Recovery of acid sulfate sediments in the Lower Lakes

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Recovery of acid sulfate sediments in the Lower Lakes

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Cover Photograph

Sediment sampling at Waltowa, Lower Lakes. Photographer: Leigh Sullivan.

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LIST OF ABREVIATIONS

ANC – acid neutralising capacity
AVS – acid volatile sulfide
$CaCO_3$ – calcium carbonate
Cl – chloride
CRS – chromium reducible sulfur
EC – electrical conductivity
Eh – redox potential
Fe – iron
Fe^{2+} – ferrous iron
Fe^{3+} – ferric iron
FIA – flow-injection analysis
HCl – hydrochloric acid
HPLC – high-performance liquid chromatography
ICP-MS – inductively coupled plasma - mass spectrometry
MBO – monosulfidic black ooze
NATA – National Association of Testing Authorities
PSA – potential sulfidic acidity
RA – retained acidity
RIS – reduced inorganic sulfur
SRR – sulfate (SO_4^{2-}) reduction rates
S^0 – elemental sulfur
SO_4^{2-} – sulfate
TAA – titratable actual acidity
TAAlk – titratable actual alkalinity
TOC – total organic carbon

Executive Summary

The primary aim of this project was to monitor the biogeochemical state of the Lower Lake sediments approximately 2.5 years after lake refilling especially in relation to vegetation management of the lake acid sulfate sediments. It builds on the results of the previous studies to allow a more accurate assessment of the progression of remediation of these sediments according to bioremediation strategy and whether the potential hazards that often arise during sulfate reduction in sediments are being avoided.

This project has developed a more detailed understanding of the rates of recovery of acidic sediments and the processes that are driving this recovery in different sediment types and locations around the Lower Lakes. Recovery has been examined mainly in terms of increased pHs away from severe acidification as well as in terms of increased alkalinity and hence resilience to any future acidification events. The processes that cause or maintain extremely low pHs in these dry sediments have been examined in using the Acid Base Accounting (ABA) approach of Ahern *et al.* (2004) which allows examination of a number of different acidity pools in soils and sediments. The methodology followed in this study continues the general assessment and analytical strategy used in Sullivan *et al.* (2011, 2012b). Following this methodology allowed maximum benefit in terms of assessing temporal trends by building onto the existing biogeochemical knowledge of these sediments.

Accordingly this project focused on four locations in the Lower Lakes (two on Lake Alexandrina (Poltalloch and Tolderol) and two on Lake Albert (Waltowa and Campbell Park)), and included two control sites and a range of revegetation treatments (in terms of both the vegetation species and the date of establishment of these vegetated treatments).

An additional component of this project was a scoping investigation to examine the likely mobility and uptake by vegetation of metals (particularly nickel and zinc) from the acidified lake sediments as affected by bioremediation. This was undertaken to assess ongoing environmental risks posed by the presence of very high bio-accessible concentrations of potentially toxic trace metals as identified in a previous study of bioremediating formerly strongly acidified Lower Lakes sediments (Sullivan *et al.* 2012a).

The key findings of this study are:

1) There has been a continuing increase in pH at all sites mainly in the surficial sediment layers since re-inundation. Given the lakes pH and the trajectories in pH it would appear that most sediments are back to or near the pH expected prior to the lake drawdown and sediment desiccation. However, it is also clear that especially for the jarositic sediment layers that a much longer period of inundation will be required to restore the pH conditions of these sediments.

2) Despite the regular and consistent increases in sediment pH across all sites there were no consistent appreciable decreases in Net Acidity across these sites. The lack of consistent and appreciable decreases in Net Acidity across these sites in line with the regular and consistent increases in sediment pH indicates that other processes apart from addition of ‘external’ (i.e. to the sediment) alkalinity such as from the overlying lake waters have been important factors affecting the Net Acidities of the sediments. Most importantly the lack of consistent change in the Net Acidities of the lake sediments indicates that they have largely retained their acidity stores, have not built up stores of acid neutralising capacity and hence remain vulnerable to acidification in any future drying event.

3) There has been a considerable accumulation of Potential Sulfidic Acidity in the surface layers under *Phragmites* at the Waltowa site including an accumulation of Monosulfidic Black Ooze. The accumulation of these reduced inorganic sulfides in the surface layers of the sediments under the *Phragmites* also represents the accumulation of environmental hazards including acidity, and localised deoxygenation should these sediments be mobilised and or exposed to drying in the future.

4) There have clearly been transfers between different acidity pools in the sediment as a result of the prolonged inundation. A clear example of this is the transfer of acidity in the Retained Acidity pool to the Titratable Actual Acidity pool in sediments that were jarositic via the production of soluble Fe^{2+} as a result of reductive dissolution.

5) Despite the lack of overall trends in Net Acidity of these sediments after prolonged inundation, there were some site-specific trends observed. In the surface layers of the initially severely acidic, jarositic sites, Net Acidity has generally decreased due to a decrease in Retained Acidity, sometimes a decrease in Titratable Actual Acidity, and often an increase in the Acid Neutralisation Capacity. On the other hand at the initially neutral or limed sites, the Net Acidity has generally increased due to a decrease in Acid Neutralisation Capacity, sometimes an increase in Potential Sulfidic Acidity, and despite small decreases in Actual Acidity.

6) The data on EC and the chloride (Cl) concentrations clearly demonstrate that there has been considerable movement of solutes out of these sediment profiles. Given this, it is surprising that there is no clear indication in the data that the bicarbonate (HCO_3^-) from lake waters has been a major source of alkalinity to the underlying acidic sediments despite considerable increases in the pHs of this sediment.

7) The ABA method of Ahern *et al.* (2004) was shown to be useful for the purpose it was employed for in this study. This method was able to sensibly and clearly quantify changes in the acidity pools at each site. However, this study has identified and confirmed recent criticisms of the ABA's methods for quantifying the Retained Acidity pool as this method failed to identify and quantify the presence of jarosite even when jarosite from field examination was clearly present in appreciable amounts.

8) Although universally accepted critical metal contents for wetland vegetation are not available, this study has clearly shown elevated levels of some metals in the vegetation growing in the formerly acidified sediments. These include manganese, nickel and aluminium. Such elevated metal concentrations are important as even moderate concentrations of metals have been shown to disrupt aquatic ecologies.

9) There were clearly effects of soil on the accumulation of metals by the vegetation with the metal contents much higher in vegetation growing on the former acid sulfate soils compared to that growing on the foredunes. Additionally, the vegetation growing at sites with clayey textured surface soils had lower metal concentrations than the vegetation growing at sites with sandy surface soils.

10) There were clear differences in the uptake and accumulation of metals by different vegetation types around the lake. The *Bolboschoenus* species sampled had much lower accumulations of metals than most of the other vegetation types.

Recommendations

1) We recommend that these sites continue to be monitored over the next few years for changes to their acidity pools to adequately ascertain acidity recovery trajectories and to understand how management can impact on these changes.

2) We recommend that further studies at these sites continue on the inter-related aspects of organic matter chemistry, metal mobilisation (in sediments, vegetation and other critical sectors of the lake's ecology) and sulfate reduction processes in the sediments.

3) We recommend a further detailed study examining both of the above aspects on:
    i. the effectiveness of the different vegetation types (especially differences between different annual vegetation species) and strategies used for bioremediation, and
    ii. the un-bioremediated lake sediment behaviour.

Such an understanding is required in order to understand in sufficient detail the reasons for these different sediment behaviours and to provide a factual basis to optimise lake bioremediation strategies and to understand the lake's geochemical process to assist with ecological restoration programs.

1.0 Project Overview

Recent studies of the sediments of the Lower Lakes and of the effects of bioremediation [i.e. with the South Australian Environment Protection Authority (EPA) and Department of Environment, Water and Natural Resources (DEWNR)] (Sullivan *et al*. 2010a, 2011, 2012b) have highlighted the hazard of acid sulfate soils and their potential to impact on ecological processes. The studies showed that the role of sulfate reduction and associated processes during the re-inundation of the acidified Lower Lakes' sediments that have been exposed during the drying event from 2007-2010 is critical for on-going management.

Several key locations around the Lower Lakes were examined showing a range of vegetation treatments (in terms of both the vegetation species and timing of plantings), as well as unvegetated control sites. The results of the recent studies by Sullivan *et al*. (2011, 2012b) indicate that bioremediation of the exposed acidified lake sediments by vegetation produced substantial environmental benefits from a combination of vegetation-associated processes including the provision of alkalinity directly from plant roots, from sulfate reducing processes enabled by the ongoing production of organic matter by vegetation, as well as from the vegetation minimising soil erosion and hence preventing the exposure of severely acidic subsoils (i.e. below surface soils) that occurred under unvegetated sites.

At the same time, the studies highlighted that several of the likely future hazards associated with a strategy of enhancing organic matter input into sediments to stimulate sulfate reduction and the beneficial co-production of alkalinity, had been substantially avoided in the initial refilling period of the Lower Lakes (i.e. first 19 months), particularly where annual vegetation was too short to survive inundation. This hazard avoidance was due to the characteristic nature of the sulfur cycling occurring in these sediments, the consequent lack of accumulation in the surficial lake sediments of sulfide minerals such as monosulfides and pyrite and their associated hazards of acidification, metal and metalloid mobilisation, and deoxygenation.

However, the recent study by Sullivan *et al.* (2012b) showed when *Phragmites* (a species that survived lake re-filling and continued to grow vigorously when inundated) was used to bioremediate these sediments, there was considerable accumulation of both pyrite and monosulfide (as Monosulfidic Black Ooze (MBO)) in the uppermost sediment layers. These accumulated sulfides indicated that alkalinity had also been produced via sulfate reducing processes enabled by the ongoing production of organic matter by *Phragmites.* In addition, these uppermost sediments under *Phragmites* appeared likely to act as sources of soluble phosphate that could lead to increased nutrient flux/accumulation to lake water. The Sullivan *et al.* (2012b) study strongly indicated a number of potentially important hazards would have arisen if *Phragmites* were to be used for bioremediation of exposed lake sediments (e.g. increased accumulation of sulfides); such hazards were avoided almost completely when inundation intolerant vegetation was used.

It was recognised in the study by Sullivan *et al*. (2012b) that 19 months of re-inundation was too short a time to adequately assess whether these possible future biogeochemically-driven hazards associated with bioremediation will continue to be avoided over the longer term as the broad range of biogeochemical regimes (e.g. from highly acidic and oxic, right through to alkaline and highly anoxic) inevitably sweep through the Lower Lake sediments over the years post lake refilling.

This project builds on the results of the Sullivan *et al*. (2011, 2012b) study to allow a more accurate assessment of the progression of remediation of these sediments according to the bioremediation strategy and whether the potential hazards that often arise during sulfate reduction in sediments continue to be avoided. This project develops a more detailed understanding of the rates of recovery of acidic sediments and the processes that are driving this recovery in different sediment types and locations around the Lower Lakes. Recovery is examined mainly in terms of increased pHs away from severe acidification as well as in terms of increased alkalinity and hence resilience to any future acidification events. The processes that cause or maintain extremely low pHs in these dry sediments are examined in detail, as are the processes that provide alkalinity to overcome these severely acidified conditions.

The methodology followed in this study continues the general assessment and analytical strategy used in Sullivan *et al.* (2011, 2012b). Following this methodology allows maximum benefit in terms of assessing temporal trends by 'building onto' the existing knowledge of the biogeochemistry of these sediments. An additional component of this project is a preliminary scoping investigation to examine the likely mobility and uptake by vegetation of metals (particularly nickel and zinc) from the acidified lake sediments as affected by bioremediation. This will be undertaken to assess ongoing environmental risks posed by the presence of very high bio-accessible concentrations of these two potentially-toxic trace metals as identified in a previous study of bioremediating formerly strongly acidified Lower Lakes sediments (Sullivan *et al.* 2012a).

Accordingly this project focused on four locations in the Lower Lakes (two on Lake Alexandrina (Poltalloch and Tolderol) and two on Lake Albert (Waltowa and Campbell Park)), and included two control sites and a range of revegetation treatments (in terms of both the vegetation species and the date of establishment of these vegetated treatments).

2.0 Aim

The primary aim of this project is to monitor the biogeochemical state (with respect to sulfate reduction and associated processes) of the Lower Lake sediments approximately 2.5 years after lake refilling especially in relation to vegetation management of the lake sediments. The findings are aimed at informing key management decisions on the effectiveness and limitations of bioremediation options in managing acid sulfate soils in the Lower Lakes.

3.0 Introduction

3.1 Background on Acid Sulfate Soils

3.1.1. General

Acid sulfate soil materials are distinguished from other soil materials by having properties and behaviour that have either: 1) been affected considerably (mainly by severe acidification) by the oxidation of reduced inorganic sulfides (RIS), or 2) the capacity to be affected considerably (again mainly by severe acidification) by the oxidation of their RIS constituents.

A wide range of environmental hazards can be generated by the oxidation of RIS. These include: 1) severe acidification of soil and drainage waters (below pH 4 and often < pH 3), 2) mobilisation of metals (e.g. iron, aluminium, copper, cobalt, zinc), metalloids (e.g. arsenic), nutrients (e.g. phosphate), and rare earth elements (e.g. yttrium, lanthanum), 3) deoxygenation of water bodies, 4) production of noxious gases (e.g. hydrogen sulfide (H_2S)), and, 5) scalding (i.e. de-vegetation) of landscapes. Some of these hazards are caused directly or indirectly by the severe acidification that can occur as a result of the oxidation of RIS, whereas some can also be the result of other simultaneous processes occurring in the environment.

Waters draining from acid sulfate soil materials may be enriched in a wide range of potential toxicants, including metals and metalloids, endangering aquatic life and public health. Crops, trees, pastures and aquaculture may also be severely affected by acid sulfate soil materials. Acid sulfate soils can have detrimental impacts on their surrounding environments as well as on communities who live in landscapes containing these soils.

3.1.2 Characteristics and Formation

It is useful to distinguish between sulfidic soil materials that, if disturbed sufficiently, will become severely acidified, and sulfuric soil materials that have already become severely acidic as a result of the oxidation of RIS minerals.

Sulfidic materials may be current or former marine and estuarine sediments, sediments in brackish lakes and lagoons, peats that originally formed in freshwater but which have been inundated subsequently by brackish water, or accumulations of sediment in water bodies such as drains or wetlands affected by salinity (especially when sulfate is an appreciable component of that salinity). The required conditions for the formation and accumulation of RIS are: (1) a supply of organic matter, (2) reducing conditions sufficient for sulfate reduction brought about by continuous waterlogging, (3) a supply of sulfate from tidewater or other saline groundwater or surface water, (the sulfate is reduced to sulfides by bacteria decomposing the organic matter), and (4) a supply of iron from the sediment for the accumulation of iron sulfides which make up the bulk of the RIS.

These conditions are found in tidal swamps and salt marshes where, over the last 10,000 years, thick deposits of sulfidic clay have accumulated in many locations around the globe (Pons and van

Breemen 1982; Dent and Pons 1995). Sulfidic layers vary greatly in appearance but often have the gleyed colours typical of soil materials that are dominated by reduced waterlogged conditions.

Disturbance of sulfidic soils by, for example, drainage or excavation often causes dramatic changes in the properties of these soil materials and the draining waters. If there are insufficient effective neutralising materials (such as fine-grained calcium carbonate) in the sediment to neutralise the acidity generated by the oxidation of sulfides, extreme acidity can develop within weeks or months, resulting in sulfuric soil material. Sulfuric soil material is characterised by acidic pHs (e.g. pHs < 4), and usually presents yellow segregations of jarosite around pores and on ped faces. Acid sulfate soils of peaty constitution do not usually have visible jarosite segregations, presumably because these soil materials contain only minor amounts of the phyllosilicate clays that act as the main source, upon acid dissolution, of the potassium (K^+) necessary for jarosite precipitation.

Acid sulfate soil drainage waters can often have pH < 3.5 and can be the cause of massive fish kills, the death of invertebrates and benthic organisms, the development of chronic fish diseases, and impaired fish recruitment (Sammut *et al.* 1993).

Acid sulfate soils can also present health hazards to people living in landscapes containing these soils. Ljung *et al.* (2009) found that acid sulfate soils could impact detrimentally on human health. The human health issues were related mainly to the increased mobility of acid and metals from these soils affecting drinking water quality, food production and quality, but also to other issues such as increased dust generation causing respiratory health issues and acidic pools of surface water in acid sulfate soil landscapes providing suitable environments for mosquito breeding.

3.1.3 Occurrence

Estimates of the extent and distribution of acid sulfate soils globally suffer from scant field surveys, inadequate laboratory data, and also the lack of uniform, widely accepted definitions of these materials. Improvements in these areas have, however, led to better quantification of their extent and, in Australia at least, to better mapping of their distribution. The recent Australian Atlas of Acid Sulfate Soils (Fitzpatrick *et al.* 2008b) has greatly improved our understanding of the extent and distribution of acid sulfate soils within Australia.

The location of these soils is even more significant than their extent. Acid sulfate soils are often concentrated in otherwise densely settled coast and floodplains where development pressures are intense and little suitable alternative land exists for the expansion of farming or urban and industrial development. Recent studies have shown acid sulfate soils are widely distributed within the Lower Lakes region of South Australia (e.g. Fitzpatrick *et al.* 2008a; Simpson *et al.* 2008; Sullivan *et al.* 2008, 2010a).

Although acid sulfate soils are often thought of as almost exclusively a coastal issue, acid sulfate soils are also widely distributed in inland areas wherever the general conditions for RIS formation - a ready source of sulfate, iron, and organic matter in reducing waterlogged sediments - are met. In Australia, the large areas affected by human-induced salinity caused by over-clearing of trees and sub-optimal irrigation practices have also been found to be areas affected by the contemporary formation of acid sulfate soil materials (Fitzpatrick *et al.* 1996; Sullivan *et al.* 2002; Fitzpatrick *et al.* 2009).

3.1.4 Analysis

Quantitative methods of analysis are required to support soil survey programs and to provide essential data for modelling the likely response of the land to management options. The required analyses must either be performed in a timely fashion before gross chemical changes take place, or the samples must be preserved quickly by methods such as rapid oven drying or ideally freezing, otherwise, the pH may fall markedly to < 4 within days or weeks.

The methods of sampling, sample preparation, and analysis of acid sulfate soil materials vary widely according to the purpose of the study and the corresponding properties required. The methods of analysis vary from standard wet chemical methods (an authoritative, readily-available reference for these methods is Ahern *et al.* (2004)), standard soil physical methods for properties such as texture, hydraulic conductivity, and bulk density, to X-ray diffraction, X-ray fluorescence, analytical electron microscopy, through to advanced synchrotron-based techniques. In terms of management of acid sulfate soil materials, the Acid-Base Accounting approach has significant advantages over other routine analytical approaches as discussed below.

3.1.5 Acid-Base Accounting

The long period of time usually required to gain results from the incubation method (i.e. at least several weeks) along with the need for quantitative analyses for calculating liming rates, has resulted in the development of an Acid-Base Accounting (ABA) approach (Ahern *et al.* 2004). The ABA approach has gained widespread use for acid sulfate soil management. One of the benefits of ABA is that it is quantitative and provides data on acidification hazard that are suitable for purposes such as acidity hazard prioritisation, determination of liming requirements prior to oxidation, verification of liming quantities post treatment, etc.

While many ABA approaches have been used for acid sulfate soil assessment, they all share a common underlying principle of whereby the acidity hazard is the difference of alkalinity sources from the acidity as shown below:

$$\text{Acidity Hazard} = \text{Acidity} - \text{Alkalinity} \qquad [3.1]$$

There are several sources of acidity and alkalinity in soil materials and in practice the determination of several of these acidity sources are determined and expressed separately in the ABA. A commonly used ABA for acid sulfate soils is the one of Ahern *et al.* (2004) as shown below:

$$\text{Net Acidity} = \text{Potential Sulfidic acidity} + \text{Existing Acidity} - \text{Acid Neutralising Capacity} \qquad [3.2]$$

In this ABA the Existing Acidity is defined as follows:

$$\text{Existing Acidity} = \text{Actual Acidity} + \text{Retained Acidity} \qquad [3.3]$$

The Net Acidity in this ABA represents the acidity hazard of the soil material. The Potential Sulfidic Acidity refers to the potential for acidity to develop from oxidation of pyrite and is estimated from the RIS determination that assumes both the RIS to be pyritic sulfur and that the following overall oxidation reaction occurs to completion (i.e. one mole of pyrite produces 4 moles of (H^+) acidity as shown in Eq. 3.4).

$$FeS_2 + 3.75O_2 + 3.5H_2O \rightarrow Fe(OH)_3 + 4H^+ + 2SO_4^{2-} \qquad [3.4]$$

The Existing Acidity comprises both Actual Acidity (readily available soluble and exchangeable acidity) and Retained Acidity (the more slowly available acidity contained within minerals such as jarosite and schwertmannite).

Actual Acidity is a measure of the readily available soluble and exchangeable acidity in the soil material.

Retained Acidity is a measure of the more slowly available acidity contained within minerals such as jarosite and schwertmannite: the acidity in these minerals represents incomplete oxidation (cf. the equation above for pyrite oxidation). This acidity can be realized and released when these minerals decompose and this process can be a major source of acidity in waterways draining. For example, one mole of jarosite releases 3 moles of acidity as described by the following reaction:

$$KFe_3(SO_4)_2(OH)_6 + 3H_2O \rightarrow 3Fe(OH)_3 + 2H_2O + 3H^+ + K^+ \qquad [3.5]$$

The Acid Neutralising Capacity (ANC) refers to the effective neutralizing sources. In acid sulfate soils acceptable sources of ANC include calcium and magnesium carbonates, exchangeable alkalinity, organic matter but sources of buffering that do not act above pH 6.5 are usually considered ineffective (Ahern *et al.* 2004).

In Australia, the minimum critical Net Acidity levels that initiate the development of detailed management plans should acid sulfate soil materials be disturbed, vary according to soil texture and the amount of soil disturbed. For sandy soil materials, and where large amounts of soil are to be disturbed, the critical Net Acidity level is > 0.03 %S (or alternatively > 18 moles (H^+) tonne^{-1} when expressed as acidity) (Ahern *et al.* 2004).

Limitations of the ABA approach stem from our incomplete understanding of the acidifying and neutralising processes that take place in these soil materials, and from the lack of reliable methods available currently to quantify effective acidity and acid neutralising capacity. These limitations include:

1) It is not clear what proportion of the potential acidity capable of being produced by pyrite oxidation eventually becomes expressed. Similarly, our understanding of the kinetics of acidification processes is limited.

2) The currently available methods for quantifying acid neutralizing capacity in acid sulfate soil materials require improvement to provide accurate determinations (Ahern *et al.* 2004). These methods may either overestimate or underestimate the 'real' acid neutralizing capacity due to a number of reasons, including:

    i. Overestimation may be due to the inclusion of finely-ground shell materials in the test sample deriving from large shell materials in the field samples. Large shell materials are generally ineffective as a neutralizing agent but if finely ground, such components will be included in the acid neutralising capacity determination inflating the true capacity of the original field samples (Ahern *et al.* 2004).
    ii. Overestimation of the acid neutralizing capacity may also result from the imposition of extremely low pHs (e.g. pHs < 2) when using acid back titration methods (Ahern *et al.* 2004) and hence the inclusion of acid neutralizing mechanisms such as clay mineral dissolution that may not occur in natural ASS environments with less extreme pHs.

The ABA method of Ahern *et al.* (2004) also includes a consideration of the potential acidity residing in minerals such as jarosite and schwertmannite. Whilst this is defensible on the basis that such acidity may pose an environmental hazard, the lack of an accurate quantification procedure for the content of these commonly occurring minerals (Vithana *et al.* 2013) provides another limitation that could lead to an overestimation or underestimation of the acidity hazard posed by acid sulfate soil materials.

Despite these limitations, the quantitative capability of the ABA method provides distinct advantages for the purposes of managing ASS materials over the other ASS identification methods used for soil taxonomic purposes.

3.1.6 Minerals and Reductive Processes

A defining characteristic of sulfidic acid sulfate soils is the presence of significant concentrations of RIS. RIS include iron disulfides (most commonly pyrite (FeS_2) (Pons 1973; Bloomfield and Coulter 1973; van Breemen 1973), lower amounts of other minerals such as monosulfides (e.g. Georgala 1980; Bush *et al.* 2000), greigite (Fe_3S_4) (Bush and Sullivan 1997) and elemental sulfur (S_8) (Burton *et al.* 2006a,b).

The vast majority of RIS in sulfidic acid sulfate soil materials have formed at earth-surface temperatures and pressures under waterlogged, anoxic conditions. Under such conditions, accumulation of RIS species depends on microbially-mediated sulfate reduction, which is itself dependent on organic carbon availability, supply of sulfate, and on the amount of competing electron acceptors including reactive Fe^{III} minerals (Fanning *et al.* 2002). (Note in this report solid-phase species for components with a specific redox state are indicated by superscripted Roman numerals (e.g. Fe^{III}), and individual species in solution are shown with a charge (e.g. Fe^{3+})). These variables influence the activity of dissimilatory sulfate-reducing microorganisms, which include phylogenetically diverse anaerobes that oxidise simple organic compounds or hydrogen using sulfate as an electron acceptor. The overall process of dissimilatory sulfate reduction can be shown, for example, by:

$$CH_3COO^- + SO_4^{2-} + H^+ \rightarrow H_2S + 2HCO_3^- \qquad [3.6]$$

During this process, the sulfur in sulfate is reduced from the S^{6+} oxidation state to S^{2-}. Conditions that are conducive to microbially-mediated sulfate reduction occur in organic-rich coastal and estuarine sediments, such as in tidal marshes and swamps. In such systems, tidal exchange of pore-water supplies sulfate and removes the resultant HCO_3^- produced via the reaction in Eq. 3.6. Tidal flushing thereby prevents the accumulation of pore-water alkalinity. In iron-deficient systems, this tidal flushing can also remove pore-water H_2S and lead to its subsequent oxidation to elemental S (and eventually to sulfate).

In contrast, in soils containing Fe^{2+}, often produced by the activity of ferric iron reducing microorganisms, H_2S may react rapidly to form monosulfide (FeS) precipitates as below:

$$H_2S + Fe^{2+} \rightarrow FeS + 2H^+ \qquad [3.7]$$

The initial FeS phase to form by reaction between H_2S and Fe^{2+} (Eq. 3.7) has proved difficult to characterise, even in well-defined synthetic studies (Rickard and Morse 2005). Recently, such studies have shown that nanoparticulate mackinawite (tetragonal FeS) is the first condensed phase to form through this reaction. In acid sulfate soil materials the occurrence of mackinawite as 5 – 30 nm nanoparticles has been only recently demonstrated (Burton *et al.* 2009). The strong black colour seen in some of these acid sulfate soil materials is largely due to the presence of nanoparticulate mackinawite (Burton *et al.* 2009).

The H_2S produced by microbial sulfate reduction can also react with Fe^{III} contained in ferric oxide and oxyhydroxide minerals such as goethite, to produce elemental sulfur:

$$H_2S + 2FeOOH + 2H^+ \rightarrow S_8 + 2Fe^{2+} + 3H_2O \qquad [3.8]$$

The Fe^{2+} produced via this reaction may then feed into the reaction described by Eq. 3.6 thus also resulting in mackinawite formation. This overall process, termed “sulfidisation” can be represented as:

$$3H_2S + 2FeOOH \rightarrow S_8 + FeS + 4H_2O \qquad [3.9]$$

In the presence of an oxidant, such as O_2, mackinawite is unstable and can transform readily via a solid-state process to greigite:

$$4FeS + 0.5O_2 + 2H^+ \rightarrow Fe_3S_4 + Fe^{2+} + H_2O \qquad [3.10]$$

Although frequently mentioned, there are only few studies (e.g. Bush and Sullivan 1997) that conclusively document the occurrence of greigite in acid sulfate soil materials. On the basis of the limited amount of field data it appears that greigite occurrence is limited to the oxidation front in mildly acidic soils that are subject to an oscillating groundwater table. Mackinawite and greigite are often described as “iron-monosulfide” minerals because they have an Fe:S ratio that is close to 1:1 (Rickard and Morse 2005). These mineral species are defined analytically by their dissolution in HCl to yield H_2S gas and described as acid volatile sulfide (AVS).

Both mackinawite and greigite have long been implicated as precursors to the formation of iron-disulfides such as pyrite and marcasite. For example:

$$Fe_3S_4 + 2H^+ \rightarrow FeS_2 + Fe^{2+} + H_2 \qquad [3.11]$$

Pyrite can also form without the need for precursory greigite via (1) mackinawite oxidation by polysulfide species (Rickard 1975; Luther 1991) and (2) mackinawite oxidation by H_2S (Rickard 1997; Rickard and Luther 1997). These two pathways of pyrite formation, which involve an intermediate dissolved FeS cluster complex, can be represented overall as:

Polysulfide pathway: $FeS + S_n^{2-} \rightarrow FeS_2 + S_{n-1}^{2-}$ [3.12]

Hydrogen sulfide pathway: $FeS + H_2S \rightarrow FeS_2 + H_2$ [3.13]

Whilst iron monosulfides are widely believed to be an essential precursor to pyrite formation, this is not necessarily always the case. Pyrite can form quite rapidly in the presence of suitable reactive surfaces such as bacterial surfaces (Canfield *et al.* 1998) that serve to overcome a significant supersaturation threshold by providing heterogeneous nucleation sites. Other suitable reactive surfaces include pre-existing pyrite crystals or organic substrates, such as plant material. Accumulation of pyrite in soil can occur rapidly under suitable field conditions (Howarth 1979; Rosicky *et al.* 2004a).

Pyrite is by far the most commonly observed RIS species in sulfidic acid sulfate soil materials. In these materials, pyrite presents a range of distinct crystal morphologies. The most remarkable of these morphologies are framboids (from the French term for raspberry – *framboi*s). Pyrite framboids consist of spheroidal aggregates of densely packed, individual microcrystals. Earlier research into the origin of pyrite framboids in sediments pointed towards either a bacterial influence or the magnetic aggregation of precursor greigite crystals. However, it now seems that the formation of framboids is more likely a function of the degree of solution supersaturation with regard to pyrite.

Whilst pyrite is normally the most abundant iron-disulfide in acid sulfate soil materials, marcasite (orthorhombic FeS_2) may occur in specific situations. Acidic conditions (pH < 6) are required for the initial formation of marcasite instead of pyrite. Such conditions occur in waterlogged soils and sediments that are rich in dissolved organic acids, capable of buffering the low pH. For example, marcasite is a common iron sulfide in some peaty acid sulfate soil materials in eastern Australia (Bush *et al.* 2004a).

3.1.7 Minerals and Oxidation Processes

Pyrite and other iron-sulfide minerals can persist in soils only under anoxic, waterlogged conditions. If these conditions become oxic by, for example excavation of the soils, the iron-sulfide components can undergo a series of oxidation reactions. For example, in the presence of oxygen (and water) pyrite oxidises to ultimately yield sulfuric acid and a poorly soluble Fe^{III} precipitate:

$$FeS_2 + 3.75O_2 + 3.5H_2O \rightarrow Fe(OH)_3 + 4H^+ + 2SO_4^{2-} \qquad [3.14]$$

While this reaction shows that exposure to oxygen under moist conditions is the driving force for pyrite oxidation, it neglects the great complexity of reaction steps in the overall oxidation process. This complexity includes a number of possible final iron phases as well as the formation of intermediate sulfoxyanions and elemental S. Chemolithotrophic Fe- and S-oxidising bacteria play an important role in mediating various steps in the overall oxidation process, and in determining the formation and persistence of intermediate S species.

A wide variety of potential phases play a role in determining the iron biogeochemistry following pyrite oxidation. Ferrous iron released in the initial stages of pyrite oxidation may precipitate as Fe^{II} hydroxysulfate minerals (Fanning *et al.* 2002), most importantly melanterite, rozenite and szomolnokite. These phases are readily soluble and are rarely observed in acid sulfate soil materials.

Under continuation of oxidising conditions, the Fe^{2+} released by pyrite oxidation is also subject to oxidation to Fe^{3+}. Whilst the simple oxidation process consumes some acidity, the subsequent hydrolysis of the resulting Fe^{3+} leads to the liberation of acidity. At low pH (e.g. < 4), Fe^{3+} is sufficiently soluble that it may serve as a very effective electron acceptor driving further pyrite oxidation (Moses *et al.* 1987). For this reason, it has been often suggested that rate of Fe^{2+} oxidation to Fe^{3+} may be the rate-determining step in pyrite oxidation.

Partial oxidation of Fe^{2+} to Fe^{3+} can lead to precipitates of mixed valence Fe salts, such as copiapite. This phase is one of the “soluble salts” that may form in acid sulfate soils under prolonged dry conditions (Fanning *et al.* 2002). Dissolution of these minerals during rainfall events may cause a first-flush of stored acidity.

The Fe^{3+} produced via pyrite oxidation also commonly precipitates as a range of Fe^{III} bearing minerals. In acid sulfate soil conditions at pH < 3, and/or in the presence of abundant K^+, jarosite appears to be the predominant Fe^{III} phase, whereas in the pH range of 3 – 4, schwertmannite is an important Fe^{III} phase in acid sulfate soil landscapes (Bigham *et al.* 1992; Sullivan and Bush 2004). The widespread occurrence of schwertmannite in acid sulfate soils has only been confirmed relatively recently (Sullivan and Bush 2004).

Schwertmannite is metastable and over time transforms, via dissolution-reprecipitation, to form a range of Fe^{III} oxyhydroxides (Bigham *et al.* 1996). These include ferrihydrite, lepidocrocite and goethite, with the latter being most stable. The transformation of schwertmannite (a Fe^{III} oxyhydroxysulfate) to these Fe^{III} oxyhydroxides involves the hydrolysis of Fe^{III} and the liberation of acidity. As a consequence, schwertmannite transformation can suppress pH long after the initial source of acidification (i.e. pyrite) has been consumed.

The type of secondary minerals formed from the Fe released during pyrite oxidation determines to a large extent the amount of acidity expressed (Dold and Fontbote 2001). For example, if the released Fe precipitates as goethite or ferrihydrite from the Fe^{3+} produced by sulfide oxidation, then 3.0 moles of H^+ are formed for every mole of Fe^{3+} hydrolysed from pyrite. However, if hydrolysis is incomplete and jarosite is formed, only around 2 moles of H^+ is released for every mole of Fe^{3+} hydrolysed from pyrite (van Breemen 1976). If schwertmannite is formed then approximately 2.575 moles of H^+ is released for every mole of Fe^{3+} hydrolysed from pyrite (Piene *et al.* 2000). The ‘stored’ acidity in these two minerals is important as the Fe^{3+} in both jarosite and schwertmannite can undergo further hydrolysis and result in the release of acidity into the surrounding environment (Dold and Fontbote 2001; Sullivan and Bush 2004).

3.1.8 Pyrite Oxidation

The oxidation of FeS_2 depends on factors including the supply of O_2, the availability of water, and the physical properties of FeS_2. Pyrite oxidation generates acid and releases heat; consequently, the acidity and temperature of the surrounding solution will affect the overall reaction rates. The oxidation of FeS_2 in the environment is usually ultimately determined by the supply of O_2. Models describing FeS_2 oxidation are often based on the assumption that all other constituents required for the oxidation process are freely available except for O_2, which is supplied through the porous material from the atmosphere (Dent and Raiswell 1982; Davis and Ritchie 1986; Pantelis and Ritchie 1991; Bronswijk *et al.* 1993). The rate of pyritic oxidation is often assumed to be a linear function of the dissolved O_2 concentration (Bartlett 1973; Braun *et al.* 1974) but the Michaelis-Menton equation has also been adopted (Liu *et al.* 1987; Tan 1996).

Temperature, which influences both chemical and microbial oxidation, is an important factor in determining the oxidation rate of pyritic materials. Biological oxidation only occurs between 0°C to 55°C (optimum 25-45°C) (Lundgren and Silver 1980) but chemical oxidation can take place above this temperature. Jaynes *et al.* (1984) modelling acid generation in mine spoil, took account of rates of diffusion of both O_2 and Fe^{3+} and also the activity of the bacteria generating Fe^{3+}, which was estimated from available energy and deviations from ideal temperature, solution pH and O_2 concentration. Pantelis and Ritchie (1992) introduced a ceiling temperature (100°C) above which microorganisms cease to be effective as catalysts in FeS_2 oxidation. The influence of temperature on oxidation rate follows the empirical Arrhenius equation (Ahonen and Tuovinen 1991). Because the pyritic oxidation reaction is exothermic, temperature rises depending on the rate of reaction and thermal properties of the bulk soil.

3.1.9 Hazards from Acid Sulfate Soils

3.1.9.1 Acidification

Oxidation of RIS is the primary cause of the extreme acidification that characterises sulfuric acid sulfate soil materials. By definition, the pH of sulfuric acid sulfate soil is < pH 4 (or < 3.5 according to the particular soil taxonomy being employed) but values of pH < 3 in actively oxidising soils are frequently observed (Dent 1986). Such extreme acidification significantly alters the soil chemistry, and can render it hostile to plants and create a source of contamination to groundwater and surface water run-off. The acid produced can react with clay minerals and oxides to release silica and metal ions, principally aluminium, iron, potassium, sodium and magnesium (Nriagu 1978). Other ions such as metals and metalloids can also be released (van Breemen 1973; Sammut *et al.* 1996b; Åström 2000).

The impacts of severe acid sulfate soil acidification on agricultural crops have been well documented (Dent 1986). Many crop plants are highly sensitive to low pH soil conditions and acidification can greatly reduce yields and in extreme cases, cause complete crop failure. In addition, the formation of acidic secondary iron minerals such as jarosite and schwertmannite can significantly reduce the availability of nutrients such as phosphorus and nitrogen. Farmers have tried many different approaches to ameliorate acidity by techniques, such as the addition of neutralising agents, soil amendments, organic mulch and reconfiguring plant beds to enhance the leaching of acidic products from the soil (Dent 1986). Success in cropping acid sulfate soil landscapes is mixed and highly dependent on the initial degree of acidification and capacity of the specific crop types to tolerate acidic conditions. Acidity severely constrains farming on acid sulfate soils with some exceptions (White *et al.* 1997).

Aluminium toxicity is a significant issue linked to acid sulfate soil acidification for terrestrial plants (Dent 1986) and downstream aquatic flora and fauna (Sammut *et al.* 1996a,b). The solubility of Al is critically dependent on pH, only becoming soluble at environmentally significant levels at approximately pH < 5. Soluble aluminium affects plant growth primarily by disrupting root function and is a major concern for food production and agricultural income for rural and regional communities. Severe environmental impacts can occur when acidic Al-rich leachate from acid sulfate soil enters water bodies. The more acute ecological impacts of acid sulfate soil acidification in waterways include fish kills (Sammut *et al.* 1996a,b; Callinan *et al.* 2005), loss of native aquatic macrophytes and fauna followed by invasion by acid tolerant species (Sammut *et al.* 1996a), mass mortality of crustaceans and shell fish (Simpson and Pedini 1985), and loss of benthic communities (Corfield 2000). Sub-lethal exposure of fish to acidity has also been linked to an increased susceptibility to skin diseases (Callinan *et al.* 2005), whereas depletion of alkalinity has been linked to poor shell development in crustaceans (Dove and Sammut 2007).

A range of potentially longer-term impacts on aquatic ecosystems arising from acid sulfate soil leachate include: disturbance to fish reproduction and recruitment, acidity barriers to fish migration, decline of primary food web, reduction of species diversity, and long term habitat degradation (Sammut *et al.* 1996a,b). In assessing the likely impacts of acid sulfate soil acidification on downstream aquatic environments, it is necessary to consider the vulnerability of the aquatic ecosytems, the duration and frequency of acidification episodes, the potential intensity of acidification based on the properties and quantities of the acidic leachate.

3.1.9.2 Iron Mobilisation

Ferrous iron is a primary product of pyrite oxidation. At high pH values (pH > 7), Fe^{2+} is chemically rapidly oxidised to Fe^{3+} (Cornell and Schwertmann 2003). At lower pHs (i.e. pH < 4.5), the oxidation of Fe^{2+} to Fe^{3+} is catalysed by acidophilic lithotrophic bacteria such as *Acidithiobacillus ferroxidans* (Pronk and Johnston 1992), *Thiobacillus ferroxidans* and *Leptospirillium ferroxidans* (Johnson 1993). The oxidation of Fe^{2+} has direct environmental consequences arising from the liberation of acidity and the formation of secondary iron minerals that can control soil and water geochemistry.

Accumulations of iron minerals are ubiquitous in acid sulfate soil landscapes. The precipitation and mineralogy of secondary iron minerals has been reviewed in detail by Alpers and Nordstrom (1999) and Cornell and Schwertmann (2003).

Understanding the types of iron precipitates that form in acid sulfate soil landscapes during oxidation is important as particular iron mineral phases can exercise a major influence on the environment (e.g. Dold and Fontbote 2001; Sullivan and Bush 2004). In a study of surface iron precipitate accumulations associated with waterways in acid sulfate soil landscapes, Sullivan and Bush (2004) found schwertmannite was the dominant secondary iron mineral. The schwertmannite occurred as coatings on vegetation, accumulations in low depressions and as iron flocs adhering to surfaces in acidified waterways. The potential acidity within the schwertmannite was high, ranging between 1,900 - 2,580 mol H^{+} t^{-1}, indicating that the schwertmannite was a substantial intermediate store of acidity within these acid sulfate soil landscapes. The retained acidity within both schwertannite and jarosite have recently been included into the quantitative assessment of the net acidity of sulfate soil materials (Ahern *et al.* 2004).

Iron precipitates in the form of iron flocs within the water column also are known to directly affect gilled organisms, smother benthic communities and aquatic flora (Sammut *et al.* 1996a,b), diminish the aesthetic values of recreational waterways, and threaten estuarine and marine environments (Powell and Martens 2005). The accumulation of iron flocs has also been linked to contemporary sulfur cycling and the formation of monosulfidic black ooze (MBO) accumulations in acid sulfate soil affected waterways.

3.1.9.3 Metal and Metalloid Mobilisation

Mobilisation of metals and metalloids to soil pore-waters from acid sulfate soil can constitute a major environmental hazard (e.g. Åström *et al.* 2001; Burton *et al.* 2006c, 2008a). Metals that have been reported at levels exceeding accepted environmental protection thresholds in acid sulfate soil include Al, As, Ba, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Sb, V and Zn (e.g. Åström *et al.* 2001; Macdonald *et al.* 2004a; Burton *et al.* 2006c). Metals in natural soils occur within mineral phases or as charged ions or ionic complexes sorbed to reactive surfaces (Åström 1998; Fältmarsch *et al.* 2008; Claff *et al.* 2010). Acidification can greatly enhance the solubility of metals, promoting their subsequent release from mineral phases by dissolution or cation exchange. The pH dependence of metal release has received considerable attention (Sammut *et al.* 1996b; Wilson *et al.* 1999; Åström 2001; Preda and Cox 2001; Macdonald *et al.* 2004a; Simpson *et al.* 2010), and there are strong similarities in metal release within acid sulfate soil and acid mine drainage systems (Evangelou and Zhang 1995).

Numerous studies have documented the impacts from soluble metals on crop production (e.g. Dent 1986), terrestrial habitats (van Breemen 1973), and more recently, attention has turned to their impact on aquatic environments (Sammut *et al.* 1996a,b; Wilson *et al.* 1999; Johnston *et al.* 2004; Callinan *et al.* 2005). Gilled organisms are particularly vulnerable to soluble metals and metal mobilisation can lead to rapid mortality rates in these species (Simpson and Pedini 1985; Sammut *et al.* 1995; Sammut *et al.* 1996a,b). Studies of the effects of metals on shellfish (oysters) revealed longer term, more chronic impacts on their growth and survival (Dove and Sammut 2007). However, the longer term impacts of metal release from acid sulfate soils to surrounding aquatic environments are poorly understood. Although elevated metal concentrations can be toxic to both aquatic flora and fauna, the consequences of these conditions to algal and phytoplankton production are largely unknown, as is the potential for their bioaccumulation (Macdonald *et al.* 2004a).

Most reports on the impacts arising from metal release from acid sulfate soil focus on the consequences of metal mobilisation under oxic-acidifying conditions. However, metals can also be mobilised when sulfuric acid sulfate soils are subject to prolonged inundation, resulting in the development of anoxic reducing conditions. Acid sulfate soils occur in low-lying floodplain environments and therefore, are subject to periodic water logging and oscillating redox conditions. The processes of metal mobilisation and behaviour of metals is very different under these conditions. The behaviours of iron and arsenic are a good example of metal mobilisation from acid sulfate soil materials following inundation.

Accumulations of iron minerals in acid sulfate soils are often concentrated at the ground surface and include goethite, ferrihydrite, jarosite and schwertmannite. These iron minerals often have a large surface area and are a significant sink for the sorption of metals. Under reducing conditions, these iron oxides are prone to microbial reductive dissolution (van Breemen 1973; Burton *et al.* 2007). Microbial iron reduction triggers three major changes that affect metal mobilisation. Firstly, it results in the dissolution of Fe^{3+} and transformation to Fe^{2+}, causing the co-release of other metals sorbed to the Fe mineral surfaces. Secondly, the microbial reduction process is proton-consuming and when accompanied by the formation of bicarbonate as a by-product of microbial respiration, can result in *in situ* neutralisation (Blodau 2006). The increase in pore-water pH generally reduces the solubility of divalent metals and aluminium. It also facilitates the recently identified Fe^{2+} catalysed transformation of poorly crystalline iron oxide minerals to more crystalline phases (e.g. rapid transformation of schwertmannite to goethite).

