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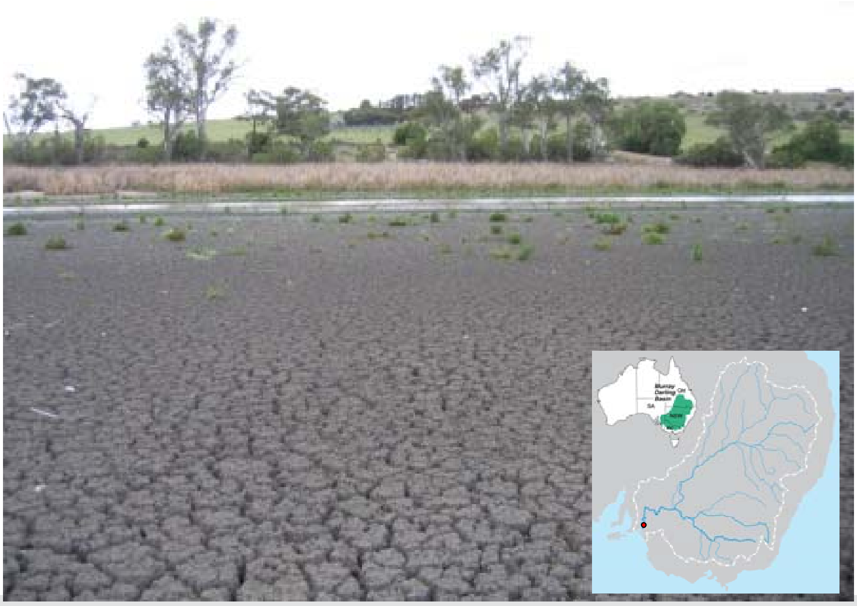

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# Assessment of Acid Sulfate Soil Materials (Phase 2)
## Forster Lagoon, South Australia

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**Cover Photograph:**

Forster Lagoon
Photographer: Gerard Grealish
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# EXECUTIVE SUMMARY

An initial Phase 1 acid sulfate soil investigation of Forster Lagoon in 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 Forster Lagoon to determine the nature, severity and the specific risks associated with acid sulfate soil materials. The wetland had dried during previous drought conditions, but contained surface water 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 below the sediment quality guidelines and soil ecological investigation levels for those elements where guidelines are available.

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 guideline values (ANZECC/ARMCANZ 2000) was used to characterise the degree of hazard. For Forster Lagoon, iron (Fe) was assigned a moderate hazard with concentrations exceeding ANZECC/ARMCANZ environmental protection guidelines by more than 10 times. Other metals and metalloids which breached ANZECC/ARMCANZ environmental protection guidelines were aluminium (Al), arsenic (As), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni) and vanadium (V). The soils displayed little trend in pH throughout the tests, with all samples being slightly acidic by day 35 of the tests. Over the duration of the analysis, there was a slight decrease in Eh in one profile, which is thought to be responsible for increasing iron (Fe) solubility over time. The metals cobalt (Co ), chromium (Cr) and nickel (Ni) showed similar trend suggesting a control by the reductive dissolution of iron oxide/oxyhydroxide minerals. The metalloids arsenic (As) and vanadium (V) also showed a tendency to increase over time probably related to dissolution of an iron-rich mineral phase.

Forster Lagoon 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 **high** risk rating for ***contaminant mobilisation*** of **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 due to **acidification** and a **medium to high r**isk 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 forming either in the wetland or during laboratory experiments.

In designing a management strategy for dealing with acid sulfate soils in Forster Lagoon, 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 to high risks to the wetland values associated with contaminant mobilisation in Forster Lagoon, a monitoring program of surface waters is recommended.

# 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).This report outlines the results of Phase 2 activities on selected surface soil samples from Forster Lagoon.

Forster Lagoon was selected for Phase 2 detailed assessment following analysis of results from the Phase 1 detailed assessment (Grealish *et al.* 2011) and the priority ranking criteria adopted by the Scientific Reference Panel of the MDB ASSRAP (see Table 1-1). Sampling sites from Phase 1 are shown on 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, 2 high priority sites based on hyposulfidic materials with $S_{CR} \geq 0.10\%$ and 2 moderate priority sites based on the presence of hyposulfidic materials with $S_{CR} < 0.10\%$. Phase 2 investigations were carried out on selected surface soil samples from sites assessed in the Phase 1 assessment.

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 Forster Lagoon is presented in Table 1-3.

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. |

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 Forster Lagoon samples analysed for Phase 2 assessment.**

| Soil Laboratory Test | Forster Lagoon samples | Sample depth (cm) | Number of samples analysed |
|---|---|---|---|
| Reactive metals | FRL 2.1 | 0-5 | 4 |
| | FRL 2.2 | 5-20 | |
| | FRL 3.1 | 0-10 | |
| | FRL 3.2 | 10-40 | |
| Contaminant and metalloid dynamics | FRL 2.1 | 0-5 | 4 |
| | FRL 2.2 | 5-20 | |
| | FRL 3.1 | 0-10 | |
| | FRL 3.2 | 10-40 | |
| Monosulfide formation potential | FRL 2.1 | 0-5 | 1 |
| Mineral identification by X-ray diffraction (XRD) | N/A | | |

