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Assessment of Acid Sulfate Soil Materials (Phase 2) Nigra Creek (12294), South Australia

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Report to the Murray-Darling Basin Authority

Water for a Healthy Country Flagship Report series ISSN: 1835-095X

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Citation: Shand P, Grocke S, Baker AK, Smith L, Fiebiger C & Cozens G 2011. Assessment of Acid Sulfate Soil Materials (Phase 2) Nigra Creek (12294). CSIRO: Water for a Healthy Country National Research Flagship

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Cover Photograph:

Nigra Creek (12294)
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ACKNOWLEDGMENTS

This work was funded by the Murray-Darling Basin Authority. We are particularly grateful to Rob Kingham and his team from the Murray-Darling Basin Authority for support throughout this project. We would like to thank the many landholders who allowed access through their properties and contributed their local knowledge about the wetland and history of the areas.

The following CSIRO staff members are thanked for their outstanding turnaround of analytical data: Julie Smith, Claire Wright, John Gouzos and Michelle Smart. We are grateful to Rob Kingham, Lucy Paterson and Lea Weekes-Randall from the Murray-Darling Basin Authority and Warren Hicks from CSIRO for constructively reviewing the report.

EXECUTIVE SUMMARY

An initial Phase 1 acid sulfate soil investigation of Nigra Creek (12294) during March 2010 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 Nigra Creek (12294) to determine the nature, severity and the specific risks associated with acid sulfate soil materials.

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 below the sediment quality guidelines (SQG) and soil ecological investigation levels (EIL) for those elements where guidelines are available. Although the metals and metalloids 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 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 Nigra Creek (12294), no contaminants were assigned a moderate or high hazard as no concentrations exceeded ANZECC/ARMCANZ environmental protection guidelines by more than 10 or 100 times. However, aluminium (Al), cobalt (Co), iron (Fe) and zinc (Zn) were above the guideline values.

Nigra Creek (12294) has been classified as high conservation status by the SA Murray-Darling Basin Natural Resources Management Board (Miles *et al.* 2010). The wetland was largely full at the time of sampling with subaqueous soils studied. The tests undertaken for the subaqueous soils in this wetland are difficult to extrapolate to a case where the soils have dried and oxidised, as the generated acidity depends on a number of complex factors. In addition, it is not possible to predict the potential impacts of metal and metalloid release, as these may be present in reduced minerals such as pyrite and thus not easily released until oxidation occurs. 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 **medium** risk rating due to ***acidification*** and a **medium *contaminant mobilisation*** 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 **medium** risk rating for ***acidification*** and **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.

In designing a management strategy for dealing with acid sulfate soils in Nigra Creek (12294), 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 data suggest that management options for acidification should be considered during any future disturbance to the soils. The data from the reactive metals and contaminant and metalloid dynamics tests show that metal and metalloid mobility are a hazard particularly

during wetting and drying periods where contaminants may be cycled between oxidised (oxide/oxyhydroxide) and reduced (sulfide) minerals. This data helps to provide information on impacts related to contaminant mobilisation. Due to the medium risks to the wetland values associated with acidification and contaminant mobilisation in Nigra Creek (12294), a monitoring program is strongly 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 Nigra Creek (12294) Phase 1 assessment (Grealish *et al.* 2010) and the priority ranking criteria adopted by the Scientific Reference Panel of the MDB ASSRAP (see Table 1-1), Nigra Creek (12294) was selected for Phase 2 detailed assessment. The Phase 1 assessment sampled from 4 sites along two transects (Figure 1-1). The Phase 1 assessment identified no high priority sites based on the presence of sulfuric materials, 3 high priority sites based on the presence of hypersulfidic materials, 1 high priority site based on hyposulfidic materials with SCR ≥ 0.10% and 4 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 1 site (12294_1) identified in the Phase 1 assessment (Grealish *et al*. 2010).

The soils comprised clays and clays overlying loamy sands. Net acidities were very variable, varying overall from 12 to 339 mol H^+/tonne. The wetland was poorly buffered, with ANC being zero in all samples.

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 ≥ 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.

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 Nigra Creek (12294) is presented in Table 1-3.