Although the overall consequences of these rapid mineral transformations on metal mobility are yet to be quantified (Burton *et al.* 2010), the mobility of some metals and metalloids can increase under these conditions. For example, arsenic is most soluble at around pH 5 and when associated with iron oxides in acid sulfate soil materials, is readily mobilised at the onset of microbially-mediated iron reduction (Burton *et al.* 2008a). Severe arsenic contamination of groundwater and surface water is occurring as the result of such processes in acid sulfate soil landscapes, such as parts of the Mekong delta. It is important to recognise that metals and metalloids can have a significant impact in acid sulfate soil landscapes both 1) when acid sulfate soil are allowed to oxidise and acidify, but 2) also following the prolonged inundation of previously oxidised, iron-enriched acid sulfate soil.

Previous studies of metal mobilisation of lake sediments (Sullivan *et al.* 2009) have demonstrated the capacity of these materials to mobilise elevated concentrations of Ni, Zn and Mn within these sediments. These studies also clearly highlighted the dynamic behaviour of these materials over a prolonged period (i.e. 130 days) of inundation. Simpson *et al.* (2010) found that Al, Fe, Cu, Ni, V, and Zn may be rapidly mobilized (i.e. within 24 hours) by re-wetting exposed Lower Lakes sediments. The rate and extent of release of these metals depended strongly on the pH of those sediments with the lower the pH the greater the release of metals.

3.1.9.4 Deoxygenation of Waterbodies

Acute deoxygenation of estuaries, lakes, rivers and drainage channels is a major contributor to catastrophic fish kills (Johnston *et al.* 2003; Howitt *et al.* 2007; Hamilton *et al.* 1997). Many potential factors contribute to deoxygenation events, and they are known to impact a very wide range of environments. Severe deoxygenation of waterways within acid sulfate soil landscapes have been linked directly to the behaviour of acid sulfate soil materials (e.g. Sullivan and Bush 2000).

Deoxygenation results when solids and aqueous compounds with a capacity to react with dissolved oxygen, enter water bodies and consume oxygen more rapidly than it can be replenished. The magnitude of deoxygenation depends on the spatial scale of the event, its persistence and its intensity. Aquatic ecosystems require dissolved oxygen concentrations generally greater than 85% saturation for lowland rivers (e.g. ANZECC/ARMCANZ 2000). Native fish and other large aquatic organisms are known to survive on dissolved oxygen concentration of as little as 2 mg L^{-1}, but may become stressed below 4 - 5 mg L^{-1} (Hladyz and Watkins 2009). In recent studies of a major estuarine river system in Eastern Australia affected by deoxygenation, Wong *et al.* (2010) found deoxygenation was confined to downstream acid sulfate soil confluences and occurred during the later phase of the flood recession.

Anaerobic decomposition of floodplain vegetation in backswamps can be a primary process leading to the deoxygenation of large volumes of waters in acid sulfate soil landscapes (e.g. Johnston *et al.* 2003; Wong *et al.* 2010). Decomposition of flood-intolerant vegetation in drained acid sulfate floodplains can lead to the formation of “blackwater” - a colloquial term used to describe anoxic stagnant floodplain water that develops a distinctive dark colour as a result of the accumulation of dissolved organic carbon compounds. Blackwater is typically anoxic, has a high chemical oxygen demand (COD) and high dissolved Fe concentrations, and rapidly consumes dissolved oxygen when it discharges to main water bodies (Johnston *et al.* 2003). Extensive floodplain drainage networks in acid sulfate soil areas can significantly enhance the transport of hypoxic backswamp blackwater to main river channels, thereby enhancing the magnitude and duration of consequent deoxygenation.

The propensity for monosulfidic black ooze (MBO) to accumulate and be mobilised by floodwaters in drainage channels has also been identified as a contributing factor to deoxygenation in acid sulfate soil areas (Sullivan *et al.* 2002; Bush *et al.* 2004b,c; Burton *et al.* 2006b,d).

The chemistry of surface waters during hypoxic events has indicated elevated concentrations of redox sensitive species associated with acid sulfate soil (e.g. Fe^{2+}, dissolved Mn, and elemental sulfur) (Wong *et al.* 2010), further implicating acid sulfate soil and MBO materials in deoxygenation events.

The role of MBO in deoxygenation and latter acidification in acid sulfate landscapes has only recently been discovered (Sullivan and Bush 2000; Sullivan *et al.* 2002). Burton *et al.* (2006c) have described the oxidation dynamics of MBO when mobilised into oxygenated water. The oxidation of MBO follows a two step process with oxygen consumption occurring with each step (after Burton *et al.* 2006c):

Step 1

$$FeS \begin{cases} Fe^{2+} + 0.5O_2 + 1.5\,H_2O \rightarrow 2H^+ + FeOOH \\ S^{2-} + 0.5O_2 + 2H^+ \rightarrow H_2O + 0.125S_8 \end{cases} \qquad [3.15]$$

Step 2

$$0.125S_8 + 1.5\,O_2 + H_2O \rightarrow SO_4^{2-} + 2H^+ \qquad [3.16]$$

The first step is a rapid chemical reaction of iron monosulfide minerals with oxygen, forming iron oxides and elemental sulfur. This initial oxygen-consuming step does not affect pH and is therefore non-acidifying. It is probably for this reason that the role of MBO in deoxygenation was overlooked until recently. Acidification associated with MBO oxidation can result from the second step, the microbially-mediated oxidation of elemental sulfur, when oxygen is available.

Elevated elemental sulfur concentrations in deoxygenated waterways in acid sulfate soil landscapes may be a useful indicator of MBOs as a contributing cause to deoxygenation, although elemental sulfur can also form as a primary product of H_2S oxidation, and may be present within MBOs prior to flood events (Burton *et al.* 2006a,b).

The presence of MBO acid sulfate soil materials in the Lower Lakes has been identified in several reports (e.g. Fitzpatrick and Shand 2008; Fitzpatrick *et al*. 2008c; Sullivan *et al*. 2008). It is very likely on the basis of the data available and given the shallow nature of these lakes that episodic localised deoxygenation events may occur in areas where MBOs are concentrated, due to mixing of these sediments with the waters of the lakes.

3.1.9.5 Production of Noxious Gases

Anthropogenic and biogenic sulfur-containing gases have important impacts on global climate change (Charlson *et al.* 1987; Lohmann and Feichter 2005), and atmospheric acid-base chemistry (Berresheim *et al.* 1995). Coastal estuarine and marine environments are major emitters of biogenic H_2S (Aneja 1990; Bates *et al.* 1992). Emissions of H_2S, and more recently sulfur dioxide (SO_2), from floodplains have been linked to acid sulfate soil management (Macdonald *et al.* 2004b).

Hydrogen sulfide is a highly noxious gas that causes distress to humans (Luther *et al.* 2003; EPA 2003) and threatens aquatic organisms (Diaz and Rosenberg 1995; Rabalais 2002). As described by Equation 3.6, H_2S is produced by sulfur-reducing bacteria under anoxic conditions. Even at small concentrations, H_2S can be detected by its characteristic rotten-egg odour. In acid sulfate soil landscapes, periodically inundated soil surfaces, shallow waterways and field drains where stratified anoxic conditions can develop, are all situations conducive to sulfate reduction and the formation of H_2S (Dent 1986). However, H_2S is an unstable phase and its persistence in water and soil and ultimate gaseous emission is highly constrained by a wide range of oxidants in natural sediments and water bodies (Jørgensen *et al.* 1991). These oxidants include O_2, NO_3, Mn and Fe oxyhydroxides (Froelich *et al.* 1979; Luther *et al.* 1997). Due to their abundance in acid sulfate soil, iron oxides (Millero *et al.* 1987) are a particularly effective oxidant of H_2S, a process that can lead to the formation of iron sulfides as described previously. Hydrogen sulfide becomes a problem when the rate of its formation exceeds the catalytic oxidative capacity of the sediments and water bodies to eliminate its gaseous emission. An excess of labile carbon and stagnant water bodies create conditions that favour H_2S emissions in acid sulfate soil landscapes (Rozan *et al.* 2002).

Partially oxidised RIS-containing acid sulfate soil materials are a known source of SO_2. Macdonald *et al.* (2004b) quantified SO_2 flux from agricultural acid sulfate soils using both ground chamber and micro metrological methods. In this study, the rates of SO_2 emission from the soil was closely linked to soil moisture and evaporative flux, leading the authors to conclude that acidic dissociation of sulfite (SO_3^{2-}) occurring within the near-surface soil pore-water was probably the major source of SO_2. The precise mechanisms for SO_2 formation in acid sulfate soil require resolution: bacterial processes that utilise sulfate (Saltzman and Cooper 1989) or organo-sulfur compounds (Freney 1961) are both possibilities. From relatively few measurements, Macdonald *et al.* (2004b) estimated global SO_2 emissions from acid sulfate soil to be 3.0 Tg S yr^{-1}, ~ 3% of global anthropogenic emissions.

3.1.9.6 Scalding of Acid Sulfate Soil Landscapes

Scalded (i.e. non-vegetated) land surfaces can be an extreme symptom of land degradation and in low-lying acid sulfate soil landscapes can extend for thousands of hectares, impacting the environment, and those who live and rely on these areas. Scalded acid sulfate soil land is environmentally damaging, agriculturally unproductive and difficult to rehabilitate. There are a multitude of causes for the complete and prolonged failure of vegetation to establish. In acid sulfate soil landscapes, extreme acidification and/or salinisation are often involved with the initiation of scalds (Rosicky *et al.* 2004a,b). Peat fires arising from the desiccation of low-lying backswamps can also lead to the formation of scalds, as can the prolonged inundation of low-lying areas with acidic-aluminium-iron rich and shallow surface waters.

The size and condition of scalds vary considerably, spatially and temporally. In a broad study of scalds along the east coast of Australia, Rosicky *et al.* (2004a), found that even relatively minor changes such as a shift to wetter conditions, could instigate the rapid growth of acid tolerant plants such as spike-rush (*Eleocharis acuta*). The establishment of such re-vegetation typically would advance from the edge of scald, only to die off and recede when drier conditions returned.

Rosicky *et al.* (2004a,b) found that the surface soil layers of scalds experienced extreme acidification (pH < 3), evaporative accumulation of acidic salts and metals (Al, Fe), high salinities caused by the accumulation of evaporative salts (e.g. gypsum), and accumulations of iron minerals (e.g. schwertmannite, ferrihydrite, goethite and jarosite). Combined with other stresses such as grazing pressure and frosts, such soil conditions generally prevent the long-term establishment of vegetation.

The primary goal for restoring scalds is to establish persistent vegetation. Strategies for revegetating scalds generally revolve around improving the surface soil layers by practical agricultural intervention. Techniques that have been demonstrated to work include: the exclusion of stock, the use of ridges and furrows, mulching, liming, addition of fertiliser, pre-treating seed with nutrients and neutralising agents, and more recently water management practices that create and maintain wetter conditions.

More recently, landholders have begun experimenting with watertable manipulation to provide more persistent wetter conditions to enable plant establishment on scalds. Excessive drainage is generally the most important primary driver of acid sulfate soil scald formation and strategies that reduce evaporation from bare areas and maintain or raise watertables in the near vicinity of scalds, can contribute to their restoration and revegetation. The shallow ponding of freshwater can trigger rapid and complete re-vegetation of scalds (Rosicky *et al.* 2004b).

Around the former coastal lake sediments of the Lower Lakes of the River Murray in South Australia, the extensive acid sulfate soil landscapes comprised of lake sediments exposed and acidified during droughts, large-scale revegetation programs have proven to be successful in ameliorating acidification and providing protection during re-inundation after lake refilling (Sullivan *et al*. 2011).

3.1.10 Inundation of Acid Sulfate Soils

Inundation has often been proposed to improve the water quality in acid sulfate soil landscapes (Dent 1986), however, the response of acid sulfate soils to submergence is reported to be highly variable (Ponnamperuma *et al.* 1973; Tuong 1993; Konsten *et al.* 1994; Johnston *et al.* 2005). In addition to aiming to prevent further sulfide oxidation, inundation often removes the acidity in partially-oxidised sediments as the acidity gets consumed from the reduction of iron (III) oxides, sulfates and other oxidised species by anaerobic bacteria (Dent 1986). In most moderate acid soils, reduction causes the pH to rise to approximately 7 within a few weeks. However, some acid sulfate soils may not reach a pH of more than 5 after months of submergence (Ponnamperuma 1972). Factors which have been identified as being responsible for slow reduction, and hence a slow increase in pH, include a low content of easily oxidisable organic matter, a low content of easily reducible iron, a low dissolved sulfate concentration, the adverse effect of low pH on activity of microbes, and a poor nutrient status (Ponnamperuma 1973; van Breemen 1976; Berner 1984).

While the increase in pH from reduction may improve water quality, recent studies have shown that the inundation of sulfuric soil materials from the Lower Lakes with freshwater was capable of mobilising high concentrations of contaminants (Simpson *et al.* 2008, 2010; Sullivan *et al.* 2008). The inundation of sulfuric soil materials from the Lower Lakes lead to the chemical reduction of iron minerals and caused the mobilisation of high concentrations of metals (i.e. Al, As, Cu, Mn, Ni, Ag, Cd, Cr, Co) and nutrients (i.e. NH_3, NO_X) (Sullivan *et al.* 2008). Sullivan *et al.* (2008) also found that while oxic suspensions of MBOs from the Lower Lakes did not result in acidification, there was still the mobilisation of various metals and nutrients to high concentrations.

A recent study by Sullivan *et al*. (2010a) examined the response of exposed Lower Lakes soils to rewetting with seawater and River Murray water. The study found the response of the inundating waters to the underlying soils varied considerably in terms of pH and alkalinity. While the inundation of most sediments did not appreciably acidify the inundating waters, inundation by seawater generally had a greater initial acidification effect than by River Murray water suggesting that the higher alkalinity of the seawater was insufficient (under the experimental conditions) to overcome the additional exchange of acidity from the lake soils caused by the higher salinity of the seawater.

By simulating inundation of Lower Lakes soil materials, Sullivan *et al*. (2010a) showed that the availability of organic carbon was a major limiting factor to sulfate reduction. Bioremediation of Lower Lakes sites commenced in 2009 through enhancing organic carbon availability and has now been supported as a realistic management option. The current study examines various revegetation methods aimed at increasing the availability of organic carbon so as to facilitate sulfate reduction and, consequently, enable improved management of acid sulfate soil materials in the Lower Lakes whilst achieving complementary environmental objectives.

3.2 Introduction to this Study

As a result of prolonged drought, combined with management practices upstream in the Murray-Darling catchment, the Lower Lakes of Lake Alexandrina and Lake Albert have recently experienced their first major drying phase since the introduction of barrages more than 50 years ago (Simpson *et al*. 2008; Sullivan *et al.* 2008). Concurrently, it was identified that the Lower Lakes were also being impacted by the presence of acid sulfate soil materials (Fitzpatrick *et al.* 2008a). As a consequence of unprecedented low water levels, extensive areas of acid sulfate soils were exposed in the Lower Lakes which resulted in soil acidification ($pH<4$) over large areas and localised acidification of surface waters (DENR 2010).

To inform management decision making, a research program was undertaken to fill critical knowledge gaps related to the risks posed by exposure of acid sulfate soils in the Lower Lakes (DENR 2010). The research areas examined in this program included:

- an acid sulfate soil spatial heterogeneity/mapping survey;
- measurement of acid generation rates;
- assessment of the in-situ contaminant generation, transport and neutralisation processes;
- laboratory and field studies of the potential for mobilisation of contaminants following inundation with seawater compared to river water ; and
- geochemical modelling of lake water quality.

A study by Sullivan *et al.* (2010a) examined the response of exposed Lower Lakes’ soil materials to wetting with seawater and river water. Among other key findings, Sullivan *et al.* (2010a) identified that the major factor limiting sulfate reduction in the Lower Lakes sediments was the availability of organic carbon. Given the potential importance of microbially-mediated sulfate reduction in relation to critical sediment/water aspects (e.g. the development of alkalinity in the sediments), confirmation that the availability of organic carbon in the Lower Lakes environment was a limiting factor, supported the approach undertaken by the South Australian government. The bioremediation of Lower Lakes’ sites via enhancing organic carbon availability was supported as a feasible management option.

Sullivan *et al*. (2011, 2012b) examined the effects of various bioremediation options carried out by the Department for Environment, Water, and Natural Resources (DEWNR) aimed at facilitating sulfate reduction and, consequently, remediation of often strongly acidified acid sulfate soil materials around the drought-exposed margins of the Lower Lakes. The results of these studies indicate that bioremediation of the exposed acidified lake sediments by certain types of vegetation produced substantial environmental benefits from a combination of vegetation-associated processes including the provision of alkalinity from plant roots, provision of organic carbon for sulfate reducing bacteria and the role of vegetation in minimising soil erosion and hence preventing further exposure of severely acidic subsoils that occurred under unvegetated sites.

The studies also highlighted that several of the likely future hazards associated with a strategy of enhancing organic matter input into sediments to stimulate sulfate reduction and the beneficial co-production of alkalinity, had been substantially avoided in the initial refilling period of the Lower Lakes (i.e. first 19 months), particularly where annual vegetation was too short to survive inundation. This hazard avoidance was due to the characteristic nature of the sulfur cycling occurring in these sediments, the consequent lack of accumulation in the surficial lake sediments of sulfide minerals such as monosulfides and pyrite and their associated hazards of acidification, metal and metalloid mobilisation, and deoxygenation.

However, the recent study by Sullivan *et al.* (2012b) showed when *Phragmites* (a species that survived lake re-filling and continued to grow vigorously when inundated) was used to bioremediate these sediments, there was considerable accumulation of both pyrite and monosulfide (as Monosulfidic Black Ooze (MBO)) in the uppermost sediment layers. These accumulated sulfides indicated that alkalinity had also been produced via sulfate reducing processes enabled by the ongoing production of organic matter by *Phragmites.* In addition, these uppermost sediments under *Phragmites* appeared likely to act as sources of soluble phosphate that could lead to increased nutrient flux/accumulation to lake water. The Sullivan *et al.* (2012b) study strongly indicated a number of potentially important hazards would have arisen if *Phragmites* were to be used for bioremediation of exposed lake sediments (e.g. increased accumulation of sulfides); such hazards were avoided almost completely when inundation intolerant vegetation was used.

It is recognised that 19 months of re-inundation is too short a time to adequately assess the longer term impact on the biogeochemistry (e.g. Sullivan *et al*. 2012b; Moreno-Mateos *et al.* 2012). This project aims to monitor the biogeochemical state (with respect to sulfate reduction and associated processes) of the Lower Lake sediments approximately 2.5 years after lake refilling, and 12 months after the last detailed research of these sediments (Sullivan *et al.* 2012b), to allow a more accurate assessment of the progression of remediation of these sediments according to bioremediation strategy.

The methodology followed in this project continues the general assessment and analytical strategy used in Sullivan *et al.* (2011, 2012b). Following this methodology allows maximum benefit in terms of assessing temporal trends by ‘building onto’ the existing knowledge of the biogeochemistry of these sediments. An additional component of this project is a preliminary scoping investigation to examine the likely mobility and uptake by vegetation of metals (particularly nickel and zinc) from the acidified lake sediments as affected by bioremediation.

3.3 Sampling Strategy

3.3.1 Sediment and Surface Water Collection

The sediment sampling strategy undertaken in both the previous studies by Sullivan *et al.* (2011, 2012b) and this project addresses contemporary conditions in the lakes and assesses sulfate reduction and alkalinity generation in the subsurface (i.e. below surface) sediments arising from leaching of soluble organic matter - derived from bioremediation - into the subsoil.

In this project sediments were collected from the same four study areas as sampled by Sullivan *et al.* (2011, 2012b). The four study areas around the Lower Lakes sampled included Waltowa, (east Lake Albert), Poltalloch (east Lake Alexandrina), Tolderol (west Lake Alexandrina) and Campbell Park (west Lake Albert). The locations of the sediment sampling study areas are shown in Figure 3-1.

In the initial study by Sullivan *et al.* (2011) nine treatment sites were examined in detail between May 2010 and February 2011. Eight of these sites were re-examined in March 2012 (Sullivan *et al.* 2012b) and March 2013. Only one site at the Poltalloch study area was sampled from March 2012 as the sites sampled in the initial study by Sullivan *et al.* (2011) yielded similar results. A summary of the treatments examined in this study are presented in Table 3-1.

Surface waters were collected for water quality determination from the four study areas at the time of soil sampling.

Table 3-1. Summary of the treatments examined at each study area in the Lower Lakes (March 2013).

Study Area	Treatment
Campbell Park	i. Scald (no bioremediation)
	ii. 2010 seeded with Bevy rye and *Puccinellia* bioremediation
Tolderol	i. Scald (no bioremediation)
	ii. 2010 planted *Juncus* into 2009 plantings of Bevy rye bioremediation
Poltalloch	i. 2009 plantings of Bevy rye bioremediation
Waltowa	i. *Phragmites* bioremediation
	ii. *Cotula* bioremediation
	iii. *Juncus* bioremediation

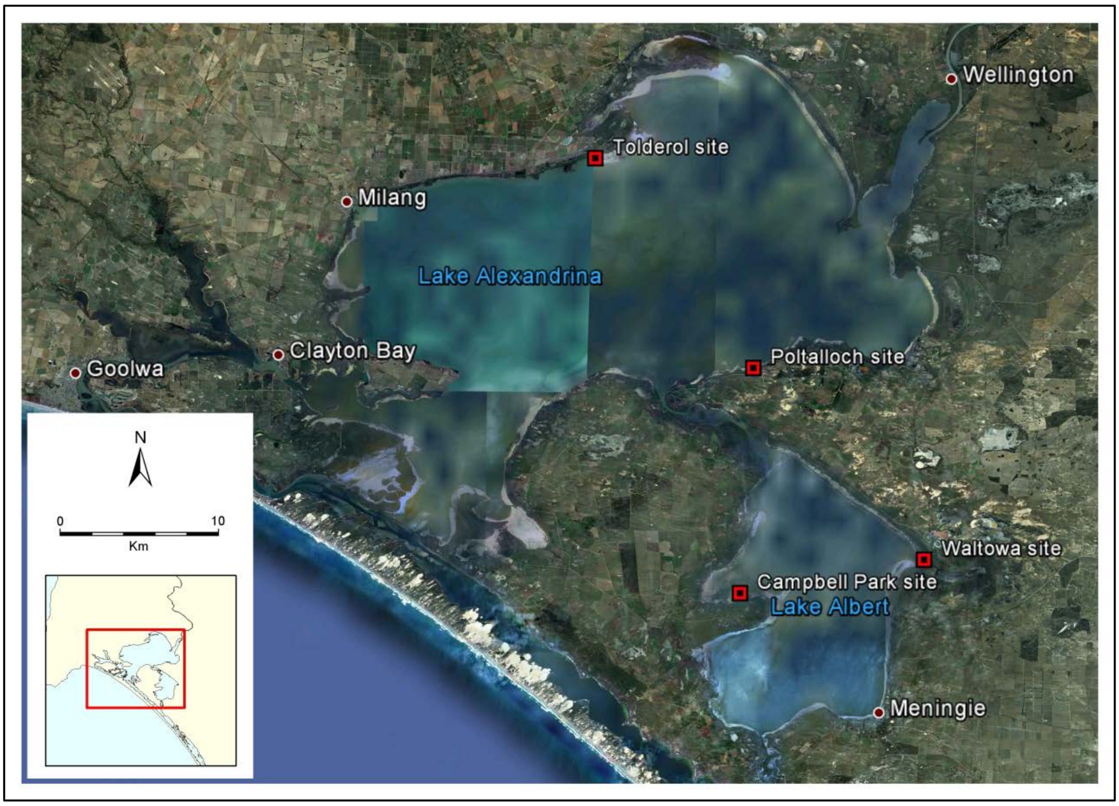

Figure 3-1. Map showing study areas in the Lower Lakes (Source: Google Maps).

Maps showing the sampling locations in each study area and selected photographs are presented in Sections 3.3.3.1 to 3.3.3.4. Bathymetry maps of each study area are also presented in Appendix 6 (Figures 9-18 to 9-21). Historical water level and salinity data for Lake Alexandrina and Lake Albert are also included in Appendix 6 (Figures 9-22 and 9-23).

3.3.1.1 Campbell Park, West Lake Albert Study Area Characteristics

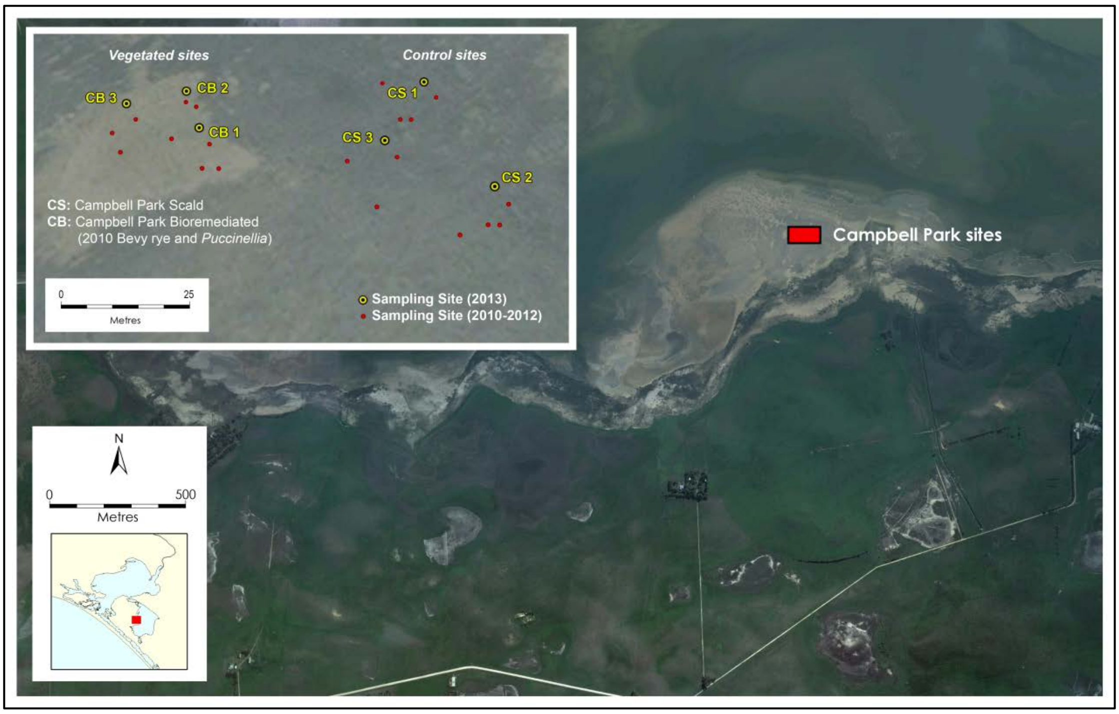

Figure 3-2. Campbell Park sampling locations (Source: Google Maps).

Figure 3-3. Water quality sampling at Campbell Park (March 2013).

3.3.3.2 Tolderol, West Lake Alexandrina Study Area Characteristics

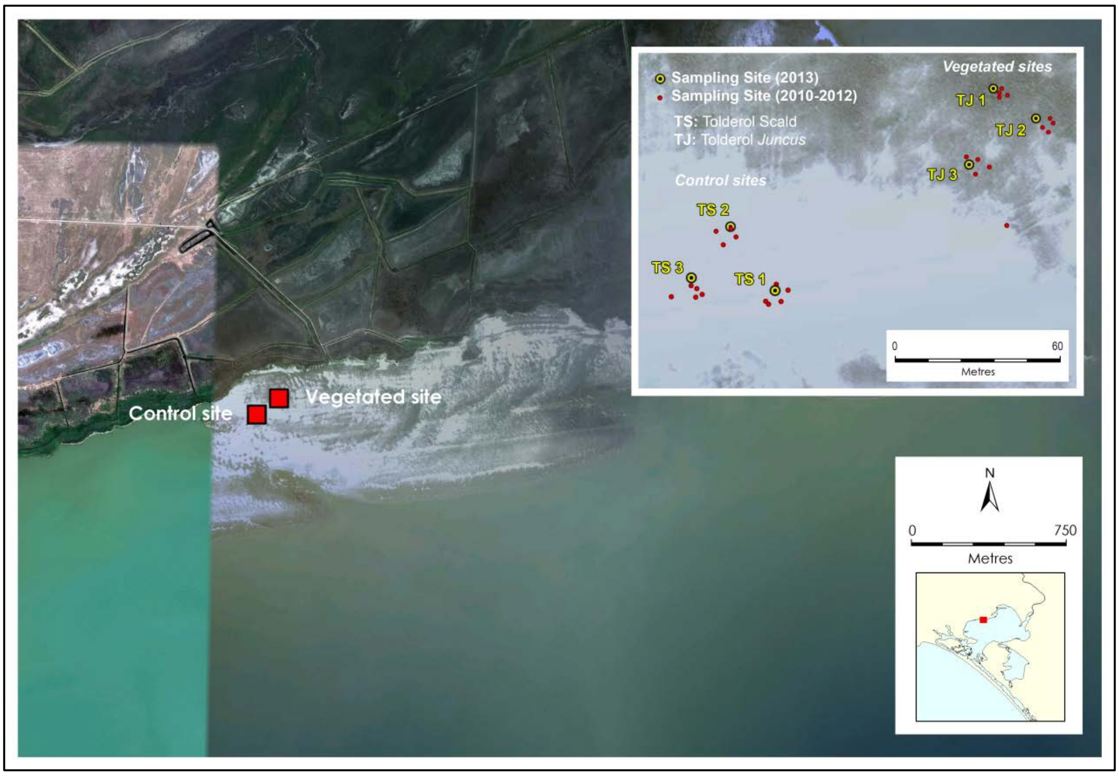

Figure 3-4. Tolderol sampling locations (Source: Google Maps).

3.3.3.3 Poltalloch, East Lake Alexandrina Study Area Characteristics

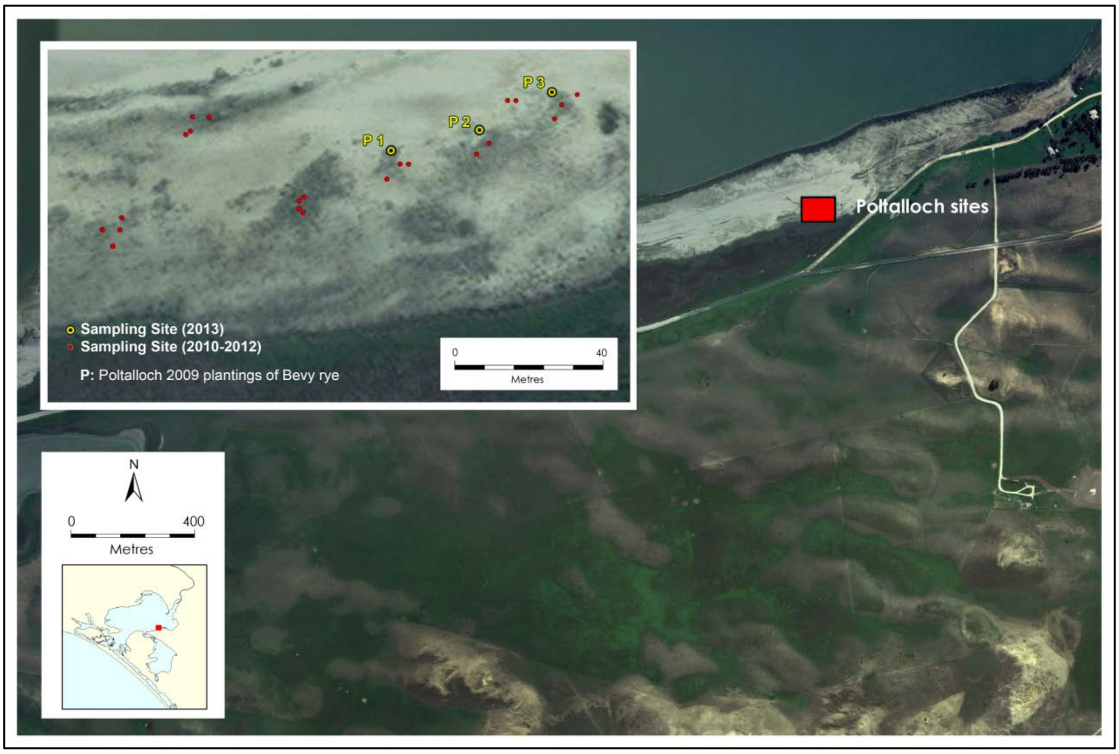

Figure 3-5. Poltalloch sampling locations (Source: Google Maps).

3.3.3.4 Waltowa, East Lake Albert Study Area Characteristics

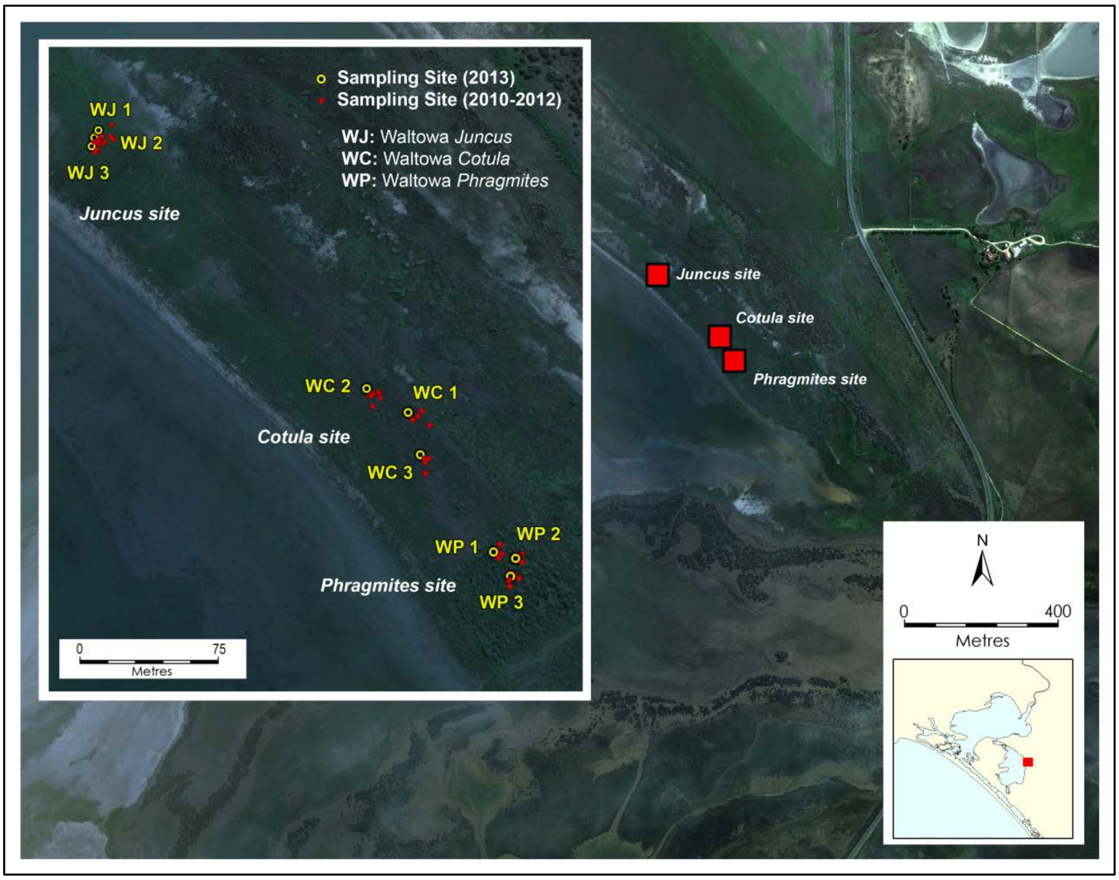

Figure 3-6. Waltowa sampling locations (Source: Google Maps).

Figure 3-7. Sediment sampling at Waltowa (March 2013).

Figure 3-8. *Phragmites* site at Waltowa (March 2013).

Figure 3-9. Sampling at the *Phragmites* site at Waltowa (March 2013).

3.3.2 Plant Material Collection

Plant materials (including leaves, stems and flowers) were collected from a total of 15 sites on the fringes of Lake Alexandria and Lake Albert that had been exposed during the recent drought. The locations sampled around the Lower Lakes in March 2013 are presented in Figure 3-10, and included plant materials collected from the four sediment study areas.

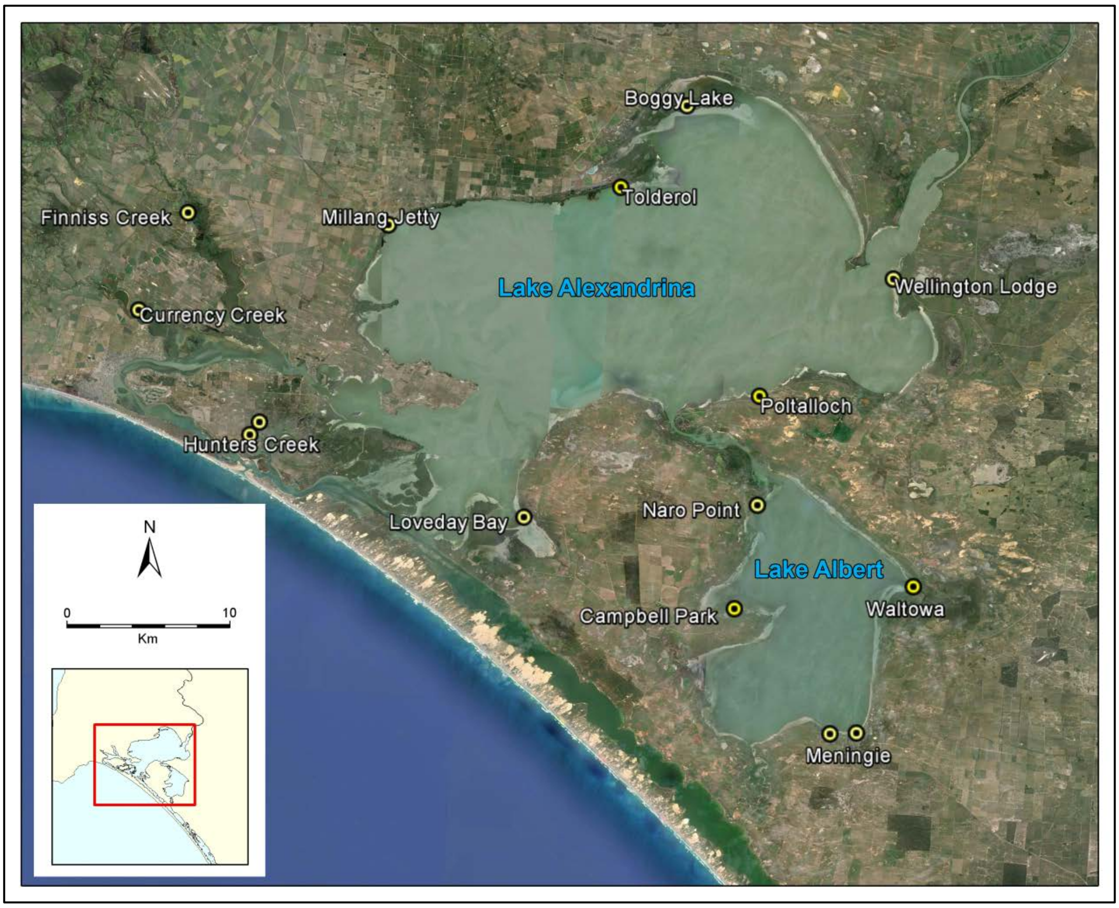

Figure 3-10. Vegetation sampling locations (Source: Google Maps).

A summary of the vegetation types collected from each site is presented in Table 3-2. At selected sites the same species of vegetation were collected from more than one location. Vegetation samples were collect from both the shoreline and further within the lake at Meningie (*Schoenoplectus valaidus)* and Naro Point *(Phragmites australis)*. *Phragmites australis* samples were also collected from the foredune (i.e. the dune that runs parallel to the shoreline of the lake) in addition to the inundated former scald at the Tolderol site.

Table 3-2. Summary of the vegetation types collected from each site around the Lower Lakes (March 2013).

Location	Vegetation Types Collected
Waltowa	*Phragmites australis*, bull rush, *Schoenoplectus valaidus* and unidentified *juncus*
Poltalloch	*Phragmites australis*
Campbell Park	*Phragmites australis* and *Schoenoplectus valaidus*
Tolderol	*Phragmites australis (*from both the foredune and the inundated former scald)
Wellington Lodge	*Phragmites australis* and *Schoenoplectus valaidus*
Meningie	Couch, shoreline T*ypha orientalis*, *Phragmites australis*, unidentified reed and *Schoenoplectus valaidus* (from shoreline and 50 m from shore)
West Meningie	*Phragmites australis* and sedge
Loveday Bay	*Phragmites australis*, *Schoenoplectus valaidus* and *Typha orientalis*
Naro Point	*Phragmites australis* (from shoreline and further in lake), *Schoenoplectus valaidus* and *Typha orientalis*
Boggy Lake	*Phragmites australis*, unidentified grass*, Typha orientalis* and unidentified rush
Milang Jetty	*Phragmites australis*
Currency Creek	*Phragmites australis* and *Schoenoplectus valaidus*
Hunters Creek	*Bolboschoenus* and *Schoenoplectus valaidus*
Finniss Creek	*Phragmites australis*, *Schoenoplectus valaidus* and *Typha orientalis*

Figure 3-11. *Schoenoplectus valaidus* next to the *Phragmites* site at Waltowa.

Figure 3-12. *Bolboschoenus* at the Hunters Creek site.

Figure 3-13. Bull rush next to the *Phragmites* site at Waltowa.

Figure 3-14. Insects eating *Phragmites* at Waltowa.

Figure 3-15. Insects eating *Phragmites* at Waltowa.

Figure 3-16. Insects eating *Phragmites* at Waltowa.

4.0 Materials and Methods

4.1 Field Sampling of Soils, Surface Waters and Vegetation

4.1.1 Field Sampling of Soils

Field sampling at the four Lower Lakes study areas was undertaken between 22nd and 24th March 2013. In the previous studies by Sullivan *et al.* (2011, 2012b), field sampling at the same study areas was undertaken before seeding/planting in May 2010, and then undertaken on four separate occasions (i.e. August 2010, November 2010, February 2011 and March 2012). A summary of the sampling dates for this and the previous studies are presented below in Table 4-1. Note a field sulfate reduction rate assessment was not undertaken in this study.

Table 4-1. Sampling dates for the field sulfate reduction rate assessment and soil profile sampling (May 2010 – March 2013).

Season (Date)	**Field Sulfate Reduction Rate Assessment**	**Soil Profile Sampling**
Late Autumn (21st - 23rd May 2010)		✓
Late Winter (28th - 31stAugust 2010)	✓	✓
Late Spring (21st – 24th November 2010)		✓
Late Summer (14th – 17th February 2011)	✓	✓
Mid Autumn (29th – 31st March 2012)	✓	✓
Mid Autumn (22nd – 24th March 2013)		✓

The sampling dates in the Sullivan *et al.* (2011) study were originally chosen to coincide with four growth stages of the annual vegetation to be planted during 2010: before planting, early-growth, near-maturity and post-maturity. However, flooding in the lakes during June-August 2010 impeded the establishment of the seeded/planted areas in Lake Alexandrina, and the development of the seeded areas at the Campbell Park study area beyond early-growth stage after the inundation of Lake Albert that occurred post-August 2010.

In this study quadruplicate intact sediment cores were collected using a 5 cm diameter push-tube coring device from three replicate sampling sites from each treatment/location to a maximum depth of 60 cm. Each core was collected within approximately 4 m of the initial site sampled in the previous studies (Sullivan *et al.* 2011, 2012b) to ensure that the detection of any changes in soil properties since the last sampling time was optimised. As observed in the previous Sullivan *et al.* (2012b) study, a surficial monosulfidic black ooze (MBO) was identified at one of the Waltowa *Phragmites* sites and was sampled separately. All sediment samples were frozen after sub-sampling and field measurements.

A soil description together with pH/Eh data for each horizon collected is presented in Appendix 1 (Table 9-1). The pH and Eh were determined using calibrated electrodes linked to a TPS 90-FLMV multi-parameter meter; Eh measurements are presented versus the standard hydrogen electrode. The global positioning system (GPS) coordinates for each site are also presented in Appendix 1 (Table 9-1).

4.1.2 Field Sampling of Surface Waters

Surface waters were collected for water quality determination from the four study areas at the time of soil sampling. Duplicate surface water samples were collected at a depth of approximately 30 cm using polypropylene containers that were rinsed three times prior to use. The surface water temperature, pH, redox potential (Eh), electrical conductivity (EC) and dissolved oxygen (DO) concentrations were determined using calibrated electrodes linked to a TPS 90-FLMV multi-parameter meter (Table 9-10, Appendix 3).

In addition, known volumes of filtered (0.45 µm) surface water samples were added to a variety of traps in the field for the laboratory determination of the concentration of iron species, dissolved sulfide and alkalinity (see Section 4.2.3 for further details). Filtered (0.45 µm) water samples were collected for further laboratory analysis of major cations, anions and nutrients, and stored frozen until analysis (see Section 4.2.3).

