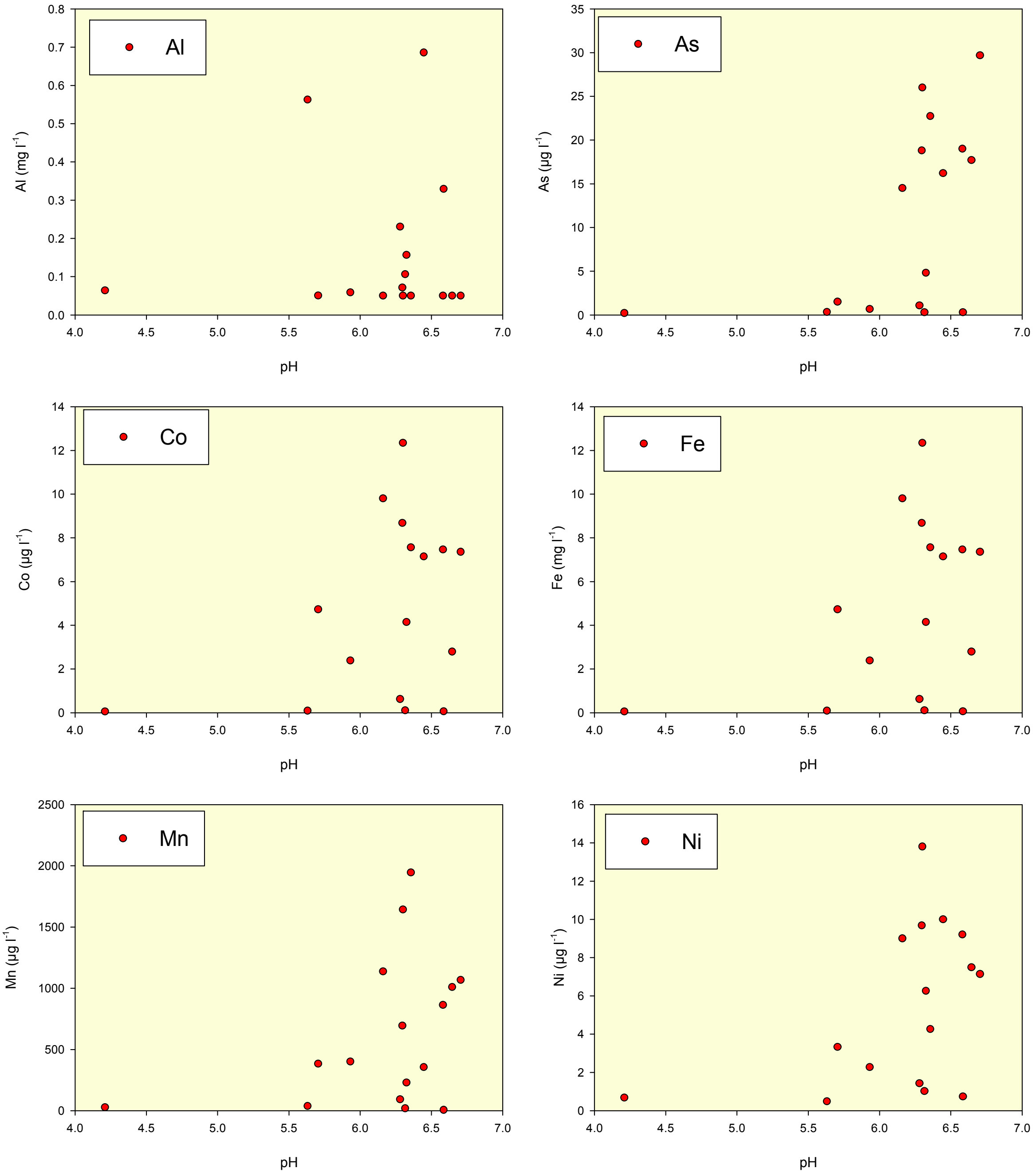

Figure 3-4 Selected trace elements plotted against pH.

3.1.3. Monosulfide formation potential data

The monosulfide formation potential data for sample SRE 2.1 are shown in Table 3-3. The pH of the soil water changed from 5.60 to 4.61 over the seven week incubation period. The decrease in pH is different from the contaminant and metalloid dynamics tests, where pH increased from 5.93 to 6.36 (Figure 3-1). This may be due to fermentation of organic substrate added (sucrose) which caused acidification of the pore-waters.

The Eh decreased significantly from 438 to 166 mV indicating a significant change to more reducing conditions. The Eh data are generally consistent with the Eh trend in the contaminant and metalloid dynamics experiments, where Eh decreased from 367 to 29 mV. The differences were possibly due to the lower pH in the monosulfide formation potential tests.

Table 3-3 Summary of monosulfide formation potential data for the Swan Reach Complex wetland surface soil material SRE 2.1 after 7 weeks (3.6 g/L sucrose).