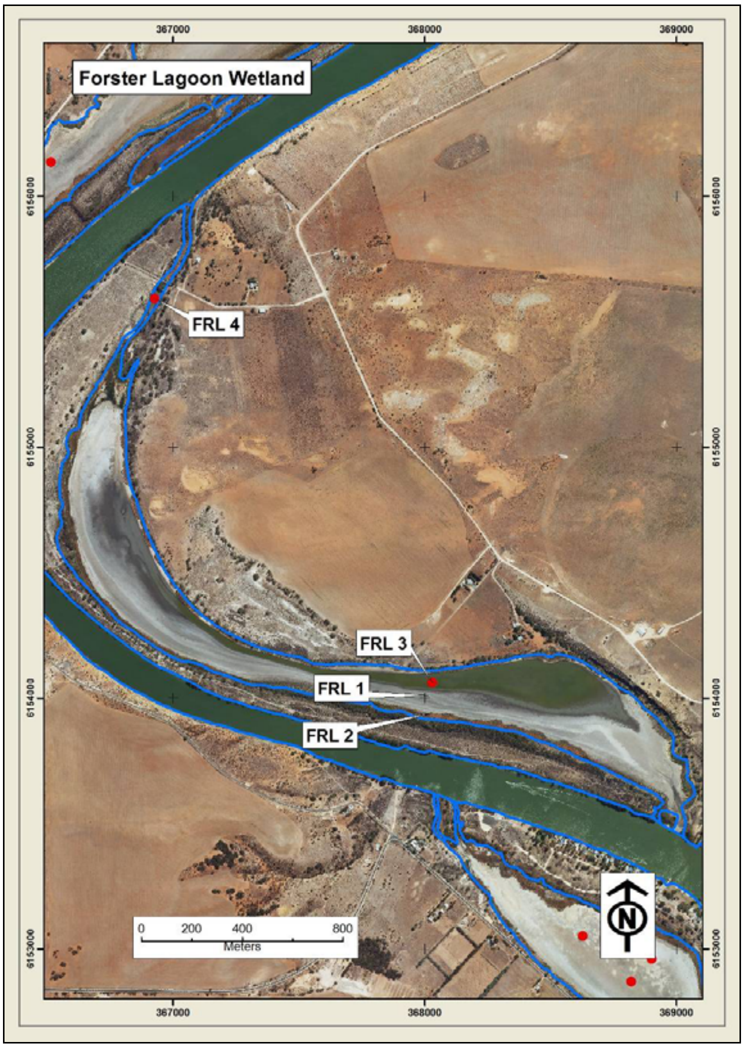

Figure 1-1 Forster Lagoon aerial photograph with Phase 1 sampling sites identified.

# 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 and 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 (CMD) 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 most metals were below sediment quality guideline (SQG) values and soil ecological investigation levels (EIL) for those elements for which guidelines are available (Table 3-1). Nevertheless, the concentrations of a number of elements are considered sufficiently high for extractions, that they may pose a hazard if released to soil pore-waters and surface waters, including aluminium (Al), iron (Fe) and manganese (Mn).

**Table 3-1 Forster Lagoon reactive metals data.**

Concentrations in mg $kg^{-1}$ and μg $kg^{-1}$ as indicated.

| Sample | Ag* | Al | As | Cd* | Co | Cr* | Cu | Fe | Mn | Ni | Pb | Sb* | Se* | V | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| FRL 2.1 | 6.4 | 315 | 0.59 | 56 | 2.7 | 52 | 3.6 | 510 | 74 | 5.5 | 2.1 | < 1.9 | 25 | 8.1 | 3.5 |
| FRL 2.2 | 7.7 | 309 | 0.38 | 50 | 0.50 | 31 | 2.2 | 353 | 27 | 4.2 | 1.5 | < 2.2 | 22 | 6.8 | 2.2 |
| FRL 3.1 | 12 | 378 | 1.6 | 54 | 0.87 | 95 | 6.8 | 877 | 50 | 5.3 | 4.1 | < 3.2 | 37 | 16 | 3.5 |
| FRL 3.2 | 8.8 | 338 | 1.5 | 48 | 1.8 | 66 | 4.1 | 679 | 107 | 5.5 | 7.2 | < 3.3 | 37 | 14 | 3.2 |
| | | | | | | | | | | | | | | | |
| [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 four Forster Lagoon 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 | Forster Lagoon | | |
|---|---|---|---|---|---|
| | | | Min. | Median | Max. |
| pH | | 6.5-8.0 | **4.6** | **5.7** | **6.4** |
| EC* | μS cm$^{-1}$ | 2200 | 138 | 268 | 464 |
| Eh | mV | - | 74 | 440 | 524 |
| Ag | μg l$^{-1}$ | 0.05 | <0.01 | <0.01 | <0.02 |
| Al[A] | mg l$^{-1}$ | 0.055 | <0.05 | <0.05 | **0.37** |
| As[B] | μg l$^{-1}$ | 13 | <0.2 | 0.40 | **32** |
| Cd | μg l$^{-1}$ | 0.2 | <0.01 | <0.06 | <0.06 |
| Co | μg l$^{-1}$ | 2.8 | 0.31 | **3.6** | **19** |
| Cr[C] | μg l$^{-1}$ | 1 | <0.07 | <0.3 | **2.5** |
| Cu[H] | μg l$^{-1}$ | 1.4 | <0.4 | 0.80 | **3.6** |
| Fe[I] | mg l$^{-1}$ | 0.3 | <0.10 | <0.10 | **12** |
| Mn | μg l$^{-1}$ | 1700 | 45 | 362 | 772 |
| Ni[H] | μg l$^{-1}$ | 11 | <1 | 4.4 | **20** |
| Pb[H] | μg l$^{-1}$ | 3.4 | 0.20 | <0.8 | <1.0 |
| Sb | μg l$^{-1}$ | 9 | <0.60 | <2 | <4 |
| Se | μg l$^{-1}$ | 11 | <0.04 | 0.07 | 0.56 |
| V | μg l$^{-1}$ | 6 | <0.30 | 1.0 | **14** |
| Zn[H] | μg l$^{-1}$ | 8 | <2 | 2.6 | 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 the soil materials were slightly acidic after day 1, ranging from pH 5.35 to 6.36, which is less than the lower ANZECC/ARMCANZ environmental protection guideline of pH 6.5 (Figure 3-1). All samples showed a decrease by day 7, followed by an increase on days 14 and 35 with a range from pH 5.86 to 6.28.

The Eh was initially high, and generally decreased over the duration of the experiments (Figure 3-1). The samples from profile 2 in Phragmites reeds at the edge of the wetland (Figure 1-1), had slightly lower Eh by day 35. The SEC increased in all samples up to day 14, subsequently decreasing to values slightly lower than those on day 1 (Figure 3-1). The surface soils had higher SEC, consistent with the observation of white salts on the soil surface during sampling (Grealish *et al*. 2011).

Iron (Fe) concentrations were initially low or below the detection limit. Samples from the marginal profile FRL 2 displayed a large increase on day 35 (Figure 3-2), associated with the lowest Eh. Manganese (Mn) concentrations were relatively low, showing a general increase over time, but remaining well below the ANZECC/ARMCANZ environmental protection guideline (Figure 3-2)

A number of other trace elements were above the ANZECC/ARMCANZ environmental protection guidelines including aluminium (Al), arsenic (As), cobalt (Co), chromium (Cr), iron (Fe), nickel (Ni) and vanadium (V). However, only iron exceeded guideline values by more than ten times (Table 3-2). Aluminium (Al) concentrations were above the ANZECC/ARMCANZ environmental protection guideline in sample FRL 2.2, and on days 1 and 14 in sample FRL 2.1, but remained below detection limit in profile FRL 3. It is likely that the aluminium (Al) exists as colloidal material or associated with organic matter, since equilibrium concentrations should be very low at the pH levels measured in the samples.

Arsenic (As) increased in profile FRL 2 on day 35, similar to iron (Fe) (Figure 3-1), and its behaviour is probably closely linked to the reductive dissolution of iron (Fe). Cobalt (Co), nickel (Ni) and vanadium (V) also displayed an identical pattern with concentrations above the ANZECC/ARMCANZ environmental protection guideline (Figure 3-2 and Figure 3-3). Chromium (Cr) was above the ANZECC/ARMCANZ environmental protection guideline only in one sample on day 35 (Figure 3-2).