Individual unfiltered and filtered (0.45 µm) water samples were collected from each study area for the analysis of total organic carbon (TOC) and dissolved organic carbon (DOC), respectively. All samples collected for TOC and DOC analysis were acidified with concentrated sulfuric acid on collection and stored frozen until analysis (see Section 4.2.3).

4.1.3 Field Sampling of Vegetation

Plant materials were placed in sealed plastic bags and stored refrigerated until analysis. A comprehensive analysis of metals in the plant tissues (including leaves, stems and flowers) was undertaken (see Section 4.2.4).

The surface sediment pH was also measured at the time of sampling by direct insertion of a calibrated pH electrode into the sediment sample linked to a TPS WP-80 meter. Further details of the locations of the vegetation sampling sites, the types of vegetation collected and the sediment characteristics at each site are presented in Table 9-22, Appendix 5.

4.2 Laboratory Analysis Methods

4.2.1 General Comments

All laboratory glassware and plastic-ware were cleaned by soaking in 5% (v/v) HCl for at least 24 hours, followed by repeated rinsing with deionised water. Reagents were analytical grade and all reagent solutions were prepared with deionised water (milliQ). All solid-phase results are presented on a dry weight basis (except where otherwise noted).

4.2.2 Sediment Analyses

The parameters measured on the sediment/soil layers collected included:

- Moisture content
- Bulk density
- pH (1:5 soil:water)
- EC (1:5 soil:water)
- RIS (CRS, S(0) and AVS)
- Total C and N (by LECO)
- pH (1:40 soil: I.0 M KCl)
- TAA (only if pHKCl is <6.5)
- ANC (only if pHKCl is >6.5)
- TAAlk (only if pHKCl is >6.5)
- RA (only if pHKCl is <4.5)
- HCl extractable metals/metalloids
- Organic matter availability and quantity

The sediment moisture content was determined by weight loss due to drying at 105°C. The bulk density was determined following weighing a known volume of each sediment layer (70 cm^3) before and after oven-drying at 80°C. Sediments for further analysis (with the exception of sediments analysed for reduced inorganic sulfur (RIS) and sulfate reduction rates) were oven-dried at 80°C and sieved (< 2 mm) prior to being ring mill ground.

The acid volatile sulfide (AVS), elemental sulfur (S(0)) and pyritic sulfur fractions were determined using a sequential extraction procedure on duplicate frozen sub-samples. The AVS fraction was initially extracted via a cold diffusion procedure, with the use of ascorbic acid to prevent interferences from ferric iron (Fe (III)) (Burton *et al.* 2007). The solid phase S(0) fraction was extracted using methanol as a solvent and quantified by high-performance liquid chromatography (HPLC) (McGuire and Hamers 2000). The remaining RIS fraction (i.e. pyritic sulfur) was determined using the chromium reduction analysis method of Burton *et al.* (2008b).

Electrical conductivity (EC) and pH were determined by direct insertion of calibrated electrodes into a 1:5 soil:water extract linked to a TPS WP-81 meter. Total carbon (%C) and total nitrogen (%N) were measured on powdered oven-dried samples by combustion using a LECO-CNS 2000 analyser. The potassium chloride (KCl) extractable pH (pH_{KCl}) was measured in a 1:40 1.0 M KCl extract (Method Code 23A), and the titratable actual acidity (TAA) (i.e. sum of soluble and exchangeable acidity) was determined by titration of the KCl extract to pH 6.5 (Method Code 23F) (Ahern *et al.* 2004). Titratable actual acidity is a measure of the actual acidity in soil materials. The titratable actual alkalinity (TAAlk) was measured on samples where pH_{KCl} was >6.5 (Sullivan *et al.* 2010b). Titratable actual alkalinity where the suspension is titrated with 0.05 M hydrochloric acid (HCl) down to pH 6.5 is the reverse of the TAA method. The acid neutralising capacity (ANC_{BT}) was quantified on the <0.5 mm sieved soil fraction (only if pH_{KCl} is >6.5) using a standard back-titration determination (Method Code 19A2) (Ahern *et al.* 2004). The retained acidity (RA) was determined from the difference between 4.0 M HCl extractable sulfur (S_{HCl}) and 1.0 M KCl extractable sulfur (S_{KCl}) when the sample pH_{KCl} was < 4.5 (Method Code 20J) (Ahern *et al.* 2004). The retained acidity identifies stored soil acidity in the form of jarosite and similar relatively insoluble iron and aluminium hydroxy sulfate compounds (Ahern *et al.* 2004). The net acidity was estimated by the acid-base account method of Ahern *et al.* (2004). Reactive metals and metalloids (Fe, Al, Ag, As, Pb, Cd, Cr, Cu, Mn, Ni, Se and Zn) were extracted using 1.0 M HCl and analysed using ICP-MS (Inductively Coupled Plasma - Mass Spectrometry).

The organic matter availability and quantity (i.e. total organic C, hydrolysable C and non-hydrolysable C) were measured after the 1.0 M HCl method described by Silveira *et al.* (2008). The total organic carbon (TOC) content was determined by a LECO-CNS 2000 analyser following the removal of inorganic carbon by treatment with 1.0 M HCl. The non-hydrolysable organic carbon content was determined by a LECO-CNS 2000 analyser following treatment with 6.0 M HCl at 105°C for 2 hours. The hydrolysable organic carbon content was determined from the difference between the TOC and the non-hydrolysable carbon fractions.

All sediment data are presented in Appendix 2 (Tables 9-2 to 9-9).

4.2.3 Surface Water and Pore-Water Analyses

Selected surface water quality parameters were measured in the field, and filtered (0.45 µm) samples were either frozen or added to traps for later analysis (see Section 4.1.2). Pore-waters were extracted after centrifuging the soil samples at 3,000 rpm for 10 minutes. The parameters measured in the laboratory on the surface water and pore-water samples collected included:

- Redox potential (Eh) (*pore-water only*)
- pH (*pore-water only*)
- Electrical conductivity (EC) (*pore-water only*)
- Alkalinity
- Dissolved sulfide
- Dissolved ferrous iron(Fe^{2+}) and total dissolved iron ($Fe^{3+} + Fe^{2+}$)
- Soluble chloride and sulfate
- Soluble cations (Ca, Mg, Na, K)
- Nutrients (orthophosphate, nitrate, nitrite, and ammonia)
- Total organic carbon (TOC) and dissolved organic carbon (DOC) (*surface water only*)

Pore-water redox potential, pH and electrical conductivity were immediately measured on unfiltered samples, and all other properties were determined on filtered (0.45 µm) samples. Redox potential (Eh) was determined using a calibrated electrode linked to a TPS smart*CHEM*-LAB laboratory analyser; Eh measurements are presented versus the standard hydrogen electrode. Electrical conductivity (EC) and pH were determined using calibrated electrodes linked to a TPS WP-81 meter.

Ferrous iron (Fe^{2+}), total iron ($Fe^{2+} + Fe^{3+}$), alkalinity and dissolved sulfide were fixed immediately after sampling. The Fe^{2+} 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). Bromophenol blue traps were used for alkalinity (Sarazin *et al.* 1999) and alkalinity standards were determined with 0.01 M HCl using the Gran procedure (Stumm and Morgan 1996). The dissolved sulfide fraction was trapped in an alkaline zinc acetate trap prior to determination by the spectrophotometric method of Cline (1969). The iron species, alkalinity and dissolved sulfide were all quantified colorimetrically using a Varian Cary 50 UV-Visible spectrophotometer.

Major cations and anions (Na^+, K^+, Ca^{2+}, Mg^{2+}, SO_4^{2-}, Cl^-) were analysed by ICP-OES (Inductively Coupled Plasma - Optical Emission Spectrometry). Nutrients (orthophosphate, nitrate, nitrite, and ammonia) were analysed turbidimetrically using flow-injection analysis (FIA) colorimetry (Lachat QuikChem 8000) (APHA 2005). The total organic carbon (TOC) and dissolved organic carbon (DOC) content were analysed using a Shimadzu TOC-L Analyzer following the APHA 5310 B high-temperature combustion method (APHA 2005).

All surface water and pore-water data are presented in Appendix 3 (Tables 9-10 to 9-21).

4.2.4 Vegetation Analyses

A comprehensive analysis of metals in the plant tissues (including leaves, stems and flowers) was undertaken at selected sites around the Lower Lakes (see Figure 4-2). Plant materials were initially washed thoroughly in tap water followed by deionised water (milliQ) to remove any potential contamination (i.e. dust). The plant materials were then dried at 70˚C for 24 hours prior to being ground. The metal concentrations were determined using ICP-MS (Inductively Coupled Plasma - Mass Spectrometry) following microwave digestion with nitric acid (HNO_3).

Plant material analysis data are presented in Appendix 5 (Table 9-23).

4.2.5 Expression of Results

The means (Av.) are presented in tables in this document with graphs given to illustrate certain points. The standard errors (SE) are presented on many of the graphs.

4.2.6 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 followed included the monitoring of blanks, duplicate analysis of at least 1 in 10 samples, and the inclusion of standards in each batch.

Blanks were collected for laboratory or field samples to examine whether contaminants had been introduced to the sample. 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.

Calibrations were performed on matrix-matched solutions and these were analysed along with standard solutions and the tested analytes. These calibrations and checks confirmed the methodology and the proper functioning of the analytical instruments.

Duplicates were prepared for all experiments and analysed separately. Selected analytical duplicate samples were prepared by dividing a test sample into two, then analysing these sub-samples separately. On average, the frequencies of quality control samples processed were: 10% blanks, ≥ 10% laboratory duplicates, and 5% laboratory controls. The analytical precision was usually ±10% for all analyses.

5.0 Results

5.1 General Sediment Condition

5.1.1 Campbell Park

5.1.1.1 pH$_{(1:1, \text{soil:water})}$

Both sites initially had acidic surface soil layers (0 – 10 cm) prior to the inundation that took place after the August 2010 sampling (Figures 5-1 – 5-2). For the control site this acidic layer was severely acidic (i.e. pH < 3) down to 30 cm depth whereas for the vegetated site only the 10 – 40 cm was severely acidified: the surface layer (0 – 10 cm) under this site initially had a pH of ~6.0. As the treatment site was not able to be sampled separately prior to the establishment of the vegetation, it is not possible to ascribe the difference in the pH of the surface soil layers directly to the presence of the vegetation. Indeed it was noticed that the control treatment had suffered from severe erosion post the establishment of the vegetation whereas the vegetated treatment was protected from the erosion. Therefore differences in the surficial pHs of the vegetated site and the control treatments are complicated in this study area due to erosion exposing acidic subsoils (i.e. below surface soils) in the case of the control sites.

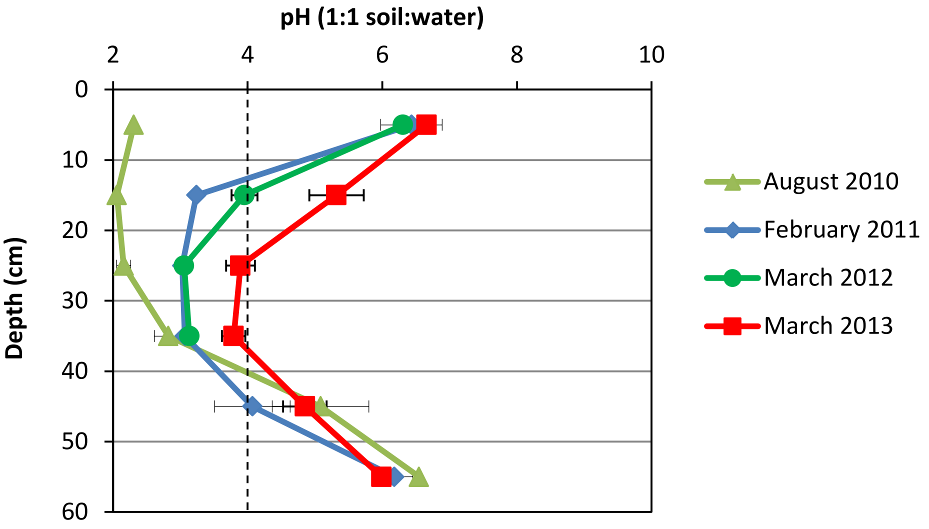

Figure 5-1. Campbell Park field pH dynamics at the control site (August 2010 – March 2013).

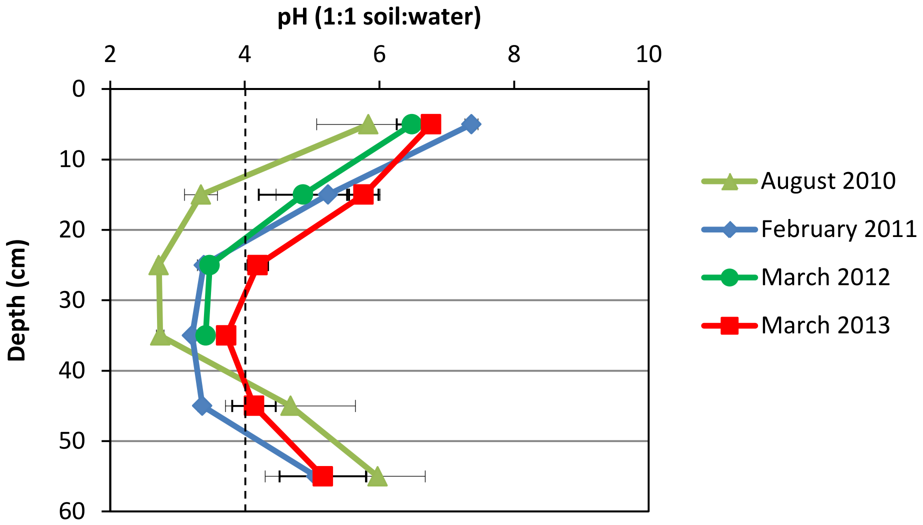

Figure 5-2. Campbell Park field pH dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2013).

By March 2013, the pH of the surface soil layers of the control site and the formally vegetated site had increased after inundation to a pH of 7.2. The pH of the subsoil layers from 10 – 40 cm has continued to increase over the inundation period. However, at both sites there remains an acidic layer with pH of ~ 4 between 20 and 40 cm. This layer at both sites corresponds to where jarosite accumulations around old root holes in particular are still visible and it is likely that the presence of this mineral is controlling the severely acid conditions at this site. It is significant that even though it is most likely that jarosite is controlling the geochemistry in these layers, that the pHs of these layers are now around 4 (1:1 soil:water) in the field when they were originally < 3 prior to inundation. This demonstrates a considerable increase in pH during inundation even though jarosite is no doubt still controlling the geochemistry.

5.1.1.2 Redox Potential (Eh)

Each site initially (i.e. in August 2010) had oxic conditions from 400 – 700 mV in the top 40 cm sandy-textured layers (Figures 5-3 – 5-4), but during the inundation process increasingly reductive conditions developed throughout these layers especially in the top 20 cm of the sediment where the Eh decreased down to around 200 mV at the March 2013 sampling.

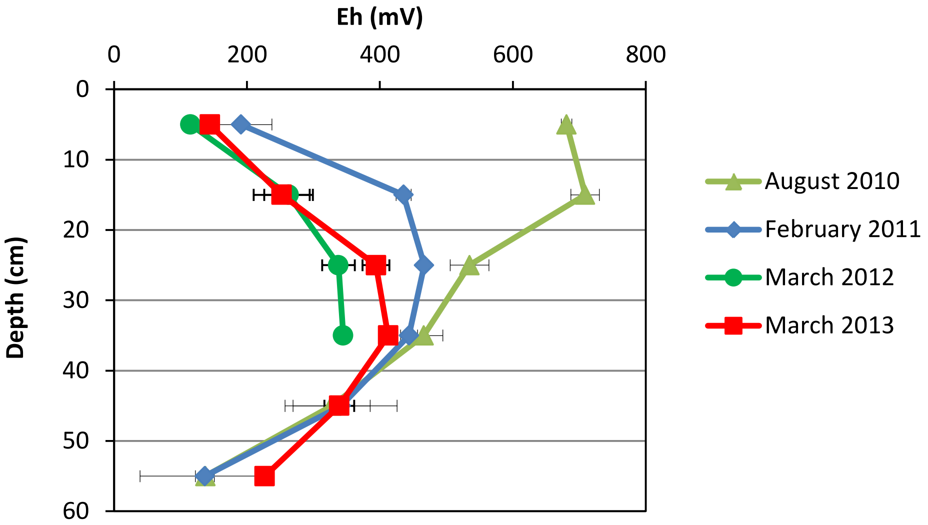

Figure 5-3. Campbell Park field Eh dynamics at the control site (August 2010 – March 2013).

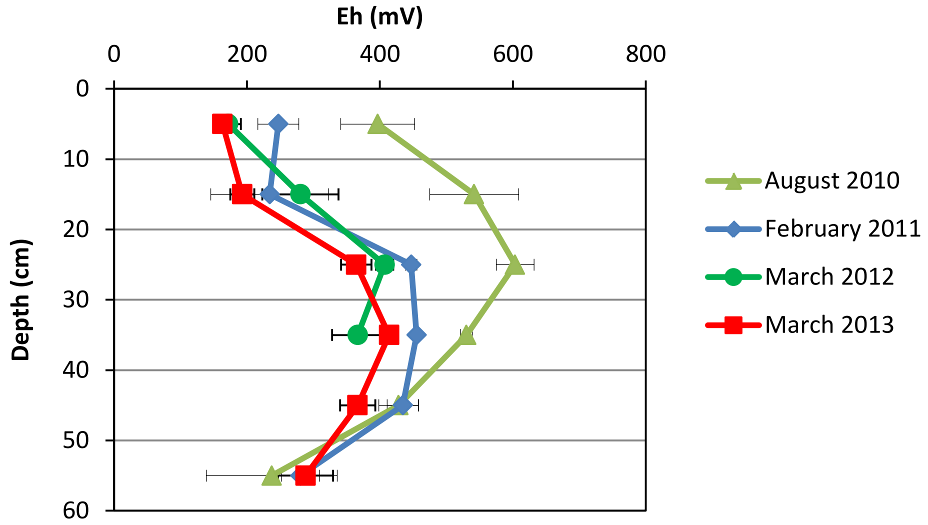

Figure 5-4. Campbell Park field Eh dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2013).

5.1.1.3 Electrical Conductivity (EC)

The salinity (i.e. EC) has continued to decrease during inundation as shown in Figures 5-5 – 5-6. By March 2013 the salinity in the sediments under both treatments gradually increased with depth from ~ 180 – 230 µS cm^{-1} to ~ 2,000 µS cm^{-1}. This likely reflects ongoing diffusion of salts out of the sediments into the overlying lake water (see Figure 9-23, Appendix 6).

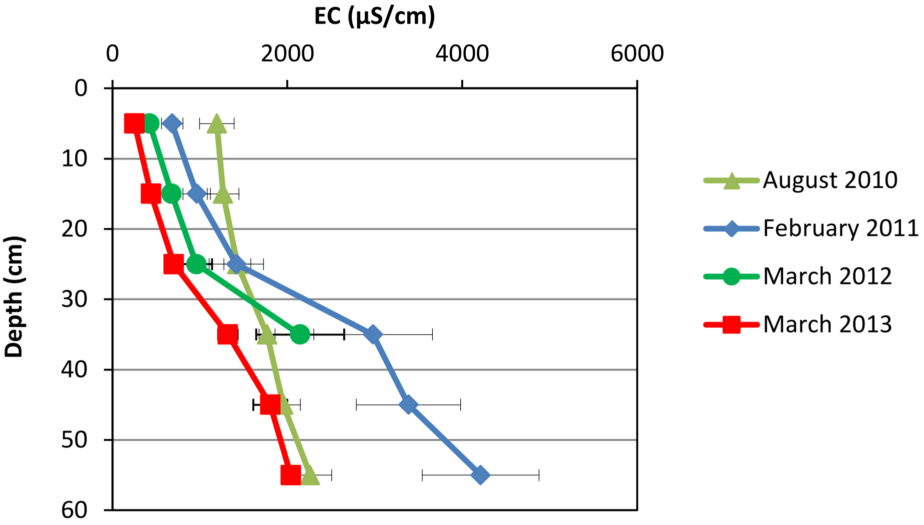

Figure 5-5. Campbell Park EC dynamics at the control site (August 2010 – March 2013).

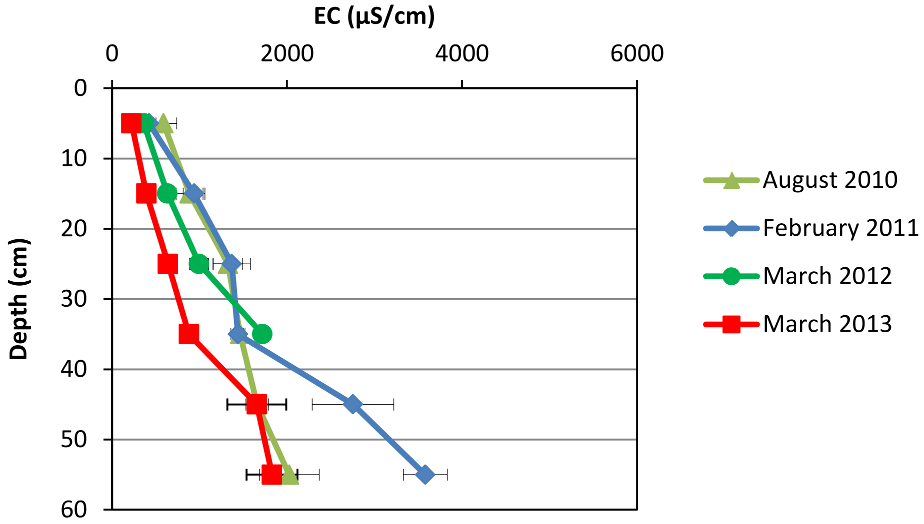

Figure 5-6. Campbell Park EC dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2013).

5.1.1.4 Acid-Base Accounting

5.1.1.4.1 Potential Sulfidic Acidity (PSA)

Graphs showing the PSA in August 2010 and March 2013 for the two sites are presented in Figures 5-7 and 5-8. These graphs show that the Reduced Inorganic Sulfur components have not increased appreciably in the surficial layers at these sites as a result of inundation.

The pyritic sulfur contents as determined by the chromium reducible sulfur (CRS) were very low (i.e. < 0.02% S) in the surficial layers (0 – 20 cm) at both sites (Figures 9-1 – 9-2, Appendix 4) prior to inundation. The apparent variations in pyritic S concentrations (i.e. up to 0.70% S as pyrite) in the 30 – 40 cm layer in the both sites is most likely the result of sediment erosion caused by wave action in the lake waters effectively bringing residual reduced inorganic sulfides, formerly more deeply buried, closer to the sediment surface. In March 2013 there was still no evidence of the appreciable accumulation of reduced inorganic sulfides in the surficial sediments after inundation.

The concentration of Acid Volatile Sulfide (i.e. monosulfides) remained below the limit of detection (i.e. <0.01% S) in all layers. Elemental sulfur (Figures 9-12 – 9-13, Appendix 4) was present in the surficial layers at each site but in very low concentrations (i.e. ≤ 0.004% S).

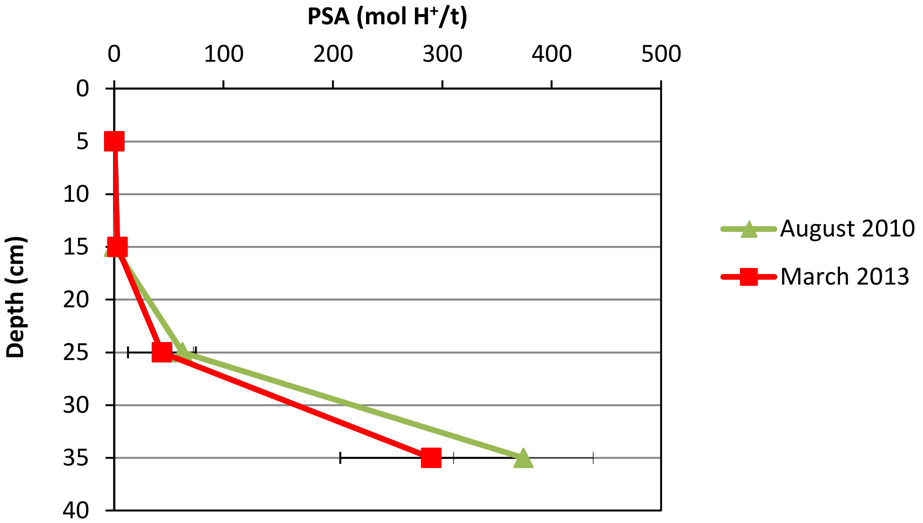

Figure 5-7. Campbell Park Potential Sulfidic Acidity dynamics at the control site (August 2010 and March 2013).

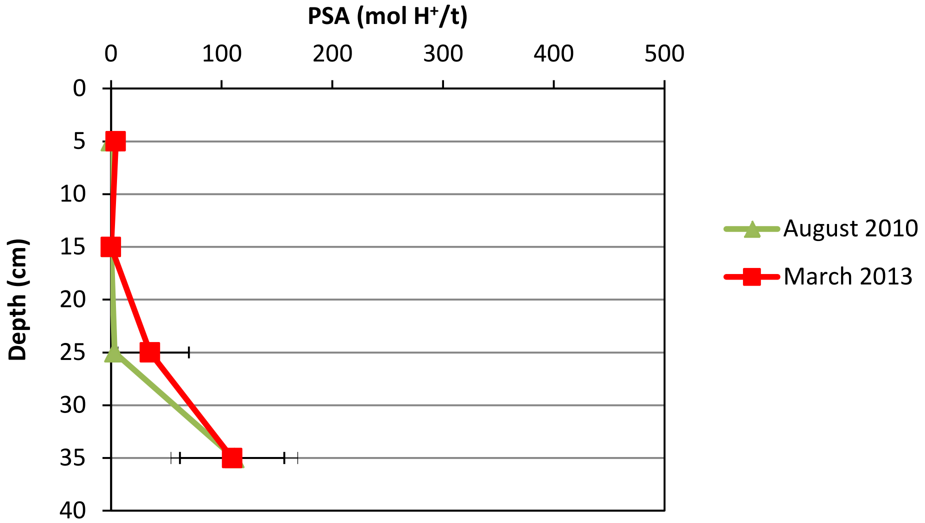

Figure 5-8. Campbell Park Potential Sulfidic Acidity dynamics at the Bevy rye/*Puccinellia* site (August 2010 and March 2013).

5.1.1.4.2 Titratable Actual Acidity (TAA)

Given the considerable increases in pH in the top 40 cm layers at both sites since re-inundation, it is perhaps surprising that there has not been a corresponding decrease in the measured TAAs in these layers. The TAAs (Figures 5-9 – 5-10) were all low in the surface soil layers (i.e. < 18 mol H^+ t^{-1}) but increased up to 35 mol H^+ t^{-1} in the 30 - 40 cm layers of each site both prior to and during inundation. This is the zone that contains appreciable quantities of jarosite.

There are several likely reasons for the maintenance of TAAs in these surficial layers despite appreciable increases in field pH.

1) The TAA measurement has recently been shown to include inputs arising from the dissolution of iron precipitate minerals especially schwertmannite and jarosite in 1 M KCl (Vithana *et al.* (2013). Previously both of these minerals not been considered as contributors to the TAA acidity pool in acid sulfate soils. They were thought to contribute only to the Retained Acidity pool.
2) The soluble Fe^{2+} contents in the sediment pore waters arising from the reductive dissolution of jarosite (iron mobilization from jarosite dissolution is evident by orange iron staining around many of the remaining jarosite segregations in these layers and the Eh data in the next section) would likely also be measured as TAA after these samples are dried prior to TAA analysis. The increases in Fe^{2+} in line with the measured TAA are shown in Figures 5-11 and 5-12 below. Figures 5-13 and 5-14 show very strong positive relationships between total soluble Fe in the pore-waters (which is mainly in the Fe^{2+} form) and the measured TAA at each of these sites supporting this proposition.

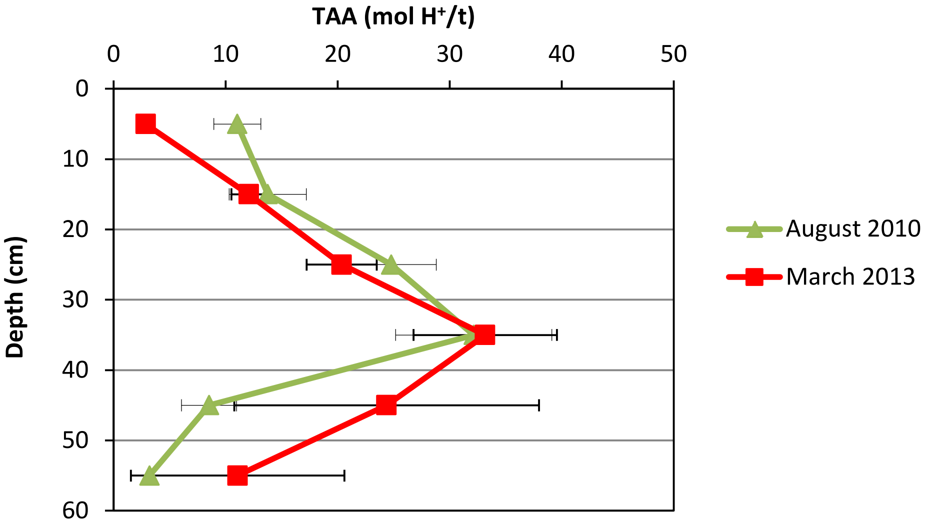

Figure 5-9. Campbell Park TAA dynamics at the control site (August 2010 and March 2013).

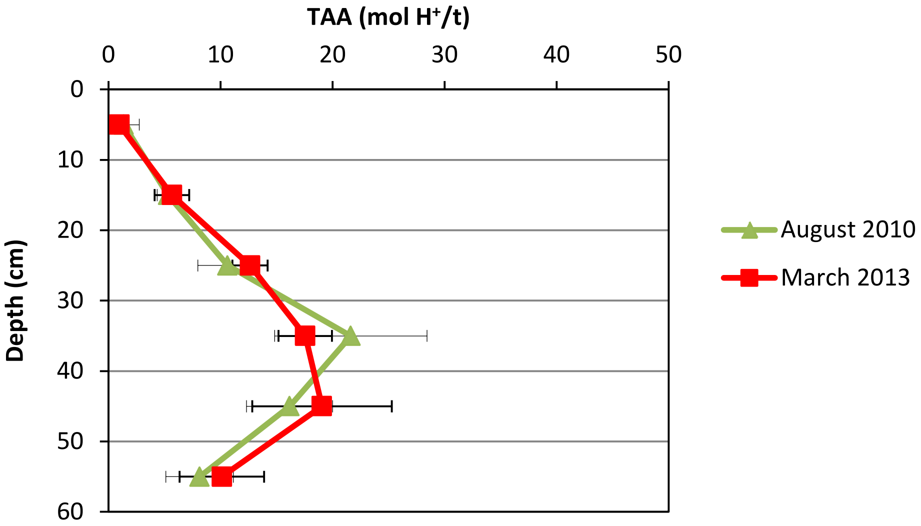

Figure 5-10. Campbell Park TAA dynamics at the Bevy rye/*Puccinellia* site (August 2010 and March 2013).

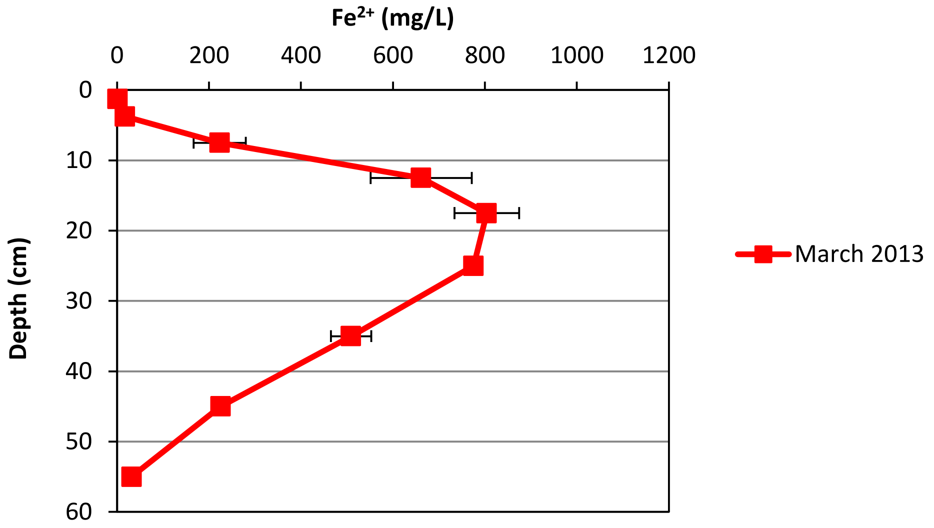

Figure 5-11. Campbell Park pore-water dissolved ferrous iron (Fe^{2+}) characteristics at the control site (March 2013).

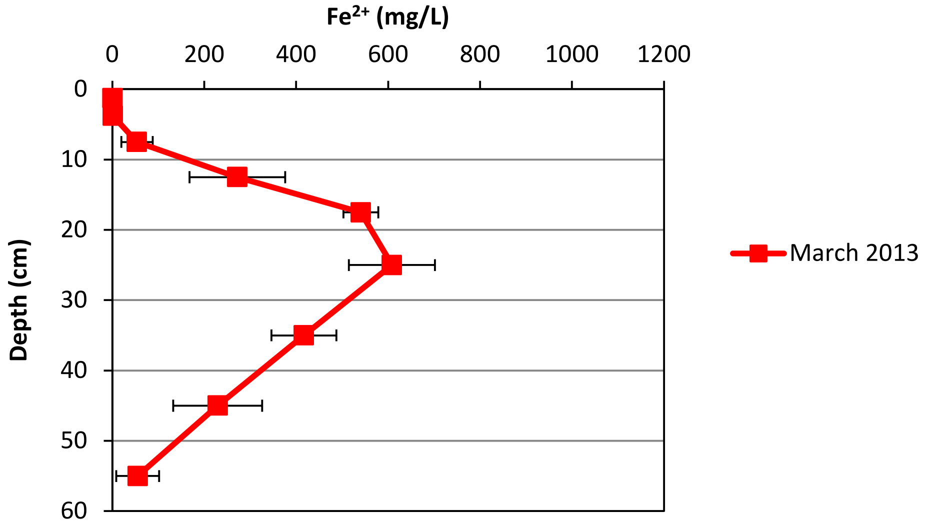

Figure 5-12. Campbell Park pore-water dissolved ferrous iron (Fe^{2+}) characteristics at the Bevy rye/*Puccinellia* site (March 2013).

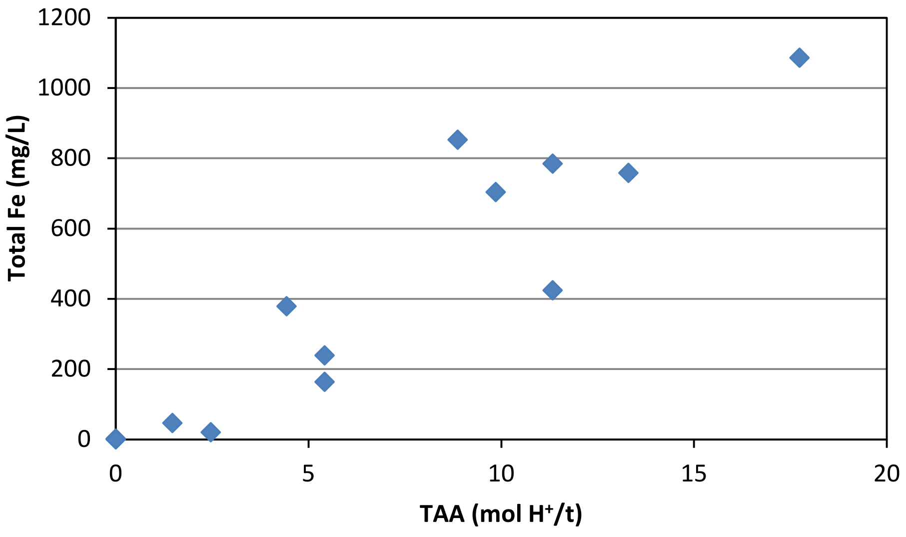

Figure 5-13. Pore-water total dissolved soluble iron vs TAA for Campbell Park control site (0-20 cm layers) (March 2013).

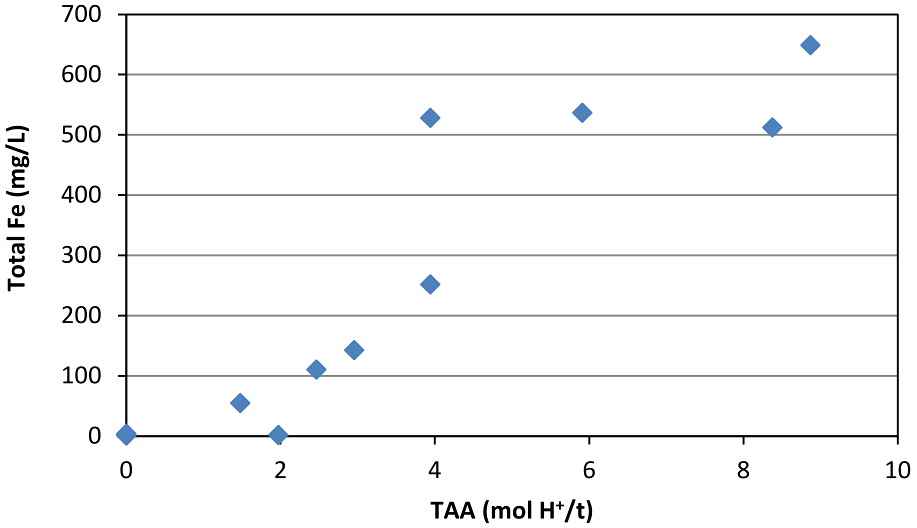

Figure 5-14. Pore-water total dissolved soluble iron vs TAA for Campbell Park Bevy rye/*Puccinellia* site (0-20 cm layers) (March 2013).

5.1.1.4.3 Acid Neutralising Capacity (ANC)

The graphs of ANC for the two sites are presented in Figures 5-15 and 5-16. Figure 5-15 shows a slight increase in the ANC of the surface layer (0-10 cm) at the control site between August 2010 and March 2013, whereas the vegetated site shows a decrease. The ANC accumulated in the surface layer under the vegetated site prior to inundation had most likely decreased during re-inundation, by either alkaline mineral dissolution or via mobilisation of alkaline sediment.

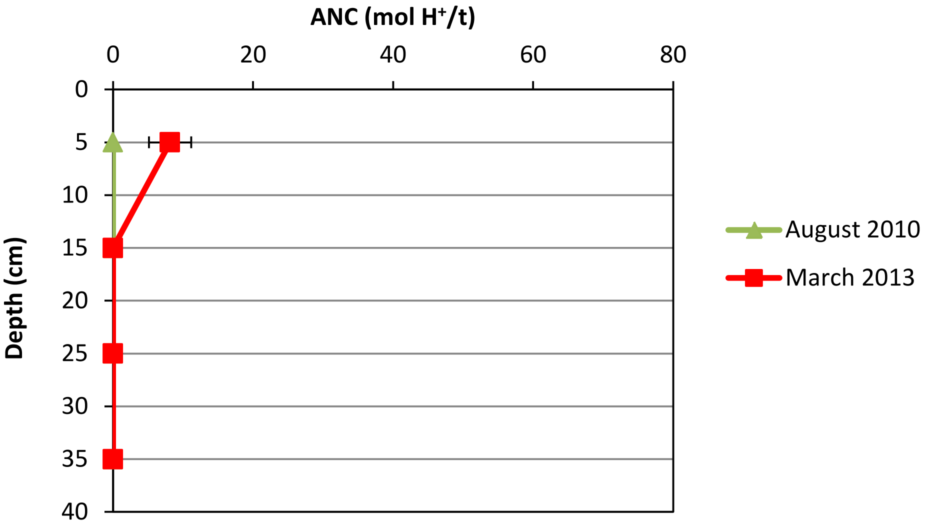

Figure 5-15. Campbell Park Acid Neutralising Capacity dynamics at the control site (August 2010 and March 2013).

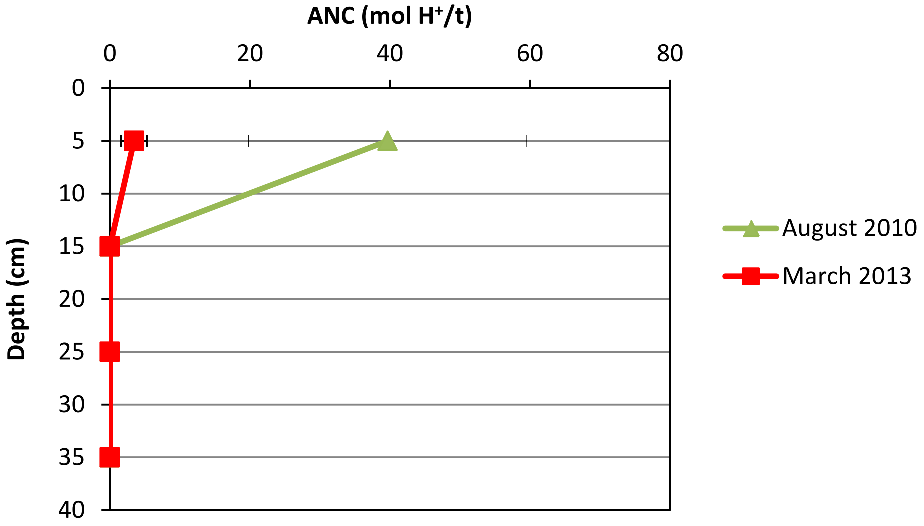

Figure 5-16. Campbell Park Acid Neutralising Capacity dynamics at the Bevy rye/*Puccinellia* site (August 2010 and March 2013).

5.1.1.4.4 Retained Acidity

The graphs of Retained Acidity for the two sites (Figures 5-17 and 5-18) show that the Retained Acidity of the surficial layers of these sites has decreased considerably during inundation presumably via the substantial (yet partial) dissolution of the jarosite in these sediment layers.

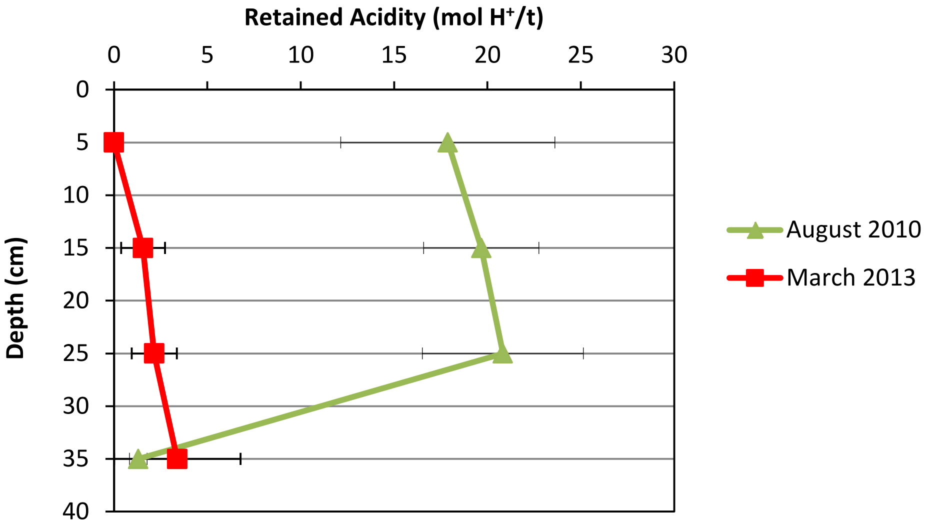

Figure 5-17. Campbell Park Retained Acidity dynamics at the control site (August 2010 and March 2013).

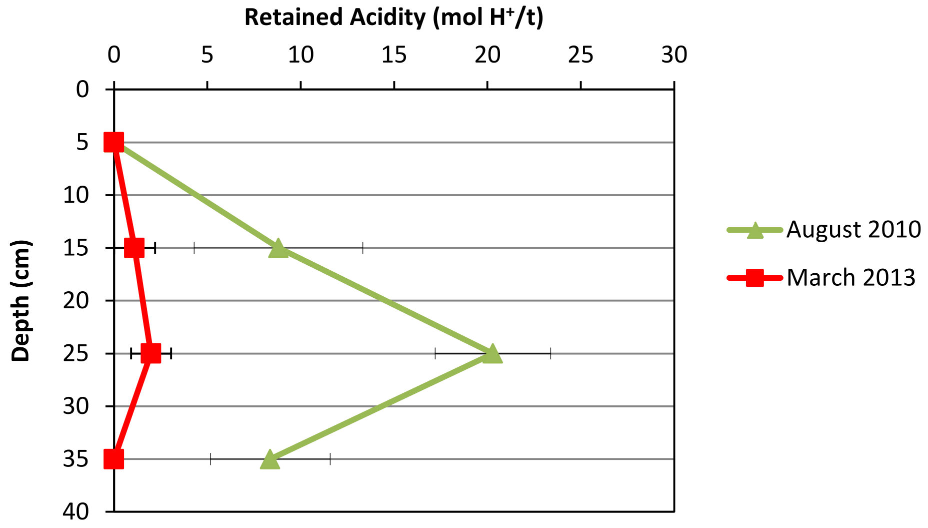

Figure 5-18. Campbell Park Retained Acidity dynamics at the Bevy rye/*Puccinellia* site (August 2010 and March 2013).