Inundation Time	Parameter	Units	SRE 2.1
Day 0	Total Fe	mg/kg	4934
	$Fe(II)^-$	mg/kg	608
	Sulfate*	mg/kg	1980
	pH		5.60
	Eh	mV	438
Week 7	pH		4.61
	Eh	mV	166
	S_{AV}	Wt. %S	<0.01
	S^o	Wt. %S	<0.01
	Pyrite-S	Wt. %S	0.02
	Dissolved S^{2-}	µg/L	25

*completed during Phase 1

The sample originally contained 0.01% S_{CR} (pyrite-S + S_{AV}) and no ANC (Grealish *et al.* 2011). After 7 weeks, acid volatile sulfide (S_{AV}) and elemental sulfur (S^o) were both <0.01 %, with only a trace of pyrite-S (Table 3-3). It appears, therefore, that monosulfide formation (or significant pyrite formation) has not occurred in this sample during the tests. Dissolved sulfide concentrations were low at 25 µg l^{-1} (Table 3-3), suggesting that redox conditions are not sufficiently low enough for sulfide formation.

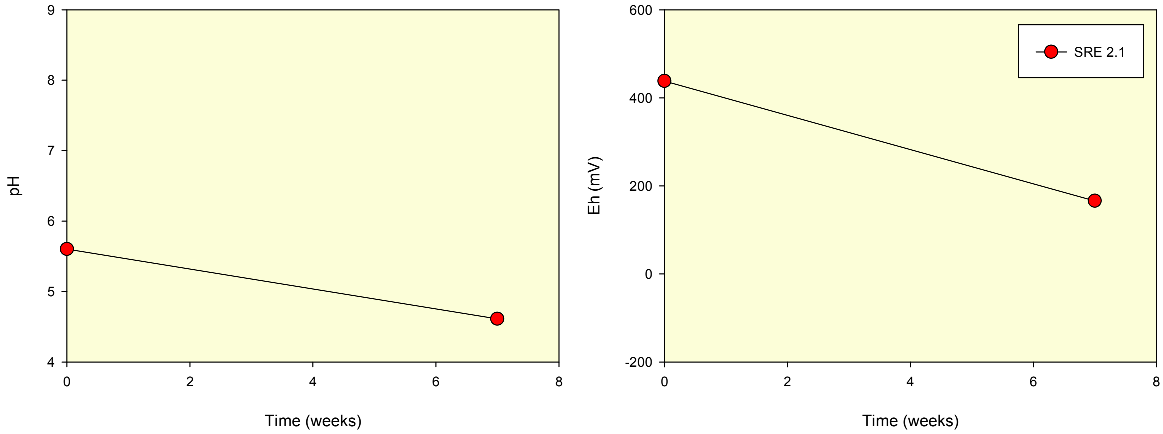

Figure 3-5 pH and Eh dynamics during monosulfide formation potential tests in surface soil sample SRE 2.1 from Swan Reach Complex wetland.

3.1.4. Mineral identification by x-ray diffraction

No surface mineral efflorescences were identified or sampled at this wetland during the Phase 1 field survey.

3.2. Interpretation and discussion of results

The reactive metals and contaminant and metalloid dynamics tests undertaken as part of this Phase 2 assessment assist in determining the impacts on water quality by simulating the release of metal and metalloid concentrations that may occur under saturated conditions.

The 24 hour **reactive metals** studies provide an indication of those metals and metalloids which are more strongly bound to minerals (or weakly soluble with an acid extraction), and thus have the potential to be released. The use of a moderately strong acid (HCl) should provide an indication of “stored metals” and metalloids associated with iron (Fe) and manganese (Mn) oxides and organic materials as well as acid soluble minerals. It is commonly found that the concentrations of metals and metalloids released using extractions are much higher than those found in solution (Gooddy *et al.* 1995). Although guideline values exist for soils and sediments, these are generally for total soil concentrations, and therefore, are not directly appropriate for the data from metal mobilisation studies. Nevertheless, they provide a basis for comparison; and concentrations close to or above guideline values indicate an elevated hazard.

The metal and metalloid concentrations were all below sediment quality guidelines and soil ecological investigation level values (Table 3-1). Nevertheless, the concentrations of many metals are sufficiently high (mg kg^{-1}) compared to water quality guidelines (generally µg kg^{-1}) that significant release could pose a hazard to soil and surface water quality.

The **contaminant and metalloid dynamics** test was designed to determine the release of metals and metalloids in soils. The data represent the availability of metals and metalloids from a weak extraction (water, and thus easily bioavailable) of saturated soils, and for dry wetland soils (especially below Lock 1), those easily mobilised from mineral surfaces and readily soluble mineral phases (such as salts). The exercise was undertaken in a batch process for time periods of 1 day, 7 days, 14 days and 35 days. This approach was aimed at understanding changes in concentrations over time. This is particularly important for dried soils which have been in contact with the atmosphere. The soil materials and the release/uptake of metals/metalloids are expected to change as the chemical environment changes from oxidising to reducing. Typical changes would be a reduction in redox potential (Eh), providing sufficient organic matter or other reducing agents are present, and an increase in pH (providing the soils contain or have the capacity to generate acid neutralising agents). The data can be compared to existing water quality guidelines, although care should be taken when extrapolating to surface waters without knowledge of hydrological conditions and natural chemical barriers. The impact on surface waters will be governed by the upward chemical flux which is a function of soil type, water flow, diffusion and the chemistry of the soils near the sediment-water interface. The mobility of most metals is commonly related to the stability of iron (Fe) and manganese (Mn) minerals. Under oxidising conditions iron (Fe) and manganese (Mn) oxide minerals are important sorbents for trace metals, whilst under very reducing conditions they may be incorporated into sulfide minerals. However, under moderately reducing conditions i.e. during the transition (suboxic) from oxidising to reducing conditions, iron (Fe) and manganese (Mn) are soluble and this is the period where metals may be released into solution and pose the greatest hazard.