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). Overall, the samples displayed a negative correlation with pH (Figure 3-4) suggesting that pH is not the major control on dissolved concentrations. The time series data in Figure 3-1 to Figure 3-3 indicate a much closer link between high metal and metalloid concentrations and Eh. The reductive dissolution of iron (Fe) is probably the dominant control on the release of the metals cobalt (Co), chromium (Cr) and nickel (Ni), as well as the metalloids arsenic (As) and vanadium (V). The future trends are difficult to predict, but if iron continued to increase then it is likely that these metals would follow a similar trend. Although the higher metal and metalloid concentrations were found only in the marginal soils, this is probably due to the slightly higher Eh in the main wetland soils, and further reduction may release iron and associated metals similar to the marginal soils. The low concentrations of other metals and metalloids are likely to indicate low abundances or availability in the soils of Forster Lagoon.

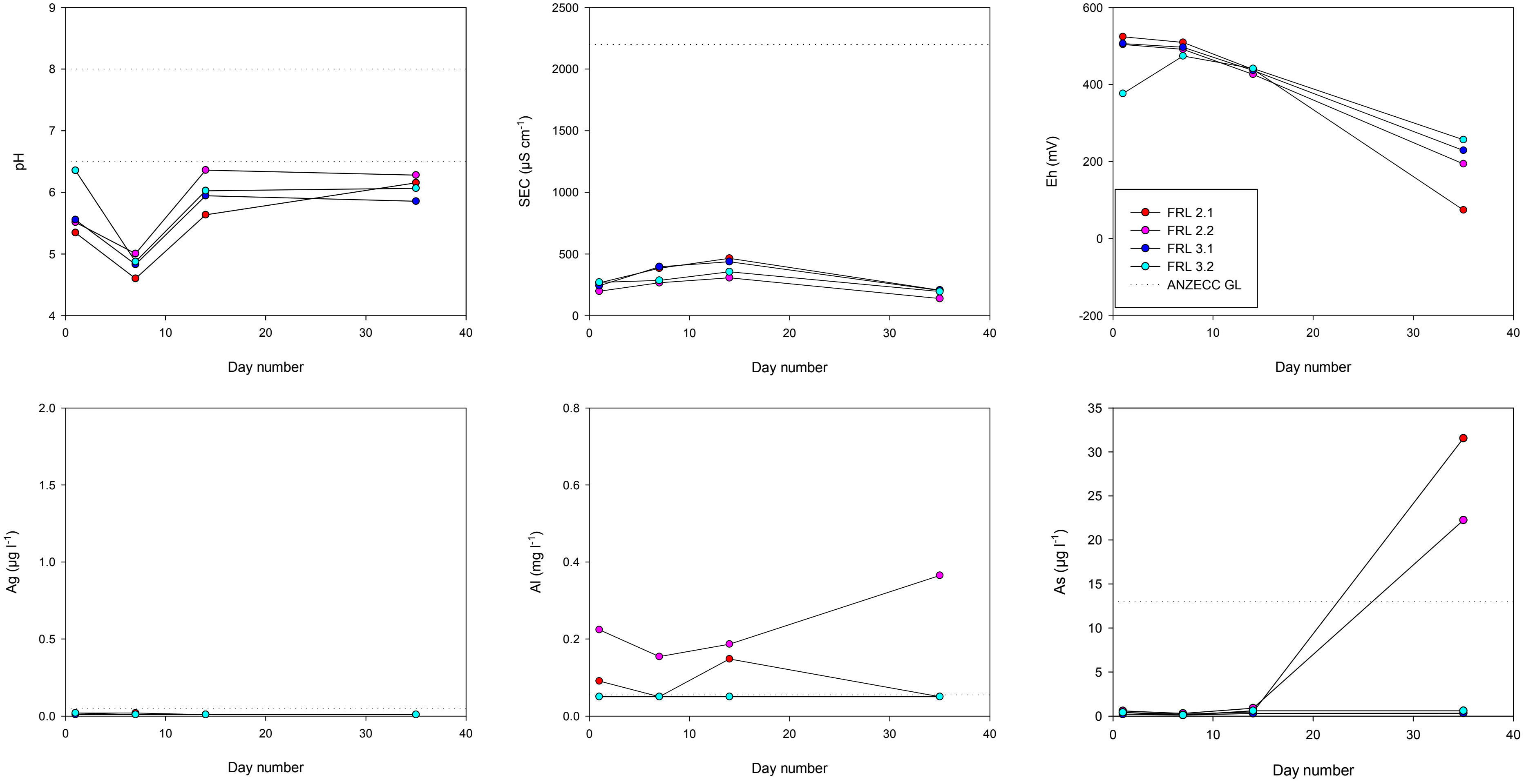

**Figure 3-1 Contaminant and metalloid dynamics results for Forster Lagoon soil materials for pH, SEC, Eh, silver (Ag), aluminium (Al) and arsenic (As).**