5.1.1.4.5 Net Acidity

Perhaps surprisingly, despite the appreciable increases in pH in the surficial layers at both of these sites since re-inundation, there are no consistent trends in the Net Acidity of these layers as measured by the Ahern *et al.* (2004) method (see Figures 5-19 and 5-20). For example, in the scald site (Figure 5-19) whilst there were decreases in the Net Acidities in the 0 - 30 cm depth layer over time since inundation, at the neighbouring vegetated site was an increase in Net Acidity in the 0 - 10 cm layer over this period of inundation (Figure 5-20).

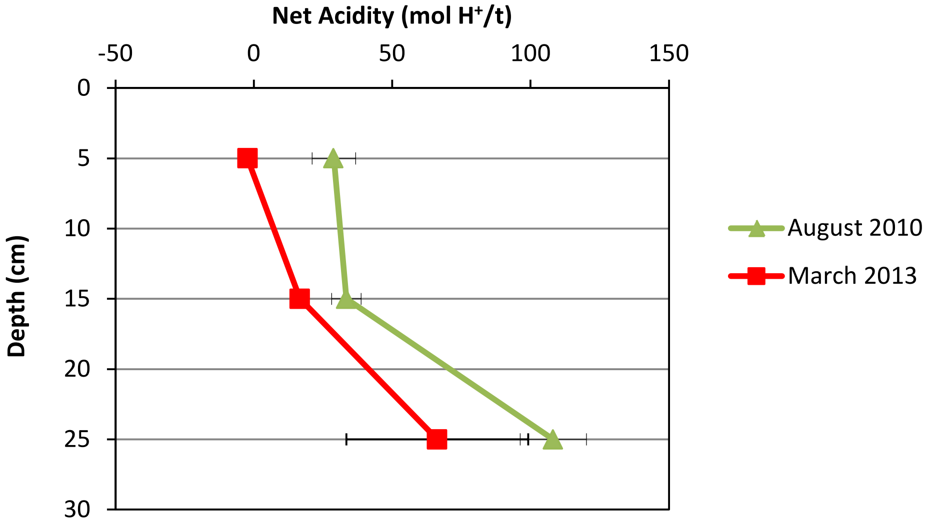

Figure 5-19. Campbell Park Net Acidity dynamics at the control site (August 2010 and March 2013).

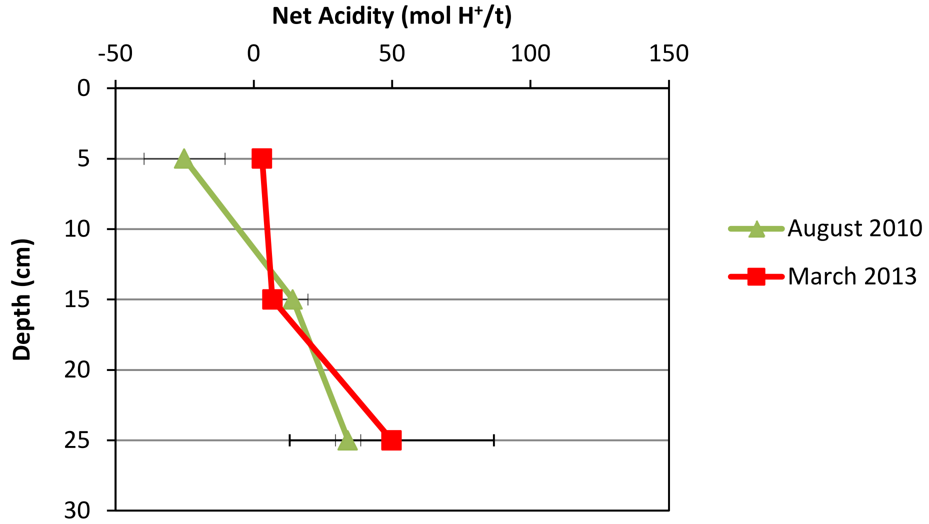

Figure 5-20. Campbell Park Net Acidity dynamics at the Bevy rye/*Puccinellia* site (August 2010 and March 2013).

Given that the Reduced Inorganic Sulfur components have not changed appreciably in the surficial layers at these sites as a result of re-inundation, the differences in trends in Net Acidity observed at these sites cannot be explained by accumulation or oxidation of Potential Sulfidic Acidity.

The Retained Acidity at both sites in the 0 – 30 cm layer for the control and the 10 - 40 cm layer for the vegetated site, decreased appreciably since re-inundation presumably as a result of the dissolution of jarosite (as evidenced by the observation in March 2013 of orange segregations around jarosite segregations and the increase in Fe^{2+} concentrations in these layers (Figures 5-11 and 5-12)).

Some of the differences in the changes to the Net Acidity between these two sites can be explained by the changes in the ANC that have occurred since inundation. Figure 5-16 for example, shows that the ANC that had accumulated in the surface layer under the vegetated site prior to re-inundation had decreased during re-inundation. The Net Acidity in the surficial layer at the vegetated site increased largely as a result of this decrease in ANC. In contrast, the increase in ANC in the surficial layer of the control site during inundation has contributed only slightly to the observed decrease in Net Acidity in this layer as a result of prolonged inundation.

In summary the re-inundation has caused substantial decrease in Net Acidity of the surficial 30 cm layers on the scald site and a slight increase in surficial 10 cm layer in the initially less acidic vegetated site. On the scald site the appreciable reduction in Net Acidity was due to appreciable decreases in Retained Acidity and the TAA, and an increase in the ANC. On the vegetated site the slight increase in Net Acidity was due to an appreciable decrease in the ANC that was accompanied by a lesser decrease in the Retained Acidity.

5.1.2 Tolderol

5.1.2.1 pH$_{(1:1, soil:water)}$

The control site (from 0 - 60 cm) and the Bevy rye site (only 10 - 60 cm) initially were highly acidic (i.e. pH < ~4) prior to inundation (Figures 5-21 – 5-22). Upon lake filling in August 2010 the pHs of the surface soil layers down to 40 cm depth in the control site became even lower likely due to the exchange of acidity from the soil by the inundating waters. This exchange acidification due to inundation effect was confined to the 20 – 50 cm layer in the Bevy rye site.

Prolonged inundation of both sites since August 2010 has increased the pHs of sediments at both sites, although this was initially most pronounced in the surficial sediment layers at the control site. Since last sampling in March 2012 the pHs of the sediments at both sites have largely stabilised.

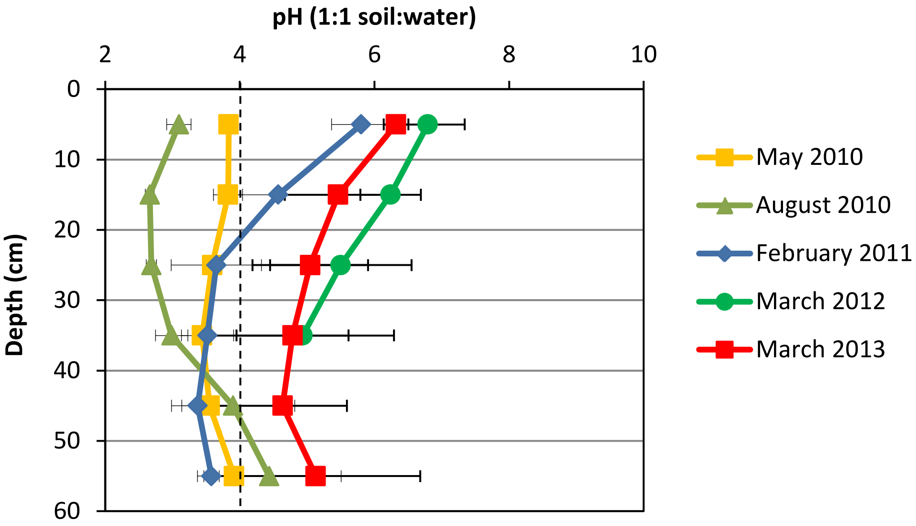

Figure 5-21. Tolderol field pH dynamics at the control site (May 2010 – March 2013).

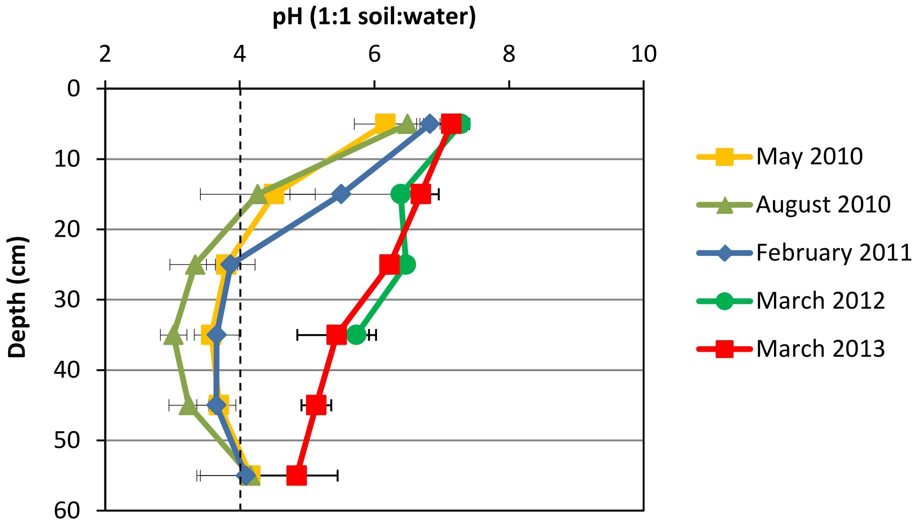

Figure 5-22. Tolderol field pH dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2013).

5.1.2.2 Redox potential (Eh)

All treatments initially (i.e. in May 2010) had oxic conditions (Figures 5-23 – 5-24), but during the inundation increasingly reductive conditions have developed throughout the whole profile down to 60 cm during the prolonged inundation. It is noticeable that the reduction in Eh in the surficial layers occurred much earlier (e.g. by August 2010) in the Bevy rye treatment as compared to the control site. Consequently the Eh was maintained in the surficial layers under the Bevy rye site in the 13 months to March 2012 but continued to decrease over this period under the control site. Since last sampling in March 2012 the Eh in all soil layers at this study area have shown minimal change.

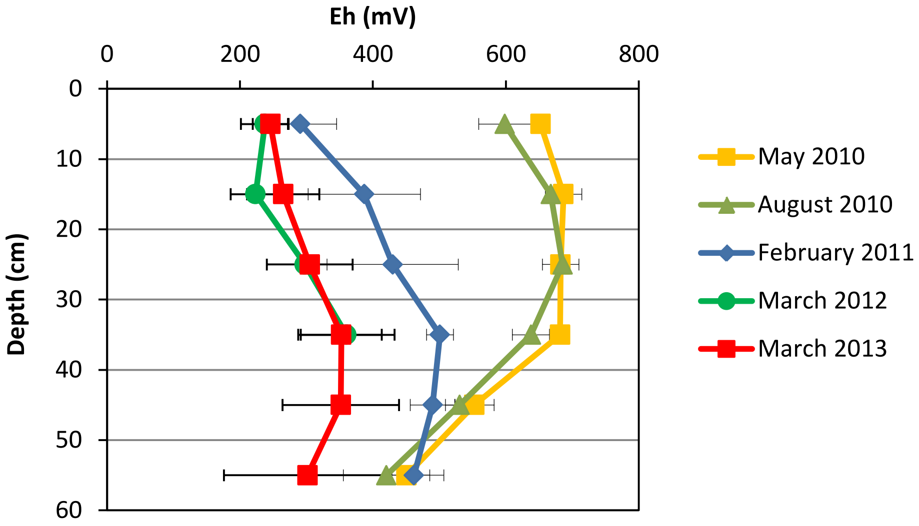

Figure 5-23. Tolderol field Eh dynamics at the control site (May 2010 – March 2013).

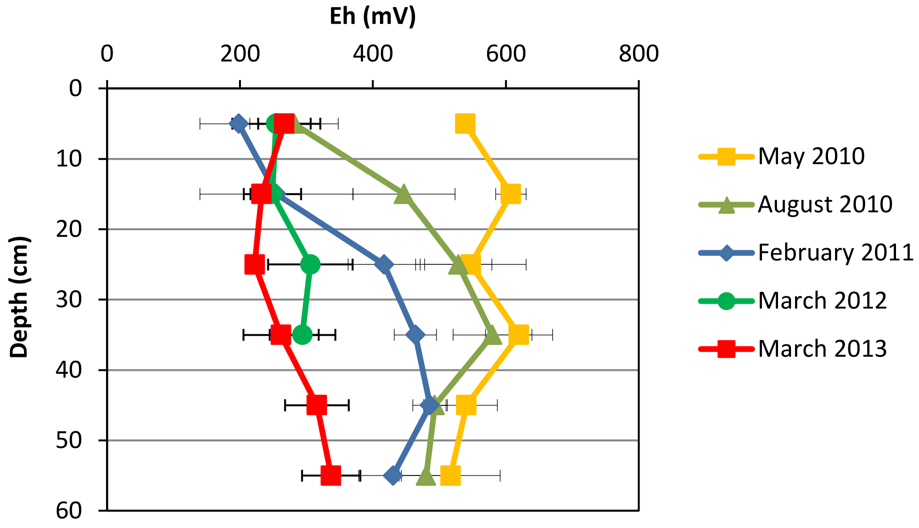

Figure 5-24. Tolderol field Eh dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2013).

5.1.2.3 Electrical Conductivity (EC)

As shown in Figures 5-25 – 5-26 the salinity (i.e. EC) decreased appreciably between February 2011 and March 2012 in the sediment layers under both treatments. Since March 2012 the salinity at both sites has not decreased any further. The loss of salinity since inundation is most likely due to diffusion of salts from the sediments to the overlying lake waters (see Figure 9-22, Appendix 6).

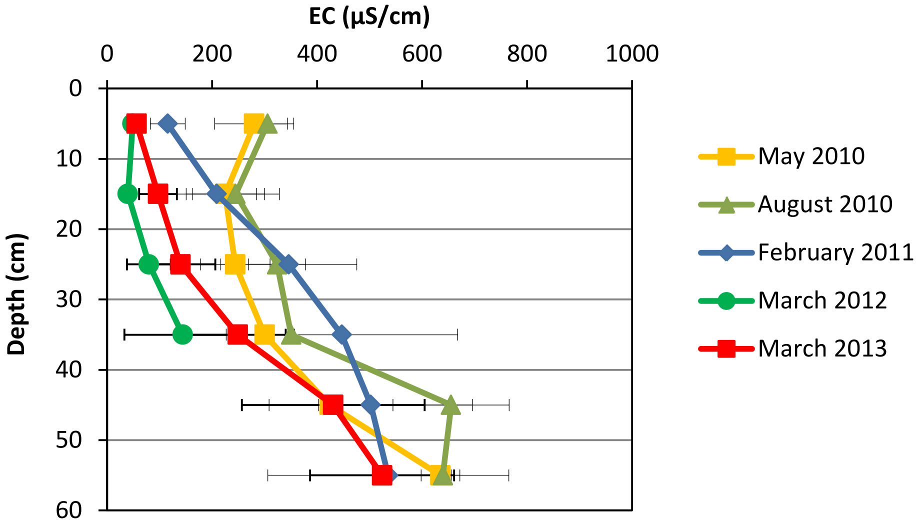

Figure 5-25. Tolderol EC dynamics at the control site (May 2010 – March 2013).

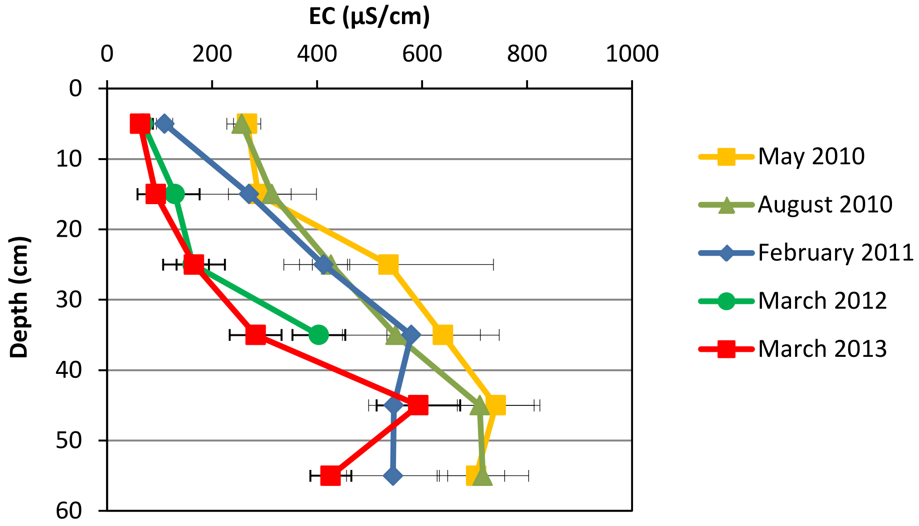

Figure 5-26. Tolderol EC dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2013).

5.1.2.4 Acid-Base Accounting

5.1.2.4.1 Potential Sulfidic Acidity (PSA)

Graphs showing the PSA in August 2010 and March 2013 for the two sites are presented in Figures 5-27 and 5-28. These graphs indicate that the Reduced Inorganic Sulfur components have increased in some of the surficial layers at these sites as a result of inundation. The large error bars observed for many of the sediment layers at the Bevy rye site in May 2010 indicate a large variation in Reduced Inorganic Sulfur contents between the duplicate sites.

The pyritic sulfur contents were very low (i.e. < 0.02% S) in the surficial layers (0 – 40 cm) at both sites prior to inundation (Figures 9-3 – 9-4, Appendix 4). These have largely remained low during the inundation period to date. At the control site there was a slight accumulation of pyrite at the February 2011 assessment and decrease from then at the March 2012 assessment. Since March 2012 there has been a slight accumulation of pyrite (i.e. < 0.02% S) at the Bevy rye site (Figure 9-4, Appendix 4). The apparent accumulation of an appreciable concentration of reduced inorganic sulfides (i.e. up to 0.07% S as pyrite) in the 30 – 40 cm layer in the Bevy rye site is most likely the result of sediment erosion caused by wave action in the lake waters effectively bringing residual reduced inorganic sulfides, formerly more deeply buried, closer to the sediment surface.

The concentration of Acid Volatile Sulfide (i.e. monosulfides) remained below the limit of detection (i.e. <0.01% S) in all layers. Elemental sulfur was only present in some of the surficial layers at each site but in very low concentrations (i.e. ≤ 0.002% S).

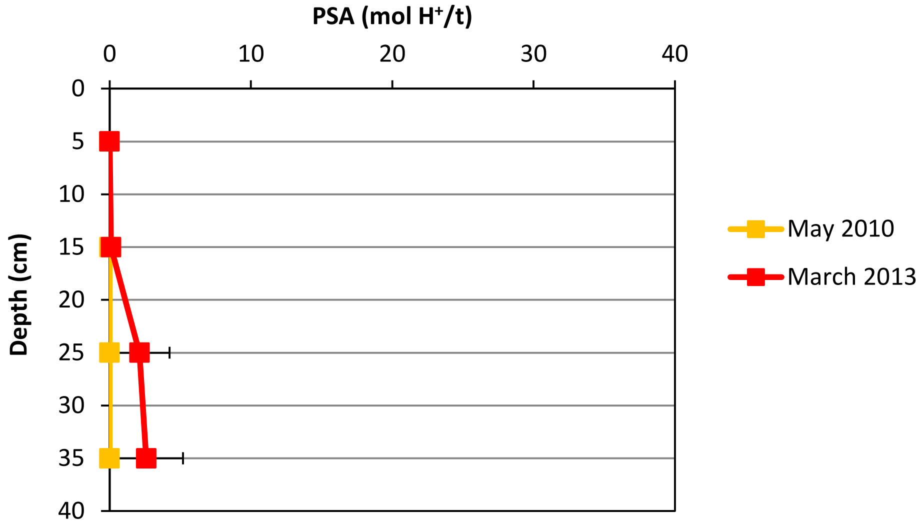

Figure 5-27. Tolderol Potential Sulfidic Acidity dynamics at the control site (May 2010 and March 2013).

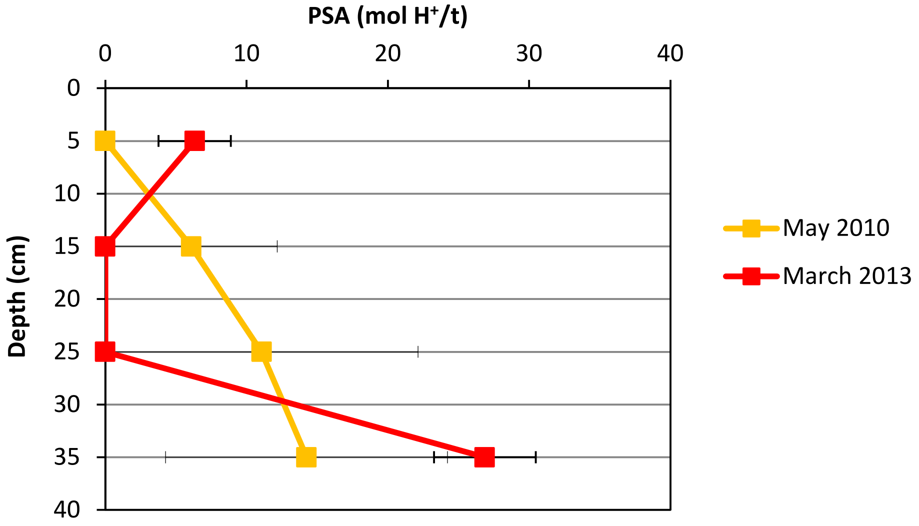

Figure 5-28. Tolderol Potential Sulfidic Acidity dynamics at the *Juncus* in Bevy rye site (May 2010 and March 2013).

5.1.2.4.2 Titratable Actual Acidity (TAA)

The TAAs (Figures 5-29 – 5-30) were all very low (i.e. < 18 mol H^+ t^{-1}) in each soil layer, and were especially low initially in the surface sediment layers of the Bevy rye treatment (i.e. initially ~3 mol H^+ t^{-1}). The TAAs have generally decreased further with prolonged inundation in all soil layers of each site.

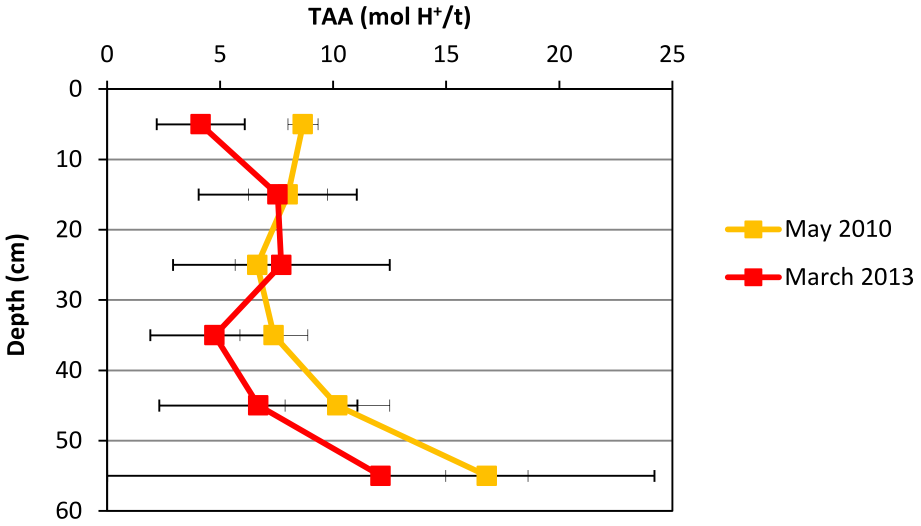

Figure 5-29. Tolderol TAA dynamics at the control site (May 2010 and March 2013).

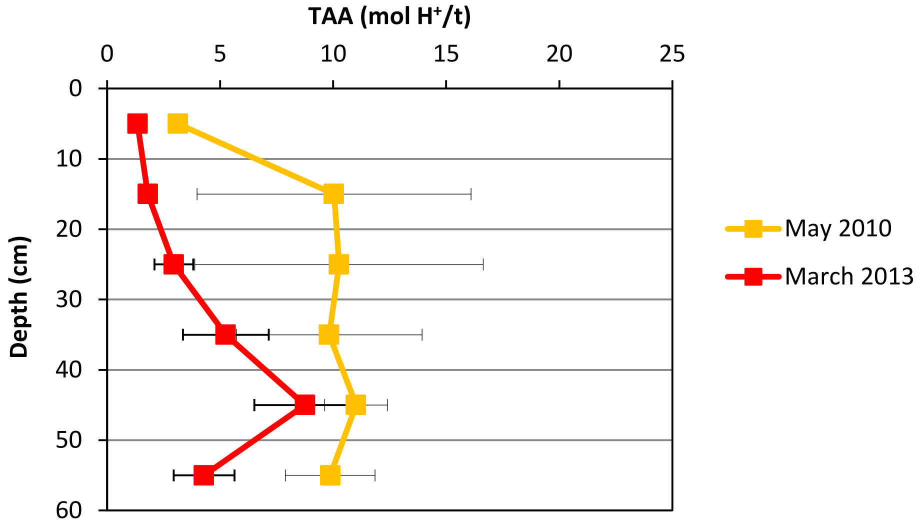

Figure 5-30. Tolderol TAA dynamics at the *Juncus* in Bevy rye site (May 2010 and March 2013).

5.1.2.4.3 Acid Neutralising Capacity (ANC)

The graphs of ANC for the two sites are presented in Figures 5-31 and 5-32. The ANCs for all sites in these surficial layers are very low. Figure 5-32 shows a slight increase in the ANC of the surface layer (0 - 10 cm) at the *Juncus* in Bevy rye site between August 2010 and March 2013.

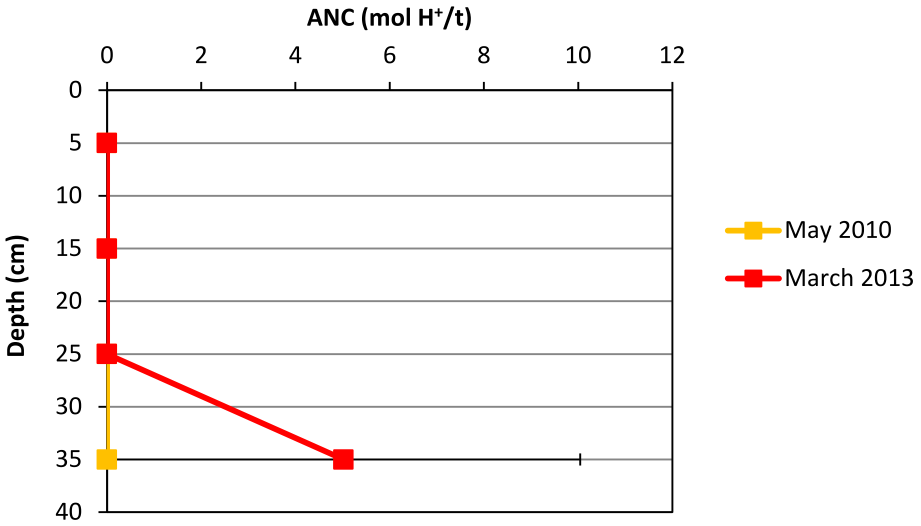

Figure 5-31. **Tolderol Acid Neutralising Capacity dynamics at the control site (May 2010 and March 2013).**

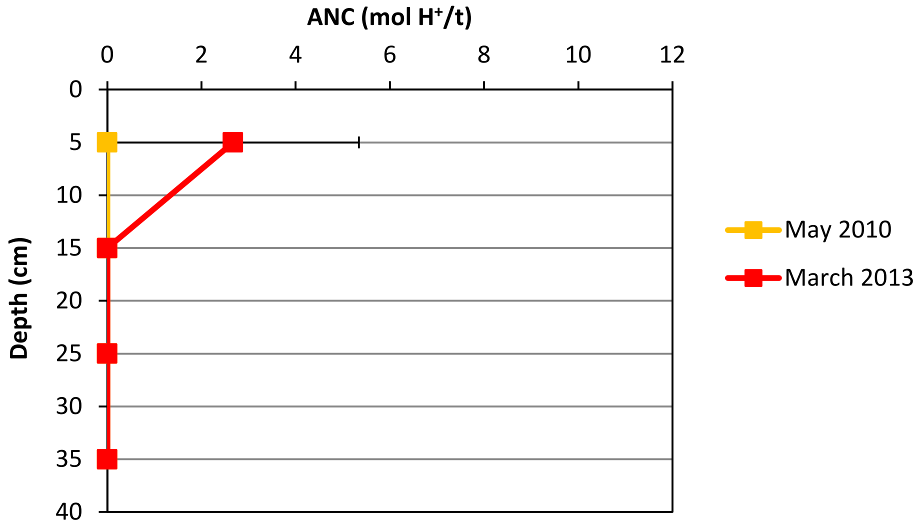

Figure 5-32. **Tolderol Acid Neutralising Capacity dynamics at the *Juncus* in Bevy rye site (May 2010 and March 2013).**

5.1.2.4.4 Retained Acidity

Only very small quantities of Retained Acidity were measured at this site. This is surprising given the presence of jarositic segregations around root holes in the subsurface sediment layers at the control site. This result may be explained by the recent findings of Vithana *et al.* (2013) who demonstrated that the Retained Acidity method is not an accurate quantification procedure for jarosite and its use can lead to an underestimation of the Retained Acidity hazard posed by acid sulfate soil materials.

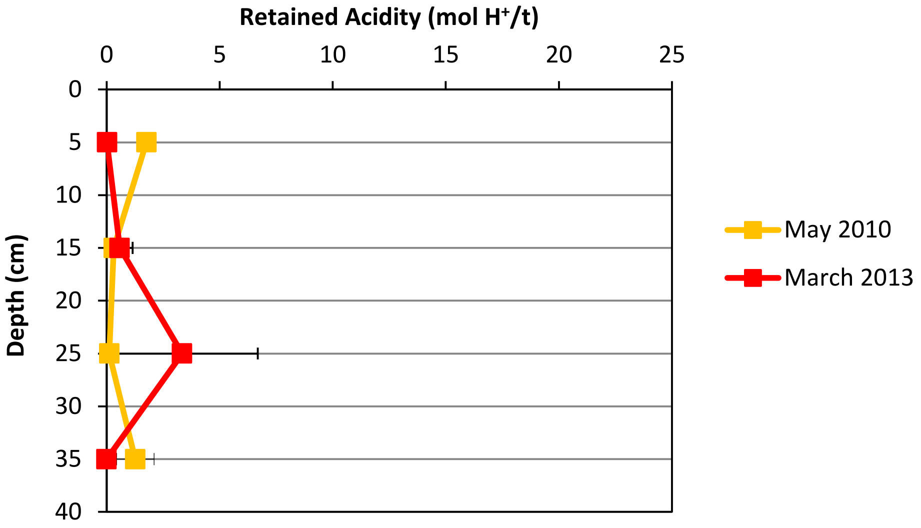

Figure 5-33. Tolderol Retained Acidity dynamics at the control site (May 2010 and March 2013).

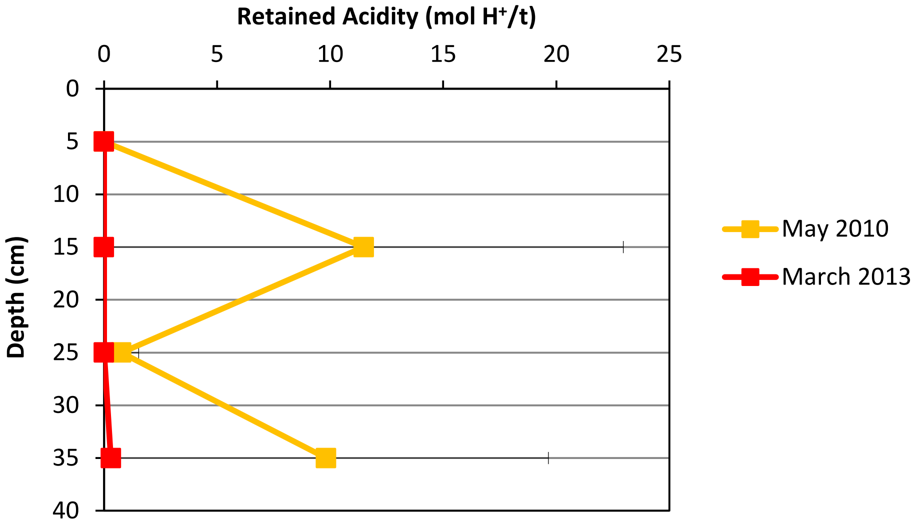

Figure 5-34. Tolderol Retained Acidity dynamics at the *Juncus* in Bevy rye site (May 2010 and March 2013).

5.1.2.4.5 Net Acidity

Again perhaps surprisingly, despite the appreciable increases in pH in the surficial layers at both of these sites since re-inundation, as for the Campbell Park site, there are no consistent trends in the Net Acidity of these layers as measured by the Ahern *et al.* (2004) method (see Figures 5-35 and 5-36) apart from a decrease in Net Acidity in the 10 – 30 cm layers at the Juncus site since inundation.

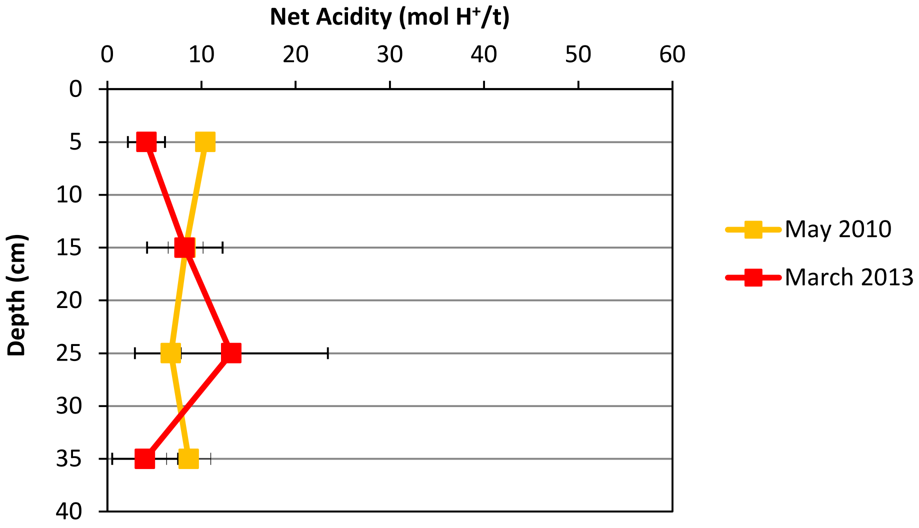

Figure 5-35. Tolderol Net Acidity dynamics at the control site (May 2010 and March 2013).

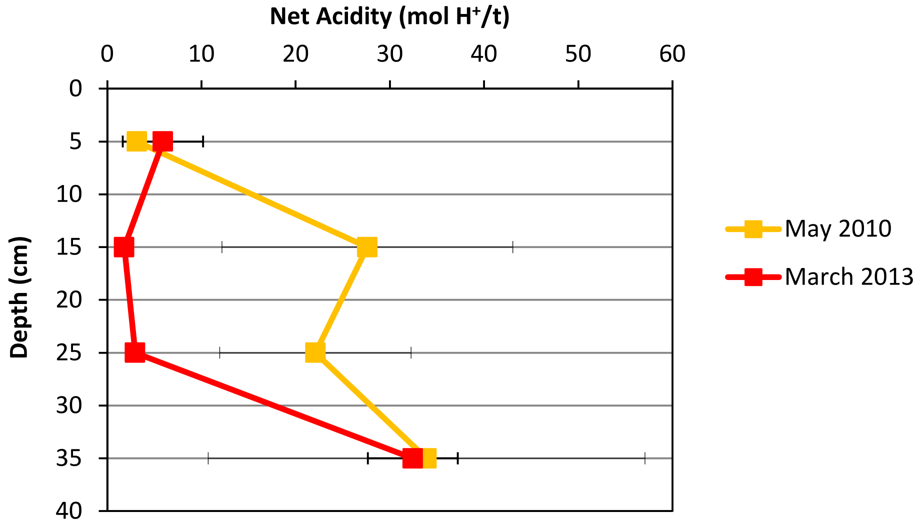

Figure 5-36. Tolderol Net Acidity dynamics at the *Juncus* in Bevy rye site (May 2010 and March 2013).

5.1.3 Poltalloch

5.1.3.1 pH(1:1, soil:water)

Since re-inundation in August 2010, the pHs of the upper 50 cm of sediment has gradually increased to a pH of 6.6 - 7.7 (Figure 5-37). The greatest change in pH over this period has occurred in the 10 - 40 cm layers which were approximately pH 4 in August 2010.

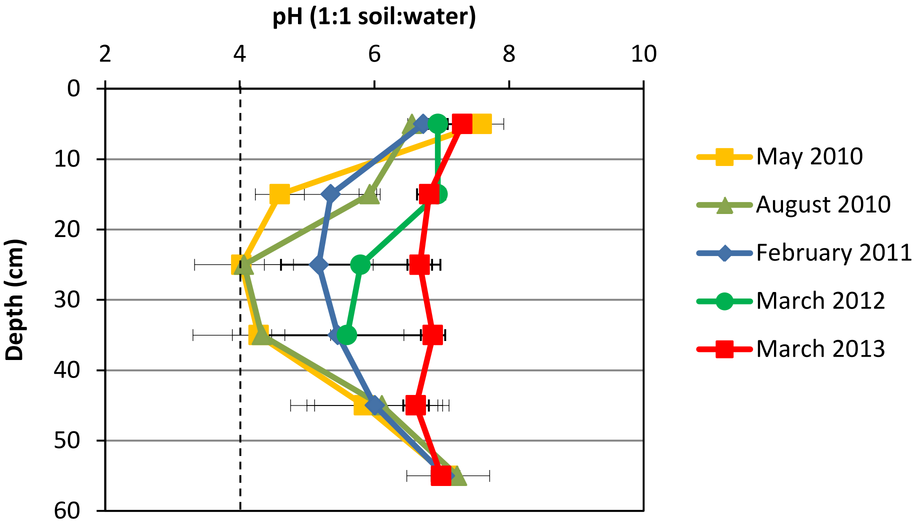

Figure 5-37. Poltalloch field pH dynamics at the Bevy rye site (May 2010 – March 2013).

5.1.3.2 Redox Potential (Eh)

Initially (i.e. in May 2010) the site had oxic conditions (Figure 5-38), but during the inundation process increasingly reductive conditions developed throughout the whole profile down to 60 cm and have further decreased during the prolonged inundation.

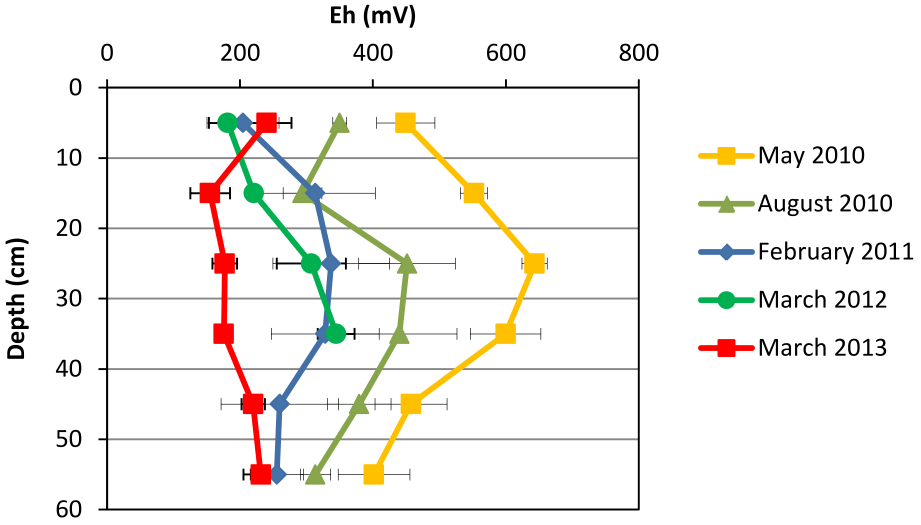

Figure 5-38. Poltalloch field Eh dynamics at the Bevy rye site (May 2010 – March 2013).

5.1.3.3 Electrical Conductivity (EC)

While the salinity (i.e. EC) did not change appreciably from before inundation until February 2011, since then the salinity has continued to decrease. As shown in Figure 5-39 the salinity in the surface layers fell from ~500 µS cm^{-1} to <100 µS cm^{-1} after prolonged inundation. The salinities of the sediment layers gradually increase with depth. At the March 2013 sampling the EC in the lowest layer was ~1000 µS cm^{-1} at 60 cm. This loss of salinity is most likely due to diffusion of salts from the sediments to the overlying lake waters (see Figure 9-22, Appendix 6).

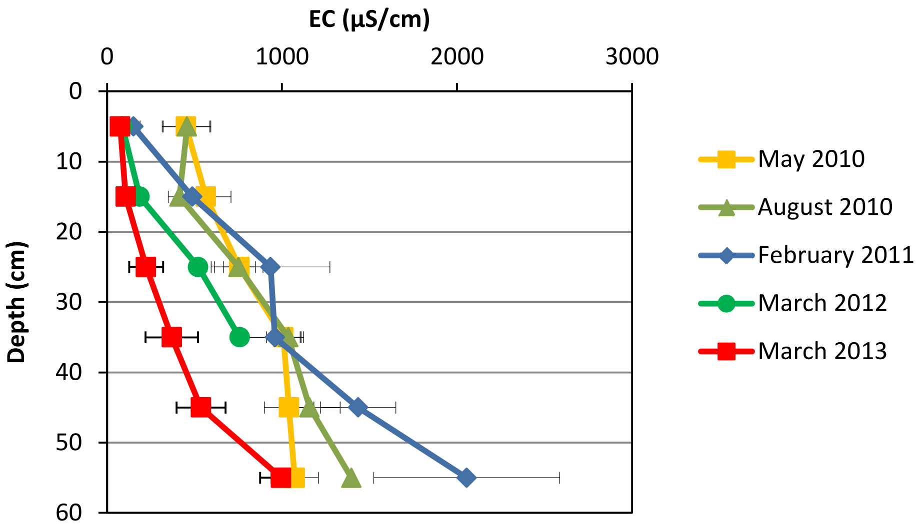

Figure 5-39. Poltalloch EC dynamics at the Bevy rye site (May 2010 – March 2013).

5.1.3.4 Acid-Base Accounting

5.1.3.4.1 Potential Sulfidic Acidity (PSA)

A graph showing the PSA in May 2010 and March 2013 at Poltalloch is presented in Figure 5-40. This graph suggests that the Reduced Inorganic Sulfur components have increased in the 30 – 40 cm layer as a result of inundation.

The pyritic sulfur contents were very low (i.e. < 0.02% S) in the upper 30 cm (Figure 9-5, Appendix 4) prior to inundation. There were low concentrations of residual reduced inorganic sulfides (i.e. up to 0.08% S as pyritic sulfur) in the 30 – 40 cm depth sediment layer. The apparent accumulation of an appreciable concentration of reduced inorganic sulfides (i.e. up to 0.08% S as pyrite) in the 30 – 40 cm layer in the Bevy rye site is most likely the result of sediment erosion caused by wave action in the lake waters effectively making residual reduced inorganic sulfides, formerly more deeply buried, closer to the sediment's new eroded surface.

The concentration of Acid Volatile Sulfide (i.e. monosulfides) remained below the limit of detection (i.e. < 0.01% S) in all layers. Elemental sulfur (Figure 9-14, Appendix 4) was still present in the surficial layers at each site but in very low concentrations (i.e. < 0.003% S).

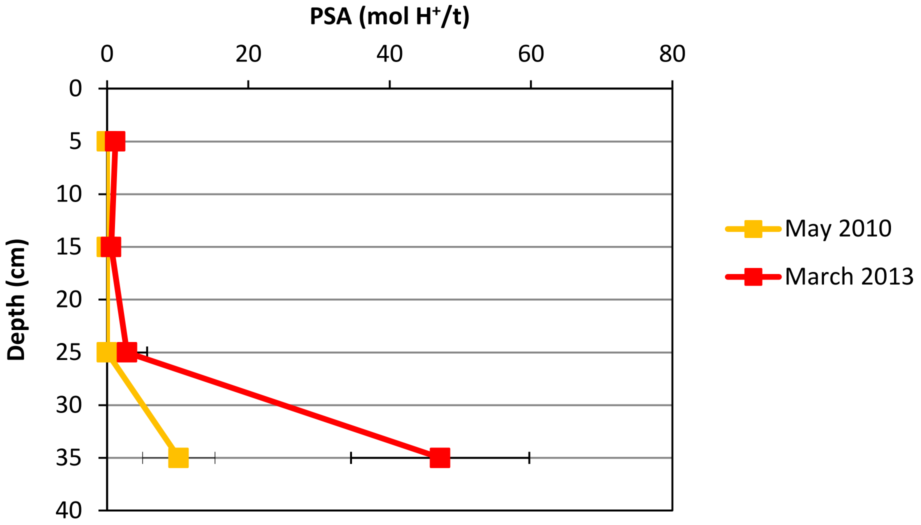

Figure 5-40. Poltalloch Potential Sulfidic Acidity dynamics at the Bevy rye site (May 2010 and March 2013).