The soils had a slightly acidic to circumneutral pH, and by day 35 of the contaminant and metalloid mobilisation tests, the pH was above or close to the lower limit for ANZECC/ARMCANZ environmental protection guideline values. The pH was high enough that reductive processes form important controls on the solubility of metals and metalloids. For three soil samples, the Eh decreased sufficiently that iron (Fe) solubility has increased significantly.

The highest concentrations of most metals and metalloids which have breached ANZECC/ARMCANZ environmental protection guideline values were associated with lower Eh in the samples, suggesting that the reductive dissolution of iron (Fe) and manganese (Mn) oxides/oxyhydroxides are a major control of metal and metalloid mobility. A number of trace elements (cobalt, nickel) reached a maximum after 14 days, subsequently decreasing by day 35. The decrease may be related to diffusion into the soil matrix or more likely adsorption onto soil mineral surface layers, as sorption of metal cations generally increases with pH.

The degree to which samples exceed guideline concentrations has been used to assign a degree of hazard (Table 3-4). For some samples which required dilution, the detection limits were slightly above ANZECC/ARMCANZ environmental protection guideline values due to required dilution. Cadmium (Cd) for one sample had a detection limit of 0.3 µg l^{-1}, which is slightly higher than the ANZECC/ARMCANZ environmental protection guideline value. It is therefore not possible to determine hazard classifications for cadmium (Cd), although it can be concluded that it either sits in the no to moderate hazard grouping.

The data are consistent with the generally low (3-55 mol H^+/tonne) net acidities noted by Grealish *et al.* (2011) which suggest that these shallow soils will take little time to recover in terms of any acidification. This appears to be largely confirmed in the contaminant and metalloid dynamics experiments where most samples reached pH values approaching neutrality during the experiments (Figure 3-1). The higher pH will limit the solubilities of most trace metals, and the concentrations in this study suggest that mobilisation will be limited and of short duration at the circumneutral pH of most samples. The main risk will be related to how quickly reducing conditions are achieved and maintained in the soils. Both iron (Fe) and manganese (Mn) were increasing by day 35 of the contaminant and metalloid dynamics tests, indicating that continued reducing conditions may lead to further release of a number of metals and metalloids to solution. At higher pH, the metalloids identified as a hazard (arsenic and vanadium) may be stable due to limited adsorption of oxyanions at high pH. These metalloids may remain at relatively high concentrations over longer timescales, at least until further reduction into the field of iron sulfide stability, in which case they may be scavenged by precipitating iron (Fe) sulfides.

Table 3-4 Summary of the degree of hazard associated with the measured contaminant and metalloid concentrations in the Swan Reach Complex wetland.

Degree of Hazard	Guideline Threshold	Metal/Metalloid
No Hazard	Value below ANZECC/ARMCANZ guideline threshold	Ag, Pb, Se, Zn
Low Hazard	Value exceeds ANZECC/ARMCANZ guideline threshold, but is less than 10x exceedance	As, Co, Cr, Cu, Mn, Ni, V
Moderate Hazard	Value exceeds ANZECC/ARMCANZ guideline threshold by 10x or more, but is less than 100x exceedance	Al, Fe
High Hazard	Value exceeds ANZECC/ARMCANZ guideline threshold by 100x or more	

Note: Cd and Sb were below detection limits in some samples due to dilutions, and therefore can only be classified as no to moderate hazard (see Appendix 2 for data).

The monosulfide formation potential test assists in determining the propensity for monosulfides to form during future inundation. The sample used for this test contained no acid volatile sulfide (S_{AV}) or elemental sulfur (S^0). The lack of acid volatile sulfide (S_{AV}) ranks the monosulfide formation potential hazard as ‘No hazard’ (Table 3-5).

Table 3-5 Guideline thresholds for the degree of hazard associated with acid volatile sulfide (S_{AV}) concentrations.

Degree of Hazard	Guideline Threshold
No Hazard	< 0.01 % S_{AV}
Low Hazard	0.01 % S_{AV}
Moderate Hazard	>0.01 – 0.05 % S_{AV}
High Hazard	> 0.05 % S_{AV}

4. RISK ASSESSMENT

4.1. Risk assessment framework

Risk is a measure of both the consequences of a hazard occurring, and the likelihood of its occurrence (MDBA 2011). According to the National Environment Protection Measures (NEPM), risk is defined as "*the probability in a certain timeframe that an adverse outcome will occur in a person, a group of people, plants, animals and/or the ecology of a specified area that is exposed to a particular dose or concentration of a hazardous agent, i.e. it depends on both the level of toxicity of hazardous agent and the level of exposure*" (NEPC 1999).