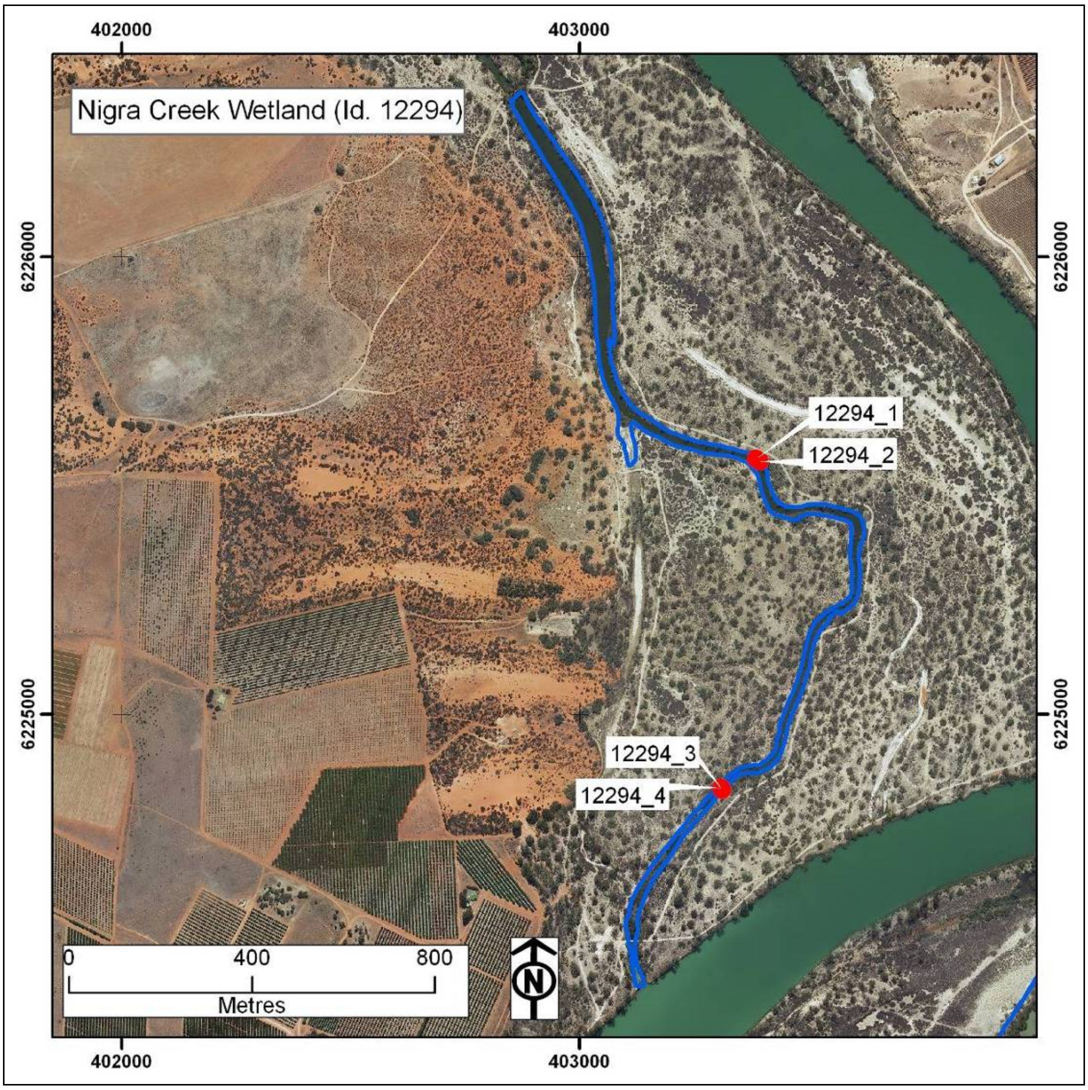

Figure 1-1 Nigra Creek (12294) aerial photograph with Phase 1 sampling sites identified.

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 Nigra Creek (12294) samples analysed for Phase 2 assessment.

Soil Laboratory Test	Nigra Creek (12294) samples	Depth of sample (cm)	Number of samples analysed
Reactive metals	12294_1.1	0-5	2
	12294_1.2	5-10	
Contaminant and metalloid dynamics	12294_1.1	0-5	2
	12294_1.2	5-10	
Monosulfide formation potential	-	-	0
Mineral identification by X-ray diffraction (XRD)	-	-	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 for which guidelines exist (Table 3-1). The highest concentrations were for iron (Fe).

Table 3-1 Nigra Creek (12294) reactive metals data.

Concentrations in mg kg^{-1}, and µg kg^{-1} as indicated by asterisk.

Sample	Ag*	Al	As	Cd*	Co	Cr*	Cu	Fe	Mn	Ni	Pb	Sb*	Se*	V	Zn
12294_1.1	1.0	169	0.37	14	1.2	36	1.3	468	53	2.4	1.4	< 25	6.5	4.5	2.7
12294_1.2	1.1	148	0.50	11	0.77	38	1.7	612	37	1.8	1.4	< 24	5.9	3.9	1.2
[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 two Nigra Creek (12294) 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). For copper (Cu), the detection limit was higher than ANZECC/ARMCANZ environmental guideline values.

Table 3-2 Summary of contaminant and metalloid dynamics data

Parameter	units	ANZECC Guidelines	Nigra Creek		
			Min.	Median	Max.
pH		6.5-8.0	**4.5**	**5.1**	**6.3**
EC*	µS cm^{-1}	2200	63	105	137
Eh	mV	-	57	363	424
Ag	µg l^{-1}	0.05	<0.01	<0.01	<0.02
Al[A]	mg l^{-1}	0.055	<0.05	<0.05	**0.11**
As[B]	µg l^{-1}	13	<0.20	<0.28	<1.1
Cd	µg l^{-1}	0.2	<0.04	<0.04	<0.08
Co	µg l^{-1}	2.8	0.06	**5.4**	**14**
Cr[C]	µg l^{-1}	1	<0.10	<0.18	<0.40
Cu[H]	µg l^{-1}	1.4	<0.20	<0.45	<2.0
Fe[I]	mg l^{-1}	0.3	<0.10	<0.10	**1.1**
Mn	µg l^{-1}	1700	261	503	1067
Ni[H]	µg l^{-1}	11	0.22	3.5	10
Pb[H]	µg l^{-1}	3.4	<0.06	<0.45	<1.4
Sb	µg l^{-1}	9	<0.50	<1.0	<7.0
Se	µg l^{-1}	11	<0.02	<0.03	<0.05
V	µg l^{-1}	6	<0.08	<0.13	0.79
Zn[H]	µg l^{-1}	8	<2	3.3	**13**

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 the soil materials was acidic for both samples, and for the duration of the 35 day tests. As the pH was lower than the pH measured in Phase 1, it appears that some oxidation of the samples has occurred during or prior to the tests. It was significantly less than the lower ANZECC/ARMCANZ environmental protection guidelines, although the surface sample (12294_1.1) increased to pH 6.25 on day 35 (Figure 3-1).