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

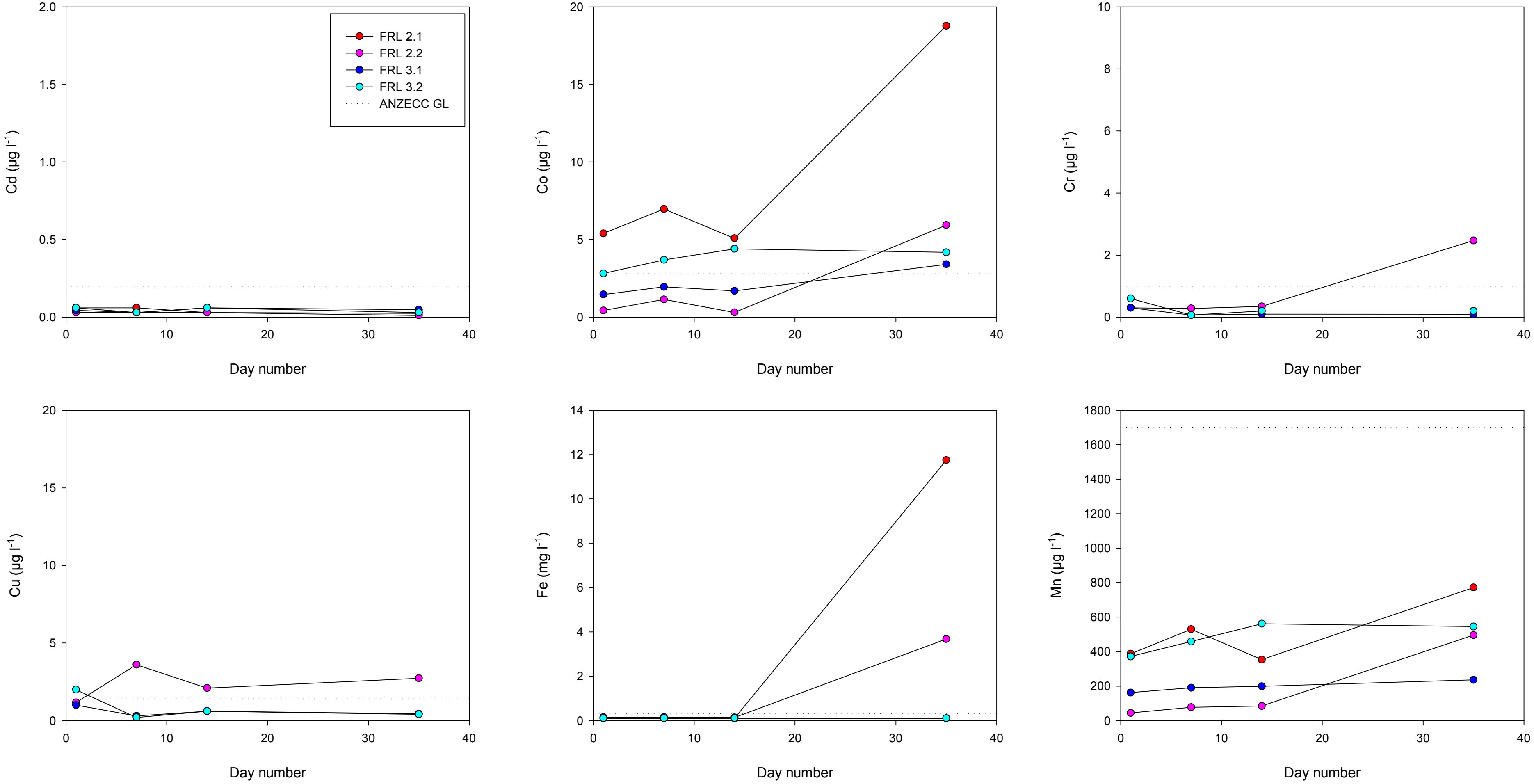

**Figure 3-2 Contaminant and metalloid dynamics results for Forster Lagoon soil materials for cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe) and manganese (Mn).**

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

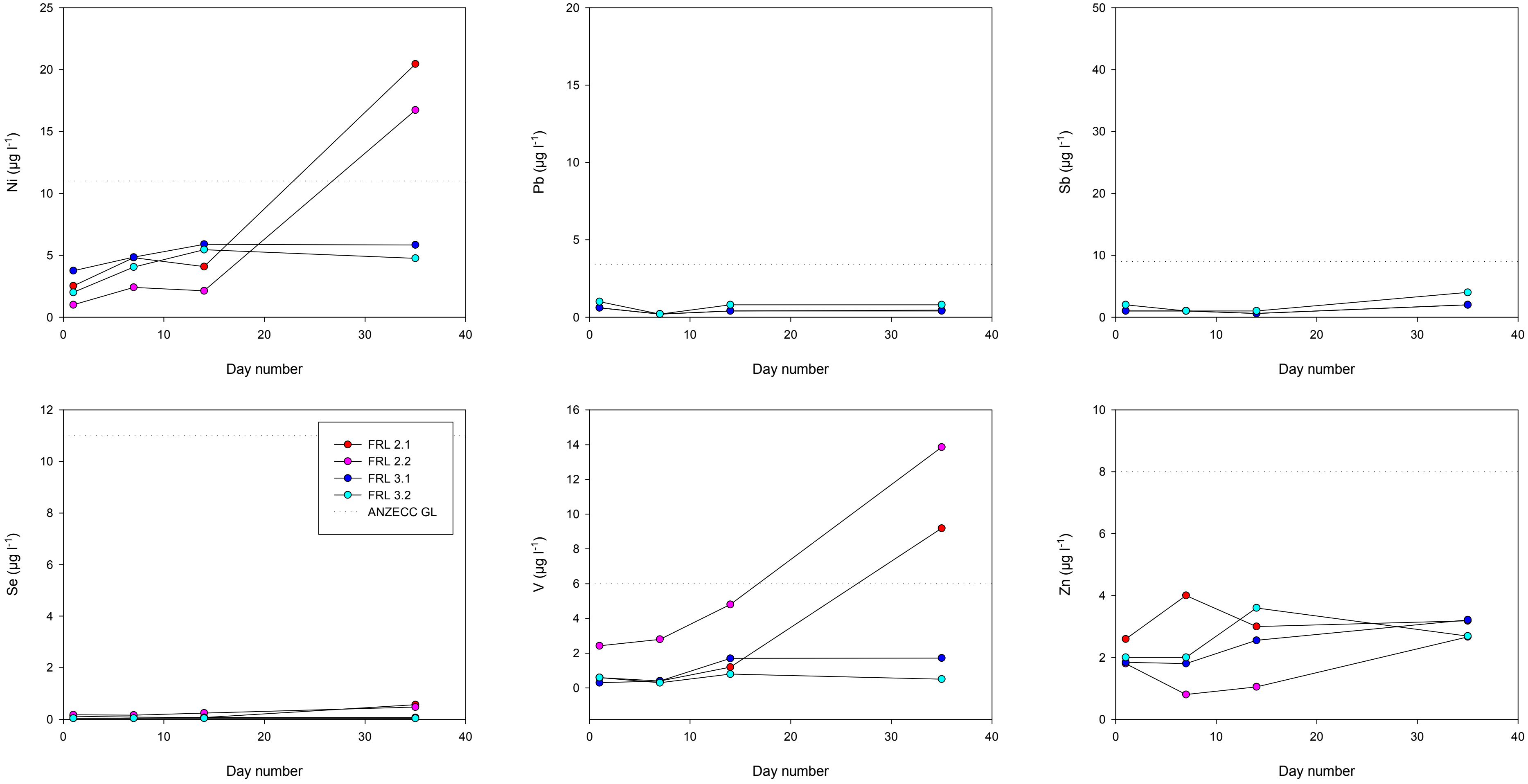

**Figure 3-3 Contaminant and metalloid dynamics results for Forster Lagoon soil materials for nickel (Ni), lead (Pb), antimony (Sb), selenium (Se), vanadium (V) and zinc (Zn).**