5.1.3.4.2 Titratable Actual Acidity (TAA)

The TAAs (Figure 5-41) were all very low (i.e. < 5 mol H^+ t^{-1}) in each soil layer, and were especially low (i.e. <2 mol H^+ t^{-1}) in the surface sediment layers prior to inundation. The TAAs have decreased further since inundation.

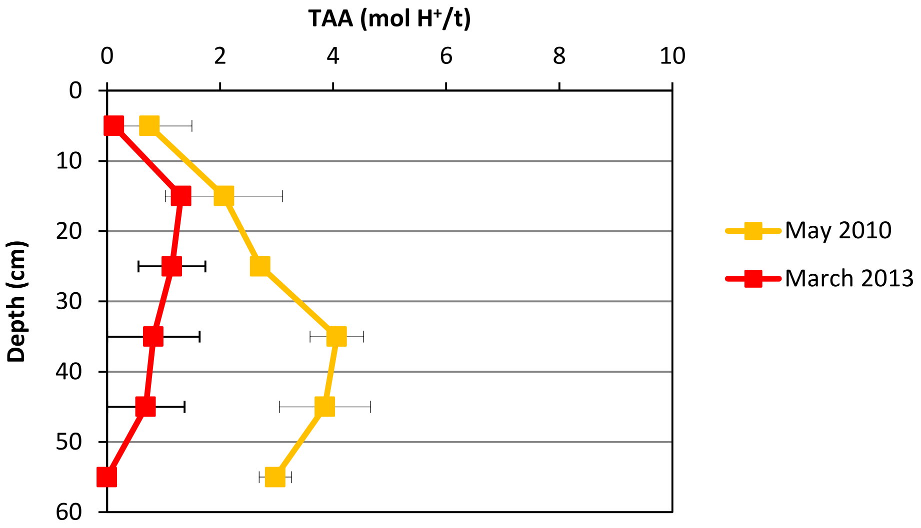

Figure 5-41. Poltalloch TAA dynamics at the Bevy rye site (May 2010 and March 2013).

5.1.3.4.3 Acid Neutralising Capacity (ANC)

The graphs of ANC are presented in Figures 5-42. The ANCs for all sites in these surficial layers are low. Figure 5-42 shows the ANCs of the surface layers (0 - 20 cm) have decreased between May 2010 and March 2013, whereas the ANC of the 30 – 40 cm layer has apparently increased. This increase is most likely (as for the increase in PSA discussed earlier) the result of sediment erosion caused by wave action in the lake waters effectively making residual ANC in formerly more deeply buried and more alkaline sediments layers, closer to the sediment's new eroded surface.

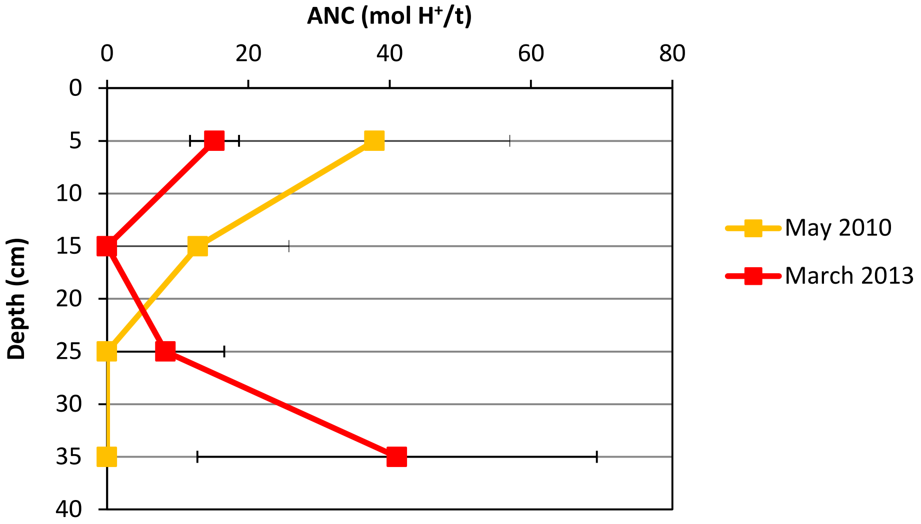

Figure 5-42. Poltalloch Acid Neutralising Capacity dynamics at the Bevy rye site (May 2010 and March 2013).

5.1.3.4.4 Retained Acidity

Retained Acidity was not measured at this site on any occasion. This is surprising given the presence of jarositic segregations around root holes in the subsurface sediment layers at this site. This result may be explained by the recent findings of Vithana et al. (2013) who demonstrated that the Retained Acidity method is not an accurate quantification procedure for jarosite and it use can lead to an underestimation of the Retained Acidity hazard posed by acid sulfate soil materials.

5.1.3.4.5 Net Acidity

Again perhaps surprisingly, despite the appreciable increases in pH in the surficial layers at this site since re-inundation, as for the Campbell Park site, there are no consistent trends in the Net Acidity of these layers as measured by the Ahern *et al.* (2004) method (see Figures 5-35 and 5-36) apart from minor increases in Net Acidity in the surficial 0 – 10 cm layer since inundation. It is important to note that the Net Acidity in this layer is still negative and that this increase was due to a decrease in the ANC of this layer since re-inundation.

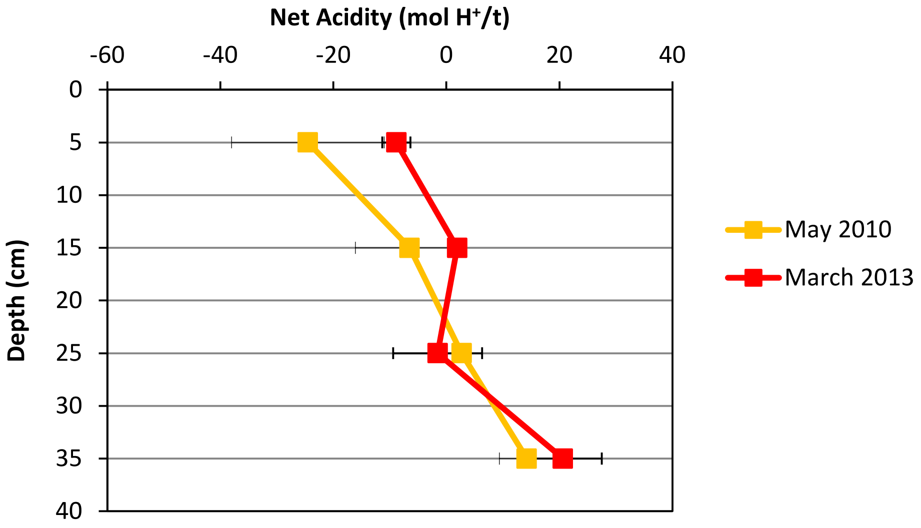

Figure 5-43. Poltalloch Net Acidity dynamics at the Bevy rye site (May 2010 and March 2013).

5.1.4 Waltowa

5.1.4.1 pH$_{(1:1, soil:water)}$

All sites initially (i.e. before refilling in August 2010) had slightly acidic subsoil layers from 10 – 40 cm, especially the site under *Cotula* where the pH in the 20 – 30 cm layer was ~4 (Figures 5-44 – 5-46). Upon near lake filling in August 2010 the pHs of these soil layers dropped considerably probably due to acidity exchange from the soil by the salts in the inundating waters.

The two treatment sites that had Aglime previously applied to the surface (i.e. the *Phragmites* and *Juncus* treatments) displayed surface pHs of about 8 - 8.5 initially, but when their sediment surfaces became and remained saturated the pHs of these layers were thereafter maintained at a pH of ~ 7.

In the unlimed *Cotula* treatment site the pH of the surface layer initially decreased from ~ 7 to ~ 5 from May 2010 to August 2010, but thereafter increased to ~pH 7 under the inundated conditions.

Under all treatments at this study area and at each depth to 40 cm the pH of the sediment had increased by ~ 0.5 of a unit since last sampled in March 2012.

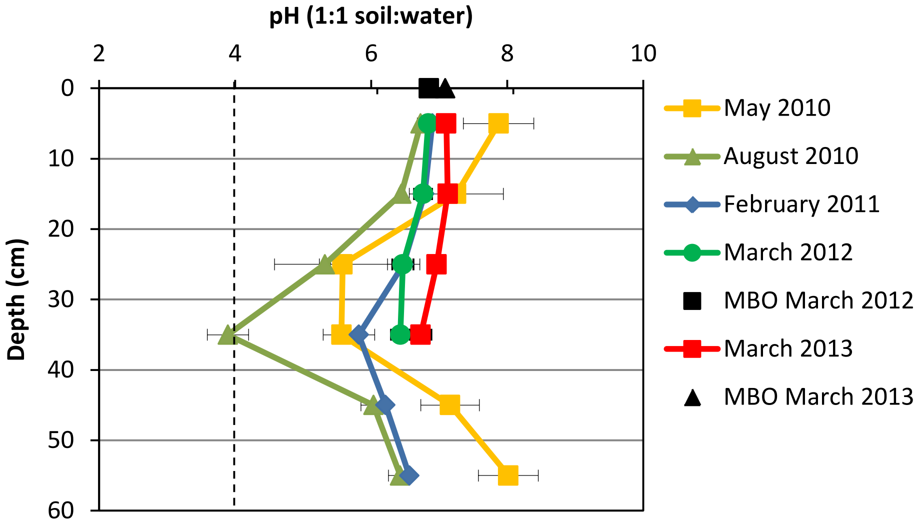

Figure 5-44. Waltowa field pH dynamics at the established *Phragmites* site (May 2010 – March 2013).

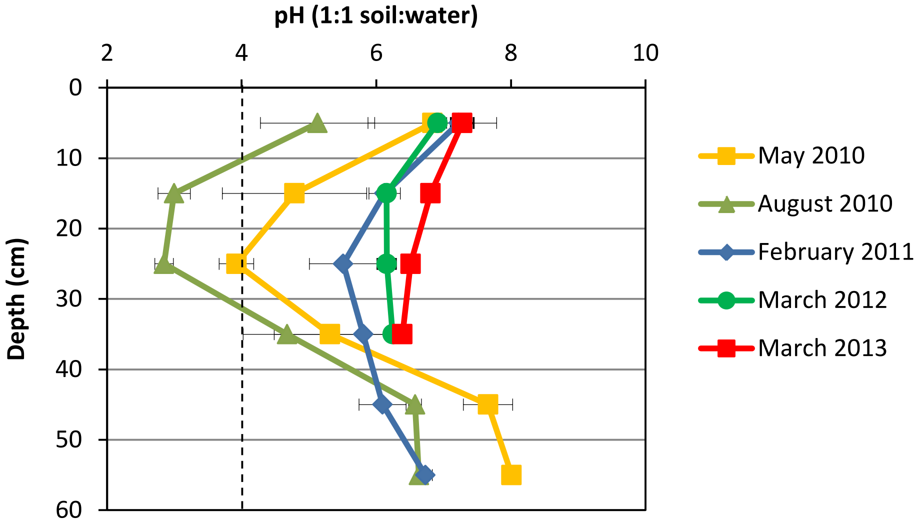

Figure 5-45. Waltowa field pH dynamics at the established *Cotula* site (May 2010 – March 2013).

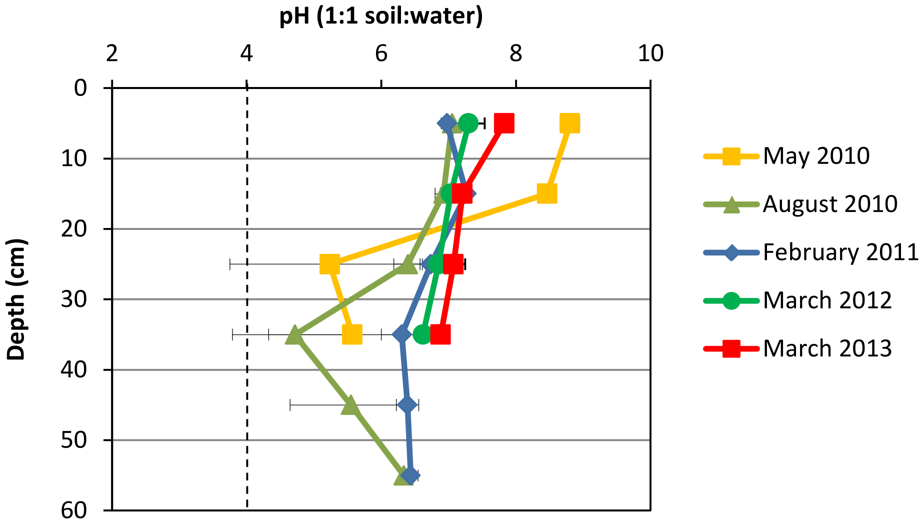

Figure 5-46. Waltowa field pH dynamics at the established *Juncus* site (May 2010 – March 2013).

5.1.4.2 Redox Potential (Eh)

All sites initially (i.e. in May 2010) had oxic conditions (Figures 5-47 – 5-49), but during inundation increasingly reductive conditions between 0 – 200 mV developed in the whole profile down to 40 cm by February 2011. Since that sampling date the 0 - 20 cm sediment depth layers under all treatments have increased slightly but are still < 200 mV (Figures 5-47 – 5-49).

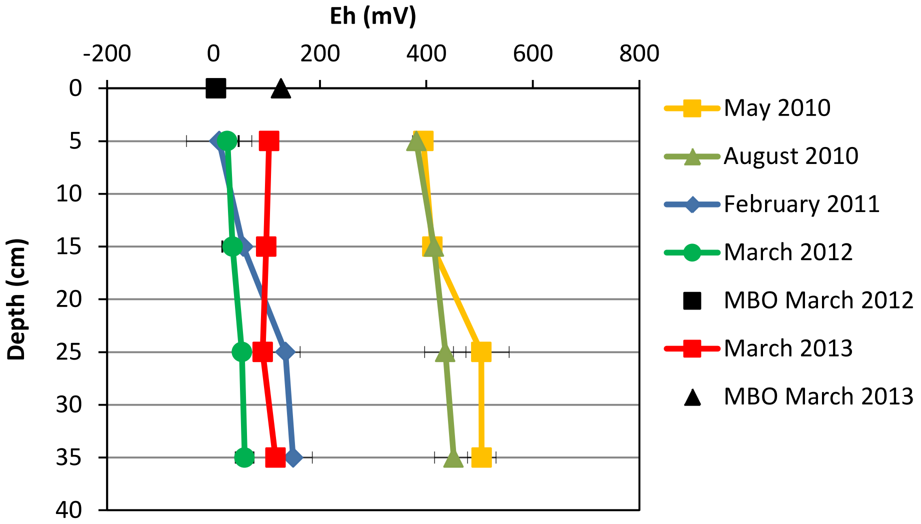

Figure 5-47. Waltowa field Eh dynamics at the established *Phragmites* site (May 2010 – March 2013).

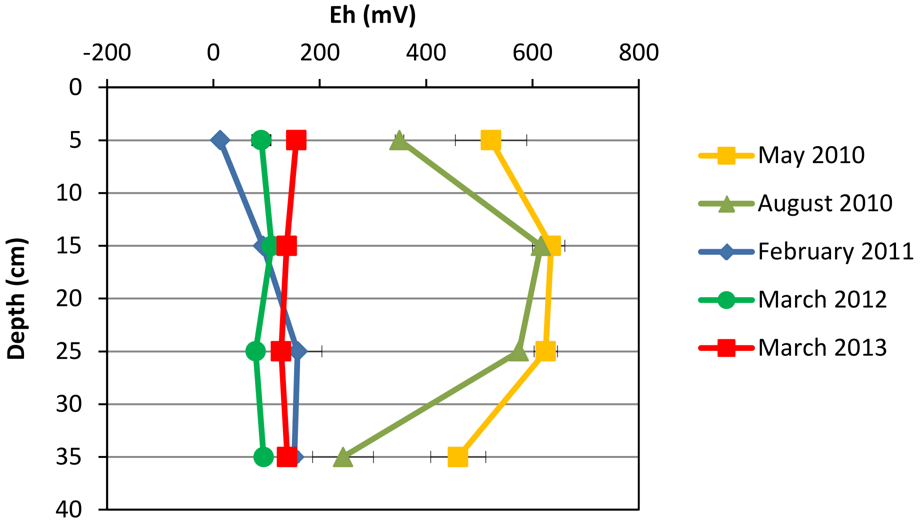

Figure 5-48. Waltowa field Eh dynamics at the established *Cotula* site (May 2010 – March 2013).

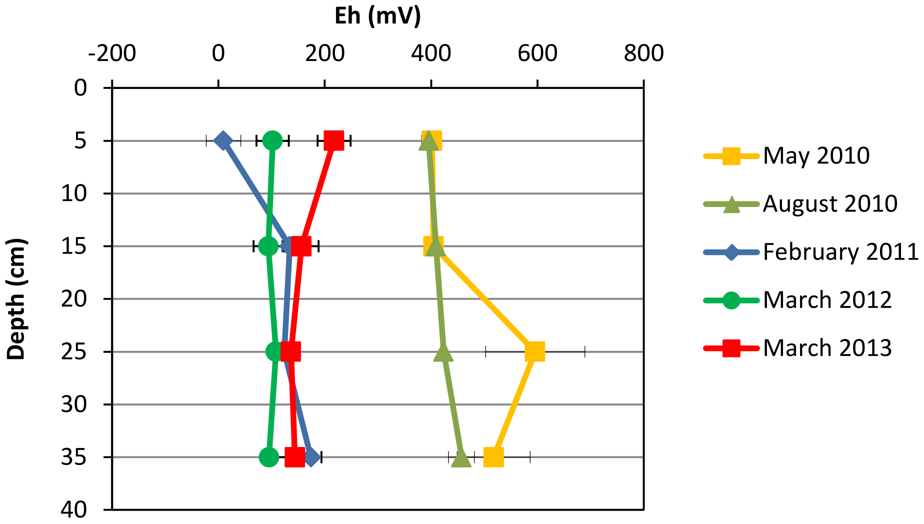

Figure 5-49. Waltowa field Eh dynamics at the established *Juncus* site (May 2010 – March 2013).

5.1.4.3 Electrical Conductivity (EC)

The salinities (i.e. EC) in the sediments have continued to decrease during inundation. As shown in Figures 5-50 – 5-52 the salinity in all treatments was between 125 and ~1000 µS cm^{-1} in the surface layers down to ~30 cm depth, but increased to up to ~4,200 µS cm^{-1} in the layers down to 60 cm. As for the sediments at the other sites this loss of salinity is most likely due to diffusion of salts from the sediments to the overlying lake waters (see Figure 9-23, Appendix 6).

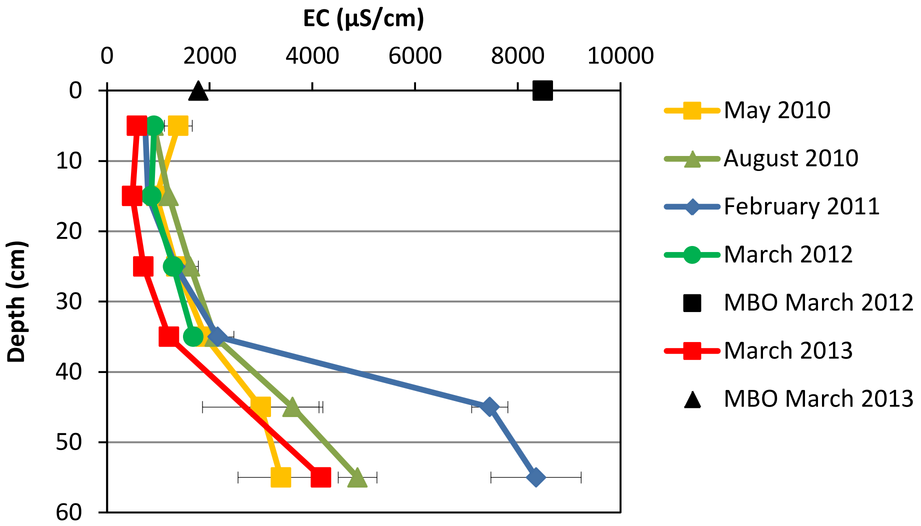

Figure 5-50. Waltowa EC dynamics at the established *Phragmites* site (May 2010 – March 2013).

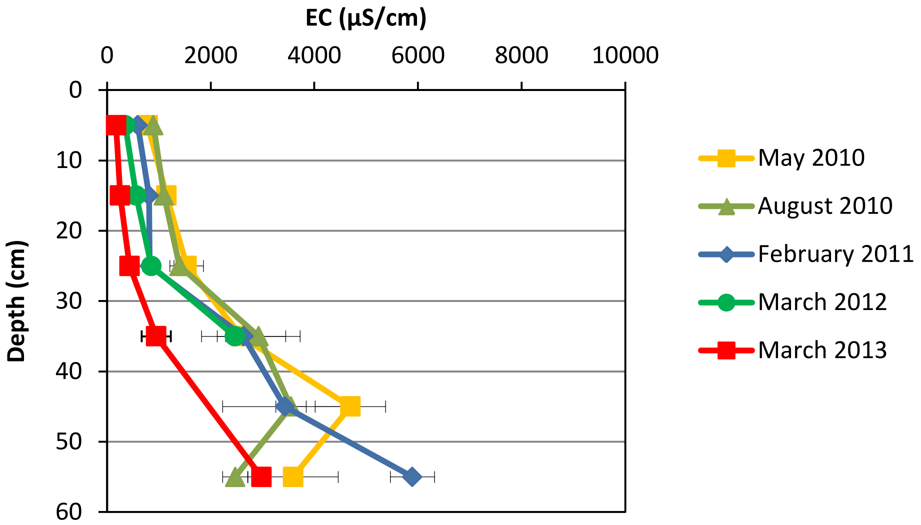

Figure 5-51. Waltowa EC dynamics at the established *Cotula* site (May 2010 – March 2013).

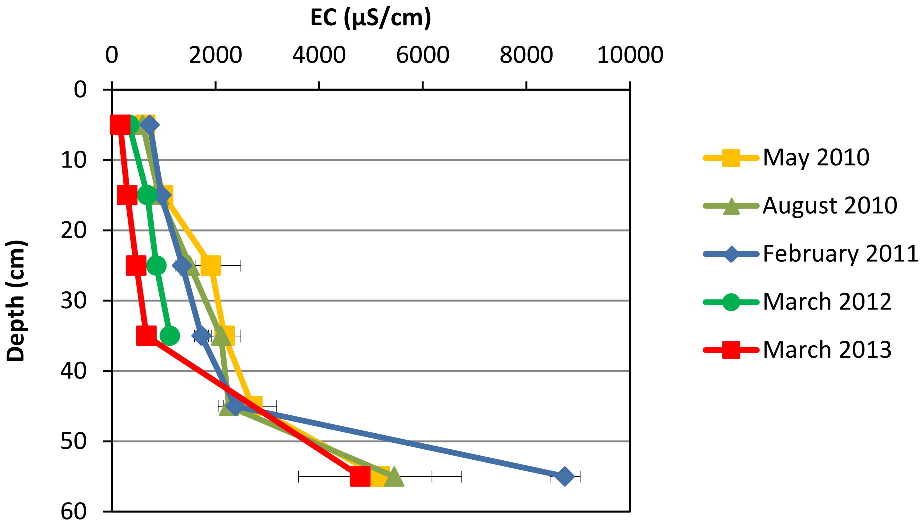

Figure 5-52. Waltowa EC dynamics at the established *Juncus* site (May 2010 – March 2013).

5.1.4.4 Acid-Base Accounting

5.1.4.4.1 Potential Sulfidic Acidity (PSA)

Graphs showing the PSA in May 2010 and March 2013 for the three sites are presented in Figures 5-53, 5-54 and 5-55. These graphs indicate that the Reduced Inorganic Sulfur components increased appreciably in some of the surficial layers at the *Phragmites* site (Figure 5-53) and *Cotula* site (Figure 5-54) as a result of inundation.

The pyritic sulfur contents in the top 30 cm of sediments were very low in all sites prior to inundation (Figures 9-6 – 9-8, Appendix 4). The deeper soil materials contained pyrite at all sites. There was evidence of accumulation of appreciable concentrations of reduced inorganic sulfides (i.e. up to 0.08% S as pyrite), especially in the *Phragmites* site profile and the *Cotula* surface layer after 6 months of inundation (i.e. the February 2011 data). With another 13 months of inundation pyrite continued to accumulate in the upper surface layers under the *Phragmites* treatment, but declined in the *Cotula* upper surface layer in which it had previously accumulated. In March 2013 the pyrite concentration continued to accumulate in the 0 - 2.5 cm surface layer under the *Phragmites* treatment with a concentration of 0.14% S. The apparent accumulation of appreciable concentrations of reduced inorganic sulfides in the 30 – 40 cm layers at the *Phragmites* and *Cotula* sites is most likely the result of sediment erosion caused by wave action in the lake waters effectively making residual reduced inorganic sulfides, formerly more deeply buried, closer to the sediment's new eroded surface.

The concentration of Acid Volatile Sulfide (i.e. monosulfides) has trended in parallel with the pyrite concentrations (Figures 9-9 – 9-11, Appendix 4). It is of importance that an appreciable depth of Monosulfidic Black Ooze (MBO) had begun to accumulate by March 2012 on the surface of the *Phragmites* treatment. The surface layers of the *Phragmites* treatment (i.e. the top 2.5 cm) contained >0.05 %S as monosulfides in March 2013.

Elemental sulfur (Figures 9-15 – 9-17, Appendix 4) was still present in the surficial layers at each site but in very low concentrations (i.e. <0.03% S).

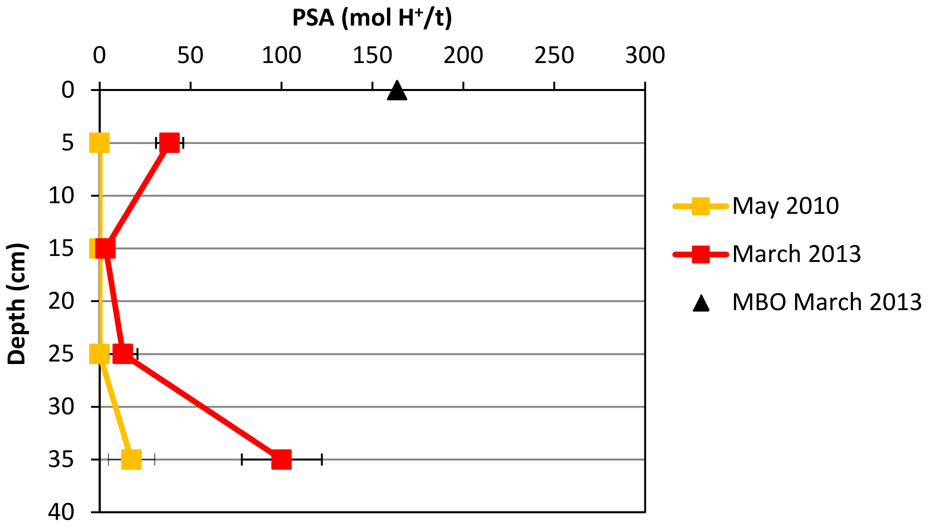

Figure 5-53. Waltowa Potential Sulfidic Acidity dynamics at the established *Phragmites* site (May 2010 and March 2013).

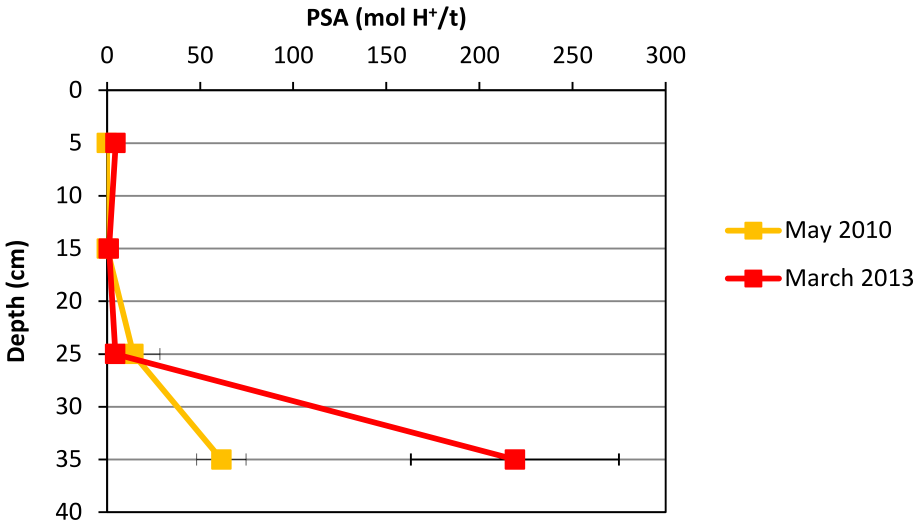

Figure 5-54. Waltowa Potential Sulfidic Acidity dynamics at the established *Cotula* site (May 2010 and March 2013).

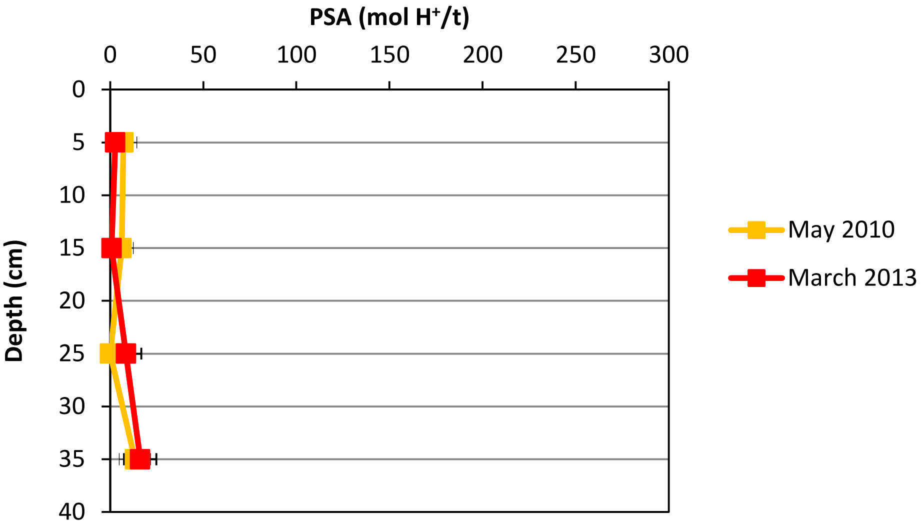

Figure 5-55. Waltowa Potential Sulfidic Acidity dynamics at the established *Juncus* site (May 2010 and March 2013).

5.1.4.4.2 Titratable Actual Acidity (TAA)

The TAAs (Figures 5-56 – 5-58) were all very low (i.e. < 18 mol H^+ t^{-1}) in each soil layer, and were especially low in the surface sediment layer of the *Cotula* site (i.e. initially ~ 2 mol H^+ t^{-1}) and the limed *Phragmites* and *Juncus* sites. In line with the observed pHs, the TAAs of the sediment had generally decreased further since last sampled in March 2012.

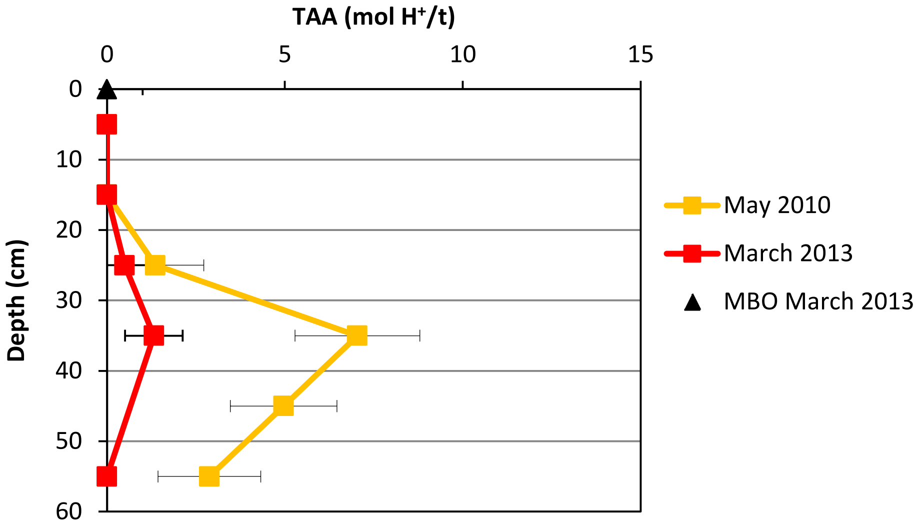

Figure 5-56. Waltowa TAA dynamics at the established *Phragmites* site (May 2010 and March 2013).

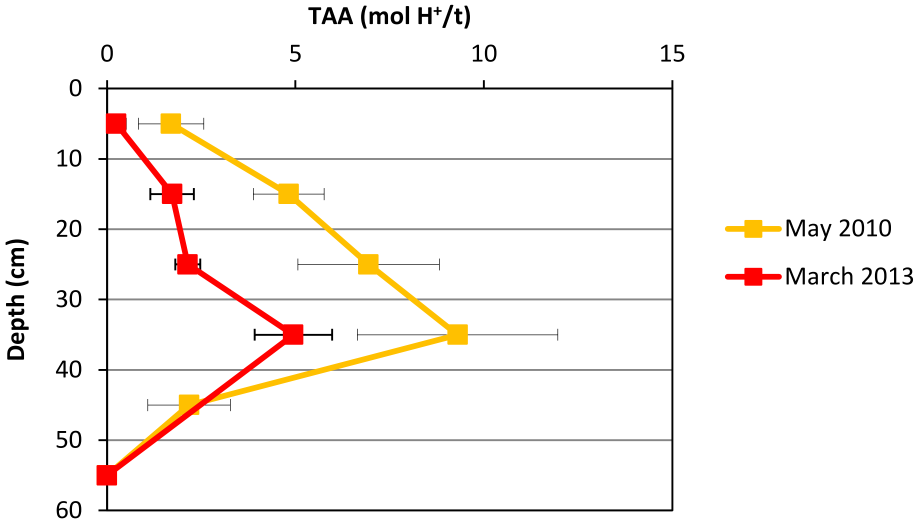

Figure 5-57. Waltowa TAA dynamics at the established *Cotula* site (May 2010 and March 2013).

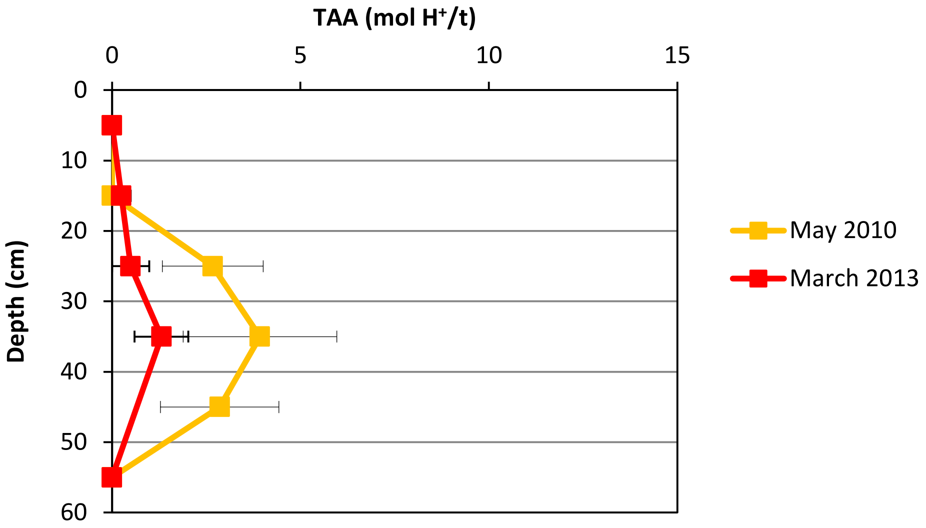

Figure 5-58. Waltowa TAA dynamics at the established *Juncus* site (May 2010 and March 2013).

5.1.4.4.3 Acid Neutralising Capacity (ANC)

The graphs of ANC are presented in Figures 5-59 to 5-61. The ANCs for the 0-10 cm layer at the Waltowa site remained high no doubt due to the initial liming when this treatment was planted and interestingly has remained high. On the other hand the 0-10 cm layer at the *Juncus* site that was also limed initially has been largely been removed. There is clear evidence of a small accumulation of ANC in the surficial sediment layer under the *Cotula* treatment.

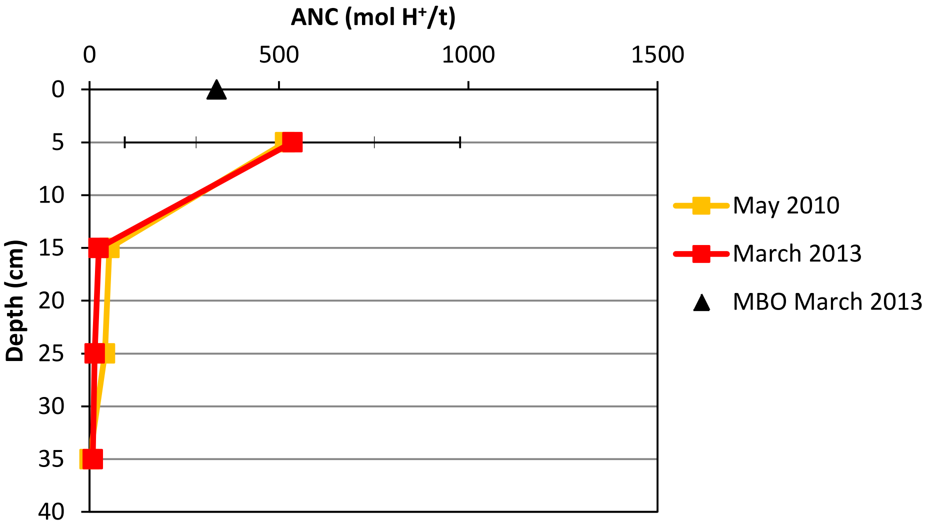

Figure 5-59. Waltowa Acid Neutralising Capacity dynamics at the established *Phragmites* site (May 2010 and March 2013).

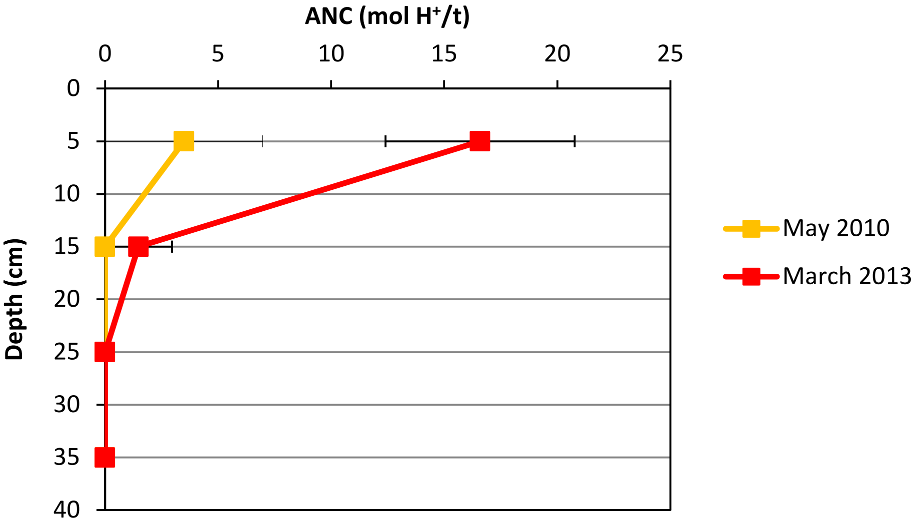

Figure 5-60. Waltowa Acid Neutralising Capacity dynamics at the established *Cotula* site (May 2010 and March 2013).

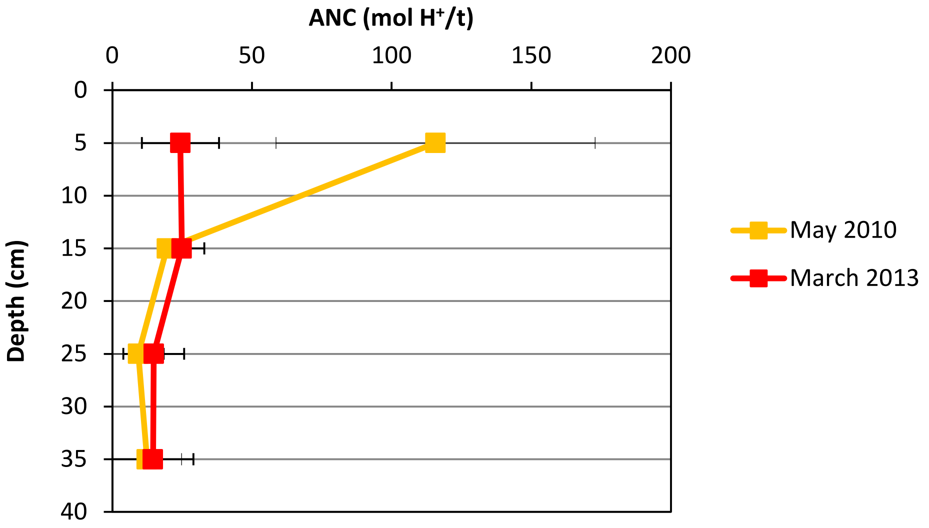

Figure 5-61. Waltowa Acid Neutralising Capacity dynamics at the established *Juncus* site (May 2010 and March 2013).

5.1.4.4.4 Retained Acidity

Only very low amounts of Retained Acidity were detected at the *Phragmites* and *Cotula* sites initially. No Retained Acidity was detected at these sites during the March 2013 sampling. Retained Acidity was not detected at the *Juncus* site at any of the sampling periods.

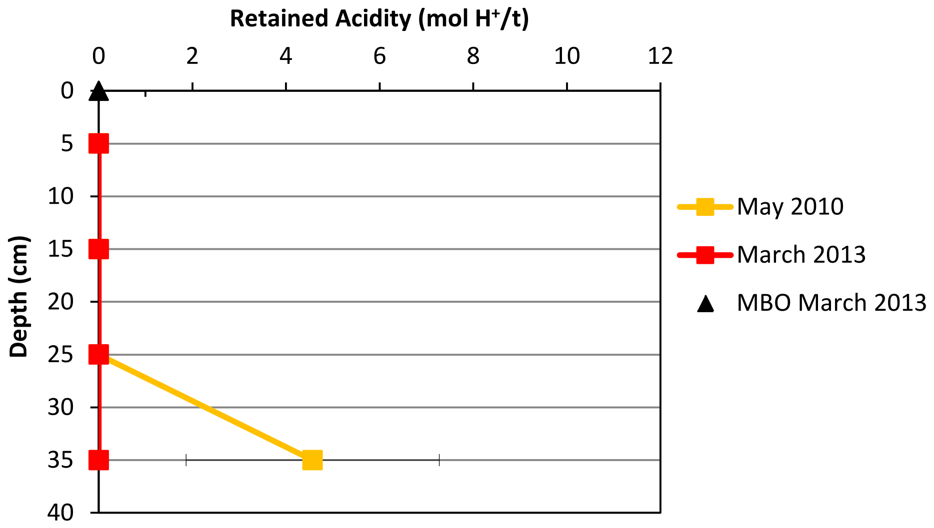

Figure 5-62. Waltowa Retained Acidity dynamics at the established *Phragmites* site (May 2010 and March 2013).

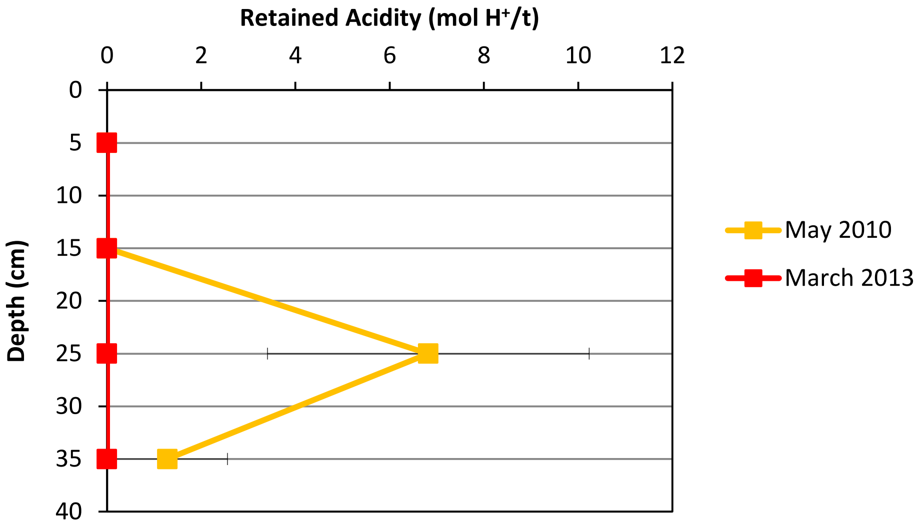

Figure 5-63. Waltowa Retained Acidity dynamics at the established *Cotula* site (May 2010 and March 2013).

5.1.4.4.5 Net Acidity

Again perhaps surprisingly, despite the appreciable increases in pH in the surficial layers at this site since re-inundation, as for the other sites, there are no consistent trends or appreciable changes in the Net Acidity of the sediments at either the *Phragmites* or *Cotula* sites as measured by the Ahern *et al.* (2004) method (see Figures 5-64 and 5-65). This is despite an appreciable increase in the PSA in the surficial layer of the *Phragmites* site.