The Eh showed a decrease over the day 35 period in both samples, from oxidising initially on day one (Figure 3-1) to moderately reducing, especially for the surface sample 12294_1.1. Salinities, as indicated by the SEC, were low (Figure 3-1), and there was a trend of decreasing SEC, although relatively small.

Iron (Fe) concentrations remained low for days 1 to 14, but increased on day 35, especially in the surface soil sample, where concentrations were much higher than the ANZECC/ARMCANZ environmental protection guideline. The data are consistent with pH-Eh relationships for the stability fields of iron (Fe). Manganese (Mn) also increased with time in both samples, but concentrations were well below the ANZECC/ARMCANZ environmental protection guideline value (Figure 3-2). The increase for manganese (Mn) is consistent with the pH-Eh relationships, where manganese (Mn) generally becomes soluble during the transition from oxidising to reducing conditions at higher Eh than iron (Fe).

Aluminium (Al) concentrations were low in most samples, increasing only on day 35 in the surface sample 12294_1.1 to 0.1 mg l^{-1}(Figure 3-1). Aluminium solubility should be higher at the low pH values, and it appears that the samples have not been acidic for long enough to dissolve aluminosilicate minerals such as clays.

Arsenic (As) concentrations were very low and well below ANZECC/ARMCANZ environmental protection guideline values in the two samples (Figure 3-1). Cobalt (Co) was initially above the ANZECC/ARMCANZ environmental protection guideline value, increasing to day 14 before decreasing, thus showing contrasting behaviour to iron (Fe) and manganese (Mn). Nickel (Ni) and zinc (Zn) both displayed similar patterns to cobalt (Co).

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 is shown on Figure 3-4. The metals cobalt (Co), nickel (Ni) and zinc (Zn) showed a tendency for higher concentrations at lower pH. Iron (Fe) and manganese (Mn) appear to be more closely correlated with the Eh of the solutions.

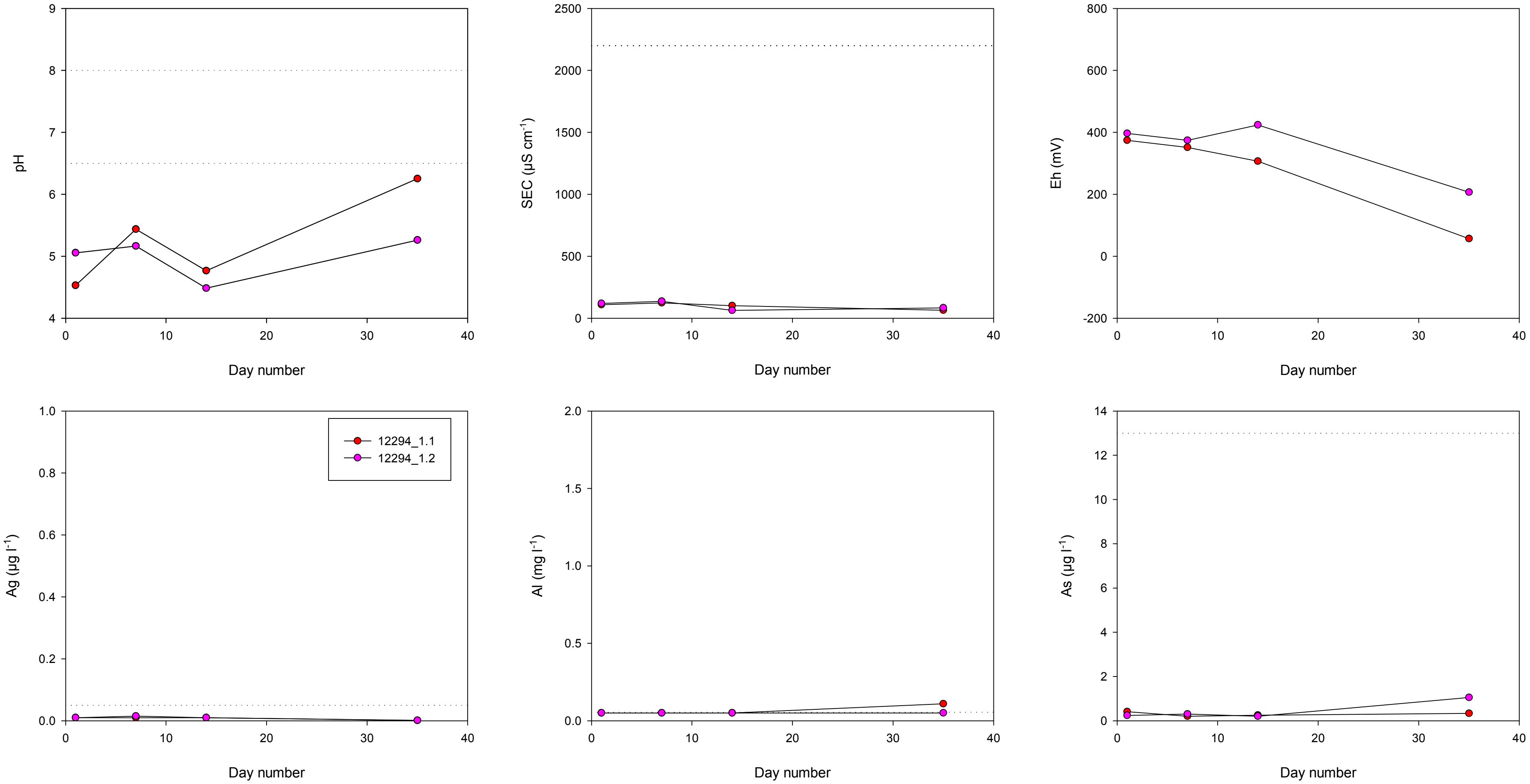

Figure 3-1 Contaminant and metalloid dynamics results for Nigra Creek (12294) soil materials for pH, SEC, Eh, silver (Ag), aluminium (Al) and arsenic (As).