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

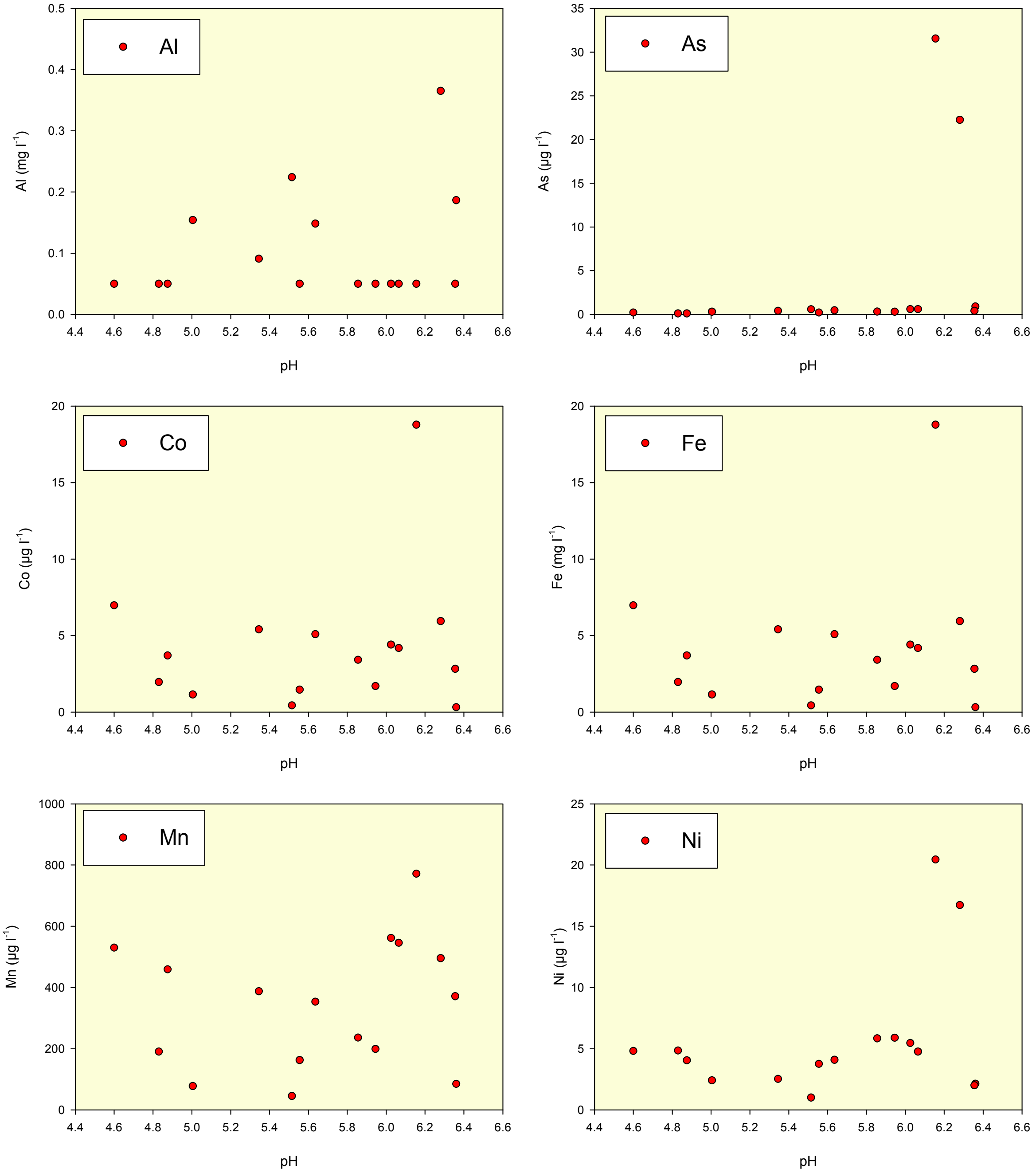

**Figure 3-4 Selected trace elements plotted against pH.**

## 3.1.3. Monosulfide formation potential data

The monosulfide formation potential data for sample FRL 2.1 are shown in Table 3-3. The pH of the soil water changed from 5.20 to 4.32 over the seven week incubation period. The decrease in pH is different from the contaminant and metalloid dynamics tests, where pH increased from 5.35 to 6.16 (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 469 to 171 mV indicating a significant change to more reducing conditions (Table 3-3). The Eh data are generally consistent with the Eh trend in the contaminant and metalloid dynamics (CMD) experiments where Eh decreased from 524 to 74 mV.

**Table 3-3 Summary of monosulfide formation potential data for the Forster Lagoon surface soil material FRL 2.1 after 7 weeks (3.6 g/L sucrose).**

| Inundation Time | Parameter | Units | FRL 2.1 |
|---|---|---|---|
| Day 0 | Total Fe | mg/kg | 3786 |
| | Fe(II)$^-$ | mg/kg | 585 |
| | Sulfate* | mg/kg | 3705 |
| | pH | | 5.20 |
| | Eh | mV | 469 |
| Week 7 | pH | | 4.32 |
| | Eh | mV | 171 |
| | $S_{AV}$ | Wt. %S | <0.01 |
| | $S^o$ | Wt. %S | <0.01 |
| | Pyrite-S | Wt. %S | 0.01 |
| | Dissolved $S^{2-}$ | µg/L | 29 |

*completed during Phase 1

The sample originally contained <0.01% $S_{CR}$ (pyrite-S + $S_{AV}$) and no acid neutralising capacity (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), therefore, monosulfide formation has not occurred in this sample during the tests. Dissolved sulfide concentrations were low at 29 µg $l^{-1}$ (Table 3-3).

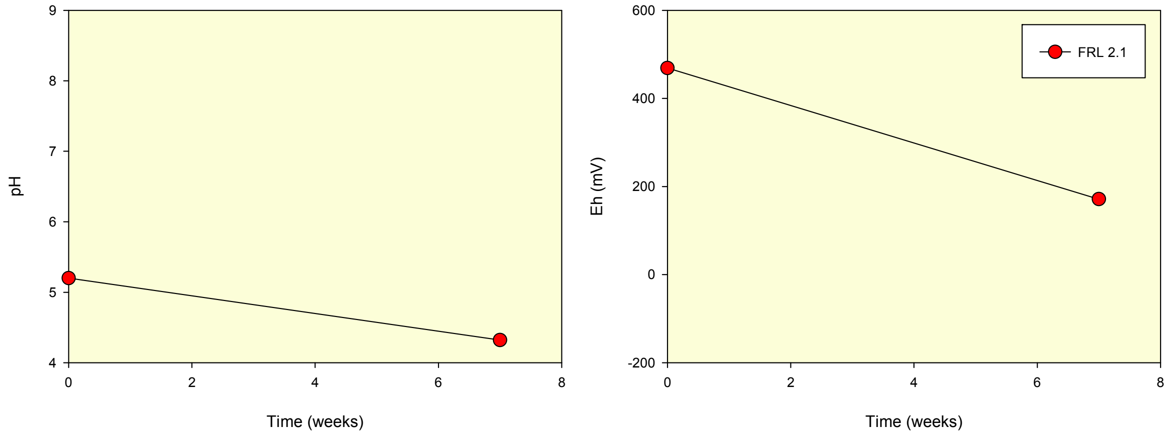

**Figure 3-5 pH and Eh dynamics during monosulfide formation potential tests in surface soil sample FRL 2.1 from Forster Lagoon.**

### 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 soil and 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 concentrations for most reactive metals and metalloids are relatively low (Table 3-1). Although concentrations do not exceed sediment quality guidelines and soil ecological investigation levels, they are generally sufficiently high (mg $kg^{-1}$), especially iron (Fe), manganese (Mn) and aluminium (Al), to pose a risk to waters if dissolved in waters where guideline values are generally order of µg $l^{-1}$.