There was a considerable increase in the Net Acidity of the surficial sediment at the *Juncus* site, largely as a result of a substantial decrease in the ANC: most likely due to dissolution of the lime applied during planting of the *Juncus* (Figure 5-66). It is important to note that the Net Acidity of this surficial layer was still negative.

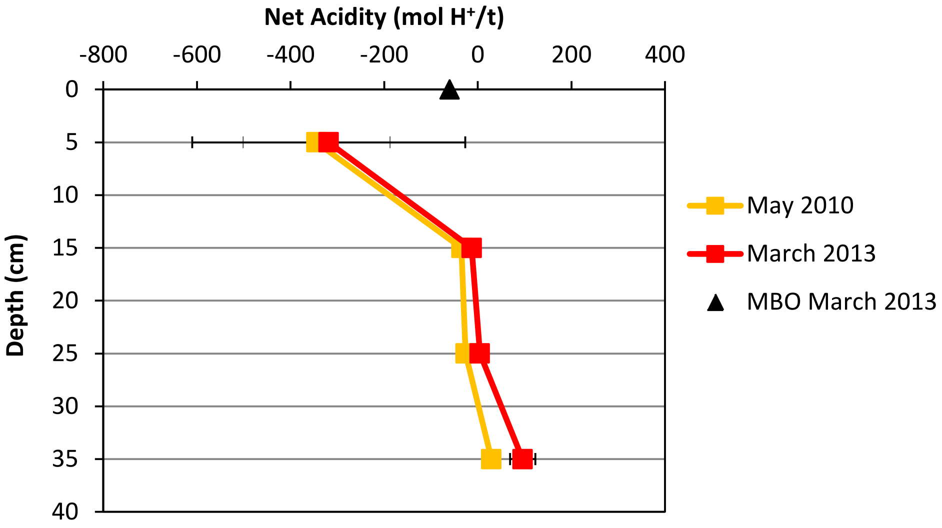

Figure 5-64. Waltowa Net Acidity dynamics at the established *Phragmites* site (May 2010 and March 2013).

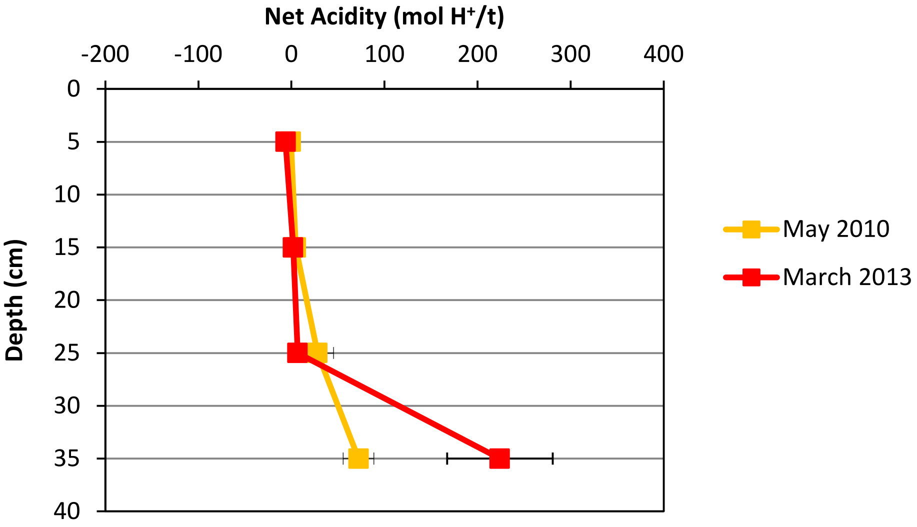

Figure 5-65. Waltowa Net Acidity dynamics at the established *Cotula* site (May 2010 and March 2013).

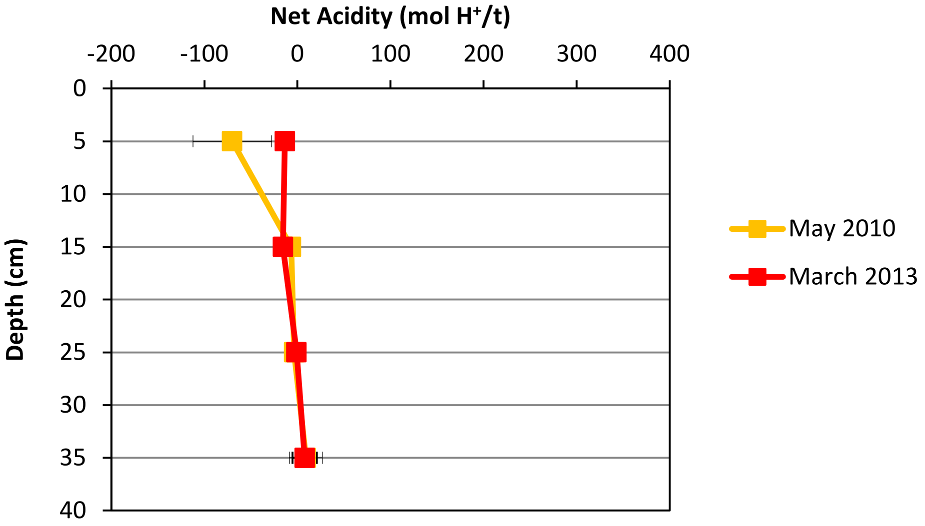

Figure 5-66. Waltowa Net Acidity dynamics at the established *Juncus* site (May 2010 and March 2013).

5.2 Metal Concentrations in Vegetation

A component of this project was a preliminary scoping investigation to examine the likely mobility and uptake by vegetation of metals (particularly nickel and zinc) from the acidified lake sediments as affected by bioremediation. Previous studies have shown that acidic sediment layers had pore-water nickel and zinc concentrations that greatly exceeded the respective water quality guidelines for ecosystem protection (e.g. Sullivan *et al.* 2011). More recently some of the vegetation surrounding the Lower Lakes has been found to contain high concentrations of nickel (Sullivan *et al.* 2012a). High metal concentrations in the vegetation of the Lower Lakes have the potential to impact on the food web of the ecosystem. The vegetation metals data for this project was examined and presented in a report by Danielle Post (Post 2013). The section largely presents the findings of Post (2013).

A summary of the metal/metalloid concentrations in the vegetation at the sites examined is presented in Table 5-1; further details of the concentrations at each site are given in Table 9-23, Appendix 5. High concentrations of some of the metals (particularly iron (Fe), aluminium (Al), manganese (Mn) and nickel (Ni)) were measured in some of the vegetation samples. Low concentrations of arsenic (As), cadmium (Cd) chromium (Cr), lead (Pb) and silver (Ag) were found in the majority of the vegetation samples analysed; the concentrations of these metals were often below the detection limits of between 0.1 and 2.0 mg/kg depending on the metal/metalloid analysed.

Table 5-1. Summary of metal/metalloid concentrations (mg/kg) observed in vegetation around the Lower Lakes (March 2013).

Metal/Metalloid	Stem Concentration	Leaf Concentration	Flower Concentration	Concentration Range
Iron (Fe)	54 – 987	133 – 1,299	81 – 1,245	54 – 1,299
Aluminium (Al)	66 – 1,036	77 – 1,152	112 – 1,053	66 – 1,152
Arsenic (As)	<2	<2	<2	<2
Cadmium (Cd)	<0.5	<0.5	<0.5	<0.5
Chromium (Cr)	<2 – 3	<2 – 9	<2 – 10	<2 – 10
Copper (Cu)	2 - 13	3 - 12	4 - 9	2 - 13
Lead (Pb)	<1 – 118	<1 - 2	<1	<1 – 118
Manganese (Mn)	27 – 1,703	59 – 2,636	69 – 270	27 – 2,636
Nickel (Ni)	5 – 135	5 – 574	6 – 173	5 – 574
Selenium (Se)	<0.1 – 4.4	<0.1 – 2.2	0.6 – 3.4	<0.1 – 4.4
Silver (Ag)	<1	<1	<1	<1
Zinc (Zn)	3 - 58	8 - 54	10 - 44	3 - 58

The distribution of the dominant metals observed in the vegetation at each site is presented in Figure 5-67. Aluminium, iron and manganese were consistently observed to be at the highest concentrations. The highest mean concentrations of aluminium and iron were found at Currency Creek, Naro Point and the former scald at Tolderol, with particularly high concentrations of manganese at West Meningie (Figure 5-67). Low concentrations of aluminium, iron and manganese were observed at the two Hunters Creek sites and the Tolderol foredune site.

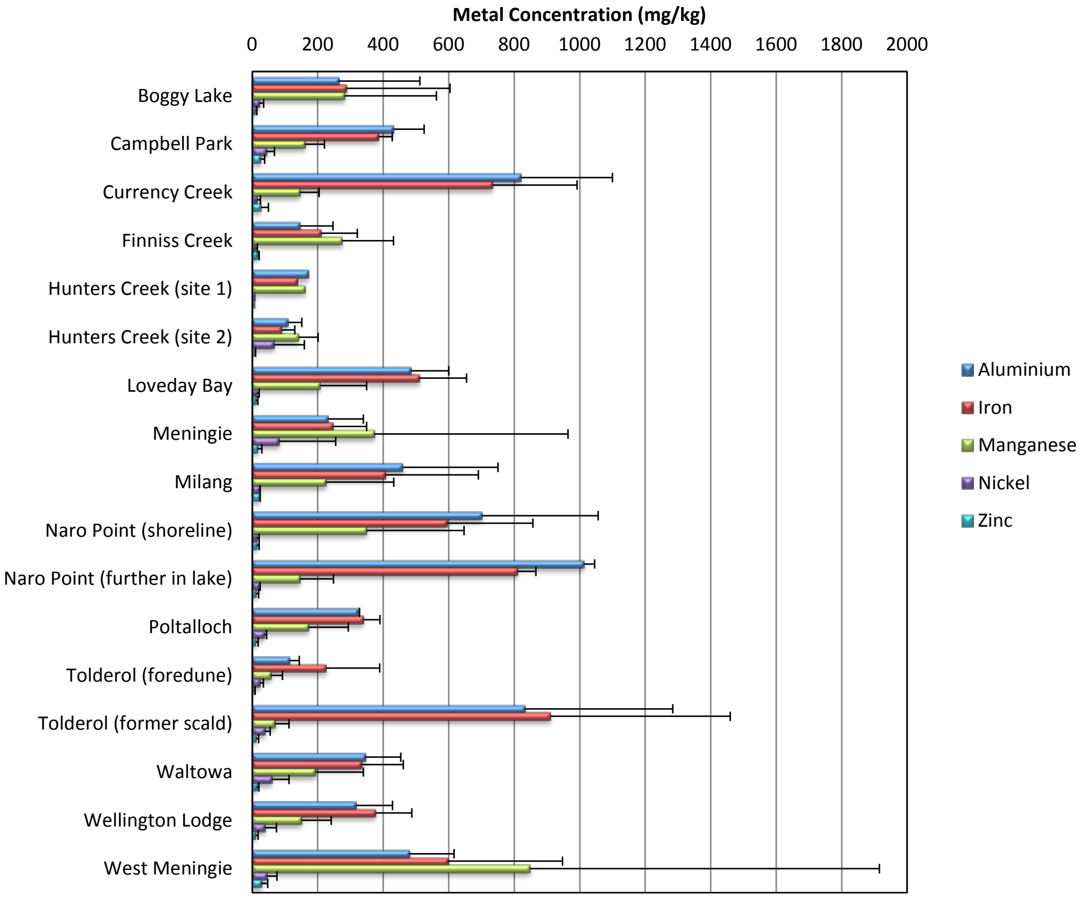

Figure 5-67. Average concentration of aluminium, iron, manganese, nickel and zinc at each site (The error bars indicate the standard deviation).

Further detail of the variation in the nickel and zinc concentrations of the vegetation between sites is presented in Figure 5-68. High mean nickel concentrations of ≥ 60 mg/kg were found at Hunters Creek (site 2), Meningie and Waltowa. Low concentrations of nickel and zinc were observed at Hunters Creek (site 1). The concentrations of nickel in the leaves and stems were often considerably higher than those reported by Sullivan *et al.* (2012a).

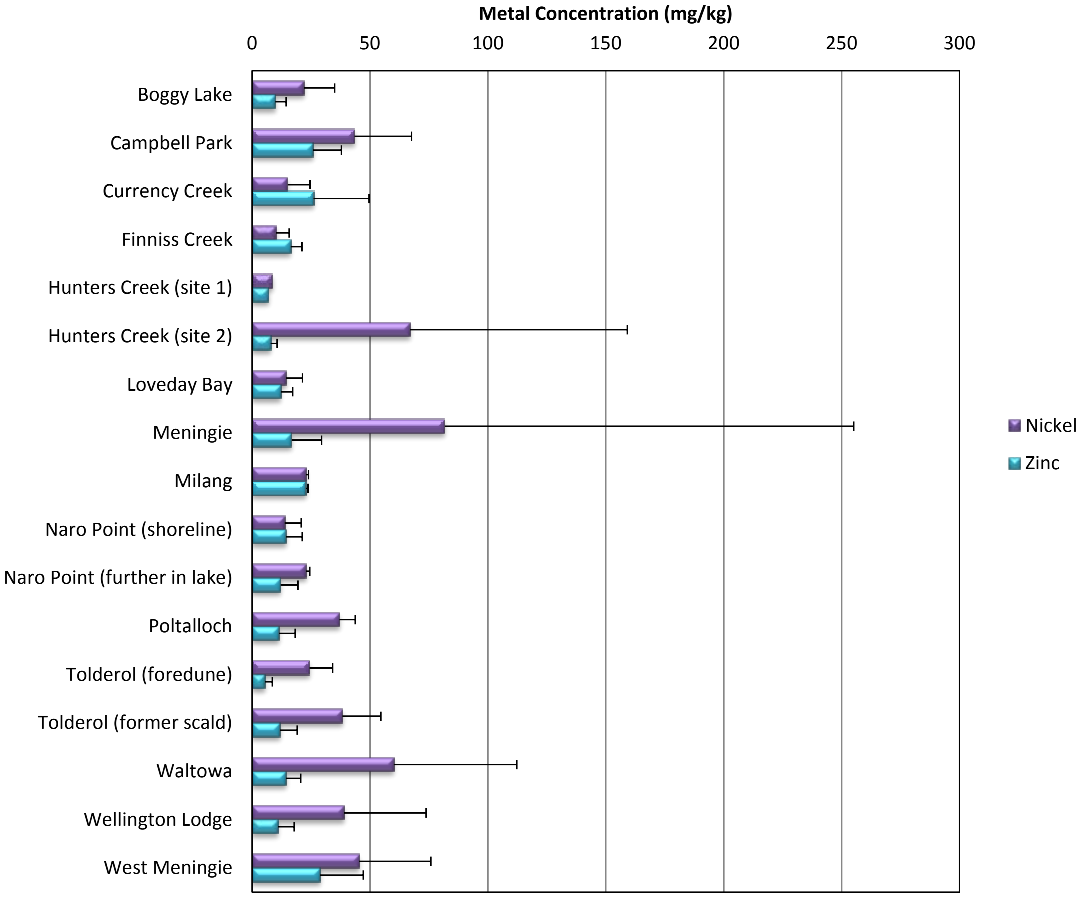

Figure 5-68. Average concentration of nickel and zinc at each site (The error bars indicate the standard deviation).

Further examination of the metal concentrations in the *Phragmites australis* collected from the shoreline and further within the lake at Naro Point shows no consistent differences in the metal concentrations between the sites (see Table 9-23, Appendix 5). However, a comparison of the metal concentrations in the *Phragmites australis* samples collected from the foredune and the inundated former scald at the Tolderol site shows substantial differences in metal uptake (see Figure 5-69). Both the aluminium and iron concentrations are much higher in both the stem and leaves collect from the former acidic scalded site (Figure 5-69). The mean nickel and zinc concentrations are also slightly higher in the vegetation collected from the former scalded site than the foredune (Figure 5-68). Figure 5-69 also indicates that many of the metals are more concentrated in the leaves rather than the stems of *Phragmites australis* at Tolderol.

A comparison between the *Schoenoplectus valaidus* stem collected from the shoreline and 50 m from the shoreline at Meningie also shows a substantially higher manganese concentration in the vegetation collected from the shoreline, and slightly higher aluminium, iron and zinc concentrations (see Table 9-23, Appendix 5).

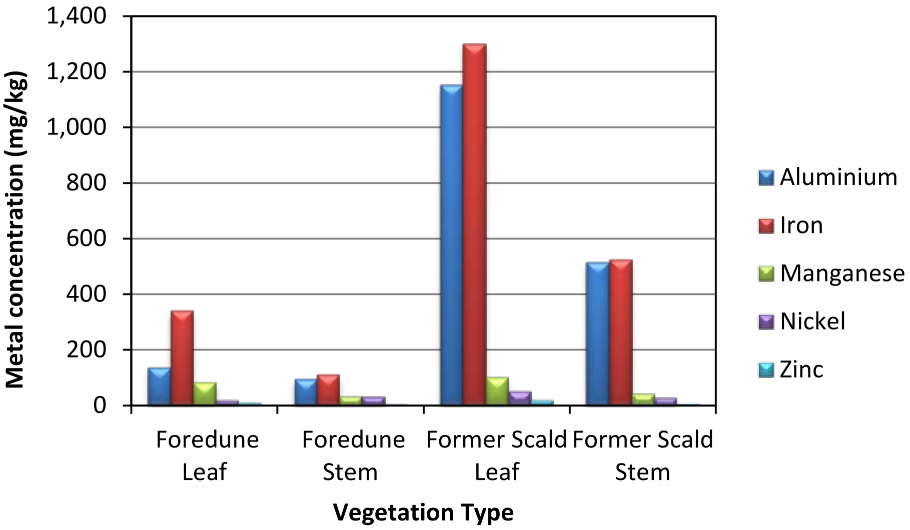

Figure 5-69. Metal concentrations in the leaves and stem of the *Phragmites australis* collected from the foredune and former scald at the Tolderol site.

Most of the sites sampled for vegetation in this study had a near neutral pH at the time of sampling (see Table 9-22, Appendix 5); only Currency Creek had soil pH values of less than 5. However, many of the sites experienced acidification during the recent drought resulting in the potential mobilisation of metals. It is expected that acidification did not occur at the Tolderol foredune site, and this site had relatively low concentrations of many of the metals analysed. Low metal concentrations were also observed at the Hunters Creek sites and acidification was known to occur in this region of the Lower Lakes. The high metal concentrations in some of the vegetation at Meningie may be from nearby potential anthropogenic sources, however, the previously acidified sediments are the likely source of many of the high concentrations observed.

It is interesting to note that the vegetation at sites with clayey surface soils rather than sandy surface soils (such as Boggy Lake, Campbell Park, Finniss Creek and Hunters Creek) tended to have lower metal concentrations. It is expected that metals would be less available in clayey soils when compared to sandy soil due to metal adsorption by clay minerals.

In addition to there being a considerable variation in the metal concentration of the vegetation between sites, a large variation was also observed between vegetation species (see Figures 5-70 and 5-71). The highest mean metal concentrations were found in the Sedge at West Meningie and the unidentified Rush at Boggy Lake (Figure 5-70). The highest mean nickel concentrations of >60 mg/kg were found in the Sedge at West Meningie and the *Schoenoplectus valaidus* (Figure 5-71). Low metal concentrations were found in the unidentified reed at Meningie and the *Bolboschoenus* at Hunters Creek (site 2) (Figures 5-70 and 5-71).

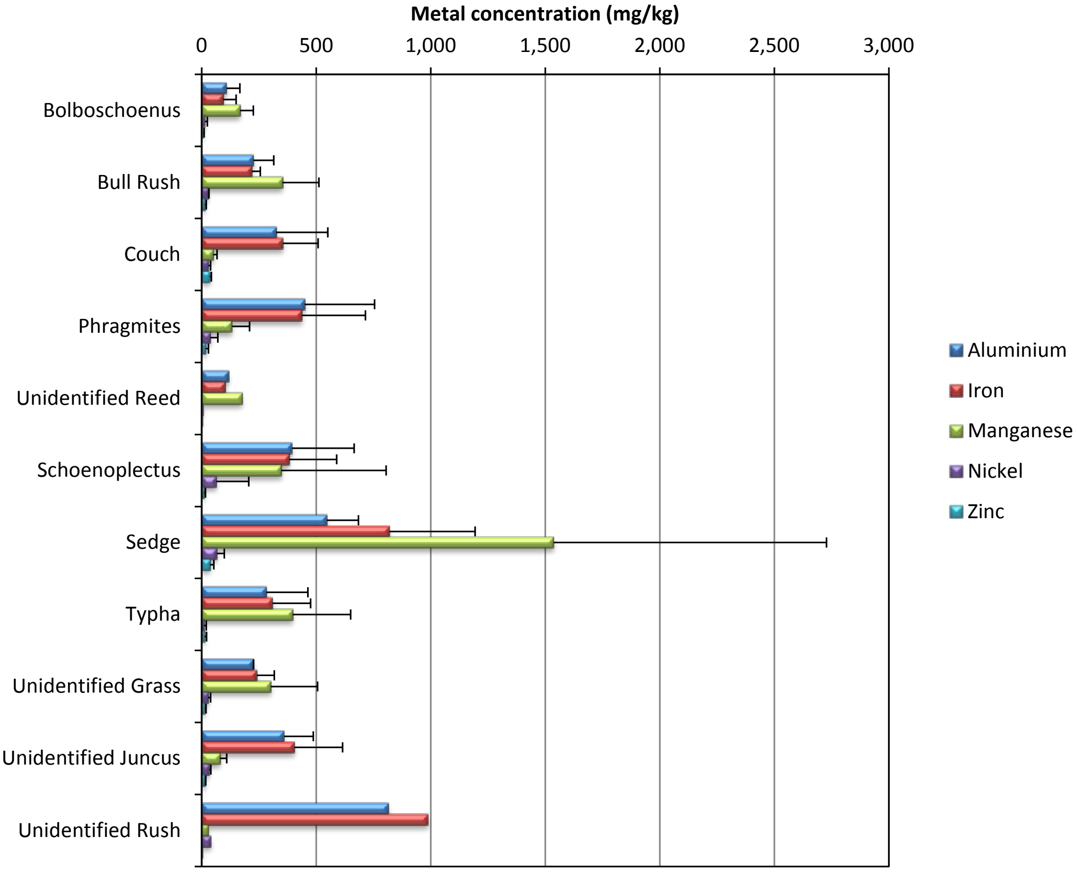

Figure 5-70. Average concentration of aluminium, iron, manganese, nickel and zinc in each vegetation type (The error bars indicate the standard deviation).

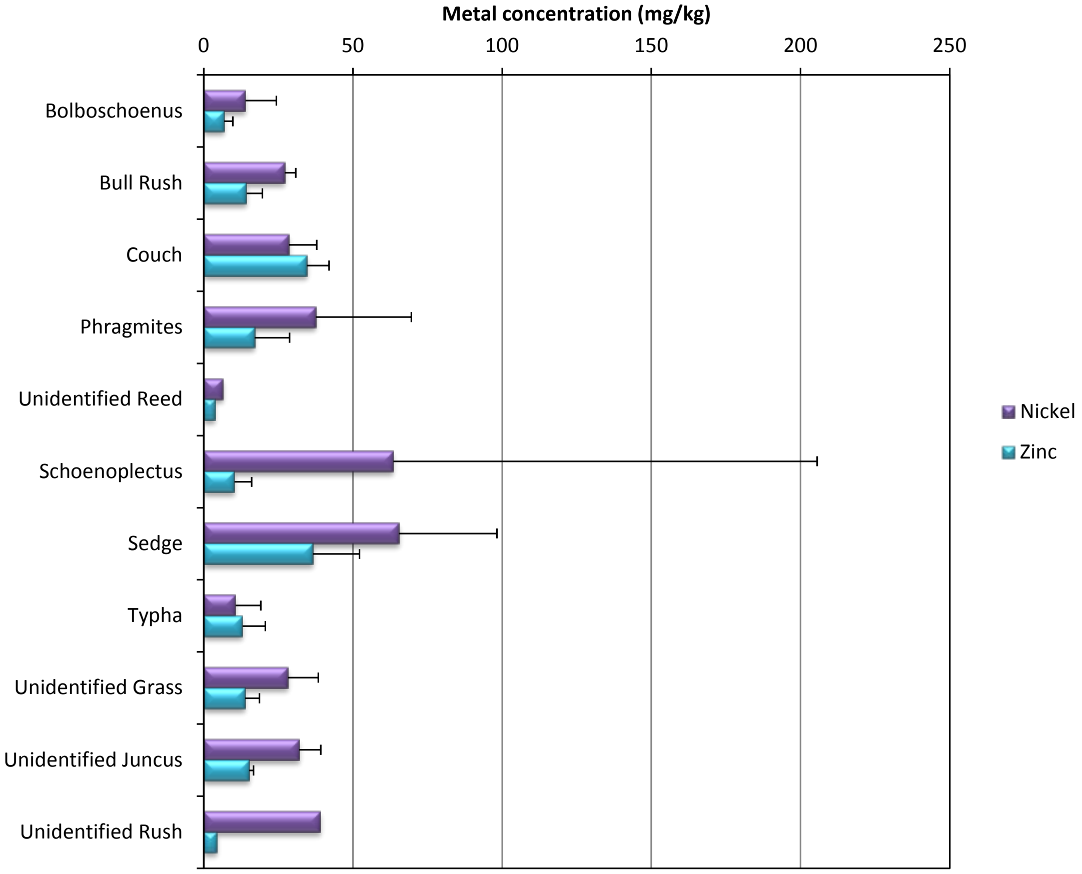

Figure 5-71. Average concentration of nickel and zinc in each vegetation type (The error bars indicate the standard deviation).

As mentioned previously, a comparison between the metal content in the vegetation collected from the former scalded site and the foredune at Tolderol indicated that many of the metals were more concentrated in the leaves rather than the stems of the *Phragmites australis* (Figure 5-69). To get an understanding of where the metals were concentrating, the metal concentrations in the leaves, stems and flower were compared for all the vegetation examined (Figure 5-72) and the three main vegetation types collected (i.e. *Phragmites australis, Schoenoplectus valaidus and Typha orientalis*) (Figures 5-73 – 5-75).

Figure 5-72 would initially suggest that aluminium and iron tend to be more concentrated in the vegetation flowers. However, only six flower samples were collected in this study and the flowers of the *Schoenoplectus valaidus* at Currency Creek and Sedge at West Meningie were particularly high in these two metals. While many of the metals tended to be more concentrated in the leaves rather than the stems (Figure 5-72), when the metal concentrations in the stems and leaves were compared within species this relationship with some of the metals was not always observed (see Figures 5-73 – 5-75).

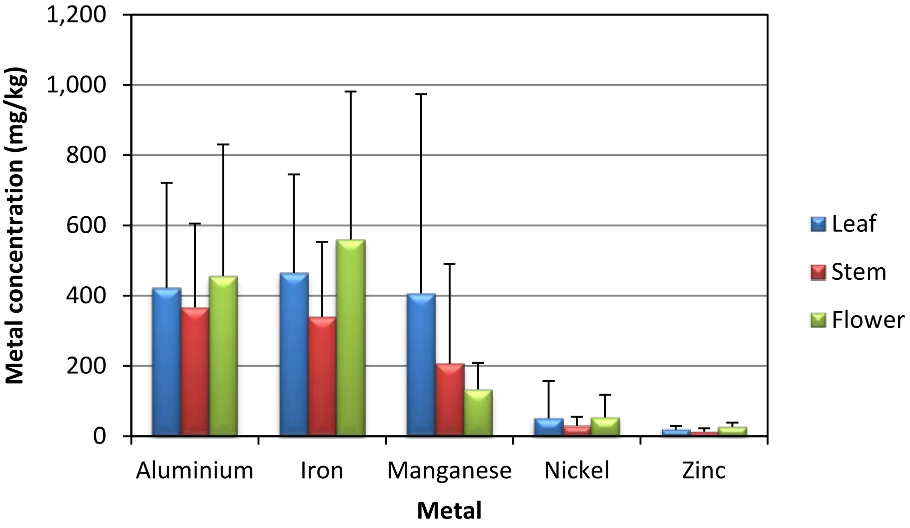

Figure 5-72. Average metal concentrations found in the leaves, stem and flower of the vegetation at all sampling sites (The error bars indicate the standard deviation).

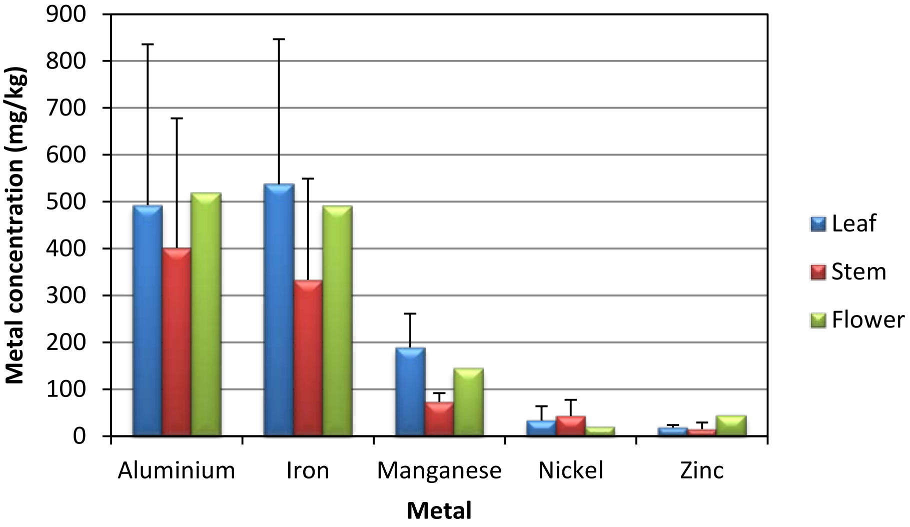

Figure 5-73. Average metal concentrations found in the leaves, stem and flower of *Phragmites australis* at all sampling sites (The error bars indicate the standard deviation).

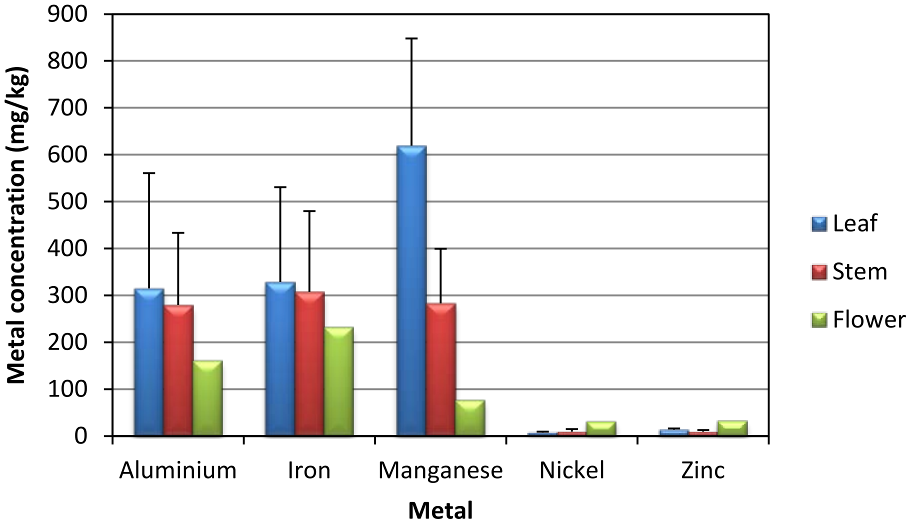

Figure 5-74. Average metal concentrations found in the leaves, stem and flower of *Typha orientalis* at all sampling sites (The error bars indicate the standard deviation).

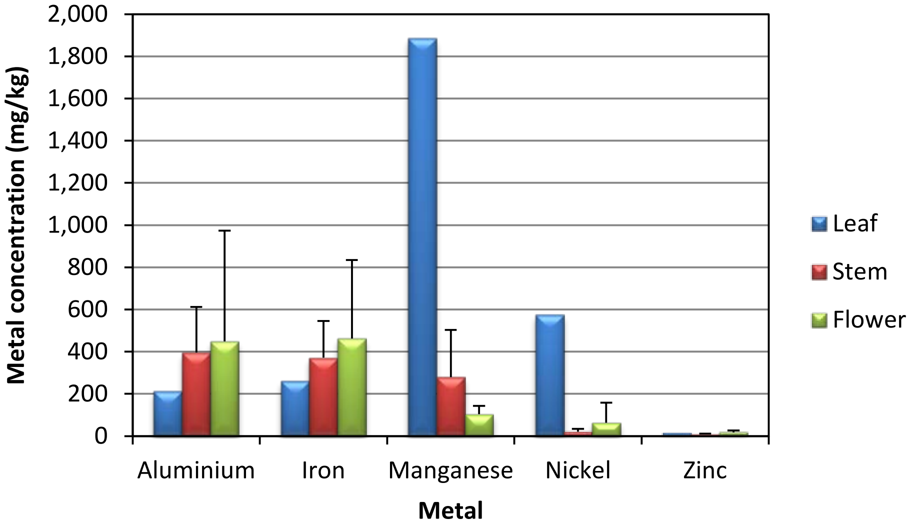

Figure 5-75. Average metal concentrations found in the leaves, stem and flower of *Schoenoplectus valaidus* at all sampling sites (The error bars indicate the standard deviation).

In conclusion, the concentrations of many of the metals and metalloids (i.e. Cu, Co, Cd, Cr, As, Pb, Se,) were generally low in the plant materials. There were some clear differences in the uptake and accumulation of metals by different vegetation types. The *Bolboschoenus* species sampled here having much lower accumulations of metals than most of the other vegetation types. Many of the other vegetation types exhibited similar metal uptake behaviour.

There were clearly effects of soil on the accumulation of metals by the vegetation. Figure 5-69 clearly shows that the metal contents of Al and Mn were much higher in the *Phragmites* growing on the former acid sulfate soils compared to the *Phragmites* growing on the foredune at Tolderol. This indicates that the soil is playing a major role in determining the uptake of these metals into the *Phragmites* with higher metal contents in the former acid sulfate soils as would be expected from the previous studies (e.g. Sullivan *et al.* 2011) that have shown elevated metal concentrations in inundated former acid sulfate sediments from the Lower Lakes. In addition, vegetation growing at sites with clayey textured surface soils tended to have lower metal concentrations than vegetation growing at sites with sandy surface soils, which is likely due to metal adsorption by clay minerals.

The nickel concentrations in most natural vegetation range from 0.05 to 5 mg/kg dry weight (NAS 1975). The vegetation around the Lower Lakes all had appreciably higher nickel concentrations that these levels. The *Schoenoplectus* and the sedge materials sampled in these sediments contained on average over an order of magnitude greater concentrations of nickel that the uppermost value for natural vegetation range (i.e. > 50 mg/kg dry weight). Normal levels of zinc in most crops and pastures range from 10 mg/kg to 100 mg/kg (WHO 2001) so the levels of zinc observed in the vegetation around the lakes is not unusual.

5.3 Discussion

5.3.1 Recovery of the Acidified Sediment Layers

The data clearly shows that there has been a continuing increase in pH at all sites since inundation. This increase in pH is most apparent in the surficial sediment layers but is also apparent in the sub-surface layers further down to the 60 cm. This is no doubt the result of processes some of which are likely to be consequent of the bioremediation, and some not. Given the lake pH and the trajectories in pH it would appear that most sediments are back to or near the pH expected prior to the lake drawdown and sediment desiccation. However, it is also clear that especially for the jarositic sediment layers that a much longer period of inundation will be required to restore the pH conditions of these sediments.

The surprising result in this study is that despite the regular and consistent increases in sediment pH across all sites there were no consistent appreciable decreases in Net Acidity across these sites.

Conventionally, in acid sulfate soil management this result means that despite better (i.e. less acidic) pHs having developed in these inundated sediments, that the acidity hazard (this is measured by the Net Acidity) has not changes appreciably nor consistently. This is surprising because we would expect that remediated acid sulfate soils would consistently experience both increases in pH from acidic conditions, as well as decreases in Net Acidity, usually from positive values to negative values.

Increases in pH from acidic conditions and decreases in Net Acidity as a result of the remediation of acid sulfate sols is usually the result of the addition of liming materials to the acid sulfate soil materials. In the case of these acid sulfate sediments fringing the lakes that have been re-inundated it would be expected that the addition of liming materials to the acid sulfate sediments may have been supplied by the alkalinity (mainly bicarbonate) being transferred from the alkaline lake waters to the acidic sediments by either mass flow or diffusion. However, the lack of consistent and appreciable decreases in Net Acidity across these sites in line with the regular and consistent increases in sediment pH indicates that other processes apart from addition of ‘external’ (i.e. to the sediment) alkalinity have been important factors affecting the Net Acidities of the sediments.

One such factor clearly is that some geochemical processes that can effect appreciable change to the pH of the sediment do not necessarily change the Net Acidity. For example, dissimilatory sulfate reduction during organic matter decomposition (that in sediments often leads to the formation of pyrite) (see equation 3.6), whilst increasing the pH of soils due to the production of HCO_3^- also often leads to the formation of reduced inorganic sulfides such as pyrite that represent a store of potential acidity. This store of acidity is included in the Net Acidity. This sulfate reduction, whilst leading to increases in sediment pH may actually have no net effect in reducing the Net Acidity of that sediment. Indeed if the relatively mobile and alkaline HCO_3^- is lost from the sediment via diffusion or mass flow, then sulfate reduction may actually result in net increases in the Net Acidity of that sediment.

The magnitude of the changes in Potential Sulfidic Acidity in the sediment layers in this study are generally not large enough to ascertain whether this process is occurring appreciably apart from under *Phragmites* at the Waltowa site. However, the Ag-lime added to this site during the establishment of the *Phragmites* unfortunately precluded an examination of whether there has been a build up of alkalinity (measurable as Acid Neutralising Capacity) commensurate with the accumulation of Potential Sulfidic Acidity (as a result of sulfide accumulation).

Another factor that is a major process in some of the sediments and that leads to changes in pH but does not appreciably affect the Net Acidity is the transfer between different acidity pools in the sediment. A clear example of this is the transfer of acidity in the Retained Acidity pool to the Titratable Actual Acidity pool in sediments that were jarositic. Effectively this is a transfer from acidity stored in jarosite (and related minerals) and that is measured in the Retained Acidity pool, to the soluble Fe^{2+} that is measured in the Titratable Actual Acidity pool, as a result of reductive dissolution. This effect is clearly shown in Figures 5-13 and 5-14. This process has led to very high concentrations of soluble iron (i.e. over 800 mg/L) in the pore waters of formerly jarositic sediments. In sediments soluble Fe^{2+} will likely be measured, after drying of those sediments prior to analysis in the Titratable Actual Acidity pool.

Despite the lack of overall trends in Net Acidity of these sediments after prolonged inundation, there were some site-specific trends observed.

In the surface layers of the initially severely acidic, jarositic sites Net Acidity has generally decreased due to a decrease in Retained Acidity, sometimes a decrease in Titratable Actual Acidity, and often an increase in the Acid Neutralisation Capacity. The decrease in Retained Acidity is clearly due to the reductive dissolution of jarosite as discussed previously. The reasons for the increase in Acid Neutralisation Capacity are beyond the boundary of this study but are likely due to either the deposition of biogenic carbonates from the lake water column onto the sediment surface, or the biogeochemical precipitation of calcareous materials within these sediments.

On the other hand at the initially neutral or limed sites, the Net Acidity has generally increased due to a decrease in Acid Neutralisation Capacity, sometimes an increase in Potential Sulfidic Acidity, and despite small decreases in Actual Acidity. The decrease in Acid Neutralisation Capacity may be due to removal (physically or via dissolution) of surface applied liming material at the *Juncus* Waltowa site where there has been a considerable lowering of Acid Neutralisation Capacity since inundation, or of carbonates originally accumulated on the surficial sediment at these sites during the drawdown of the lakes and the desiccation of these sediments.

In contrast to the decrease in the Acid Neutralisation Capacity at the *Juncus* Waltowa site, the Acid Neutralisation Capacity at the other limed site (the *Phragmites* Waltowa site) had not changed appreciably since re-inundation of the sediment. The reason for this is not apparent but if the process responsible for removal of the surface applied liming material at the *Juncus* Waltowa site was physical erosion of these sediments (little if any of the original *Juncus* material remain on site), then the surface applied liming material at the far more densely vegetated *Phragmites* site would have offered substantial protection against the physical removal of these materials.

The increases in Potential Sulfidic Acidity in these layers are certainly due to sulfate reduction processes which as has been noted previously (e.g. Sullivan *et al.* 2012b) is constrained by the amount of available organic materials present in these sediments. Thus at the *Phragmites* Waltowa site where the living vegetation is providing appreciable quantities of organic materials the accumulation of sulfides has been considerable (i.e. Potential Sulfidic Acidity of ~40 mol H^+/t). In addition to the accumulation Potential Sulfidic Acidity in the surface layers of this site there has been an accumulation of Monosulfidic Black Ooze.

The accumulation of reduced inorganic sulfides in the surface layers of the sediments under the *Phragmites* site only, also represent the accumulation of environmental hazards including acidity, and localised deoxygenation should these sediments be mobilised and or exposed to drying in the future. It is noted that the accumulation of reduced inorganic sulfides in the surface layers of the sediments under the *Phragmites* site continues and is expected to continue whilst ever there remain no constraints to this process.

The data on EC and the Cl concentrations clearly demonstrate that there has been considerable movement of solutes out of these sediment profiles. Presumably this movement has been dominated by diffusion caused by the observed considerable concentration gradients in salts towards the lake.

Similar concentration trends have also been observed for the SO_4 concentration gradients and hence also the trends in $Cl:SO_4$ ratios. However as discussed previously there has been appreciable accumulation of reduced inorganic sulfides in some layers clearly indicating that there has been sulfate reduction and that this process may in addition to solute movement have affected both the SO_4 concentration gradients and the trends in $Cl:SO_4$ ratios observed at these sites.

Given the above data that clearly shows there has been considerable movement of solutes out of these sediment profiles, it is surprising that there is no clear indication in the data that the HCO_3^- from lake waters has been a major source of alkalinity to the underlying acidic sediments despite considerable increases in the pHs of this sediment. Such a clear indication would have been a substantial decrease in the Net Acidities of these sediments. However substantial and consistent decreases in the Net Acidities of the surficial layers were not observed at the sites.

The data above necessitates an examination of the utility of the ABA method of Ahern *et al.* (2004) for the purpose it was employed for in this study. It has been acknowledged that this method has great utility for the management of acid sulfate soils, especially for the assessment of the acidity hazard and in predicting liming rates for the amelioration of acid sulfate soils that have been or will be disturbed. The ABA method has been widely employed also in the assessment of the hazards that Australian inland wetlands containing acid sulfate soils may pose. However, this study is perhaps the first to employ this method to research changes in acidity pools within wetland sediments over time under changing conditions.

Firstly, on the positive side the ABA method has been able to sensibly and clearly quantify changes in some of the acidity pools at each site. For example, it is not surprising that the Potential Sulfidic Acidity has increased in line with measured increases in reduced inorganic sulfur as each relies on the same measurement. However, the ABA method has also clearly demonstrated changes in the Retained Acidity pool in line with field observations at some sites and sediment layers with the initial presence of abundant jarosite and the subsequent dissolution of this jarosite under prolonged inundation and the imposition of reducing conditions. Under these conditions the decrease in Retained Acidity has been accompanied by an increase in the Titratable Actual Acidity pool. Additionally, the ABA method has been able to clearly identify and sensibly quantify changes in the Acid Neutralisation Capacity of sediment layers.

However, on the negative side the Retained Acidity method used for the ABA method was not able to identify and quantify the presence of jarosite especially in the Poltalloch site where the presence of jarosite in some sediment layers was, prior to re-inundation, clearly identifiable in appreciable amounts. These field results confirm the recent findings of Vithana *et al.* (2013) who show that both of the methods used for Retained Acidity quantification in the ABA method of Ahern *et al.* (2004) are unable to fully recover of jarosite and under some situations, especially low concentrations of jarosite, may not even identify the presence of jarosite. In addition it is acknowledged that the methods used for the quantification of Acid Neutralisation Capacity in acid sulfate soils need improvement.

5.3.2 Metal concentrations in lake vegetation

Universally accepted critical metal contents for wetland vegetation are not available. However, the data clearly shows elevated levels of some metals in the vegetation growing in the formerly acidified sediments. The levels of Mn in the plant tissue (mean of 240 mg/kg but up to 2,636 mg/kg) were well above the published criteria for the limit for most plants of 50 mg/kg, “above which toxicity may be observed and if it is taken above this level by animals via food chain is hazardous” (Jarvis and Whitehead 1981).

The nickel concentrations in most natural vegetation range from 0.05 to 5 mg/kg dry weight (NAS 1975). The vegetation around the Lower Lakes had appreciably higher nickel concentrations than these levels. The *Schoenoplectus* and the sedge materials sampled in these sediments contained on average over an order of magnitude greater concentrations of nickel than the uppermost value for natural vegetation range. Indeed, the levels of Ni in the plant tissue (mean of 35 mg/kg but up to 574 mg/kg) were much higher than the published ranges for fodder of 1 - 10 mg/kg (e.g. NAS 1975).

The levels of Al in the plant tissue (mean of 412 mg/kg but up to 1,500 mg/kg) were well above the published criteria for the limit for most plants of 200 mg/kg.