The MDB Acid Sulfate Soils Risk Assessment Project developed a framework for determining risks to wetland values from acid sulfate soil hazards (MDBA 2011). The risk assessment framework has been applied in this study to determine the specific risks associated with acidification, contaminant mobilisation and de-oxygenation. In this risk assessment framework, a series of standardised tables are used to define and assess risk (MDBA 2011). The tables determine the consequence of a hazard occurring (Table 4-1), and a likelihood rating for the disturbance scenario for each hazard (Table 4-2). These two factors are then combined in a risk assessment matrix to determine the level of risk (Table 4-3).

Table 4-1 determines the level of consequence of a hazard occurring, ranging from insignificant to extreme, and primarily takes account of the environmental and water quality impacts to the wetland values and/or adjacent waters.

Table 4-1 Standardised table used to determine the consequences of a hazard occurring, from MDBA (2011).

Descriptor	Definition
Extreme	Irreversible damage to wetland environmental values and/or adjacent waters; localised species extinction; permanent loss of drinking water (including stock and domestic) supplies.
Major	Long-term damage to wetland environmental values and/or adjacent waters; significant impacts on listed species; significant impacts on drinking water (including stock and domestic) supplies.
Moderate	Short-term damage to wetland environmental values and/or adjacent waters; short-term impacts on species and/or drinking water (including stock and domestic) supplies.
Minor	Localised short-term damage to wetland environmental values and/or adjacent waters; temporary loss of drinking water (including stock and domestic) supplies.
Insignificant	Negligible impact on wetland environmental values and/or adjacent waters; no detectable impacts on species.

Table 4-2 determines the likelihood (i.e. probability) of disturbance for each hazard, ranging from rare to almost certain. This requires an understanding of the nature and severity of the materials (including the extent of acid sulfate soil materials, the acid generating potential and the buffering capacity of wetland soil materials) as well as contributing factors influencing the risk (MDBA 2011). Examples of disturbance include: (i) rewetting of acid sulfate soil materials after oxidation, (ii) acid sulfate soil materials that are currently inundated and may be oxidised, or (iii) acid sulfate soil materials that are currently inundated and may be dispersed by flushing (e.g. scouring flows) (MDBA 2011). As mentioned previously, the consequence of a hazard occurring and the likelihood rating for the disturbance scenario for each hazard are then ranked using a standardised risk assessment matrix (Table 4-3).

Table 4-2 Likelihood ratings for the disturbance scenario, from MDBA (2011).

Descriptor	Definition
Almost certain	Disturbance is expected to occur in most circumstances
Likely	Disturbance will probably occur in most circumstances
Possible	Disturbance might occur at some time
Unlikely	Disturbance could occur at some time
Rare	Disturbance may occur only in exceptional circumstances

Table 4-3 Risk assessment matrix, adapted from Standards Australia & Standards New Zealand (2004).

Likelihood category	Consequences category				
	Extreme	Major	Moderate	Minor	Insignificant
Almost certain	Very High	Very High	High	Medium	Low
Likely	Very High	High	Medium	Medium	Low
Possible	High	High	Medium	Low	Low
Unlikely	High	Medium	Medium	Low	Very low
Rare	High	Medium	Low	Very low	Very low

It is suggested that:

- For very high risk immediate action is recommended.
- For high risk senior management attention is probably needed.
- Where a medium risk is identified management action may be recommended.
- Where the risk is low or very low, routine condition monitoring is suggested.

These categories of management responses have been kept quite broad to acknowledge that jurisdictional authorities and wetland managers may choose to adopt different approaches in dealing with acid sulfate soils. The imprecise nature of these management responses is intended to provide flexibility in jurisdictional and wetland manager responses to the risk ratings associated with the acid sulfate soil hazards (MDBA 2011).

4.2. Assessment of risks

Realisation of the main risks associated with acid sulfate soil hazards (acidification, contaminant mobilisation and deoxygenation) is highly dependent on transport and therefore on the surface and sub-surface hydrology. The risks are thus scenario dependent, and difficult to quantify without predicted changes of water flows and inputs and hydrogeological controls.

The consequences of a hazard, as outlined in Table 4-1, relate to reversible or irreversible damage to wetland values. Few studies have documented in sufficient detail the short or long term damage to inland wetland ecosystems and values caused by acid sulfate soil hazards, but short term consequences have been clearly illustrated e.g. for water quality and ecosystem impacts (McCarthy *et al.* 2006; Shand *et al.* 2010). Irreversible damage is difficult to assess due to lack of sufficient data over longer timescales and lack of knowledge, for example, on sub-surface soil recovery and metal mobilisation impacts on benthic organisms. Nevertheless, the following sections detail the hazards and likelihood of a number of scenarios and discuss consequences based on limited previous work (e.g. McCarthy *et al.* 2006; Shand *et al.* 2010). The risks to soil water quality and surface water quality are necessarily different. The risks to soil water quality in terms of acidification and contaminant release are easier to assess from the tests carried out in this study than the risks posed to surface water quality. The impacts on surface water quality will be largely controlled by upward flux of acidity and metals from the soils and sediments into the water column. This will be controlled by *inter alia* surface water volume and groundwater connectivity and level, soil type, hydraulic conductivity and degree and depth of soil cracking.