Note: silver (Ag) and arsenic (As) in all samples were < detection limit, data represent detection limits which vary according to required dilutions.

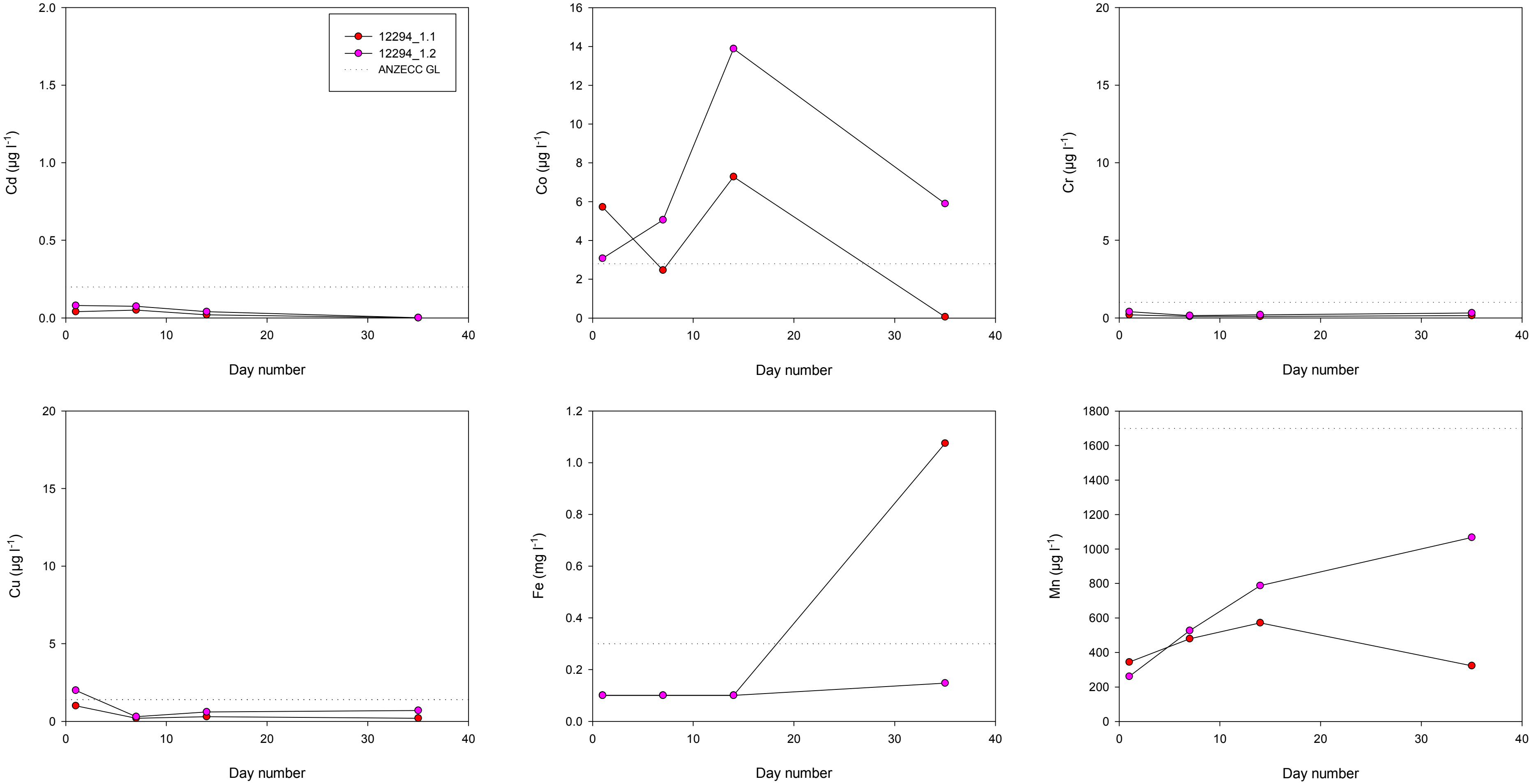

Figure 3-2 Contaminant and metalloid dynamics results for Nigra Creek (12294) soil materials for cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe) and manganese (Mn).

Note: cadmium (Cd), chromium (Cr) and copper (Cu) were < detection limit, data represent detection limits which vary according to required dilutions.