The levels of Fe in the plant tissue (mean of 475 mg/kg but up to 6,100 mg/kg) were high but generally within the published ranges for fodder plants of 18 to 1,000 mg/kg.

In contrast, the levels of Cu, Zn, Pb, As, Cd in the plant tissue were below the published elevated level criteria for most plants. This is of interest especially in the case of zinc. Along with nickel, zinc was the other metal that was indicated to be of concern in earlier studies of metal mobility in inundating acid sulfate lake sediments. However the data showed no unusually high levels of zinc in the vegetation sampled. For example, normal levels of zinc in most crops and pastures range from 10 mg/kg to 100 mg/kg (WHO 2001) so the level of zinc observed in the vegetation around the lakes was not unusual.

There were clearly effects of soil on the accumulation of metals by the vegetation. Figure 5-69 clearly shows that the metal contents of Al and Mn were much higher in the *Phragmites* growing on the former acid sulfate soils compared to the *Phragmites* growing on the foredune at Tolderol. This indicates that the soil is playing a major role in determining the uptake of these metals into the *Phragmites* with higher metal contents in the former acid sulfate soils as would be expected from the previous studies (e.g. Sullivan *et al.* 2012b) that have shown elevated metal concentrations in inundated former acid sulfate sediments from the Lower Lakes. The data indicates that the vegetation growing at sites with clayey textured surface soils had lower metal concentrations than the vegetation growing at sites with sandy surface soils: this is likely due to metal adsorption by clay minerals making the metals and metalloids less available for plant uptake.

Finally, the data indicate that there were also clear differences in the uptake and accumulation of metals by different vegetation types. The *Bolboschoenus* species sampled here having much lower accumulations of metals than most of the other vegetation types.

6.0 Conclusions

The key findings of this study are:

1) There has been a continuing increase in pH at all sites mainly in the surficial sediment layers since re-inundation. Given the lake pH and the trajectories in pH it would appear that most sediments are back to or near the pH expected prior to the lake drawdown and sediment desiccation. However, it is also clear that especially for the jarositic sediment layers (observed at all sites except Waltowa) that a much longer period of inundation will be required to restore the pH conditions of these sediments.

2) Despite the regular and consistent increases in sediment pH across all sites there were no consistent appreciable decreases in Net Acidity across these sites. This is surprising because conventionally we would expect that remediated acid sulfate soils would consistently experience both increases in pH from acidic conditions, as well as decreases in Net Acidity, usually from positive values to negative values. The lack of consistent and appreciable decreases in Net Acidity across these sites in line with the regular and consistent increases in sediment pH indicates that other processes apart from addition of ‘external’ (i.e. to the sediment) alkalinity such as from the overlying lake waters have been important factors affecting the Net Acidities of the sediments. Most importantly the lack of consistent change in the Net Acidities of the lake sediments indicates that they have largely retained their acidity stores, have not built up stores of acid neutralising capacity and hence remain vulnerable to acidification in any future drying event.

3) The magnitude of the changes in Potential Sulfidic Acidity in the sediment layers in this study were generally not large enough to ascertain whether this process is occurring appreciably apart from under *Phragmites* at the Waltowa site. In addition to the accumulation of Potential Sulfidic Acidity in the surface layers of this site there has been an accumulation of Monosulfidic Black Ooze. The accumulation of reduced inorganic sulfur in the surface layers of the sediments under the *Phragmites*, also represent the accumulation of environmental hazards including acidity, and localised deoxygenation should these sediments be mobilised and or exposed to drying in the future.

4) There have clearly been transfers between different acidity pools in the sediment as a result of the prolonged inundation. A clear example of this is the transfer of acidity in the Retained Acidity pool to the Titratable Actual Acidity pool in sediments that were jarositic via the production of soluble Fe^{2+} as a result of reductive dissolution.

5) Despite the lack of overall trends in Net Acidity of these sediments after prolonged inundation, there were some site-specific trends observed. In the surface layers of the initially severely acidic, jarositic sites (e.g. Campbell Park scald), Net Acidity has generally decreased due to a decrease in Retained Acidity, sometimes a decrease in Titratable Actual Acidity, and often an increase in the Acid Neutralisation Capacity. On the other hand at the initially neutral or limed sites (e.g. *Phragmites* and *Juncus* Waltowa sites), the Net Acidity has generally increased due to a decrease in Acid Neutralisation Capacity, sometimes an increase in Potential Sulfidic Acidity, and despite small decreases in Actual Acidity.

6) The data on EC and the chloride concentrations clearly demonstrate that there has been considerable movement of solutes out of these sediment profiles. Given this, it is surprising that there is no clear indication in the data that the bicarbonate (HCO_3^-) from lake waters has been a major source of alkalinity to the underlying acidic sediments despite considerable increases in the pHs of this sediment.

7) The ABA method of Ahern *et al.* (2004) was shown to be useful for the purpose it was employed for in this study. This method was able to sensibly and clearly quantify changes in the acidity pools at each site. However, this study has identified and confirmed recent criticisms of the ABA’s methods for quantifying the Retained Acidity pool as this method failed to identify and quantify the presence of jarosite even when jarosite from field examination was clearly present in appreciable amounts.

8) Although universally accepted critical metal contents for wetland vegetation are not available, this study has clearly shown elevated levels of some metals in the vegetation growing in the formerly acidified sediments. These include manganese, nickel and aluminium. The levels of nickel in the plant tissue (mean of 35 mg/kg but up to 574 mg/kg) were much higher than the published ranges for fodder of 1 - 10 mg/kg (e.g. NAS, 1975),

and the levels of manganese in the plant tissue (mean of 240 mg/kg but up to 2,636 mg/kg) were well above the published criteria for the limit for most plants of 50 mg/kg, "above which toxicity may be observed and if it is taken above this level by animals via the food chain is hazardous" (Jarvis and Whitehead 1981). Such elevated metal concentrations are important as even moderate concentrations of metals can disrupt aquatic ecologies.

9) There were clearly effects of soil on the accumulation of metals by the vegetation with the metal contents much higher in vegetation growing on the former acid sulfate soils compared to that growing on the foredunes. Additionally, the vegetation growing at sites with clayey textured surface soils had lower metal concentrations than the vegetation growing at sites with sandy surface soils.

10) There were clear differences in the uptake and accumulation of metals by different vegetation types around the lake. The *Bolboschoenus* species sampled had much lower accumulations of metals than most of the other vegetation types. The Schoenoplectus and the sedge materials sampled in these sediments contained on average over an order of magnitude greater concentrations of nickel that the uppermost value for natural vegetation range (i.e. > 50 mg/kg dry weight).

7.0 Recommendations

1) This study along with previous studies has shown that the sediments of the lakes are recovering slowly but have not yet fully recovered. Our results show that these trajectories are different according to both the initial properties of the sediments (especially whether jarositic or not) and whether or not there is vegetation growing on these sediments. Some of these trajectories include environmental hazards that are clearly growing in magnitude. This is especially the case under vegetation where reduced inorganic sulfides continue to accumulate in the surficial sediments. It is our recommendation that these sites continue to be monitored over the next few years to adequately ascertain recovery trajectories and to understand future management challenges and opportunities arising from these trajectories.

2) This study has also highlighted the lack of consistent change in the Net Acidities of the lake sediments which indicates that they have not built up stores of acid neutralising capacity and hence remain vulnerable to acidification in any future drying event. It is our recommendation that these sites continue to be monitored for changes to their acidity pools to adequately ascertain acidity recovery trajectories and to understand how management can impact on these changes.

3) Finally, this study has confirmed previous studies of these sediments that the behaviour of metals mobilised by acid sulfate soil processes is a matter of concern because of their potential to impact aquatic ecosystems. The metals of concern are especially nickel, manganese and aluminium. It is our recommendation that further studies at these sites continue on the inter-related aspects of organic matter chemistry, metal mobilisation (in sediments, vegetation and other critical sectors of the lake's ecology) and sulfate reduction processes in the sediments.

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9.0 Appendices

APPENDIX 1. Site and sample descriptions

Table 9-1. Lower Lakes site and profile descriptions.

Location	Treatment	Date	Profile	GPS Co-ordinates Zone East, North.	Depth (cm)	pH	Eh* (mV)	Location and Profile Remarks
Waltowa	*Juncus* bioremediation	22/03/13	WJ 1	54H 0352057, 6059362	0-2.5	8.09	350	*Juncus* site down to 40 cm.
					2.5-5	8.21	215	
					5-10	7.45	161	Top ~5 cm white/beige wave-sorted sand beneath grey and beige mottled sand.
					10-15	7.08	163	
					15-20	6.84	209	0-30 cm: beige sand with iron segregations.
					20-30	6.73	164	30-40 cm: grey sand.
					30-40	6.64	161	40-70 cm: grey clay.
		22/03/13	WJ 2	54H 0352055, 6059357	0-2.5	7.76	263	*Juncus* site down to 40 cm.
					2.5-5	8.21	248	
					5-10	7.91	248	
					10-15	7.50	229	
					15-20	7.13	109	
					20-30	7.26	129	
					30-40	6.95	221	
		22/03/13	WJ 3	54H 0352054, 6059351	0-2.5	7.86	301	*Juncus* site down to 40 cm.
					2.5-5	7.77	141	
					5-10	7.66	137	
					10-15	7.41	140	
					15-20	7.28	88	
					20-30	7.23	117	
					30-40	7.06	50	
	Cotula bioremediation	22/03/13	WC 1	54H 0352237, 6059191	0-2.5	7.39	235	*Cotula* site down to 40 cm.
					2.5-5	7.13	144	
					5-10	6.86	142	0-30 cm: beige sand with iron segregations.
					10-15	6.67	121	30-40 cm: grey sand.
					15-20	6.47	136	40-70 cm: grey clay.
					20-30	6.37	119	
					30-40	6.45	107	
		22/03/13	WC 2	54H 0352215, 6059204	0-2.5	7.60	184	*Cotula* site down to 40 cm.
					2.5-5	7.28	124	
					5-10	7.61	143	
					10-15	7.04	121	
					15-20	6.84	203	
					20-30	6.69	150	
					30-40	6.33	164	
		22/03/13	WC 3	54H 0352243, 6059168	0-2.5	7.15	206	*Cotula* site down to 40 cm.
					2.5-5	6.96	157	
					5-10	7.44	126	
					10-15	7.15	109	
					15-20	6.68	139	
					20-30	6.47	114	
					30-40	6.39	146	

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-1 (continued). Lower Lakes site and profile descriptions.

Location	Treatment	Date	Profile	GPS Co-ordinates Zone East, North.	Depth (cm)	pH	Eh* (mV)	Location and Profile Remarks
Waltowa	*Phragmites* bioremediation	22/03/13	WP 1	54H 0352279, 6059118	0-2.5	6.88	135	*Phragmites* site down to 40 cm.
					2.5-5	6.91	80	
					5-10	7.10	91	0-30 cm: beige sand with iron segregations.
					10-15	7.29	77	30-40 cm: grey sand.
					15-20	7.02	93	
					20-30	6.94	81	
					30-40	6.57	116	
		22/03/13	WP 2	54H 0352290, 6059115	0	7.09	127	*Phragmites* site down to 40 cm.
					0-2.5	7.38	134	
					2.5-5	7.04	119	A ~12 cm deep (variable) MBO observed at the surface at this site.
					5-10	7.17	100	
					10-15	7.08	116	
					15-20	7.20	89	
					20-30	7.07	99	
					30-40	6.98	124	
		22/03/13	WP 3	54H 0352287, 6059106	0-2.5	7.04	102	*Phragmites* site down to 40 cm.
					2.5-5	6.96	108	
					5-10	7.25	99	
					10-15	7.08	103	
					15-20	7.08	117	
					20-30	6.88	98	
					30-40	6.63	110	
Poltalloch	2009 plantings of Bevy Rye bioremediation	23/03/13	P 1	54H 0341265, 6070662	0-2.5	8.02	342	Bevy Rye site down to 60 cm.
					2.5-5	7.42	228	
					5-10	6.97	168	0-3 cm: wave washed beige sand.
					10-15	6.88	150	3-11 cm: dark grey sand.
					15-20	6.78	158	11-32 cm: light grey sand with frequent orange segregations.
					20-30	6.78	180	>32 cm: grey sand.
					30-40	7.11	186	
					40-50	6.98	255	
					50-60	7.19	272	
		23/03/13	P 2	54H 0341286, 6070668	0-2.5	7.51	282	Bevy Rye site down to 60 cm.
					2.5-5	7.19	152	
					5-10	6.58	172	0-3 cm: wave washed beige sand.
					10-15	6.52	62	3-11 cm: dark grey sand.
					15-20	6.47	154	11-32 cm: light grey sand with frequent orange segregations.
					20-30	6.32	144	>32 cm: grey sand.
					30-40	6.52	151	
					40-50	6.53	200	
					50-60	7.07	183	

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-1 (continued). Lower Lakes site and profile descriptions.

Location	Treatment	Date	Profile	GPS Co-ordinates Zone East, North.	Depth (cm)	pH	Eh* (mV)	Location and Profile Remarks
Poltalloch	2009 plantings of Bevy Rye bioremediation	23/03/13	P 3	54H 0341304, 6070679	0-2.5	7.58	293	Bevy Rye site down to 60 cm.
					2.5-5	7.74	309	
					5-10	7.55	299	
					10-15	7.22	180	
					15-20	7.02	228	
					20-30	6.92	208	
					30-40	6.98	191	
					40-50	6.34	205	
					50-60	6.72	240	
Campbell Park	Scald (no bioremediation)	23/03/13	CS 1	54H 0340775, 6056769	0-2.5	7.24	139	Scald (no bioremediation) site down to 60 cm.
					2.5-5	6.77	138	Jarosite still around 15-30 cm layer.
					5-10	6.59	127	
					10-15	6.20	154	0-2 cm: wave washed sand.
					15-20	6.02	191	2-10 cm: grey sand.
					20-30	4.31	355	10-30 cm: light grey sand with jarosite.
					30-40	4.14	387	30-60 cm: blue grey clay.
					40-50	5.50	294	
					50-60	6.10	217	
		23/03/13	CS 2	54H 0340786, 6056747	0-2.5	6.96	142	Scald (no bioremediation) site down to 60 cm.
					2.5-5	6.69	135	
					5-10	6.24	136	
					10-15	5.51	236	
					15-20	4.29	370	
					20-30	3.61	423	
					30-40	3.61	432	
					40-50	4.55	362	
					50-60	5.88	237	
		23/03/13	CS 3	54H 0340767, 6056756	0-2.5	7.29	212	Scald (no bioremediation) site down to 50 cm.
					2.5-5	6.62	142	
					5-10	6.33	150	
					10-15	5.80	190	
					15-20	4.10	374	
					20-30	3.76	405	
					30-40	3.63	419	
					40-50	4.50	361	

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-1 (continued). Lower Lakes site and profile descriptions.

Location	Treatment	Date	Profile	GPS Co-ordinates Zone East, North.	Depth (cm)	pH	Eh* (mV)	Location and Profile Remarks
Campbell Park	2010 seeded with Bevy rye and *Puccinellia* bioremediation	23/03/13	CB 1	54H 0340733, 6056758	0-2.5	7.18	147	Bevy rye and *Puccinellia* bioremediation site down to 60 cm.
					2.5-5	6.68	151	
					5-10	6.40	178	
					10-15	6.12	163	
					15-20	4.67	295	
					20-30	3.89	406	
					30-40	3.75	420	
					40-50	4.76	330	
					50-60	5.87	248	
		23/03/13	CB 2	54H 0340730, 6056766	0-2.5	7.22	158	Bevy rye and *Puccinellia* bioremediation site down to 60 cm.
					2.5-5	7.17	170	
					5-10	6.56	152	
					10-15	6.39	159	
					15-20	5.77	186	
					20-30	4.35	327	
					30-40	3.67	417	
					40-50	3.69	418	
					50-60	3.88	371	
		23/03/13	CB 3	54H 0340719, 6056763	0-2.5	7.36	232	Bevy rye and *Puccinellia* bioremediation site down to 60 cm.
					2.5-5	6.87	150	
					5-10	6.40	147	
					10-15	6.07	160	
					15-20	5.58	196	
					20-30	4.34	361	
					30-40	3.76	405	
					40-50	3.96	352	
					50-60	5.73	246	
Tolderol	2010 planted *Juncus* into 2009 plantings of Bevy Rye bioremediation	24/03/13	TJ 1	54H 0331146, 6083497	0-2.5	7.27	204	*Juncus* into plantings of Bevy Rye bioremediation site down to 60 cm.
					2.5-5	7.21	192	
					5-10	7.44	217	0-30 cm: beige sand with lots of iron segregations at 20-50 cm, iron band at 30 cm.
					10-15	7.14	238	30-45 cm: beige sand with jarosite band at 40-45 cm.
					15-20	7.01	203	45-60 cm: grey sand with some iron segregations.
					20-30	6.42	205	60-80 cm: grey sand.
					30-40	6.27	166	
					40-50	5.57	227	
					50-60	4.33	343	

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-1 (continued). Lower Lakes site and profile descriptions.

Location	Treatment	Date	Profile	GPS Co-ordinates Zone East, North.	Depth (cm)	pH	Eh* (mV)	Location and Profile Remarks
Tolderol	2010 planted *Juncus* into 2009 plantings of Bevy Rye bioremediation	24/03/13	TJ 2	54H 0331161, 6083484	0-2.5	6.63	289	*Juncus* into plantings of Bevy Rye bioremediation site down to 60 cm.
					2.5-5	6.76	254	
					5-10	6.64	237	
					10-15	6.28	200	
					15-20	6.07	239	
					20-30	6.01	215	
					30-40	5.74	258	
					40-50	5.01	330	
					50-60	6.06	259	
		24/03/13	TJ 3	54H 0331136, 6083464	0-2.5	7.66	335	*Juncus* into plantings of Bevy Rye bioremediation site down to 60 cm.
					2.5-5	7.56	351	
					5-10	7.23	336	
					10-15	6.90	255	
					15-20	6.74	262	
					20-30	6.25	247	
					30-40	4.31	362	
					40-50	4.83	391	
					50-60	4.15	410	
	Scald (no bioremediation)	24/03/13	TS 1	54H 0331071, 6083415	0-2.5	6.50	356	Scald (no bioremediation) site down to 50 cm.
					2.5-5	6.59	281	
					5-10	6.46	218	Iron-rich crust on some scald surface layers.
					10-15	6.29	220	Jarosite still around 25 cm and lower layers.
					15-20	6.15	194	
					20-30	5.68	268	0-40 cm: beige sand with very occasional jarosite in roots.
					30-40	4.49	374	40-50 cm: beige sand with abundant jarosite in roots.
					40-50	3.86	429	50-60 cm: dark grey sandy clay with abundant jarosite in roots.
								60-80 cm: dark grey sandy clay but no jarosite.
		24/03/13	TS 2	54H 0331054, 6083438	0-2.5	6.08	180	Scald (no bioremediation) site down to 60 cm.
					2.5-5	6.16	178	
					5-10	5.80	237	Iron-rich crust on some scald surface layers.
					10-15	4.16	343	Jarosite still around 25 cm and lower layers.
					15-20	3.60	405	
					20-30	3.35	431	
					30-40	3.51	446	
					40-50	3.51	450	
					50-60	3.58	428	

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-1 (continued). Lower Lakes site and profile descriptions.

Location	Treatment	Date	Profile	GPS Co-ordinates Zone East, North.	Depth (cm)	pH	Eh* (mV)	Location and Profile Remarks
Tolderol	Scald (no bioremediation)	24/03/13	TS 3	54H 0331043, 6083419	0-2.5	6.62	363	Scald (no bioremediation) site down to 60 cm.
					2.5-5	6.58	259	
					5-10	6.40	213	
					10-15	6.16	219	
					15-20	6.41	211	
					20-30	6.11	217	
					30-40	6.35	238	
					40-50	6.54	177	
					50-60	6.68	176	

* Eh measurements are presented versus the standard hydrogen electrode

APPENDIX 2. Characteristics of soil materials

Table 9-2. Characteristics of the Waltowa soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm³)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH_{KCl}	TAA (mol $H^+ t^{-1}$)	ANC (% $CaCO_3$)	TAAlk (mol $OH^- t^{-1}$)	Retained acidity (mol $H^+ t^{-1}$)	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide ($\%S_{AV}$)	Net acidity (mol $H^+ t^{-1}$)	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
WJ 1	0-2.5	1.45	18.40	9.02	116	9.06	0.00	0.09	7.98	0.00	<0.01	0.002	<0.01	-10.39	0.12	0.02	0.10	0.02
WJ 1	2.5-5	1.41	18.92	9.04	119	8.96	0.00	0.14	7.73	0.00	<0.01	0.008	<0.01	-14.27	0.16	0.02	0.14	0.01
WJ 1	5-10	1.49	18.73	8.42	239	6.92	0.00	0.01	3.01	0.00	<0.01	0.005	<0.01	0.98	0.11	0.02	0.10	0.01
WJ 1	10-15	1.46	19.18	7.79	337	6.75	0.00	0.11	0.84	0.00	<0.01	0.002	<0.01	-14.08	0.15	0.02	0.14	0.02
WJ 1	15-20	1.38	20.04	8.51	445	8.25	0.00	0.19	5.65	0.00	<0.01	<0.001	<0.01	-25.85	0.17	0.03	0.11	0.01
WJ 1	20-30	1.45	19.55	7.42	531	6.03	1.48	0.00	0.00	0.00	<0.01	0.001	<0.01	2.28	0.20	0.03	0.16	0.02
WJ 1	30-40	1.39	20.19	5.79	700	6.05	1.48	0.00	0.00	0.00	0.05	0.001	<0.01	32.24	0.20	0.03	0.18	0.02
WJ 2	0-2.5	1.43	18.86	8.93	138.7	9.30	0.00	0.22	9.91	0.00	<0.01	0.003	<0.01	-27.74	0.12	0.02	0.10	0.02
WJ 2	2.5-5	1.47	19.44	9.10	138.2	9.72	0.00	0.67	44.35	0.00	<0.01	0.004	<0.01	-86.79	0.25	0.03	0.13	0.01
WJ 2	5-10	1.36	19.79	8.61	131	7.02	0.00	0.01	2.02	0.00	<0.01	0.004	<0.01	1.33	0.13	0.03	0.12	0.01
WJ 2	10-15	1.27	19.52	8.15	161	6.46	1.48	0.00	0.00	0.00	<0.01	0.002	<0.01	2.50	0.11	0.03	0.10	0.02
WJ 2	15-20	1.40	19.98	8.47	228	7.49	0.00	0.24	3.86	0.00	<0.01	0.001	<0.01	-31.20	0.15	0.03	0.14	0.02
WJ 2	20-30	1.37	20.95	7.99	405	7.18	0.00	0.18	2.05	0.00	0.04	<0.001	<0.01	0.77	0.22	0.03	0.20	0.02
WJ 2	30-40	1.38	19.90	7.47	493	6.35	2.46	0.00	0.00	0.00	<0.01	0.001	<0.01	3.15	0.22	0.04	0.17	0.03
WJ 3	0-2.5	1.41	19.29	9.01	121.1	8.97	0.00	0.08	7.90	0.00	<0.01	0.002	<0.01	-8.93	0.16	0.03	0.14	0.04
WJ 3	2.5-5	1.48	18.52	9.10	173.1	8.80	0.00	0.07	6.68	0.00	<0.01	0.006	<0.01	-5.83	0.15	0.03	0.10	<0.01
WJ 3	5-10	1.36	20.29	8.96	220.9	8.14	0.00	0.07	3.00	0.00	<0.01	0.005	<0.01	-6.23	0.12	0.03	0.08	<0.01
WJ 3	10-15	1.38	20.76	8.43	280	7.17	0.00	0.11	2.02	0.00	<0.01	0.002	<0.01	-13.04	0.12	0.03	0.10	0.02
WJ 3	15-20	1.42	19.67	8.26	368	7.04	0.00	0.09	2.01	0.00	<0.01	0.001	<0.01	-11.55	0.15	0.03	0.14	0.02
WJ 3	20-30	1.42	20.35	8.84	496	8.02	0.00	0.04	3.31	0.00	<0.01	<0.001	<0.01	-5.60	0.13	0.02	0.12	<0.01
WJ 3	30-40	1.36	23.24	8.15	819	7.37	0.00	0.22	2.66	0.00	0.03	<0.001	<0.01	-11.98	0.35	0.05	0.32	0.02
WJ 3	50-60	0.40	60.52	8.20	4800	7.71	0.00	0.98	11.69	0.00	0.70	<0.001	<0.01	304.98	2.77	0.28	2.63	0.61
WC 1	0-2.5	1.43	18.51	9.03	156	8.50	0.00	0.11	6.57	0.00	<0.01	0.006	<0.01	-11.43	0.22	0.02	0.19	0.03
WC 1	2.5-5	1.45	19.96	8.03	167	6.68	0.00	0.10	1.07	0.00	0.02	0.008	<0.01	3.71	0.16	0.02	0.13	0.03
WC 1	5-10	1.45	20.09	7.61	201	6.05	1.48	0.00	0.00	0.00	0.01	0.004	<0.01	11.92	0.16	0.01	0.15	0.04
WC 1	10-15	1.24	23.98	6.38	294	5.72	2.47	0.00	0.00	0.00	<0.01	0.002	<0.01	3.81	0.32	0.03	0.27	0.03
WC 1	15-20	1.32	16.82	6.59	302	5.98	2.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.97	0.22	0.02	0.19	0.01
WC 1	20-30	1.34	22.28	6.21	408	5.66	2.47	0.00	0.00	0.00	<0.01	0.001	<0.01	3.12	0.21	0.02	0.17	<0.01
WC 1	30-40	1.14	28.42	5.83	407	5.54	3.46	0.00	0.00	0.00	0.19	<0.001	<0.01	120.73	0.54	0.05	0.47	0.11

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-2 (continued). Characteristics of the Waltowa soil materials, March 2010.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH$_{KCl}$	TAA (mol H$^+$ t^{-1})	ANC (% CaCO$_3$)	TAAlk (mol OH$^-$ t^{-1})	Retained acidity (mol H$^+$ t^{-1})	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide (%S$_{AV}$)	Net acidity (mol H$^+$ t^{-1})	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
WC 2	0-2.5	1.47	18.42	8.91	126	9.03	0.00	0.14	8.68	0.00	<0.01	0.002	<0.01	-16.75	0.16	0.02	0.15	0.01
WC 2	2.5-5	1.44	19.03	8.85	175	8.22	0.00	0.08	4.89	0.00	<0.01	0.005	<0.01	-7.42	0.14	0.02	0.10	<0.01
WC 2	5-10	1.39	20.91	8.15	188	6.85	0.00	0.05	3.12	0.00	<0.01	<0.001	<0.01	-6.52	0.17	0.02	0.15	0.01
WC 2	10-15	1.35	20.94	7.57	190	6.59	0.00	0.04	0.48	0.00	<0.01	<0.001	<0.01	-5.93	0.15	0.01	0.13	<0.01
WC 2	15-20	1.39	21.96	7.16	239	6.00	1.98	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.98	0.31	0.03	0.27	0.09
WC 2	20-30	1.38	21.34	6.82	330	6.13	1.48	0.00	0.00	0.00	0.02	<0.001	<0.01	13.36	0.21	0.02	0.17	0.02
WC 2	30-40	0.73	45.17	6.10	1094	5.70	6.92	0.00	0.00	0.00	0.50	<0.001	<0.01	316.72	1.29	0.12	1.20	0.01
WC 3	0-2.5	1.39	19.92	9.02	133	9.19	0.00	0.17	11.49	0.00	<0.01	0.003	<0.01	-20.01	0.19	0.02	0.11	0.01
WC 3	2.5-5	1.50	20.21	8.93	192	8.61	0.00	0.09	2.40	0.00	<0.01	0.006	<0.01	-7.86	0.21	0.02	0.16	0.06
WC 3	5-10	1.41	19.88	8.12	199	6.83	0.00	0.11	2.85	0.00	<0.01	0.003	<0.01	-12.61	0.12	0.01	0.10	<0.01
WC 3	10-15	1.45	20.37	7.52	226	6.27	1.48	0.00	0.00	0.00	<0.01	0.004	<0.01	3.68	0.15	0.02	0.12	0.02
WC 3	15-20	1.42	19.98	7.32	263	6.44	1.48	0.00	0.00	0.00	<0.01	0.004	<0.01	4.21	0.13	0.01	0.13	<0.01
WC 3	20-30	n.a.	28.47	6.54	573	5.88	2.47	0.00	0.00	0.00	<0.01	0.002	<0.01	3.46	0.54	0.06	0.52	0.31
WC 3	30-40	0.76	41.77	6.30	1353	5.96	4.45	0.00	0.00	0.00	0.37	<0.001	<0.01	234.56	1.30	0.13	1.27	0.08
WC 3	50-60	0.42	61.65	7.99	2980	8.20	0.00	1.38	30.97	0.00	0.72	<0.001	<0.01	266.10	2.94	0.30	2.69	0.14
WP 1	0-2.5	0.71	43.12	8.39	985	8.93	0.00	1.63	79.23	0.00	0.13	0.022	0.02	-108.72	2.06	0.17	1.71	0.38
WP 1	2.5-5	1.24	24.54	8.95	517	9.19	0.00	0.13	11.28	0.00	0.01	0.002	<0.01	-9.21	0.35	0.04	0.22	<0.01
WP 1	5-10	1.38	20.90	8.90	516	8.33	0.00	0.09	5.15	0.00	<0.01	0.007	<0.01	-7.08	0.19	0.03	0.15	0.01
WP 1	10-15	1.29	21.28	8.90	582	8.37	0.00	0.13	2.84	0.00	<0.01	0.009	<0.01	-12.22	0.18	0.02	0.11	0.01
WP 1	15-20	1.34	21.58	8.57	645	7.91	0.00	0.13	2.59	0.00	<0.01	0.008	<0.01	-12.53	0.20	0.03	0.15	0.01
WP 1	20-30	1.34	22.94	7.50	867	6.27	1.48	0.00	0.00	0.00	0.03	0.002	<0.01	22.03	0.35	0.04	0.31	0.11
WP 1	30-40	1.13	27.76	7.14	1240	6.43	1.98	0.00	0.00	0.00	0.13	0.003	<0.01	85.36	0.62	0.07	0.59	0.06
WP 2	MBO	n.a.	71.01	8.28	1779	8.76	0.00	1.68	65.90	0.00	0.17	0.020	0.08	-59.64	2.37	0.26	2.27	0.80
WP 2	0-2.5	0.59	50.32	8.60	1335	8.97	0.00	23.35	73.21	0.00	0.17	0.035	0.07	-2941	6.14	0.25	3.19	0.01
WP 2	2.5-5	1.39	21.41	9.38	400	9.81	0.00	0.53	35.59	0.00	0.01	0.006	0.01	-51.81	0.26	0.02	0.22	0.14
WP 2	5-10	1.27	22.31	9.21	502	9.11	0.00	0.14	10.38	0.00	<0.01	0.003	<0.01	-16.66	0.17	0.02	0.12	0.03
WP 2	10-15	1.33	20.93	9.08	536	8.47	0.00	0.14	5.28	0.00	<0.01	0.002	<0.01	-17.21	0.15	0.02	0.15	0.06
WP 2	15-20	1.34	21.08	8.72	608	8.00	0.00	0.11	3.35	0.00	<0.01	<0.001	<0.01	-15.21	0.19	0.02	0.14	0.01
WP 2	20-30	1.38	20.10	8.83	649	8.27	0.00	0.11	4.90	0.00	<0.01	<0.001	<0.01	-14.91	0.19	0.02	0.15	0.05
WP 2	30-40	1.24	25.23	7.92	1066	7.06	0.00	0.13	1.90	0.00	0.13	<0.001	<0.01	64.04	0.47	0.05	0.40	0.18

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-2 (continued). Characteristics of the Waltowa soil materials, March 2010.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH$_{KCl}$	TAA (mol H$^+$ t^{-1})	ANC (% CaCO$_3$)	TAAlk (mol OH$^-$ t^{-1})	Retained acidity (mol H$^+$ t^{-1})	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide (%S$_{AV}$)	Net acidity (mol H$^+$ t^{-1})	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
WP 3	0-2.5	0.59	50.21	8.54	741	8.85	0.00	4.41	122.76	0.00	0.12	0.020	0.08	-449.34	3.73	0.25	2.98	1.11
WP 3	2.5-5	1.26	23.75	9.38	317	9.76	0.00	1.42	69.45	0.00	<0.01	0.003	<0.01	-187.85	0.38	0.02	0.22	0.04
WP 3	5-10	1.35	20.53	9.18	350	9.21	0.00	0.12	12.18	0.00	<0.01	0.004	<0.01	-13.50	0.17	0.02	0.14	0.04
WP 3	10-15	1.40	20.34	8.68	405	7.73	0.00	0.12	2.56	0.00	<0.01	0.004	<0.01	-13.76	0.19	0.02	0.17	0.02
WP 3	15-20	1.47	20.09	8.83	190	7.89	0.00	0.09	3.32	0.00	0.01	<0.001	<0.01	-5.43	0.12	0.01	0.12	0.01
WP 3	20-30	1.34	22.11	8.37	617	7.36	0.00	0.09	2.02	0.00	0.03	0.002	<0.01	6.32	0.23	0.02	0.19	<0.01
WP 3	30-40	n.a.	33.04	7.16	1318	6.25	1.98	0.00	0.00	0.00	0.22	<0.001	<0.01	137.99	0.90	0.09	0.80	0.01
WP 3	50-60	0.46	59.43	7.77	4170	7.04	0.00	1.05	2.53	0.00	1.22	<0.001	<0.01	621.28	2.85	0.28	2.66	0.45

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-3. HCl extractable metal/metalloid content of the Waltowa soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)[#]		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
WJ 1	0-2.5	626	85	0.02	0.90	0.75	<0.01	1.77	0.55	8.38	0.61	0.02	19.62
WJ 1	2.5-5	691	124	0.02	0.53	1.13	<0.01	1.62	0.83	7.08	0.60	0.01	1.75
WJ 1	5-10	524	107	0.02	0.26	1.19	<0.01	1.24	0.76	6.74	0.61	0.02	0.95
WJ 1	10-15	807	166	0.01	0.48	1.58	<0.01	0.93	1.33	14.73	1.26	0.03	1.53
WJ 1	15-20	782	182	0.01	0.54	1.50	<0.01	1.22	1.65	15.52	1.01	0.01	2.06
WJ 1	20-30	558	147	0.01	0.78	1.06	<0.01	0.98	1.38	11.65	0.51	0.02	1.15
WJ 1	30-40	624	153	0.01	1.01	1.42	0.01	1.02	1.66	12.20	0.85	0.02	1.33
WJ 2	0-2.5	714	90	0.01	1.07	0.87	<0.01	1.28	0.62	12.25	0.43	0.02	17.12
WJ 2	2.5-5	1186	162	0.01	0.64	1.13	<0.01	5.18	1.03	15.56	0.73	0.02	1.52
WJ 2	5-10	648	142	<0.01	0.47	1.40	<0.01	1.17	0.97	10.14	0.73	0.01	1.25
WJ 2	10-15	715	117	0.01	0.31	1.21	<0.01	3.66	0.82	11.30	0.94	0.03	1.12
WJ 2	15-20	1150	182	<0.01	0.99	2.05	<0.01	0.94	1.26	27.58	1.26	0.02	1.55
WJ 2	20-30	1006	228	<0.01	1.18	1.78	<0.01	0.78	1.98	25.39	1.60	0.02	1.90
WJ 2	30-40	459	158	<0.01	1.27	1.24	<0.01	0.83	1.63	13.74	0.56	0.02	1.21
WJ 3	0-2.5	689	130	<0.01	0.91	0.87	<0.01	1.14	0.64	15.76	0.52	0.02	4.66
WJ 3	2.5-5	628	86	<0.01	0.56	0.75	<0.01	1.12	0.66	6.68	0.43	0.01	1.23
WJ 3	5-10	592	90	<0.01	0.21	0.86	<0.01	1.31	0.69	5.85	0.48	0.02	0.84
WJ 3	10-15	526	98	<0.01	0.30	1.22	<0.01	1.01	0.69	8.40	0.74	0.01	0.97
WJ 3	15-20	812	158	0.08	0.55	1.31	<0.01	0.96	0.90	20.42	0.62	0.03	1.29
WJ 3	20-30	549	122	0.06	0.50	1.21	<0.01	1.02	0.77	11.44	0.69	0.01	1.22
WJ 3	30-40	1029	314	0.04	1.09	1.86	<0.01	0.96	2.50	24.08	1.24	0.01	1.61
WJ 3	50-60	2889	1933	0.03	3.39	7.58	0.02	5.21	13.35	146.89	4.98	0.04	11.29
WC 1	0-2.5	781	157	0.04	0.93	0.95	<0.01	1.88	0.92	12.33	0.71	0.02	6.24
WC 1	2.5-5	526	134	0.03	0.53	1.24	<0.01	1.26	1.05	5.10	0.81	0.02	2.91
WC 1	5-10	570	173	0.02	0.63	1.54	<0.01	1.08	1.37	5.25	0.73	0.01	1.62
WC 1	10-15	660	201	0.02	0.91	1.60	<0.01	1.17	1.87	8.07	0.64	0.02	1.95
WC 1	15-20	944	213	0.01	0.73	1.25	<0.01	4.74	1.46	11.88	0.66	<0.01	1.42
WC 1	20-30	543	169	0.01	0.82	1.56	<0.01	1.21	1.29	8.44	0.46	0.02	1.22

** See Table 9-1 in Appendix 1 for further details on the treatment.*

[#] *The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)*

Table 9-3 (continued). HCl extractable metal/metalloid content of the Waltowa soil materials, March 2010.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)[#]		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
WC 1	30-40	770	322	0.01	1.28	1.85	0.01	1.23	2.85	21.82	1.96	0.03	3.19
WC 2	0-2.5	686	97	0.01	0.95	0.81	<0.01	1.40	0.66	12.05	0.37	0.01	2.92
WC 2	2.5-5	599	118	0.01	0.48	1.05	<0.01	1.34	0.77	6.16	0.66	0.02	2.73
WC 2	5-10	642	203	0.01	0.56	3.18	<0.01	1.17	1.54	6.90	0.68	0.02	2.06
WC 2	10-15	460	95	0.01	0.65	0.73	<0.01	1.40	0.88	5.35	0.30	0.01	1.66
WC 2	15-20	910	188	<0.01	1.25	1.29	<0.01	3.33	1.68	11.08	0.48	0.02	2.61
WC 2	20-30	487	160	0.01	1.02	1.04	<0.01	0.89	1.40	7.98	0.33	0.02	1.62
WC 2	30-40	1480	532	0.01	3.16	3.97	0.01	2.83	7.30	41.69	3.60	0.03	5.05
WC 3	0-2.5	773	116	0.01	1.00	0.89	<0.01	2.26	0.86	13.28	0.77	0.01	7.89
WC 3	2.5-5	729	128	<0.01	0.60	0.94	<0.01	1.58	0.95	7.99	0.71	0.02	4.39
WC 3	5-10	566	180	<0.01	0.44	1.62	<0.01	1.18	0.90	4.91	1.16	0.03	2.87
WC 3	10-15	538	150	0.01	0.71	1.41	<0.01	0.95	1.44	6.23	0.56	0.02	1.82
WC 3	15-20	466	104	<0.01	0.70	0.83	<0.01	1.29	0.98	5.93	0.39	0.02	1.13
WC 3	20-30	767	218	0.06	1.47	1.76	<0.01	1.46	2.61	14.87	0.59	0.03	2.31
WC 3	30-40	1361	567	0.05	2.73	3.36	0.02	4.18	7.25	43.33	3.81	0.02	5.78
WC 3	50-60	2132	1192	0.03	3.65	6.67	0.03	2.33	13.80	145.64	5.30	0.06	7.83
WP 1	0-2.5	2755	770	0.11	1.88	3.87	0.02	16.90	5.73	67.54	3.48	0.02	16.71
WP 1	2.5-5	793	182	0.05	0.54	1.60	<0.01	2.31	1.46	10.66	0.67	0.02	10.47
WP 1	5-10	851	205	0.04	0.73	1.61	<0.01	2.19	1.51	10.01	1.03	0.02	5.18
WP 1	10-15	1040	178	0.03	0.64	1.39	<0.01	4.04	1.32	12.36	0.87	0.03	4.80
WP 1	15-20	856	186	0.02	0.82	1.38	<0.01	1.78	1.47	8.84	0.71	0.02	3.82
WP 1	20-30	743	246	0.02	1.53	1.88	<0.01	1.45	2.31	12.44	0.65	0.02	3.71
WP 1	30-40	1433	503	0.02	1.30	2.36	0.02	6.53	3.20	26.33	1.73	0.02	4.56

** See Table 9-1 in Appendix 1 for further details on the treatment.*

[#] *The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)*

Table 9-3 (continued). HCl extractable metal/metalloid content of the Waltowa soil materials, March 2010.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)#		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
WP 2	MBO	3738	1165	0.05	1.88	4.67	0.03	20.08	8.44	77.47	3.72	0.01	82.91
WP 2	0-2.5	3070	1375	0.03	2.98	5.27	0.03	11.39	7.76	134.45	3.79	0.05	13.54
WP 2	2.5-5	883	144	0.02	0.56	1.17	<0.01	3.26	0.90	8.82	0.64	0.01	2.35
WP 2	5-10	792	167	0.01	0.55	1.63	0.01	1.82	0.97	7.05	0.60	0.02	2.01
WP 2	10-15	709	168	0.01	0.57	1.76	<0.01	1.60	1.04	7.57	0.62	0.02	2.07
WP 2	15-20	879	227	0.01	0.86	1.80	0.02	1.45	1.63	10.60	1.04	0.03	2.22
WP 2	20-30	900	198	0.01	0.79	1.38	<0.01	3.76	1.41	12.39	0.66	0.03	1.45
WP 2	30-40	1254	403	0.01	1.49	2.13	0.01	4.95	2.84	23.19	1.68	0.01	2.62
WP 3	0-2.5	3442	1030	0.02	2.52	5.20	0.03	16.39	8.54	80.11	4.71	0.02	15.96
WP 3	2.5-5	735	163	0.01	0.42	1.41	<0.01	1.84	1.00	14.58	0.69	0.01	2.68
WP 3	5-10	1382	230	0.01	0.51	1.78	<0.01	6.93	1.29	12.10	0.94	0.01	3.23
WP 3	10-15	796	206	0.01	0.69	1.66	0.01	1.75	1.51	8.72	1.16	0.02	4.28
WP 3	15-20	428	128	0.01	0.44	1.10	0.01	1.27	0.95	6.49	0.59	0.02	1.71
WP 3	20-30	616	189	0.01	0.99	1.35	<0.01	1.48	1.64	9.20	0.60	0.02	1.43
WP 3	30-40	887	429	0.07	2.23	2.90	0.03	2.69	4.65	22.34	2.27	0.03	3.18
WP 3	50-60	1835	1968	0.03	5.29	9.17	0.03	3.33	14.29	132.82	5.26	0.05	14.11

** See Table 9-1 in Appendix 1 for further details on the treatment.*

 # The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)

Table 9-4. Characteristics of the Poltalloch soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH_{KCl}	TAA (mol $H^+ t^{-1}$)	ANC (% $CaCO_3$)	TAAlk (mol $OH^- t^{-1}$)	Retained acidity (mol $H^+ t^{-1}$)	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide ($\%S_{AV}$)	Net acidity (mol $H^+ t^{-1}$)	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
P 1	0-2.5	1.39	17.66	8.87	57.7	7.41	0.00	0.08	2.29	0.00	<0.01	<0.001	<0.01	-10.27	0.10	0.02	0.09	0.01
P 1	2.5-5	1.36	18.05	8.51	57.9	6.93	0.00	0.04	2.80	0.00	<0.01	0.001	<0.01	-4.93	0.11	0.02	0.10	0.01
P 1	5-10	1.34	17.99	8.84	124	8.17	0.00	0.14	5.36	0.00	<0.01	0.003	<0.01	-17.17	0.15	0.03	0.14	0.02
P 1	10-15	1.48	17.89	7.85	134	6.30	0.98	0.00	0.00	0.00	<0.01	0.001	<0.01	1.71	0.10	0.03	0.09	<0.01
P 1	15-20	1.44	18.84	7.51	161.5	6.20	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.07	0.02	0.06	0.01
P 1	20-30	1.42	18.33	8.65	382	8.53	0.00	0.12	4.80	0.00	<0.01	<0.001	<0.01	-16.60	0.09	0.02	0.08	0.01
P 1	30-40	1.38	17.98	8.82	568	8.97	0.00	0.14	5.81	0.00	0.06	<0.001	<0.01	21.90	0.10	0.02	0.02	<0.01
P 1	40-50	1.10	18.41	8.35	698	7.04	0.00	0.00	2.77	0.00	0.06	<0.001	<0.01	37.34	0.02	0.04	0.01	<0.01
P 1	50-60	1.04	25.52	8.59	1151	8.98	0.00	0.26	14.16	0.00	0.18	<0.001	<0.01	77.74	0.18	0.02	0.16	0.02
P 2	0-2.5	1.21	18.46	8.33	44.5	6.74	0.00	0.04	1.50	0.00	<0.01	<0.001	<0.01	-5.61	0.10	0.02	0.07	0.01
P 2	2.5-5	1.49	18.02	8.68	73.6	7.51	0.00	0.05	3.54	0.00	<0.01	0.003	<0.01	-5.36	0.15	0.02	0.13	0.01
P 2	5-10	n.a.	19.04	7.39	92	6.66	0.00	0.07	0.84	0.00	<0.01	0.004	<0.01	-6.61	0.12	0.02	0.07	<0.01
P 2	10-15	1.18	19.15	6.40	121	5.71	1.48	0.00	0.00	0.00	<0.01	0.002	<0.01	2.84	0.14	0.02	0.13	0.03
P 2	15-20	1.41	18.59	6.21	164	5.61	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.13	0.02	0.12	0.03
P 2	20-30	1.38	19.43	5.97	239	5.49	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.09	0.02	0.05	0.01
P 2	30-40	1.33	20.09	8.38	466	9.47	0.00	0.48	29.58	0.00	0.11	<0.001	<0.01	8.06	0.20	0.02	0.09	0.01
P 2	40-50	1.16	20.74	8.65	654	9.33	0.00	0.13	15.59	0.00	0.08	<0.001	<0.01	30.36	0.11	0.01	0.05	<0.01
P 2	50-60	1.24	19.48	8.65	760	9.08	0.00	0.08	11.39	0.00	0.08	<0.001	<0.01	40.79	0.11	0.01	0.09	0.01
P 3	0-2.5	n.a.	16.82	8.58	69	7.14	0.00	0.06	2.04	0.00	<0.01	<0.001	<0.01	-8.26	0.10	0.02	0.08	0.03
P 3	2.5-5	1.50	17.19	7.59	35	6.23	1.48	0.00	0.00	0.00	<0.01	0.001	<0.01	2.22	0.12	0.02	0.10	0.01
P 3	5-10	1.56	17.52	9.08	64	9.38	0.00	0.11	12.69	0.00	<0.01	0.002	<0.01	-13.21	0.13	0.02	0.06	0.01
P 3	10-15	1.45	17.80	7.70	33	6.24	0.98	0.00	0.00	0.00	<0.01	0.002	<0.01	2.43	0.07	0.02	0.02	<0.01
P 3	15-20	1.22	18.16	7.04	27	5.95	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.06	0.02	0.04	0.01
P 3	20-30	n.a.	17.97	5.86	49	5.94	1.48	0.00	0.00	0.00	0.01	<0.001	<0.01	10.00	0.07	0.02	0.04	<0.01
P 3	30-40	1.52	18.55	4.94	77	5.05	2.46	0.00	0.00	0.00	0.05	<0.001	<0.01	31.87	0.08	0.02	0.06	<0.01
P 3	40-50	1.38	20.27	8.71	259	5.33	2.06	0.00	0.00	0.00	0.05	<0.001	<0.01	32.72	0.03	<0.01	0.03	0.02
P 3	50-60	1.15	26.65	6.32	1073	9.00	0.00	0.55	17.42	0.00	0.27	<0.001	<0.01	97.22	0.29	0.02	0.21	0.03

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-5. HCl extractable metal/metalloid content of the Poltalloch soil materials, March 2013.