The **contaminant and metalloid dynamics** method 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.

It would appear that reductive processes have begun in the samples with a slight decrease in Eh for most samples, but the pH of the samples remained relatively constant implying some pH buffering by soil minerals and dissolved species. The soil water pH was variable in the samples studied, but increased slightly over time throughout the 35 day period. All samples showed a significant decrease in Eh, and for one profile, this was sufficient to allow iron (Fe) to be soluble. Iron (Fe) concentrations increased over time in samples from profile FRL 2, and appear to be strongly controlled by the decrease in Eh. Manganese (Mn) was relatively low, and the moderately low Eh suggests that manganese has limited availability in this wetland. The risks associated with metal and metalloid release are probably related to the dissolution of iron (Fe) and to a lesser degree manganese (Mn) compounds (Figure 3-1 to Figure 3-3). For the metalloids, it appears that redox processes are also the dominant influence, and these may continue to be a problem at higher pH, where these metalloids form oxyanions which are sorbed more at lower pH on metal oxides.

The degree to which samples analysed exceed guideline concentrations has been used to assign a degree of hazard (Table 3-4). The data are shown in Appendix 1 which displays the detection limits for individual analyses.

The data are consistent with the low to moderate to high net acidities noted by Grealish *et al.* (2011). The contaminant and metalloid dynamics data suggest that most soils may take little time to recover in terms of acidity. The higher pH in these surface soils studied will help minimise or ameliorate any upward flux of acidity and metals from this area. Higher pH will also limit the solubilities of most trace metals. The main control on metal mobility appears to be Eh in Forster Lagoon soils. The source of many metals appears to be closely related to iron (Fe), which may imply a source from iron oxides/oxyhydroxides. The metalloids also show a strong similarity to iron (Fe), therefore, a link to dissolution of an iron-bearing mineral appears likely. The mobility of both metals and metalloids may be a significant risk over longer timescales, at least until further reduction into the field of iron sulfide stability, where they may be scavenged by precipitating iron sulfides.

**Table 3-4 Summary of the degree of hazard associated with the measured contaminant and metalloid concentrations in Forster Lagoon.**

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

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 ($S^0$), with the dominant form of reduced S being only a trace of pyrite-S. 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 "t*he 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 soils, the acid generating potential and the buffering capacity of the 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 4-3, 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.

Forster Lagoon 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 low to moderate net acidities in samples analysed from Forster Lagoon show that the acidification hazard was variable but relatively low. Net acidities were highest in the surface soil materials (Grealish *et al*. 2011). Net acidity comprised both titratable actual acidity (TAA) and chromium reducible sulfur ($S_{CR}$), with TAA dominant in the marginal soils.

The probability of soil acidification is considered low to moderate as suggested by the contaminant and metalloid dynamics (CMD) experiments. After 35 days, the pH of all the samples was similar and slightly acidic. Combined with a decrease in Eh, it is unlikely that the soils would decrease in pH further. Due to the wetlands location adjacent to the river and narrow connections, the likelihood of disturbance is considered ***almost certain*** as flows return to normal in the future. The consequences are not likely to be significant for soil ecology, but the timescale for soil recovery from acidification cannot be assessed with existing information. Field scale studies in other wetlands e.g. Nelwart Lagoon (Shand *et al.* 2010) indicate that in areas with strongly acidic soils, the timescale is likely to be months to years, but much less when the pH is circumneutral or slightly acidic. An ***insignificant*** rating is therefore applied for consequence as long-term damage to soil water chemistry is considered unlikely. 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. 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. However, with an insignificant rating for soil acidification, a similar ***insignificant*** rating is applied to surface water. The *risk to surface water acidification* is therefore **low** (Table 4-4).

### 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) and manganese (Mn) oxides and hydroxides which are important sorbents of metal and metalloids species. The variable but generally low acidification hazard due to the oxidation of sulfide minerals means that metals and metalloids are not likely to pose a high risk to soil pore-waters and possibly surface waters. Although reduction processes may lead to reincorporation of metals and metalloids into sulfide minerals (following sulfate reduction), at intermediate redox potentials mobility may be high where iron (Fe) and manganese (Mn) are soluble. The reactive metals and contaminant and metalloid dynamics (CMD) results attest to the availability and mobility of a number of metals, particularly iron (Fe). Although iron (Fe) was the only element to breach the ANZECC/ARMCANZ environmental protection guidelines by more than ten times, it and a number of other elements were released during the final stages of the contaminant and metalloid dynamics (CMD) tests and with further decrease in Eh may increase further. As with assessing the acidification risk, the timescales cannot be assessed with existing information. The presence of a number of trace metals and metalloids above the ANZECC/ARMCANZ environmental protection guidelines and no evidence for a decrease in the contaminant and metalloid dynamics (CMD) experiments suggests that a **minor to *moderate*** rating should be applied for consequence. This provides a risk rating for contaminant mobilisation in soils of **medium to high** (Table 4-4).

A rating for surface water impacts from metals and metalloids will depend on surface and sub-surface hydrology. The consistently low pH values in this study, however, means that short term impacts are likely if hydrological conditions allow a flux of metals towards overlying surface water. The highest risk is likely to be during low flows where the soil to water ratio is high: metals will be most concentrated. The minimum risk to surface water acidification is considered lowest where high flows are available to both dilute metal and metalloid concentrations and transport these downwards in the soil profile. Taking a range in consequence from ***minor to moderate***, the risk rating will be **medium to high**.

### 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 (MBO) was not identified in the wetland prior to the Phase 1 survey (Grealish *et al.* 2011). The water soluble sulfate concentrations were high in all samples (164 – 10,392 mg kg$^{-1}$ $SO_4$), above the trigger value for monosulfide formation (MDBA 2010).

While water soluble sulfate concentrations were high, there was no acid volatile sulfide measured from the monosulfide formation potential test, and therefore it is concluded that there is no identified deoxygenation hazard. An **insignificant** consequence rating is therefore applied due to the lack of evidence for monosulfide formation either in the wetland or during the laboratory experiments. This provides a **low** risk rating for deoxygenation from monosulfide disturbance.