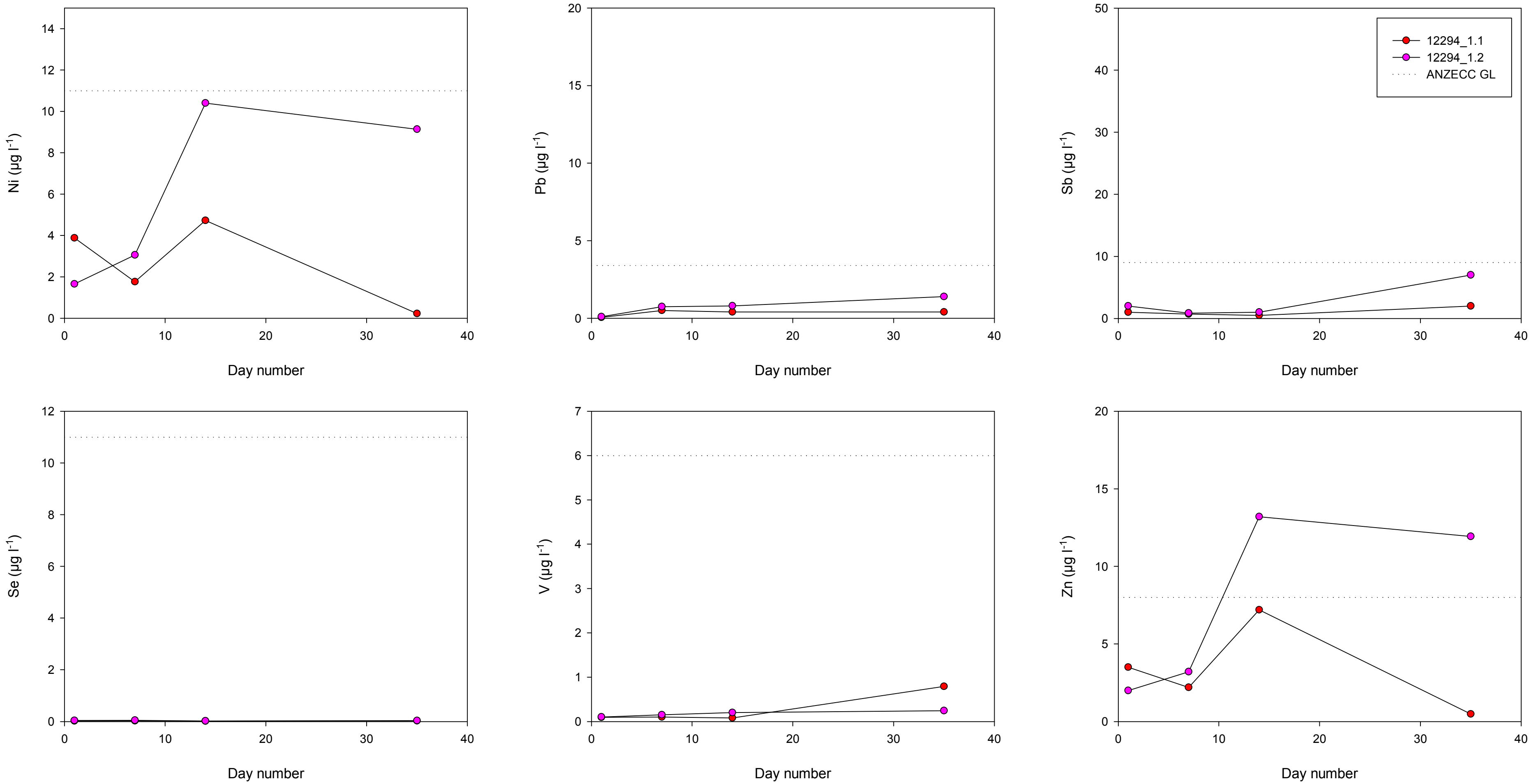

Figure 3-3 Contaminant and metalloid dynamics results for Nigra Creek (12294) soil materials for nickel (Ni), lead (Pb), antimony (Sb), selenium (Se), vanadium (V) and zinc (Zn).