The Swan Reach Complex wetland has been classified as medium conservation status by the SA Murray-Darling Basin Natural Resources Management Board (Miles *et al.* 2010).

4.2.1. Risks associated with acidification

The variable net acidities in shallow samples from Swan Reach Complex wetland studied here (11-55 mol H^+/tonne) and overall (3-55 mol H^+/tonne, Grealish *et al*. 2011) suggest that the overall acidification hazard is likely to be low. Furthermore, the ageing of soils during Phase 1 showed that no soil materials changed to sulfuric materials (although pH's were relatively low in some samples, ranging overall from pH 4.0 to 5.0). The acidification hazard is therefore considered to be moderately low.

Due to the dry nature of the wetland during sampling, the likelihood of disturbance is considered ***almost certain*** as flows return to normal in the future. The consequences for soil ecology are likely to be minor in the soil layers although the timescale for soil recovery in the surface soils from acidification will probably be quite rapid as indicated in the contaminant and metalloid dynamics experiments. The small changes in most samples noted in the contaminant and metalloid dynamics tests suggests that an ***insignificant*** rating is best applied for consequence as short-term damage to soil water chemistry is considered likely. This provides a *risk rating for soil acidification* of **low**. A rating for surface water acidification will depend on surface and sub-surface hydrology. The highest risk is likely to be during low flows where the soil to water ratio is high: acidity will be most concentrated. The risk to surface water acidification is considered lowest where high flows are available to both dilute acidity and transport acidity downwards in the soil profile. Surface water acidification is likely to be lower than soil acidification, due to limited transport and buffering reactions at the soil/water interface, therefore an ***insignificant*** categorisation is also given for consequence. The *risk to surface water acidification* is therefore likely to be **low**.

4.2.2. Risks associated with contaminant mobilisation

The risks of metal and metalloid mobilisation are controlled primarily by metal abundance and availability, geochemical controls on speciation and transport mechanisms. The master variables pH and Eh exert a direct major influence on the solubility of individual metals and metalloids and minerals such as iron (Fe), iron (Fe) and manganese (Mn) oxides and hydroxides which are important sorbents of metal and metalloids species. The low acidification hazard due to the oxidation of sulfide minerals means that metals and metalloids are not likely to be present at high concentrations. Reduction processes may lead to reincorporation of metals and metalloids into sulfide minerals (following sulfate reduction), but at intermediate redox potentials mobility may be high where iron (Fe) and manganese (Mn) are soluble. The reactive metals results attest to the limited availability and mobility of a number of metals, the exceptions being manganese (Mn) and to a lesser degree aluminium (Al) and iron (Fe). The release of iron (Fe) and manganese (Mn) during a return to reducing conditions in the dried soils suggests that these metals as well as a number of metals and metalloids (Table 3-2) pose a potential hazard to soil and surface waters. These metalloids are mobile at high pH and their increase over the 35 day period suggests that these pose a significant hazard. Aluminium (Al) was also high but it was high mainly in the sub-surface acidic soil layer. Aluminium (Al) is likely to precipitate rapidly as pH increases during any transport to the surface layers and is unlikely to be impacted by a return to reducing conditions since it is not redox-sensitive.

Although the timescales cannot be assessed with existing information, the data suggest that metal availability is significant for some metals. Nevertheless, although the ANZECC/ARMCANZ environmental protection guidelines were breached for a number of metals, many were only slightly above these guideline values. Comparisons with other studies (e.g. Nelwart Lagoon, Shand *et al.* 2010), suggest that at the pH levels of the surface layers after 35 days of the contaminant and metalloid mobilisation tests, reductive processes may occur rapidly once initiated, and soil recovery may be rapid. Taking into account the metal mobility assessed, as well as mobilisation in the sub-surface soils for a number of metals, a ***minor*** rating is applied for consequence. This provides a risk rating for contaminant mobilisation in soils of **medium** (Table 4-4).

A rating for surface water impacts from metals and metalloids will depend on surface and sub-surface hydrology. The slightly acidic to circumneutral pH values in this study, however, means that longer term impacts are unlikely. Chemical reactions with soils and interactions at the soil/water interface are likely to diminish any minor hazards from metal flux. The highest risk is likely to be during low flows where the soil to water ratio is high: metals will be most concentrated. The risk to surface metal and metalloid flux is considered lowest where high flows are available to both dilute metal and metalloid concentrations and transport these downwards in the soil profile. Due to enhanced mobility of metalloids at higher pH, the hazard cannot be assumed to be insignificant with the limited time series data available in this study, hence an ***insignificant to minor*** rating for consequence is applied. The risk to surface waters from metal mobilisation is therefore considered to be **low to medium** (Table 4-4).

4.2.3. Risks associated with de-oxygenation

Monosulfidic materials are considered the main cause of deoxygenation risk in acid sulfate soils. Monosulfidic black ooze was not identified in the wetland during the Phase 1 survey (Grealish *et al*. 2011). The water soluble sulfate concentrations were, however, high and above the trigger value for monosulfidic black ooze formation (MDBA 2010). One surface sample (SRE 2.1) with a water soluble sulfate concentration of 1980 mg kg^{-1} was selected for monosulfide formation potential analysis.