**Table 4-4 Summary of risks associated with acid sulfate soil materials in Forster Lagoon.**

| Acidification Risk | | Contaminant mobilisation | | Deoxygenation |
|---|---|---|---|---|
| *Soil* | *Water* | *Soil* | *Water* | |
| Low | Low | Medium-High | Medium-high | 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 major risks identified in this study are due to metal mobility. 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, short term risks from metal mobilisation are likely if hydrological conditions are such that there is a significant flux of metals from the soils to the overlying water column.

The wetlands below Lock 1, such as Forster Lagoon, are subject to whole-of-reach river level fluctuations. Prevention or mitigation measures should include consideration of not allowing river levels to drop to a point that the wetlands are impacted by severe drying. Nevertheless, Forster Lagoon had dried completely during the recent drought (although pools of surface water were present when sampling). As the wetland had 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, but the highest risks identified are from metal and metalloid mobilisation. 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:**<br>• Lowering saline water tables<br>• Maintaining the freshwater lens between saline groundwater and the aquatic ecosystem<br>• Stopping the delivery of irrigation return water<br>• Incorporating a more natural flow regime. |
| **2. Preventing oxidation of acid sulfate soils or controlled oxidation to remove acid sulfate soils** | **Preventing oxidation:**<br>• Keep the sediments covered by water<br>• Avoid flow regimes that could re-suspend sediments.<br>**Controlled oxidation:**<br>• Assess whether neutralising capacity of the sediments and water far exceeds the acidity produced by oxidation<br>• 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<br>• Add organic matter to promote bioremediation by micro-organisms<br>• 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<br>• Neutralise and dilute surface water<br>• Treat discharge waters by neutralisation or biological treatment. |
| **5. Limited further intervention** | • Assess risk<br>• Communicate with stakeholders<br>• Undertake monitoring<br>• Assess responsibilities and obligations and take action as required. |

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# APPENDICES

# APPENDIX 1 REACTIVE METALS DATA

## Forster Lagoon

| Sample | Depth | Analysis | Ag* | Al | As | Cd* | Co | Cr* | Cu | Fe | Mn | Ni | Pb | Sb* | Se* | V | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| FRL 2.1 | 0-5 | a | 6.1 | 309 | 0.57 | 54 | 2.6 | 52 | 3.5 | 483 | 72 | 5.4 | 2.0 | < 1.9 | 25 | 8.0 | 3.4 |
| | | b | 6.7 | 320 | 0.60 | 57 | 2.8 | 52 | 3.7 | 537 | 77 | 5.5 | 2.1 | < 1.9 | 25 | 8.2 | 3.6 |
| FRL 2.2 | 5-20 | a | 8.3 | 331 | 0.40 | 54 | 0.55 | 35 | 2.4 | 397 | 29 | 4.7 | 1.7 | < 2.2 | 23 | 7.6 | 2.3 |
| | | b | 7.2 | 288 | 0.36 | 46 | 0.46 | 26 | 2.1 | 309 | 25 | 3.7 | 1.4 | < 2.2 | 21 | 5.9 | 2.1 |
| FRL 3.1 | 0-10 | a | 12 | 377 | 1.5 | 50 | 0.84 | 89 | 6.9 | 839 | 43 | 4.7 | 4.3 | < 3.2 | 35 | 16 | 3.4 |
| | | b | 12 | 380 | 1.6 | 59 | 0.91 | 101 | 6.7 | 915 | 56 | 5.9 | 4.0 | < 3.2 | 38 | 16 | 3.5 |
| FRL 3.2 | 10-40 | a | 9.0 | 352 | 1.5 | 51 | 1.9 | 66 | 4.2 | 675 | 111 | 5.4 | 7.7 | < 3.3 | 37 | 15 | 3.4 |
| | | b | 8.6 | 324 | 1.5 | 45 | 1.8 | 66 | 3.9 | 684 | 103 | 5.5 | 6.7 | < 3.3 | 37 | 14 | 3.0 |