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

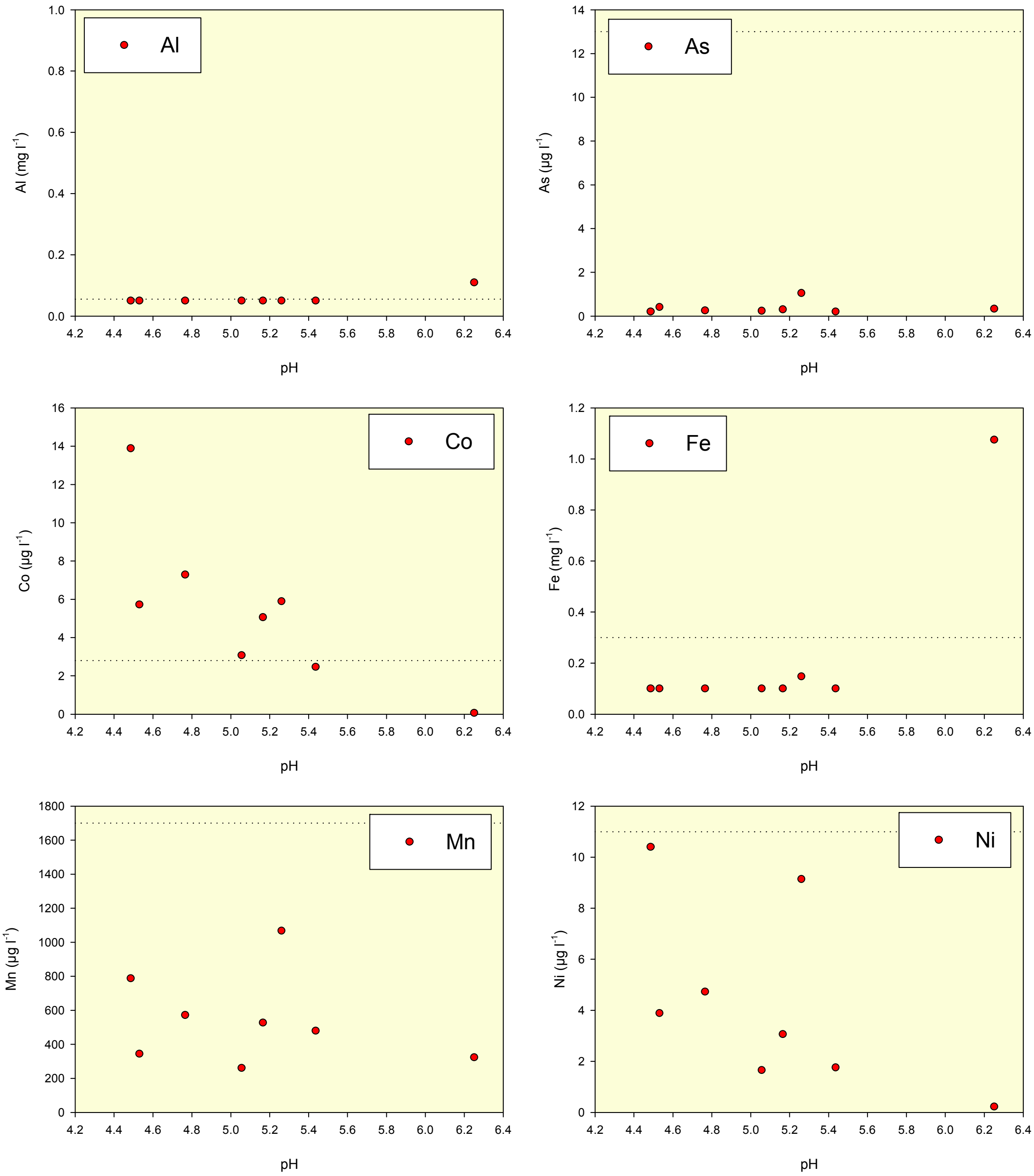

Figure 3-4 Selected trace elements plotted against pH.

3.1.3. Monosulfide formation potential data

No samples were selected from this wetland for monosulfide formation potential studies, as monosulfidic black ooze was not identified at this site.

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 generally below sediment quality guidelines and soil ecological investigation level values for those elements for which guideline values are available (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 moderately acidic pH on day one of the contaminant and metalloid dynamics tests, increasing slightly by day 35, but were below the pH range for ANZECC/ARMCANZ environmental protection guideline values (Figure 3-1). The pH in these soils appears to be poorly buffered in agreement with the lack of buffering capacity (ANC) of the soil materials (Grealish *et al*. 2010). It does appear that the soil pH has decreased either prior to or during the experiments as the pH after 24 hours was lower than the soil pH measured in Phase 1 of the study. This is supported by the Eh results on day 1 which were slightly oxidising and decreased significantly over time.

The concentrations of metals and metalloids were generally very low, the exceptions being iron (Fe) and manganese (Mn) which increased as expected with decreasing Eh. The other metals of note are the transition metals cobalt (Co), nickel (Ni) and zinc (Zn) which showed a general increase up to day 14, before subsequently decreasing. Their solubility appears to be more related to pH, and the decrease may be related to the slight increase in pH over time.

The data suggest that metal concentrations are typically low in the soil pore-waters, consistent with the reducing (pyrite present) conditions and moderately acidic to circumneutral pH of the waters. Those metals initially detected in the more acidic waters are probably related to release during partial oxidation (possibly during the experiment) of the soils, but the decrease in pH has not occurred sufficiently long for aluminium to dissolve. Metal and metalloid release during drying and oxidation of the wetland soils cannot be predicted from the contaminant and metalloid dynamics tests as any metals and metalloids will be locked up in reduced minerals such as pyrite and/or monosulfides.

The degree to which samples exceed guideline concentrations has been used to assign a degree of hazard (Table 3-3). The data are consistent with the pH and assumed Eh conditions of these subaqueous soils. The higher pH will limit the solubilities of most trace cation metals, but this will depend to a degree on redox conditions. Any assessment on the degree of risk should take the abovementioned information into account.

Table 3-3 Summary of the degree of hazard associated with the measured contaminant and metalloid concentrations in Nigra Creek (12294).

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

Note: The detection limit for Cu was above the ANZECC/ARMCANZ environmental protection guideline in one sample due to dilution, and is therefore likely to be in the low or no hazard classification.

The monosulfide formation potential test assists in determining the propensity for monosulfides to form during future inundation. Water soluble sulfate concentrations were moderate to high in most samples (varying from 138 to 1244 mg kg^{-1}). Monosulfidic black ooze was not identified in the field and therefore not considered to be a hazard for this wetland.

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.

Nigra Creek (12294) has been classified as high conservation status by the SA Murray-Darling Basin Natural Resources Management Board (Miles *et al.* 2010).

4.2.1. Risks associated with acidification

The low to high net acidities in the soils of Nigra Creek (12294) (12 to 339 mol H^+/tonne) suggest that, although pH at the time of sampling was not a hazard, there is potential for the wetland soils to acidify if dried. This was also indicated in the ageing pH work completed in Phase 1 (Grealish *et al*. 2010), where 5 of 11 samples aged to pH < 4 (pH 2.74 to 3.99). The contaminant and metalloid dynamic tests were completed on largely unoxidised material and therefore provide little information to inform the risk from the overall acidification hazard. The decrease in pH, however, supports the Phase 1 data suggesting that the hazard may be realised quickly. Taking into account the information above, the acidification hazard is therefore, considered to be high.