The hazard for monosulfide formation potential is ‘No hazard’ due to the concentration of acid volatile sulfide (S_{AV}) being less than detection limit of 0.01% S. The consequence of deoxygenation is considered to be ***insignificant*** due to the lack of an identified hazard. As such, the risk associated with deoxygenation was determined to be **low** as there was no identified hazard associated with monosulfide formation and no evidence of monosulfides either in the wetland at the time of the Phase 1 field survey or forming during laboratory experiments (Table 4-4).

Table 4-4 Summary of risks associated with acid sulfate soil materials in the Swan Reach Complex wetland.

Acidification Risk		Contaminant mobilisation		Deoxygenation
Soil	*Water*	*Soil*	*Water*	
Low	Low	Medium	Low-Medium	Low

5. BROAD ACID SULFATE SOIL MANAGEMENT OPTIONS

The options available for rehabilitation of inland waterways containing acid sulfate soils has recently been reviewed (Baldwin & Fraser 2009) and incorporated into the *National guidance on managing acid sulfate soils in inland aquatic ecosystems* (EPHC & NRMMC 2011; see Table 5-1). The national guidance document provides a hierarchy of management options for managing acid sulfate soils in inland aquatic ecosystems including:

1. *Minimising the formation of acid sulfate soils in inland aquatic ecosystems.*
2. *Preventing oxidation of acid sulfate soils, if they are already present in quantities of concern or controlled oxidation to remove acid sulfate soils if levels are a concern but the water and soil has adequate neutralising capacity.*
3. *Controlling or treating acidification if oxidation of acid sulfate soils does occur.*
4. *Protecting connected aquatic ecosystems/other parts of the environment if treatment of the directly affected aquatic ecosystem is not feasible.*
5. *Limited further intervention.*

In designing a management strategy for dealing with acid sulfate soils in affected inland wetlands, other values and uses of a wetland need to be taken into account to ensure that any intervention is compatible with other management plans and objectives for the wetland. The medium conservation status for this wetland suggests that the management responses required should align with those suggested following the risk assessment ratings (Table 4-3).

A number of options for treating acid sulfate soils in inland wetlands have been identified (see Table 5-1). By far the best option is not to allow acid sulfate soils to build up in the first instance. This requires removing the source of sulfate from the wetland, for example, by lowering saline water tables and/or introducing frequent wetting and drying cycles to the wetland so that the amount of sulfidic material that can build up in the sediments during wet phases is limited, hence reducing the likely environmental damage (acidification, metal release or deoxygenation) that would occur as a consequence of drying.

If acid sulfate soils have formed, prevention of oxidation, usually by keeping the sediments inundated to sufficient depth, is a potential strategy. If oxidation of acid sulfate soils occurs and the sediment and/or water column acidifies, neutralisation may be necessary.

The medium risks identified in this study are due to soil and water contaminant mobilisation. The likelihood of water refilling the wetland is high as flows return to normal levels. The limited number of case studies on refilling wetlands makes prediction of risk difficult in terms of determining whether reversible or irreversible damage is likely to occur. However, at the pH’s observed, the increase in pH over the 35 day period and limited contaminant mobilisation imply that any risks are likely to be localised in nature.

As the wetland has previously dried and undergone oxidation, management options 1 and 2 in Table 5-1 are not relevant to the current study, although minimising further oxidation could have been an option prior to recent high flows down the River Murray. Treatment options currently remain a viable option should water quality impacts e.g. acidification of surface water and/or high metal concentrations be seen. Since the risks are scenario dependent, it is recommended that surface water monitoring be undertaken at this wetland. Based on the data from this study and elsewhere (Shand *et al.* 2010), it is likely that soil recovery will be relatively rapid as pH is relatively high and increased in experiments in this study over time. The impacts on surface and sub-surface ecosystems are not well understood and are worthy of further work, particularly long term impacts on ecosystem functionality and diversity.

Table 5-1 Summary of management options and possible activities, from EPHC & NRMMC (2011).

Management Objective	Activities
1. Minimising the formation of acid sulfate soils in inland aquatic ecosystems	**Reduce secondary salinisation through:** • Lowering saline water tables • Maintaining the freshwater lens between saline groundwater and the aquatic ecosystem • Stopping the delivery of irrigation return water • Incorporating a more natural flow regime.
2. Preventing oxidation of acid sulfate soils or controlled oxidation to remove acid sulfate soils	**Preventing oxidation:** • Keep the sediments covered by water • Avoid flow regimes that could re-suspend sediments. **Controlled oxidation:** • Assess whether neutralising capacity of the sediments and water far exceeds the acidity produced by oxidation • Assess the risk of deoxygenation and metal release. Monitor intervention and have a contingency plan to ensure avoidance of these risks.
3. Controlling or treating acidification	• Neutralise water column and/or sediments by adding chemical ameliorants • Add organic matter to promote bioremediation by micro-organisms • Use stored alkalinity in the ecosystem.
4. Protecting adjacent or downstream environments if treatment of the affected aquatic ecosystem is not feasible	• Isolate the site • Neutralise and dilute surface water • Treat discharge waters by neutralisation or biological treatment.
5. Limited further intervention	• Assess risk • Communicate with stakeholders • Undertake monitoring • Assess responsibilities and obligations and take action as required.