Units are in 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

## Forster Lagoon

| 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 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| FRL 2.1 | 1 | 0-5 | a | 529 | 250 | 5.05 | <0.02 | 0.05 | <0.4 | <0.06 | 5.0 | <0.6 | <2 | <0.1 | 332 | 3.0 | <1 | <2 | 0.15 | <0.6 | 2.7 |
| | | | b | 519 | 280 | 5.64 | <0.02 | 0.13 | <0.4 | <0.06 | 5.8 | <0.6 | <2 | <0.1 | 443 | <2 | <1 | <2 | 0.09 | <0.6 | 2.5 |
| | 7 | | a | 514 | 385 | 4.45 | <0.02 | <0.05 | 0.20 | <0.06 | 7.4 | <0.1 | <0.4 | <0.1 | 550 | 5.4 | <0.4 | <2 | <0.08 | <0.4 | 4.0 |
| | | | b | 504 | 382 | 4.75 | <0.02 | <0.05 | 0.20 | <0.06 | 6.5 | <0.1 | <0.4 | <0.1 | 510 | 4.2 | <0.4 | <2 | <0.08 | <0.4 | 4.0 |
| | 14 | | a | 439 | 303 | 5.54 | <0.01 | 0.25 | 0.60 | <0.03 | 4.6 | 0.10 | 0.60 | 0.10 | 306 | 3.6 | <0.4 | <0.6 | 0.06 | 1.4 | 2.7 |
| | | | b | 439 | 626 | 5.73 | <0.01 | 0.05 | <0.3 | 0.03 | 5.6 | <0.1 | 0.30 | <0.1 | 401 | 4.6 | <0.4 | <0.6 | 0.08 | 1.0 | 3.3 |
| | 35 | | a | 59 | 200 | 5.88 | <0.01 | <0.05 | 35 | 0.03 | 19 | 0.97 | 1.8 | 11 | 779 | 20 | <0.4 | <2 | 0.43 | 7.6 | 5.6 |
| | | | b | 89 | 214 | 6.43 | <0.01 | <0.05 | 28 | 0.02 | 18 | 1.3 | 1.6 | 13 | 764 | 21 | <0.4 | <2 | 0.69 | 11 | 0.71 |
| FRL 2.2 | 1 | 5-20 | a | 499 | 215 | 5.65 | <0.01 | 0.27 | 0.95 | <0.03 | 0.26 | <0.3 | 1.3 | 0.21 | 34 | <1 | <0.6 | <1 | 0.18 | 4.2 | 2.6 |
| | | | b | 509 | 178 | 5.38 | <0.01 | 0.18 | <0.2 | <0.03 | 0.59 | <0.3 | <1 | <0.1 | 56 | <1 | <0.6 | <1 | 0.16 | 0.70 | <1 |
| | 7 | | a | 504 | 299 | 4.54 | <0.01 | <0.05 | <0.1 | <0.03 | 2.1 | <0.07 | <0.2 | <0.1 | 142 | 3.1 | <0.2 | <1 | 0.12 | 1.2 | 0.80 |
| | | | b | 479 | 232 | 5.47 | <0.01 | 0.26 | 0.50 | <0.03 | 0.17 | 0.49 | 7.0 | 0.20 | 13 | 1.7 | <0.2 | <1 | 0.20 | 4.4 | 0.80 |
| | 14 | | a | 429 | 332 | 6.84 | <0.01 | 0.19 | 0.30 | <0.03 | 0.39 | 0.20 | 1.2 | 0.10 | 99 | 1.6 | <0.4 | <0.6 | 0.20 | 4.2 | 0.90 |
| | | | b | 424 | 282 | 5.88 | <0.01 | 0.19 | 1.5 | <0.03 | 0.23 | 0.50 | 3.0 | 0.17 | 70 | 2.7 | <0.4 | <0.6 | 0.28 | 5.4 | 1.2 |
| | 35 | | a | 234 | 133 | 6.34 | <0.01 | 0.51 | 34 | 0.01 | 5.4 | 2.1 | 3.4 | 2.6 | 473 | 14 | 0.49 | <2 | 0.47 | 14 | 3.4 |
| | | | b | 154 | 143 | 6.22 | <0.01 | 0.22 | 10 | <0.01 | 6.5 | 2.9 | 2.0 | 4.8 | 518 | 19 | <0.4 | <2 | 0.47 | 14 | 1.9 |
| FRL 3.1 | 1 | 0-10 | a | 509 | 307 | 5.31 | <0.01 | <0.05 | <0.2 | 0.06 | 1.6 | <0.3 | <1 | <0.1 | 175 | 4.1 | <0.6 | <1 | 0.05 | <0.3 | 1.9 |
| | | | b | 504 | 175 | 5.80 | <0.01 | <0.05 | <0.2 | 0.03 | 1.3 | <0.3 | <1 | <0.1 | 150 | 3.4 | <0.6 | <1 | 0.03 | 0.30 | 1.8 |
| | 7 | | a | 494 | 379 | 4.90 | <0.01 | <0.05 | 0.10 | <0.03 | 1.7 | <0.07 | 0.40 | <0.1 | 177 | 4.6 | <0.2 | <1 | <0.04 | 0.40 | 1.6 |
| | | | b | 499 | 412 | 4.76 | <0.01 | <0.05 | <0.1 | 0.03 | 2.2 | <0.07 | 0.20 | <0.1 | 204 | 5.1 | <0.2 | <1 | <0.04 | 0.40 | 2.0 |
| | 14 | | a | 434 | 494 | 5.78 | <0.01 | <0.05 | <0.3 | 0.06 | 2.0 | <0.1 | 0.60 | <0.1 | 220 | 6.2 | <0.4 | <0.6 | 0.08 | 1.3 | 2.7 |
| | | | b | 439 | 380 | 6.11 | <0.01 | <0.05 | <0.3 | 0.06 | 1.4 | <0.1 | 0.60 | <0.1 | 178 | 5.5 | <0.4 | <0.6 | 0.06 | 2.1 | 2.4 |
| | 35 | | a | 224 | 204 | 5.35 | <0.01 | <0.05 | <0.3 | 0.05 | 3.7 | 0.09 | 0.42 | <0.1 | 253 | 6.2 | <0.4 | <2 | 0.06 | 1.3 | 3.5 |
| | | | b | 234 | 203 | 6.36 | <0.01 | <0.05 | 0.32 | 0.04 | 3.1 | 0.11 | 0.47 | <0.1 | 219 | 5.5 | <0.4 | <2 | 0.07 | 2.1 | 2.9 |
| FRL 3.2 | 1 | 10-40 | a | 379 | 280 | 5.60 | <0.02 | <0.05 | <0.4 | <0.06 | 2.6 | <0.6 | <2 | <0.1 | 370 | <2 | <1 | <2 | 0.04 | <0.6 | <2 |
| | | | b | 374 | 261 | 7.11 | <0.02 | <0.05 | <0.4 | <0.06 | 3.0 | <0.6 | <2 | <0.1 | 373 | <2 | <1 | <2 | <0.04 | <0.6 | <2 |
| | 7 | | a | 469 | 281 | 4.77 | <0.01 | <0.05 | <0.1 | <0.03 | 3.7 | <0.07 | <0.2 | <0.1 | 463 | 3.9 | <0.2 | <1 | <0.04 | <0.2 | 2.0 |
| | | | b | 479 | 289 | 4.98 | <0.01 | <0.05 | <0.1 | 0.03 | 3.7 | <0.07 | <0.2 | <0.1 | 455 | 4.2 | <0.2 | <1 | <0.04 | 0.40 | 2.0 |
| | 14 | | a | 439 | 355 | 5.00 | <0.01 | <0.05 | <0.6 | <0.06 | 4.4 | <0.2 | <0.6 | <0.1 | 560 | 5.3 | <0.8 | <1 | 0.04 | 0.80 | 4.2 |
| | | | b | 444 | 355 | 7.05 | <0.01 | <0.05 | <0.6 | <0.06 | 4.4 | <0.2 | <0.6 | <0.1 | 563 | 5.6 | <0.8 | <1 | <0.04 | 0.80 | 3.0 |
| | 35 | | a | 254 | 193 | 5.26 | <0.01 | <0.05 | <0.6 | 0.03 | 4.3 | <0.2 | <0.4 | <0.1 | 561 | 4.8 | <0.8 | <4 | 0.03 | 0.47 | 2.8 |
| | | | b | 259 | 197 | 6.87 | <0.01 | <0.05 | <0.6 | 0.03 | 4.1 | <0.2 | <0.4 | <0.1 | 531 | 4.7 | <0.8 | <4 | 0.03 | 0.54 | 2.6 |

< value is below detection limit

# APPENDIX 3 MONOSULFIDE FORMATION POTENTIAL DATA

**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** | +/- |
| 23 | csiro | Forster Lagoon | FRL2.1 | 3476 | 4096 | **3786** | *310* | 638 | 531 | **585** | *54* | 464 | 474 | **469** | *5* | 5.23 | 5.16 | **5.20** | 0.04 |
| 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** | *+/-* |
| 23 | csiro | Forster Lagoon | FRL2.1 | 0.00 | 0.00 | **<0.01** | *<0.01* | 0.01 | 0.01 | **0.01** | *<0.01* | 0.00 | 0.00 | **<0.01** | *<0.01* | 4.35 | 4.29 | **4.32** | *0.03* | 70 | 272 | **171** | *101* | <0.1 | 57 | **29** | *29* |
| 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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## Your CSIRO