The wetland is close to the river, and permanently connected to the river at the southern end, and to Schillers Lagoon at the northern end. It also has a structure and can, therefore, be managed. The likelihood of disturbance is therefore selected as ***likely***, as the wetland has been isolated a number of times over the past few years (Grealish *et al*. 2010). The consequences for soil ecology from acidification are likely to be significant if the wetland is allowed to dry, for both surface and sub-soil waters as they are generally poorly buffered with low ANC. The pH results from the contaminant and metalloid dynamics experiments suggest that an overall rating for consequence would be ***moderate*** if oxidation of the soils occurred. This provides a *risk rating for soil acidification* of **medium** if the soils are dried. A rating for surface water acidification will most likely also be significant. The smallest risk to surface water acidification would be where high flows were available to both dilute any locally derived acidity and induce transport of acidity downwards in the soil profile where ANC is more abundant. In the case of Nigra Creek (12294), acidification of surface water would potentially be ***moderate***, and therefore the *risk to surface water acidification* is therefore classed as **medium.**

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 medium acidification hazard due to the oxidation of sulfide minerals means that metals and metalloids mobility is possible. 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 iron (Fe) and manganese (Mn). A number of elements were above the ANZECC/ARMCANZ environmental protection guidelines, including aluminium (Al), cobalt (Co) and zinc (Zn). The dissolved toxic trivalent form of aluminium (Al^{3+}) is not likely to be present, except under acidic conditions ($pH < 5.5$), and aluminium is unlikely to be impacted by a return to reducing conditions since it is not redox-sensitive. It is not known how long the Eh would continue to decrease, but further decreases would allow the reductive dissolution of iron (Fe) and manganese (Mn) oxyhydroxides and any associated adsorbed metals and metalloids.

Although the timescales cannot be assessed with existing information, the data suggest that metal availability is significant for a number of metals and metalloids. 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 metalloid mobilisation tests, reductive processes may occur rapidly once initiated if there is sufficient organic matter available, and soil recovery may be rapid. The risks of metal and metalloid mobilisation are dependent on the degree of oxidation of soils and the depth to which oxidation occurs, hence related to the time of drying. For the soils of Nigra Creek (12294), the risks due to rewetting cannot be fully assessed as the soils were largely un-oxidised, although the wetland had dried previously (Grealish *et al.* 2010) and the availability may relate to previous oxidation events.

The main solutes identified as hazards were aluminium (Al), cobalt (Co), iron (Fe) and zinc (Zn) as these exceeded ANZECC/ARMCANZ environmental protection guidelines. However, the risks from oxidation of these subaqueous soils, as with the risk of acidification, is difficult to assess from the Phase 2 study because the metals are currently most likely locked up in reduced minerals such as pyrite or in minerals stable at the ambient pH of the soils. The partial oxidation suggested that some metals and metalloids are present. The connection to the river also means that impacts on the river are possible, especially at high flow, but under this scenario dilution effects will also be significant. Taking into account the metal and metalloid mobility assessed and limitations discussed above, a ***minor*** rating at least should be 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 also depend on the degree of drying and oxidation, and also on surface and sub-surface hydrology. The slightly acidic to circumneutral pH values by day 35 in the contaminant and metalloid dynamics tests undertaken in this study, however, means that longer term impacts for many metals are unlikely if the soils are undisturbed. The risks for surface waters will relate to those metals and metalloids discussed above, and decrease as the soils become more reducing if the Eh falls to within the stability field for sulfide minerals (due to scavenging by the sulfide minerals). Chemical reactions with soils and interactions at the soil/water interface are likely to diminish hazards from upward soil metal flux. The highest risk is likely to be following drying and 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 a ***minor*** rating for consequence is applied. The risk to surface waters from metal mobilisation is therefore considered to be **medium** (Table 4-4). The Phase 1 study sampled two surface waters and noted high concentrations above ANZECC/ARMCANZ environmental protection guidelines for some contaminants. This included the nutrients ammonium (NH_4) and phosphate (PO_4) as well as aluminium (Al), chromium (Cr), copper (Cu), iron (Fe) and zinc (Zn). The waters were very turbid and it is likely that some of these high results are due to colloidal material that passed through the filters, particularly for aluminium (Al).

4.2.3. Risks associated with de-oxygenation

Monosulfidic materials are considered the main cause of deoxygenation risk in acid sulfate soils. There was no evidence of monosulfides being present in the wetland during the Phase 1 field survey and the water soluble sulfate concentrations in the samples were low and generally below the trigger value for monosulfidic black ooze formation (MDBA 2010). The consequence from monosulfide disturbance is therefore considered to be ***insignificant*** and as such the risk rating for deoxygenation is **low** (Table 4-4).

Table 4-4 Summary of risks associated with acid sulfate soil materials in Nigra Creek (12294).

Acidification Risk		Contaminant mobilisation		Deoxygenation
Soil	*Water*	*Soil*	*Water*	
Medium	Medium	Medium	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 high 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.