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APPENDICES

APPENDIX 1 REACTIVE METALS DATA

Swan Reach Complex wetland

Sample	Depth	Analysis	Ag*	Al	As	Cd*	Co	Cr*	Cu	Fe	Mn	Ni	Pb	Sb*	Se*	V	Zn
SRE 1.1	0-3	a	10	302	0.81	51	2.4	101	4.3	334	115	3.4	3.4	< 1.7	31	9.3	3.2
		b	11	321	0.77	50	2.5	101	4.4	309	124	3.4	3.4	< 1.7	31	9.7	3.3
SRE 1.2	(3-25)	a	9.6	269	0.66	48	0.53	36	4.0	331	79	3.2	3.1	< 1.8	33	8.5	1.9
		b	9.8	271	0.68	49	0.51	36	4.1	320	80	3.2	3.0	< 1.8	31	8.6	1.8
SRE 2.1	0-5	a	10	314	0.80	52	2.5	94	4.1	628	207	3.5	3.6	< 1.7	39	9.8	3.4
		b	11	313	0.83	56	2.8	94	3.9	503	226	3.7	3.8	< 1.7	39	9.7	3.7
SRE 8.1	0-10	a	8.2	345	1.1	50	4.5	157	4.1	441	178	3.6	3.6	3.3	44	7.5	6.0
		b	7.5	336	1.1	49	4.2	150	3.9	438	175	3.5	3.6	< 1.6	34	7.5	5.1

Units are mg kg^{-1} unless indicated otherwise as below

* Units are in µg kg^{-1}

< value is below detection limit

APPENDIX 2 CONTAMINANT AND METALLOID DYNAMICS DATA

Swan Reach Complex wetland

Sample	Day	Depth cm	Analysis	Eh mV	EC µS/cm	pH	Ag µg/L	Al mg/L	As µg/L	Cd µg/L	Co µg/L	Cr µg/L	Cu µg/L	Fe mg/L	Mn µg/L	Ni µg/L	Pb µg/L	Sb µg/L	Se µg/L	V µg/L	Zn µg/L
SRE 1.1	1	0-3	a	369	198	6.29	<0.01	0.20	1.0	<0.02	0.66	<0.2	<2	0.11	103	1.4	<0.6	<1	0.27	3.0	<0.7
			b	369	189	6.27	<0.01	0.27	1.1	<0.02	0.59	<0.2	<2	0.14	80	1.4	<0.6	<1	0.27	3.9	<0.7
	7		a	424	96	6.29	<0.01	0.09	5.2	<0.05	3.9	0.20	6.2	0.23	213	6.3	<0.5	<0.7	0.30	3.3	1.2
			b	419	99	6.36	<0.01	0.23	4.4	<0.05	4.4	0.20	5.8	0.29	244	6.2	<0.5	<0.7	0.30	2.8	0.80
	14		a	294	206	6.50	<0.01	0.85	17	<0.06	8.5	0.20	8.4	1.0	466	11	<0.8	<1	0.52	4.4	1.8
			b	289	181	6.39	<0.01	0.52	15	<0.06	5.8	0.20	14	0.56	242	9.3	<0.8	<1	0.40	6.0	1.8
	35		a	114	121	6.37	<0.01	<0.05	31	<0.02	9.0	0.57	1.1	6.4	1038	9.3	<0.8	<4	0.32	6.4	<0.6
			b	89	131	7.04	<0.01	<0.05	28	<0.02	5.7	0.52	1.3	6.5	1096	5.0	<0.8	<4	0.35	7.5	<0.6
SRE 1.2	1	(3-25)	a	364	135	5.48	<0.01	0.30	0.36	<0.02	0.09	<0.2	<2	0.15	36	0.46	<0.6	<1	0.15	2.0	<0.7
			b	369	147	5.78	<0.01	0.83	0.31	<0.02	0.09	<0.2	<2	0.41	37	0.51	<0.6	<1	0.14	2.2	<0.7
	7		a	414	122	4.11	<0.01	<0.05	<0.2	<0.05	0.05	<0.1	<0.2	<0.1	28	0.64	<0.5	<0.7	0.09	0.70	0.40
			b	414	121	4.31	<0.01	0.08	<0.2	<0.05	0.05	<0.1	0.40	<0.1	27	0.72	<0.5	<0.7	0.09	0.90	0.40
	14		a	319	189	5.78	<0.01	0.14	<0.3	<0.03	0.10	<0.1	0.90	<0.1	18	0.96	<0.4	<0.6	0.12	2.8	0.90
			b	339	196	6.85	<0.01	0.07	<0.3	<0.03	0.10	<0.1	0.90	<0.1	18	1.1	<0.4	<0.6	0.14	2.5	0.90
	35		a	219	99	6.97	<0.01	0.38	<0.3	<0.01	0.07	0.16	0.71	0.25	5.3	0.70	<0.4	<2	0.07	2.1	0.48
			b	224	91	6.20	<0.01	0.28	<0.3	<0.01	0.05	0.14	0.62	0.18	5.1	0.76	<0.4	<2	0.06	2.2	0.57
SRE 2.1	1	0-5	a	364	256	5.82	<0.01	<0.05	0.67	<0.04	2.8	<0.2	<1	<0.1	458	2.6	<0.06	<1	0.33	1.2	1.2
			b	369	229	6.04	<0.01	0.07	0.65	<0.04	2.0	<0.2	<1	<0.1	342	2.0	<0.06	<1	0.39	1.6	<0.8
	7		a	384	246	6.17	<0.04	<0.05	14	<0.3	9.7	<0.5	<1	4.1	1130	8.8	<3	<4	0.60	2.5	2.0
			b	379	236	6.15	<0.04	<0.05	15	<0.3	9.9	<0.5	<1	4.2	1144	9.2	<3	<4	0.40	2.5	<2
	14		a	229	326	6.27	<0.01	<0.05	26	<0.2	12	<0.5	<2	6.1	1637	14	<2	<3	0.70	4.0	4.0
			b	229	347	6.33	<0.01	<0.05	26	<0.2	13	<0.5	<2	5.8	1648	14	<2	<3	0.60	4.0	4.0
	35		a	39	160	6.16	<0.01	<0.05	21	<0.05	8.5	<0.5	<1	11	1964	4.8	<2	<10	0.20	3.7	2.3
			b	19	162	6.55	<0.01	<0.05	24	<0.05	6.7	<0.5	<1	11	1923	3.7	<2	<10	0.21	2.7	<2
SRE 8.1	1	0-10	a	369	292	5.55	<0.01	0.05	1.5	<0.04	4.9	<0.2	1.7	<0.1	403	3.5	<0.06	<1	0.35	4.0	1.9
			b	369	302	5.86	<0.01	<0.05	1.5	<0.04	4.6	<0.2	1.6	<0.1	363	3.2	<0.06	<1	0.32	4.2	2.0
	7		a	294	222	6.28	<0.02	0.05	20	<0.1	9.4	0.80	1.2	5.8	786	10	<1	<1	0.48	10.0	0.80
			b	279	202	6.31	<0.02	0.09	17	<0.1	7.9	0.60	1.6	3.8	602	9.0	<1	<1	0.42	9.6	0.80
	14		a	64	364	6.57	<0.01	<0.05	18	<0.06	7.8	0.80	1.2	5.0	904	9.3	<0.8	<1	0.44	13	0.60
			b	54	337	6.59	<0.01	<0.05	20	<0.06	7.2	0.80	1.2	5.1	820	9.1	<0.8	<1	0.40	13	1.2
	35		a	14	158	6.54	<0.01	<0.05	17	<0.02	2.7	1.5	0.44	6.4	991	7.4	<0.8	<4	0.38	12	2.8
			b	-6.0	160	6.75	<0.01	<0.05	18	<0.02	2.9	1.4	<0.4	6.5	1028	7.6	<0.8	<4	0.39	13	0.22