Nigra Creek (12294) contained surface water at the time of sampling. The first two options in Table 5-1 provide the best options for minimising damage to ecosystem health and costs. A medium risk was identified for soil and surface water acidification, due mainly to the results obtained in Phase 1 of the wetland study (Grealish *et al*. 2010). The other risks are related to contaminant mobilisation but this could not be assessed fully with the tests completed on unoxidised soil materials.

Prevention options, including keeping the wetland fully saturated, will depend on connectivity with the river and the availability of water for this purpose. Although acidification of soils is likely to occur during drying, the impact on surface waters is strongly dependent on surface water and groundwater hydrology. This in turn dictates the rates of change of oxidation during drying and subsequent change to reducing conditions. At lower Eh, where sulfide minerals are stable, any metals and metalloids are likely to be scavenged by sulfides and be less of a risk. However, drying and oxidation may release these contaminants again, hence wetting-drying episodes will simply cycle these contaminants between different mineral phases and solution. The data from this study do not provide results to fully inform best management options, but are valuable in showing what contaminants are available and their potential impact under different management scenarios. As this is a managed wetland, management options should also consider monitoring during any managed wetting/drying regimes.

Since the risks are so scenario dependent, it is recommended that surface water monitoring be undertaken at this wetland on the re-wetting of oxidised soils, particularly because of the acidification hazard and also a number of contaminants were identified over the 35 days of the contaminant and metalloid dynamics experiments. Based on the data from this study and elsewhere (Shand *et al.* 2010), it is likely that soil recovery from any future acidification or metal and metalloid release will be relatively quick and depend on the actual pH and Eh of the soil materials prior to recovery. 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

Nigra Creek (12294)

Sample	Depth	Analysis	Ag*	Al	As	Cd*	Co	Cr*	Cu	Fe	Mn	Ni	Pb	Sb*	Se*	V	Zn
12294_1.1	0-5	a	1.8	158	0.41	17	0.92	40	1.5	534	42	1.8	1.5	< 25	7.0	4.3	2.0
		b	< 0.25	180	0.34	11	1.4	33	1.0	401	63	3.0	1.2	< 25	6.0	4.7	3.5
12294_1.2	5-10	a	0.73	157	0.53	13	0.85	39	1.6	620	38	1.7	1.4	< 24	5.9	4.1	1.3
		b	1.5	139	0.46	8.8	0.69	37	1.8	605	36	1.8	1.4	< 24	5.9	3.6	1.1

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

Nigra Creek (12294)

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
12294_1.1	1	0-5	a	374	101	4.57	<0.01	<0.05	0.29	<0.04	4.3	<0.2	<1	<0.1	306	2.8	<0.06	<1	<0.02	0.13	2.7
			b	374	117	4.49	<0.01	<0.05	0.53	<0.04	7.1	<0.2	<1	<0.1	382	5.0	<0.06	<1	<0.02	<0.06	4.3
	7		a	344	127	5.24	<0.01	<0.05	<0.2	<0.05	2.3	<0.1	<0.2	<0.1	464	1.4	<0.5	<0.7	<0.03	0.10	2.4
			b	359	120	5.63	<0.01	<0.05	<0.2	<0.05	2.7	<0.1	<0.2	<0.1	496	2.1	<0.5	<0.7	<0.03	<0.1	2.0
	14		a	304	99	4.34	<0.01	<0.05	0.20	0.02	7.9	<0.1	0.30	<0.1	571	5.3	<0.4	<0.5	0.02	<0.08	7.6
			b	309	103	5.19	<0.01	<0.05	0.30	<0.02	6.6	<0.1	0.30	<0.1	573	4.2	<0.4	<0.5	0.01	<0.08	6.8
	35		a	74	69	6.13	<0.001	0.11	0.55	<0.001	0.07	0.18	<0.2	1.5	359	0.22	<0.4	<2	0.03	1.0	0.32
			b	39	58	6.37	<0.001	0.11	0.12	<0.001	0.06	0.12	<0.2	0.61	286	0.22	<0.4	<2	0.03	0.57	0.64
12294_1.2	1	5-10	a	394	116	5.01	<0.01	<0.05	<0.2	<0.08	2.9	<0.4	<2	<0.1	246	1.5	<0.1	<2	<0.04	<0.1	<2
			b	399	122	5.10	<0.01	<0.05	0.27	<0.08	3.2	<0.4	<2	<0.1	276	1.8	<0.1	<2	<0.04	<0.1	<2
	7		a	364	127	5.40	<0.01	<0.05	<0.2	<0.05	4.1	<0.1	<0.2	<0.1	469	2.4	<0.5	<0.7	<0.03	<0.1	2.4
			b	384	147	4.93	<0.02	<0.05	<0.4	<0.1	6.0	<0.2	<0.4	<0.1	585	3.7	<1	<1	<0.06	<0.2	4.0
	14		a	414	12	4.44	<0.01	<0.05	<0.2	<0.04	16	<0.2	<0.6	<0.1	871	13	<0.8	<1	<0.02	<0.2	14
			b	434	113	4.53	<0.01	<0.05	0.20	<0.04	12	<0.2	<0.6	<0.1	705	8.2	<0.8	<1	<0.02	<0.2	13
	35		a	209	83	4.47	<0.002	<0.05	<1.5	<0.002	11	<0.45	<1	0.20	1556	15	<2	<10	<0.05	<0.35	20
			b	204	83	6.05	0.00	<0.05	<0.6	0.00	0.62	<0.18	<0.4	<0.1	578	3.7	<0.8	<4	<0.02	<0.14	3.4

< value is below detection limit

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