< value is below detection limit

APPENDIX 3 MONOSULFIDE FORMATION POTENTIAL DATA

Swan Reach Complex wetland

MBO Formation Potential (MBO FP) - DAY 0

IRON DATA

DAY 0

				Total Reactive Fe (mg/kg)				**Fe(II) (mg/kg)**				**Eh (mV)**				**pH**			
Sample No.	org	Site Name	**Site ID**	Replicate 1	Replicate 2	**Mean**	+/-	Replicate 1	Replicate 2	**Mean**	+/-	Replicate 1	Replicate 2	**Mean**	+/-	Replicate 1	Replicate 2	**Mean**	+/-
27	csiro	Swan Reach Complex	SRE2.1	4686	5183	**4934**	*248*	582	633	**608**	*26*	429	447	**438**	*9*	5.67	5.53	**5.60**	0.07
32	-	Blank	-	0.3	0.1	**0.2**	*0.1*	<0.1	<0.1	**<0.1**	*<0.1*	183	186	**185**	*2*	6.17	6.10	**6.14**	0.04

MBO Formation Potential (MBO FP) - Week 7

				AVS (%S)				**Pyrite (%S)**				**ES (%S)**				**pH**				**Eh (mV)**				**Aqueous Sulfide (µg/L)**			
Sample No.	**org**	**Site Name**	**Site ID**	Replicate 1	Replicate 2	**Mean**	*+/-*	Replicate 1	Replicate 2	**Mean**	*+/-*	Replicate 1	Replicate 2	**Mean**	*+/-*	Replicate 1	Replicate 2	**Mean**	*+/-*	Replicate 1	Replicate 2	**Mean**	*+/-*	Replicate 1	Replicate 2	**Mean**	*+/-*
27	csiro	Swan Reach Complex	SRE2.1	0.00	0.00	**<0.01**	*<0.01*	0.02	0.02	**0.02**	*<0.01*	0.00	0.00	**<0.01**	*<0.01*	4.66	4.55	**4.61**	*0.06*	75	257	**166**	*91*	17.9	32.3	**25.1**	*7.2*
32	csiro	Blank														5.45	5.42	**5.44**	*0.02*	325	338	**331**	*6*	< 0.1	< 0.1	< 0.1	<0.1

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