Profile ID* (Site Code, Core)	**Depth Range (cm)**	**Iron (mg/Kg)**	**Aluminium (mg/Kg)**	**Silver (mg/Kg)**	**Arsenic (mg/Kg)**	**Lead (mg/Kg)**	**Cadmium (mg/Kg)**	**Chromium (mg/Kg)**	**Copper (mg/Kg)**	**Manganese (mg/Kg)**	**Nickel (mg/Kg)**	**Selenium (mg/Kg)**	**Zinc (mg/Kg)**
SQG-Low (Trigger value)[#]		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
P 1	0-2.5	769	78	0.03	0.53	0.43	<0.01	3.53	0.38	15.45	0.45	0.01	23.96
P 1	2.5-5	922	108	0.03	0.49	0.43	<0.01	5.53	0.46	10.70	0.64	<0.01	2.63
P 1	5-10	608	137	0.02	0.89	0.67	<0.01	1.11	0.69	8.60	0.60	<0.01	1.16
P 1	10-15	405	82	0.02	0.84	0.49	<0.01	1.44	0.39	5.62	0.33	0.01	0.60
P 1	15-20	126	42	0.02	0.89	0.25	<0.01	0.08	0.18	2.77	0.01	<0.01	0.61
P 1	20-30	509	81	0.03	1.16	0.42	<0.01	2.40	0.40	8.20	0.56	<0.01	0.82
P 1	30-40	328	108	0.01	1.00	0.39	<0.01	1.65	0.50	11.46	0.54	<0.01	0.81
P 1	40-50	1105	185	0.01	0.87	0.62	<0.01	8.94	0.64	23.68	0.94	<0.01	1.42
P 1	50-60	817	457	0.02	1.76	1.24	<0.01	5.24	1.05	33.54	1.54	0.01	1.98
P 2	0-2.5	594	64	0.02	0.30	0.36	<0.01	3.31	0.39	14.63	0.49	<0.01	4.35
P 2	2.5-5	379	91	0.01	0.42	0.48	<0.01	1.12	0.48	5.14	0.76	<0.01	0.79
P 2	5-10	1773	102	0.01	0.53	0.53	<0.01	19.49	0.96	16.29	1.63	<0.01	1.39
P 2	10-15	649	73	0.01	0.95	0.38	<0.01	1.67	0.53	5.90	0.24	<0.01	0.82
P 2	15-20	500	72	0.02	0.98	0.28	<0.01	0.97	0.43	4.88	0.19	<0.01	0.56
P 2	20-30	417	62	0.01	1.15	0.25	<0.01	1.05	0.32	4.34	0.36	<0.01	0.57
P 2	30-40	888	189	0.01	1.37	0.63	<0.01	3.92	2.26	15.65	0.84	<0.01	1.30
P 2	40-50	1097	172	0.01	1.35	0.61	<0.01	7.14	0.89	19.15	1.02	<0.01	0.94
P 2	50-60	812	198	0.01	0.88	0.65	<0.01	5.36	0.35	19.27	0.81	<0.01	1.06
P 3	0-2.5	1855	121	0.01	0.45	0.50	<0.01	22.14	1.06	21.64	1.41	<0.01	1.47
P 3	2.5-5	395	82	0.01	0.35	0.39	<0.01	6.66	0.70	7.22	3.04	<0.01	0.85
P 3	5-10	475	123	0.01	0.44	1.33	<0.01	1.34	0.53	7.28	0.66	0.01	1.00
P 3	10-15	333	51	0.06	0.29	0.34	<0.01	1.18	0.26	3.75	0.13	<0.01	0.39
P 3	15-20	477	49	0.04	0.55	0.31	<0.01	2.09	0.28	4.79	0.21	<0.01	0.63
P 3	20-30	651	80	0.03	0.65	0.30	<0.01	6.80	0.43	7.13	0.71	<0.01	1.02
P 3	30-40	278	73	0.02	0.71	0.29	<0.01	1.23	0.26	4.92	0.32	<0.01	0.50
P 3	40-50	347	130	0.01	0.85	0.35	<0.01	1.31	0.16	8.32	0.93	<0.01	0.88
P 3	50-60	591	477	0.01	2.32	1.36	<0.01	2.21	0.61	30.29	1.09	<0.01	2.25

** See Table 9-1 in Appendix 1 for further details on the treatment.*

[#] The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)

Table 9-6. Characteristics of the Tolderol soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH_{KCl}	TAA (mol H^+ t^{-1})	ANC (% $CaCO_3$)	TAAlk (mol OH^- t^{-1})	Retained acidity (mol H^+ t^{-1})	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide ($\%S_{AV}$)	Net acidity (mol H^+ t^{-1})	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
TJ 1	0-2.5	1.36	19.97	8.55	95	8.28	0.00	0.16	6.34	0.00	<0.01	0.002	<0.01	-20.14	0.26	0.02	0.24	0.11
TJ 1	2.5-5	1.49	19.28	7.18	52	5.83	1.48	0.00	0.00	0.00	0.01	0.001	<0.01	10.38	0.13	0.01	0.11	0.05
TJ 1	5-10	1.35	20.65	7.02	69	5.99	1.48	0.00	0.00	0.00	0.02	<0.001	<0.01	11.48	0.14	0.01	0.13	0.02
TJ 1	10-15	1.34	19.76	6.84	55	5.83	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.06	<0.01	0.06	0.01
TJ 1	15-20	1.36	20.83	7.04	59	5.95	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.06	<0.01	0.04	0.01
TJ 1	20-30	1.28	21.81	6.95	101	5.85	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.07	<0.01	0.04	<0.01
TJ 1	30-40	1.35	21.78	5.64	221	5.24	2.46	0.00	0.00	0.00	0.03	<0.001	<0.01	22.87	0.08	<0.01	0.06	<0.01
TJ 1	40-50	1.32	22.25	4.28	437	4.51	8.24	0.00	0.00	0.00	0.08	<0.001	<0.01	59.02	0.07	0.01	0.07	<0.01
TJ 1	50-60	1.38	21.49	4.51	438	4.61	6.70	0.00	0.00	0.00	0.20	<0.001	<0.01	130.61	0.08	<0.01	0.06	<0.01
TJ 2	0-2.5	1.36	19.94	7.09	46	6.14	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.07	<0.01	0.05	<0.01
TJ 2	2.5-5	1.44	18.91	6.51	62	5.88	1.48	0.00	0.00	0.00	0.02	<0.001	<0.01	14.96	0.07	0.01	0.07	0.03
TJ 2	5-10	1.38	19.17	6.31	89	5.72	1.48	0.00	0.00	0.00	0.02	0.001	<0.01	14.37	0.05	<0.01	0.03	0.01
TJ 2	10-15	1.30	20.48	6.21	138	5.67	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.11	<0.01	0.10	0.03
TJ 2	15-20	1.32	20.65	5.71	188	5.36	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.05	<0.01	0.04	<0.01
TJ 2	20-30	1.34	20.50	5.15	283	4.98	2.95	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.95	0.08	<0.01	0.07	<0.01
TJ 2	30-40	1.36	21.02	4.39	381	4.64	4.43	0.00	0.00	0.00	0.05	<0.001	<0.01	37.32	0.03	<0.01	0.03	0.01
TJ 2	40-50	1.26	25.73	4.22	697	4.34	12.88	0.00	0.00	0.00	0.29	<0.001	<0.01	190.76	0.16	0.01	0.13	<0.01
TJ 2	50-60	1.34	21.43	5.76	487	5.34	2.06	0.00	0.00	0.00	0.04	<0.001	<0.01	28.48	0.03	<0.01	0.02	0.01
TJ 3	0-2.5	1.41	18.65	7.10	43	6.16	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.06	<0.01	0.05	<0.01
TJ 3	2.5-5	1.47	18.66	6.54	44	5.95	1.48	0.00	0.00	0.00	0.01	<0.001	<0.01	8.08	0.09	0.01	0.08	0.03
TJ 3	5-10	1.33	18.83	6.51	50	5.85	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.07	0.01	0.07	0.01
TJ 3	10-15	1.32	19.35	6.30	60	5.65	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.07	<0.01	0.07	0.01
TJ 3	15-20	1.30	20.10	6.19	59	5.64	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.04	<0.01	0.04	0.02
TJ 3	20-30	1.26	21.23	5.18	114	4.86	4.43	0.00	0.00	0.00	<0.01	<0.001	<0.01	4.43	0.08	0.01	0.07	0.01
TJ 3	30-40	1.26	22.95	4.36	247	4.42	8.87	0.00	0.00	0.92	0.04	<0.001	<0.01	37.08	0.10	0.01	0.10	<0.01
TJ 3	40-50	1.36	20.53	4.67	646	4.70	5.15	0.00	0.00	0.00	0.02	<0.001	<0.01	18.79	0.04	<0.01	0.03	0.01
TJ 3	50-60	1.38	21.43	5.31	354	5.10	4.12	0.00	0.00	0.00	0.04	<0.001	<0.01	26.69	0.02	<0.01	0.02	<0.01

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-6 (continued). Characteristics of the Tolderol soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH_{KCl}	TAA (mol $H^+ t^{-1}$)	ANC (% $CaCO_3$)	TAAlk (mol $OH^- t^{-1}$)	Retained acidity (mol $H^+ t^{-1}$)	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide (%S_{AV})	Net acidity (mol $H^+ t^{-1}$)	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
TS 1	0-2.5	1.20	21.37	7.09	45	6.31	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.13	0.05	0.09	0.01
TS 1	2.5-5	n.a.	19.41	6.17	38	5.66	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.16	0.02	0.15	0.01
TS 1	5-10	1.36	19.77	5.23	33	4.85	4.43	0.00	0.00	0.00	<0.01	<0.001	<0.01	4.43	0.12	0.01	0.11	0.02
TS 1	10-15	1.32	21.06	4.98	40	4.68	6.40	0.00	0.00	0.00	<0.01	0.001	<0.01	7.04	0.11	0.01	0.09	0.01
TS 1	15-20	1.30	21.08	4.42	65	4.24	9.35	0.00	0.00	0.00	<0.01	<0.001	<0.01	9.35	0.12	0.01	0.12	0.02
TS 1	20-30	1.26	21.06	4.96	43	4.85	3.94	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.94	0.03	0.01	0.03	<0.01
TS 1	30-40	1.28	20.20	4.53	75	4.75	4.43	0.00	0.00	0.00	<0.01	<0.001	<0.01	4.43	0.03	<0.01	0.03	<0.01
TS 1	40-50	1.29	21.02	8.45	134	4.70	5.15	0.00	0.00	0.00	0.01	<0.001	<0.01	12.18	0.03	<0.01	0.01	<0.01
TS 2	0-2.5	1.23	25.83	5.49	70	5.34	3.94	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.94	0.25	0.02	0.24	0.10
TS 2	2.5-5	1.32	23.86	4.55	85	4.47	7.87	0.00	0.00	0.23	<0.01	<0.001	<0.01	8.11	0.20	0.02	0.19	0.05
TS 2	5-10	1.28	22.13	4.56	100	4.65	9.84	0.00	0.00	0.00	<0.01	<0.001	<0.01	9.84	0.13	0.01	0.13	<0.01
TS 2	10-15	1.30	22.57	3.99	152	4.09	14.76	0.00	0.00	3.47	<0.01	<0.001	<0.01	18.23	0.15	0.01	0.14	<0.01
TS 2	15-20	1.35	21.71	3.97	184	4.21	11.81	0.00	0.00	0.00	<0.01	<0.001	<0.01	11.81	0.12	0.01	0.12	0.01
TS 2	20-30	1.29	22.42	3.96	267	4.15	17.22	0.00	0.00	10.03	0.01	<0.001	<0.01	33.61	0.14	0.01	0.10	0.01
TS 2	30-40	1.29	21.41	4.25	290	4.39	9.84	0.00	0.00	0.00	<0.01	<0.001	<0.01	9.84	0.08	0.01	0.06	0.01
TS 2	40-50	1.28	23.05	4.44	422	4.39	14.94	0.00	0.00	0.00	0.08	<0.001	<0.01	67.59	0.10	<0.01	0.08	0.01
TS 2	50-60	1.23	27.13	4.16	661	4.29	24.21	0.00	0.00	0.00	0.31	<0.001	<0.01	219.95	0.22	0.02	0.19	0.01
TS 3	0-2.5	1.38	20.76	6.75	48	6.28	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.09	0.01	0.08	<0.01
TS 3	2.5-5	1.39	22.27	6.55	49	5.87	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.12	0.04	0.08	0.01
TS 3	5-10	1.29	20.84	6.19	38	5.83	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.08	0.01	0.07	<0.01
TS 3	10-15	1.18	19.73	5.95	57	5.81	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.04	0.01	0.02	0.01
TS 3	15-20	1.26	19.48	5.66	85	5.65	1.48	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.48	0.05	<0.01	0.02	<0.01
TS 3	20-30	1.29	19.70	5.74	110	5.65	1.97	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.97	0.03	<0.01	<0.01	<0.01
TS 3	30-40	1.31	20.75	7.79	383	8.64	0.00	0.08	6.52	0.00	0.01	<0.001	<0.01	-2.25	0.06	<0.01	0.04	0.01
TS 3	40-50	n.a.	28.67	8.26	737	9.14	0.00	2.90	119.05	0.00	0.25	<0.001	<0.01	-228.47	0.53	0.01	0.20	0.03
TS 3	50-60	1.26	24.09	8.51	387	9.47	0.00	0.53	31.98	0.00	0.12	<0.001	<0.01	2.44	0.17	0.01	0.09	<0.01

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-7. HCl extractable metal/metalloid content of the Tolderol soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)#		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
TJ 1	0-2.5	562	208	0.02	0.48	1.03	0.01	1.43	0.97	16.84	0.74	<0.01	3.83
TJ 1	2.5-5	356	131	0.01	0.23	0.72	<0.01	1.35	0.69	7.28	0.40	<0.01	1.01
TJ 1	5-10	320	160	0.01	0.27	0.79	<0.01	1.48	0.90	11.62	0.51	<0.01	0.90
TJ 1	10-15	343	119	0.01	0.23	0.45	<0.01	1.87	0.45	7.34	0.39	<0.01	0.77
TJ 1	15-20	300	122	0.01	0.08	0.38	<0.01	1.59	0.36	5.64	0.37	<0.01	0.85
TJ 1	20-30	338	119	0.01	0.27	0.30	<0.01	2.09	0.40	5.20	0.40	0.01	0.85
TJ 1	30-40	430	122	0.01	0.44	0.54	<0.01	1.41	0.54	5.09	0.61	0.01	1.35
TJ 1	40-50	634	163	<0.01	0.54	0.26	<0.01	1.56	0.36	7.72	0.93	<0.01	2.87
TJ 1	50-60	565	219	<0.01	0.41	0.30	0.01	1.46	0.34	8.46	1.02	<0.01	1.61
TJ 2	0-2.5	471	104	0.01	0.45	0.56	<0.01	2.18	0.41	14.92	0.44	<0.01	7.08
TJ 2	2.5-5	363	114	0.01	0.14	0.56	<0.01	2.21	0.48	5.98	0.34	0.02	0.77
TJ 2	5-10	376	122	<0.01	0.27	0.66	<0.01	2.14	0.49	5.51	0.54	<0.01	0.66
TJ 2	10-15	528	128	0.01	0.22	0.48	<0.01	3.44	0.60	7.35	0.49	<0.01	0.89
TJ 2	15-20	360	97	<0.01	0.30	0.32	<0.01	1.85	0.38	5.17	0.29	<0.01	0.69
TJ 2	20-30	524	109	<0.01	0.41	0.48	<0.01	2.12	0.56	6.47	0.25	<0.01	0.69
TJ 2	30-40	617	123	<0.01	0.41	0.35	0.01	1.78	0.55	8.37	0.49	<0.01	0.71
TJ 2	40-50	812	345	<0.01	0.44	0.90	<0.01	1.60	0.58	23.34	1.27	0.01	2.10
TJ 2	50-60	382	127	<0.01	0.11	0.24	<0.01	1.91	0.22	9.57	0.56	<0.01	0.85
TJ 3	0-2.5	551	105	<0.01	0.31	0.53	<0.01	2.63	0.40	14.16	0.42	<0.01	7.66
TJ 3	2.5-5	367	108	0.01	0.19	0.66	<0.01	2.16	0.40	5.21	0.42	<0.01	1.13
TJ 3	5-10	385	113	0.02	0.29	0.53	<0.01	2.04	0.49	5.88	0.34	<0.01	0.76
TJ 3	10-15	306	99	0.01	0.22	0.31	<0.01	1.66	0.36	4.72	0.26	<0.01	1.44
TJ 3	15-20	347	98	0.01	0.30	0.33	<0.01	1.95	0.41	5.06	0.24	<0.01	0.85
TJ 3	20-30	531	118	0.01	0.34	0.43	<0.01	1.81	0.64	6.33	0.30	0.01	0.80
TJ 3	30-40	939	168	0.01	0.62	0.68	<0.01	1.55	0.66	9.12	0.41	<0.01	0.73
TJ 3	40-50	694	144	0.06	0.41	0.30	<0.01	1.81	0.31	8.24	0.46	<0.01	0.95
TJ 3	50-60	945	198	0.03	0.31	0.36	<0.01	5.94	0.39	13.95	0.85	<0.01	1.03

** See Table 9-1 in Appendix 1 for further details on the treatment.*

 # The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)

Table 9-7 (continued). HCl extractable metal/metalloid content of the Tolderol soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)[#]		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
TS 1	0-2.5	911	125	0.02	0.51	0.46	<0.01	6.34	0.51	14.09	1.10	<0.01	9.07
TS 1	2.5-5	532	109	0.01	0.33	0.45	<0.01	2.28	0.43	5.50	0.38	<0.01	1.81
TS 1	5-10	489	117	0.01	0.56	0.51	<0.01	1.58	0.43	5.12	0.29	<0.01	1.18
TS 1	10-15	776	154	0.02	0.72	0.90	<0.01	2.88	0.60	7.51	0.37	<0.01	0.87
TS 1	15-20	677	162	0.02	0.73	0.77	<0.01	1.42	0.82	6.73	0.33	<0.01	0.86
TS 1	20-30	462	115	0.01	0.32	0.33	<0.01	2.07	0.35	5.06	0.22	<0.01	0.99
TS 1	30-40	588	94	0.01	0.27	0.46	<0.01	1.46	0.24	5.13	0.15	<0.01	0.43
TS 1	40-50	637	124	0.01	0.21	0.53	<0.01	2.33	0.83	8.90	0.62	<0.01	0.54
TS 2	0-2.5	1164	208	0.01	0.91	1.13	<0.01	1.99	1.14	13.33	0.38	<0.01	2.56
TS 2	2.5-5	630	177	0.01	0.85	0.77	<0.01	1.17	0.93	7.47	0.39	0.01	1.63
TS 2	5-10	1265	214	0.01	0.78	0.95	<0.01	5.59	0.83	12.05	0.60	<0.01	1.39
TS 2	10-15	864	224	0.01	0.86	0.82	<0.01	1.46	1.16	8.32	0.35	<0.01	1.28
TS 2	15-20	914	212	0.01	0.61	0.72	<0.01	1.36	0.94	8.14	0.34	<0.01	1.14
TS 2	20-30	1247	236	0.01	0.59	0.77	<0.01	1.22	0.84	9.29	0.26	<0.01	1.27
TS 2	30-40	789	214	<0.01	0.62	0.71	<0.01	1.27	0.55	10.61	0.26	<0.01	1.12
TS 2	40-50	675	291	0.01	0.40	0.84	<0.01	1.92	1.47	17.94	1.05	<0.01	1.58
TS 2	50-60	804	593	0.01	0.87	1.67	0.01	2.09	1.63	32.84	2.49	<0.01	3.54
TS 3	0-2.5	526	117	0.01	0.54	0.46	<0.01	1.25	0.66	9.98	0.24	<0.01	2.19
TS 3	2.5-5	387	132	0.01	0.34	0.43	<0.01	1.30	0.49	4.68	0.21	<0.01	2.06
TS 3	5-10	349	95	0.07	0.25	0.40	<0.01	1.52	0.27	4.03	0.16	<0.01	1.55
TS 3	10-15	512	99	0.06	0.28	0.27	<0.01	3.37	0.30	6.01	0.20	<0.01	1.43
TS 3	15-20	492	98	0.03	0.29	0.39	<0.01	2.20	0.28	6.38	0.16	<0.01	2.20
TS 3	20-30	404	99	0.03	0.18	0.20	<0.01	1.69	0.29	6.11	0.20	<0.01	1.09
TS 3	30-40	568	152	0.02	0.53	0.48	<0.01	1.68	0.47	10.16	0.55	<0.01	1.50
TS 3	40-50	1108	598	0.01	0.80	1.07	<0.01	4.30	1.04	44.59	3.03	<0.01	4.00
TS 3	50-60	785	339	<0.01	0.20	0.53	<0.01	4.11	0.64	26.11	1.39	<0.01	2.15

** See Table 9-1 in Appendix 1 for further details on the treatment.*

[#] The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)

Table 9-8. Characteristics of the Campbell Park soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH_{KCl}	TAA (mol $H^+ t^{-1}$)	ANC (% $CaCO_3$)	TAAlk (mol $OH^- t^{-1}$)	Retained acidity (mol $H^+ t^{-1}$)	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide ($\%S_{AV}$)	Net acidity (mol $H^+ t^{-1}$)	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
CS 1	0-2.5	1.39	19.94	8.85	215	7.84	0.00	0.13	4.02	0.00	<0.01	<0.001	<0.01	-17.88	0.11	0.01	0.11	0.05
CS 1	2.5-5	1.25	23.11	6.95	273	5.78	2.46	0.00	0.00	0.00	<0.01	0.002	<0.01	3.48	0.25	0.02	0.24	0.04
CS 1	5-10	1.29	15.92	5.21	321	4.88	5.42	0.00	0.00	0.00	<0.01	<0.001	<0.01	5.42	0.31	0.03	0.28	0.12
CS 1	10-15	1.12	32.13	4.55	446	4.44	11.33	0.00	0.00	0.00	<0.01	<0.001	<0.01	11.33	0.35	0.03	0.33	0.07
CS 1	15-20	1.22	21.97	4.17	458	4.28	9.85	0.00	0.00	1.63	0.03	<0.001	<0.01	29.66	0.23	0.02	0.23	0.09
CS 1	20-30	1.04	31.12	3.78	868	3.96	26.61	0.00	0.00	0.00	0.17	<0.001	<0.01	130.11	0.43	0.04	0.43	0.16
CS 1	30-40	0.72	41.88	3.85	1545	4.10	33.01	0.00	0.00	0.00	0.73	<0.001	<0.01	488.96	0.64	0.05	0.59	0.16
CS 1	40-50	0.63	46.76	5.71	1809	5.20	6.18	0.00	0.00	0.00	0.91	<0.001	<0.01	575.48	0.81	0.07	0.74	0.05
CS 1	50-60	0.53	51.57	6.76	2114	6.24	1.55	0.00	0.00	0.00	0.86	<0.001	<0.01	535.42	1.27	0.10	1.22	0.36
CS 2	0-2.5	1.19	20.09	8.75	216	7.94	0.00	0.08	3.35	0.00	<0.01	<0.001	<0.01	-10.88	0.07	0.01	0.06	<0.01
CS 2	2.5-5	1.16	19.40	7.77	197	6.46	1.47	0.00	0.00	0.00	<0.01	0.002	<0.01	2.94	0.10	0.06	0.11	0.02
CS 2	5-10	1.46	19.56	5.29	243	4.93	4.43	0.00	0.00	0.00	<0.01	<0.001	<0.01	4.43	0.20	0.11	0.20	0.03
CS 2	10-15	1.25	21.34	4.41	354	4.39	8.87	0.00	0.00	0.00	<0.01	<0.001	<0.01	8.87	0.21	0.11	0.19	0.01
CS 2	15-20	1.19	24.68	4.11	471	4.17	13.30	0.00	0.00	0.00	<0.01	<0.001	<0.01	13.30	0.33	0.10	0.28	0.09
CS 2	20-30	1.12	26.51	3.92	585	4.11	17.74	0.00	0.00	2.29	0.04	<0.001	<0.01	47.69	0.36	0.09	0.28	0.03
CS 2	30-40	0.77	42.73	3.77	1207	3.99	44.35	0.00	0.00	10.16	0.33	<0.001	<0.01	257.85	0.67	0.12	0.61	0.01
CS 2	40-50	0.49	56.25	4.10	2140	3.97	50.99	0.00	0.00	0.00	1.33	<0.001	<0.01	881.59	1.38	0.11	1.25	0.10
CS 2	50-60	0.54	53.17	4.73	1965	4.39	20.60	0.00	0.00	0.00	0.97	<0.001	<0.01	626.68	1.40	0.11	1.27	0.23
CS 3	0-2.5	n.a.	21.27	8.61	251	8.59	0.00	0.17	7.73	0.00	<0.01	<0.001	<0.01	-22.07	0.12	0.06	0.11	0.05
CS 3	2.5-5	1.15	20.98	8.29	223	6.89	0.00	0.11	1.65	0.00	<0.01	0.001	<0.01	-13.51	0.17	0.05	0.16	0.04
CS 3	5-10	1.36	21.54	5.35	261	4.80	5.42	0.00	0.00	0.00	<0.01	<0.001	<0.01	5.42	0.24	0.05	0.24	0.09
CS 3	10-15	1.22	22.00	4.37	385	4.22	11.33	0.00	0.00	0.00	<0.01	<0.001	<0.01	11.33	0.29	0.05	0.26	<0.01
CS 3	15-20	1.23	25.14	3.84	531	4.03	17.74	0.00	0.00	7.71	<0.01	<0.001	<0.01	25.45	0.39	0.04	0.32	0.16
CS 3	20-30	1.14	25.84	3.86	653	4.08	16.75	0.00	0.00	4.18	<0.01	<0.001	<0.01	20.93	0.38	0.04	0.32	0.14
CS 3	30-40	0.83	38.08	3.94	1210	4.10	22.17	0.00	0.00	0.00	0.34	<0.001	<0.01	232.51	0.55	0.05	0.51	0.05
CS 3	40-50	0.61	43.92	4.96	1468	4.61	15.97	0.00	0.00	0.00	0.53	<0.001	<0.01	344.48	0.69	0.06	0.64	0.18

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-8 (continued). Characteristics of the Campbell Park soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Bulk Density (g/cm^3)	Moisture Content (%)	pH 1:5 soil: water	EC 1:5 soil:water (µS/cm)	pH_{KCl}	TAA (mol H^+ t^{-1})	ANC (% $CaCO_3$)	TAAlk (mol OH^- t^{-1})	Retained acidity (mol H^+ t^{-1})	Pyritic Sulfur (%S)	Elemental Sulfur (%S)	Acid Volatile Sulfide ($\%S_{AV}$)	Net acidity (mol H^+ t^{-1})	Total C (%C)	Total N (%N)	Total Organic C (%C)	Hydrolysable C (%C)
CB 1	0-2.5	1.28	19.99	8.45	198	7.44	0.00	0.02	3.65	0.00	<0.01	<0.001	<0.01	-3.02	0.06	<0.01	0.06	<0.01
CB 1	2.5-5	1.33	19.75	7.37	206	6.68	0.00	0.01	0.41	0.00	<0.01	0.001	<0.01	-0.34	0.13	0.01	0.13	0.04
CB 1	5-10	1.31	20.12	5.97	255	5.40	2.46	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.46	0.17	0.01	0.16	0.06
CB 1	10-15	1.22	22.79	4.54	423	4.49	8.38	0.00	0.00	0.00	<0.01	<0.001	<0.01	8.38	0.34	0.03	0.31	0.07
CB 1	15-20	1.33	21.41	4.21	446	4.32	8.87	0.00	0.00	6.58	<0.01	<0.001	<0.01	15.45	0.25	0.02	0.24	0.11
CB 1	20-30	1.20	23.95	3.92	618	4.25	15.77	0.00	0.00	2.25	0.17	<0.001	<0.01	123.73	0.21	0.01	0.20	0.09
CB 1	30-40	1.01	31.31	3.91	963	4.28	19.71	0.00	0.00	0.00	0.33	<0.001	<0.01	222.66	0.35	0.03	0.32	0.12
CB 1	40-50	n.a.	60.24	5.95	2329	5.31	9.79	0.00	0.00	0.00	1.45	<0.001	<0.01	915.58	1.29	0.11	1.17	0.23
CB 1	50-60	n.a.	60.03	6.33	2348	5.66	7.73	0.00	0.00	0.00	1.35	<0.001	<0.01	848.52	1.44	0.12	1.32	0.02
CB 2	0-2.5	1.47	16.84	8.51	177	7.33	0.00	0.06	2.47	0.00	<0.01	0.001	<0.01	-6.76	0.10	<0.01	0.09	0.04
CB 2	2.5-5	1.49	18.47	7.99	211	6.88	0.00	0.08	2.06	0.00	<0.01	0.006	0.02	5.57	0.13	<0.01	0.13	0.05
CB 2	5-10	1.38	20.01	6.61	261	5.82	1.97	0.00	0.00	0.00	<0.01	0.006	<0.01	5.73	0.18	0.01	0.18	0.08
CB 2	10-15	1.33	20.32	5.21	351	4.97	3.94	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.94	0.27	0.02	0.24	0.08
CB 2	15-20	1.43	19.84	4.50	433	4.49	5.91	0.00	0.00	0.00	<0.01	<0.001	<0.01	5.91	0.25	0.01	0.19	0.04
CB 2	20-30	1.24	22.72	4.07	646	4.21	11.33	0.00	0.00	3.69	<0.01	<0.001	<0.01	15.02	0.29	0.02	0.25	0.01
CB 2	30-40	0.98	28.68	4.06	921	4.23	20.20	0.00	0.00	0.00	0.08	<0.001	<0.01	71.78	0.37	0.02	0.34	0.05
CB 2	40-50	0.95	36.67	4.49	1336	4.41	30.90	0.00	0.00	0.00	0.43	<0.001	<0.01	299.23	0.52	0.04	0.45	0.20
CB 2	50-60	n.a.	46.38	5.31	1795	4.78	17.51	0.00	0.00	0.00	0.87	<0.001	<0.01	561.60	0.78	0.06	0.72	0.31
CB 3	0-2.5	1.47	17.53	8.38	168	7.03	0.00	0.00	1.72	0.00	<0.01	0.001	<0.01	0.86	0.06	<0.01	0.05	0.01
CB 3	2.5-5	1.62	16.98	8.28	180	6.71	0.00	0.04	2.32	0.00	<0.01	0.004	<0.01	-3.14	0.09	<0.01	0.08	0.04
CB 3	5-10	1.39	20.74	6.97	258	6.01	1.48	0.00	0.00	0.00	<0.01	0.006	0.01	13.24	0.22	0.01	0.19	0.08
CB 3	10-15	1.40	19.91	5.71	312	5.21	2.96	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.96	0.19	0.01	0.18	0.02
CB 3	15-20	1.37	19.79	4.84	397	4.73	3.94	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.94	0.14	<0.01	0.14	0.02
CB 3	20-30	1.25	24.20	4.23	655	4.27	10.84	0.00	0.00	0.00	<0.01	<0.001	<0.01	10.84	0.28	0.02	0.25	<0.01
CB 3	30-40	1.25	24.94	4.03	754	4.26	12.81	0.00	0.00	0.00	0.12	<0.001	<0.01	86.68	0.25	0.01	0.21	0.05
CB 3	40-50	0.85	37.79	4.61	1300	4.38	16.48	0.00	0.00	0.00	0.53	<0.001	<0.01	348.26	0.56	0.05	0.49	0.15
CB 3	50-60	0.79	39.93	6.40	1345	5.66	5.15	0.00	0.00	0.00	0.67	<0.001	<0.01	421.56	0.56	0.05	0.52	0.22

** See Table 9-1 in Appendix 1 for further details on the treatment.*

Table 9-9. HCl extractable metal/metalloid content of the Campbell Park soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)[#]		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
CS 1	0-2.5	691	75	0.01	0.69	0.67	<0.01	1.64	0.49	8.27	0.36	<0.01	2.68
CS 1	2.5-5	809	147	0.01	0.63	1.26	<0.01	1.76	1.07	6.89	0.33	<0.01	1.05
CS 1	5-10	754	152	0.01	0.45	1.10	<0.01	1.55	1.27	6.90	0.34	<0.01	1.21
CS 1	10-15	1113	228	0.01	0.67	1.24	<0.01	4.68	1.56	11.35	0.54	<0.01	1.22
CS 1	15-20	767	155	<0.01	0.36	0.74	<0.01	1.42	1.13	7.34	0.42	<0.01	0.85
CS 1	20-30	1073	390	0.01	0.58	1.47	<0.01	1.34	2.79	15.79	0.82	<0.01	1.85
CS 1	30-40	1282	786	0.02	2.37	3.53	0.01	2.02	4.28	39.85	3.11	<0.01	4.71
CS 1	40-50	1044	1046	0.04	2.20	3.91	0.01	2.30	5.72	51.55	2.92	0.01	4.91
CS 1	50-60	915	1398	0.02	3.07	4.30	0.01	2.97	6.07	72.47	3.41	0.02	5.72
CS 2	0-2.5	794	83	<0.01	0.56	0.62	<0.01	1.77	0.57	10.49	0.40	<0.01	2.36
CS 2	2.5-5	725	98	<0.01	0.49	0.57	<0.01	1.73	1.03	6.17	0.49	<0.01	1.32
CS 2	5-10	531	143	0.01	0.47	0.69	<0.01	1.04	1.83	5.76	0.58	<0.01	1.57
CS 2	10-15	707	165	0.01	0.56	0.65	<0.01	1.56	1.55	8.53	0.40	<0.01	1.99
CS 2	15-20	996	218	0.01	0.47	0.78	<0.01	1.23	1.68	11.31	0.29	<0.01	1.03
CS 2	20-30	962	277	0.01	0.56	0.80	<0.01	1.38	1.95	14.06	0.44	<0.01	1.59
CS 2	30-40	1694	766	0.01	1.31	2.03	<0.01	3.86	4.33	44.14	1.37	<0.01	3.31
CS 2	40-50	1798	1512	0.03	2.84	6.15	0.03	2.18	8.85	81.41	5.60	0.01	8.89
CS 2	50-60	1621	1380	0.03	3.44	4.70	0.01	2.73	7.49	92.31	5.67	0.02	8.91
CS 3	0-2.5	1801	165	0.07	0.88	0.85	<0.01	13.94	0.87	21.14	1.41	<0.01	2.21
CS 3	2.5-5	901	135	0.03	0.75	0.75	<0.01	2.50	0.84	9.43	0.72	0.01	1.93
CS 3	5-10	667	171	0.02	0.63	0.79	<0.01	1.19	1.34	6.89	0.57	<0.01	1.48
CS 3	10-15	954	182	0.02	0.62	0.87	<0.01	1.24	1.24	8.81	0.41	<0.01	0.98
CS 3	15-20	1324	252	0.02	0.78	0.92	<0.01	1.95	1.51	13.32	0.42	0.01	1.22
CS 3	20-30	920	232	0.02	0.73	0.68	<0.01	1.24	1.27	13.40	0.45	<0.01	1.39
CS 3	30-40	1028	613	0.02	1.30	2.03	<0.01	1.48	4.72	32.79	1.82	0.01	3.21
CS 3	40-50	1281	963	0.02	1.57	3.29	0.01	6.51	5.08	48.81	3.35	0.01	5.64

** See Table 9-1 in Appendix 1 for further details on the treatment.*

[#] *The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)*

Table 9-9 (continued). HCl extractable metal/metalloid content of the Campbell Park soil materials, March 2013.

Profile ID* (Site Code, Core)	Depth Range (cm)	Iron (mg/Kg)	Aluminium (mg/Kg)	Silver (mg/Kg)	Arsenic (mg/Kg)	Lead (mg/Kg)	Cadmium (mg/Kg)	Chromium (mg/Kg)	Copper (mg/Kg)	Manganese (mg/Kg)	Nickel (mg/Kg)	Selenium (mg/Kg)	Zinc (mg/Kg)
SQG-Low (Trigger value)[#]		n.a.	n.a.	1	20	50	1.5	80	65	n.a.	21	n.a.	200
CB 1	0-2.5	722	72	0.01	0.67	0.58	<0.01	2.05	0.37	9.37	0.43	<0.01	1.67
CB 1	2.5-5	531	106	0.01	0.41	0.60	<0.01	1.58	0.68	5.54	0.56	<0.01	1.23
CB 1	5-10	430	91	0.01	0.30	0.36	<0.01	1.70	0.62	5.37	0.43	<0.01	1.09
CB 1	10-15	793	177	0.01	0.56	0.90	<0.01	1.10	1.31	7.90	0.31	0.01	1.12
CB 1	15-20	910	174	0.01	0.71	0.71	<0.01	1.21	1.02	9.17	0.29	<0.01	0.79
CB 1	20-30	847	257	0.01	0.67	0.77	<0.01	2.18	1.70	13.58	0.80	<0.01	1.06
CB 1	30-40	679	415	<0.01	0.97	1.65	<0.01	1.22	1.97	21.21	1.53	<0.01	1.75
CB 1	40-50	1495	1893	0.02	3.40	6.57	0.02	5.32	9.18	88.00	6.34	0.02	11.30
CB 1	50-60	1732	2083	0.03	3.31	6.55	0.02	9.78	9.05	96.04	6.24	0.03	10.09
CB 2	0-2.5	612	83	<0.01	0.60	0.56	<0.01	1.34	0.53	8.93	0.39	<0.01	2.54
CB 2	2.5-5	793	124	<0.01	0.52	0.67	<0.01	1.17	0.74	5.43	0.38	<0.01	1.03
CB 2	5-10	709	117	<0.01	0.51	0.83	<0.01	2.13	0.90	6.20	0.68	<0.01	1.24
CB 2	10-15	526	141	<0.01	0.43	0.68	<0.01	1.45	1.06	7.58	0.43	<0.01	1.02
CB 2	15-20	493	122	<0.01	0.39	0.70	<0.01	1.02	0.94	7.21	0.27	<0.01	0.97
CB 2	20-30	768	191	<0.01	0.52	0.70	<0.01	0.91	1.09	9.60	0.20	<0.01	0.83
CB 2	30-40	1181	422	0.06	0.62	1.24	<0.01	4.03	1.86	21.30	0.92	<0.01	1.56
CB 2	40-50	1270	773	0.01	1.26	2.27	0.02	4.38	3.43	34.49	2.57	<0.01	3.19
CB 2	50-60	1451	1218	0.02	2.48	3.98	0.01	7.28	6.06	61.57	4.15	0.01	6.89
CB 3	0-2.5	636	83	0.04	0.53	0.53	<0.01	1.74	0.48	7.77	0.40	<0.01	3.26
CB 3	2.5-5	546	92	0.02	0.50	0.53	<0.01	1.03	0.84	4.20	0.27	<0.01	1.23
CB 3	5-10	624	127	0.02	0.52	0.75	<0.01	1.34	1.14	5.96	0.43	<0.01	1.64
CB 3	10-15	495	128	0.02	0.46	0.63	<0.01	1.63	1.06	7.09	0.42	<0.01	1.52
CB 3	15-20	456	104	0.02	0.31	0.55	<0.01	1.16	0.68	7.27	0.24	<0.01	1.17
CB 3	20-30	701	229	0.02	0.40	0.93	<0.01	1.25	1.21	11.92	0.30	<0.01	1.07
CB 3	30-40	628	284	0.01	0.74	0.92	<0.01	1.46	1.92	15.36	0.95	<0.01	1.44
CB 3	40-50	738	614	0.01	1.46	2.35	0.01	1.65	2.84	31.70	2.71	<0.01	3.92
CB 3	50-60	1144	922	0.01	1.46	3.02	0.01	5.98	4.06	51.60	2.61	0.02	5.76

** See Table 9-1 in Appendix 1 for further details on the treatment.*

[#] *The ANZECC sediment quality guidelines (SQG) are for total metal concentrations (ANZECC/ARMCANZ 2000)*

APPENDIX 3. Surface water and pore-water characteristics

Table 9-10. Surface water field measurements for all sites (March 2013).

Site ID	Date	Temperature (^{o}C)	pH	Redox Potential* (mV)	Electrical Conductivity (µS/cm)	Dissolved Oxygen (% Sat.)
WC1-1	22-Apr-13	17.5	8.58	360	1533	80.0
WC1-2	22-Apr-13	17.2	8.72	349	1370	66.0
WP1-1	22-Apr-13	17.2	8.42	405	1267	67.0
WP1-2	22-Apr-13	17.3	8.38	392	1182	63.0
WJ1-1	22-Apr-13	17.6	8.67	385	1261	67.5
WJ1-2	22-Apr-13	17.7	8.62	386	1302	68.7
P1-1	23-Apr-13	16.2	8.22	368	572	67.3
P1-2	23-Apr-13	16.1	8.27	362	541	69.0
CP1-1	23-Apr-13	19.6	8.51	365	2740	83.1
CP1-2	23-Apr-13	18.9	8.62	351	3800	80.3
T-1	24-Apr-13	19.5	8.55	323	584	73.5
T-2	24-Apr-13	19.1	8.61	318	564	73.0
T-3	24-Apr-13	20.5	8.42	320	564	71.0

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-11. Surface water properties for all sites (March 2013).

Site ID	Alkalinity (mmol/L)	Ferrous Iron (mg/L)	Total Iron (mg/L)	Dissolved Sulfide (µg/L)	Total Organic C (mg/L)	Dissolved Organic C (mg/L)	SO_4 (mg/L)	Cl (mg/L)	Cl:SO_4 ratio
WC1-1	42.1	<0.1	<0.1	<10	16.2	11.0	62	310	5.0
WC1-2	39.8	<0.1	<0.1	<10	-	-	61	306	5.0
WP1-1	37.9	<0.1	<0.1	<10	17.7	9.0	50	246	4.9
WP1-2	40.0	<0.1	<0.1	<10	-	-	57	265	4.6
WJ1-1	41.1	<0.1	<0.1	<10	17.9	10.3	60	274	4.6
WJ1-2	40.2	<0.1	<0.1	<10	-	-	63	300	4.8
P1-1	27.3	<0.1	<0.1	<10	16.9	8.9	32	103	3.3
P1-2	26.9	<0.1	<0.1	<10	-	-	19	71	3.7
CP1-1	83.8	<0.1	<0.1	<10	34.4	15.9	180	1002	5.6
CP1-2	83.3	<0.1	<0.1	<10	-	-	113	643	5.7
T-1	27.1	<0.1	<0.1	<10	12.5	8.3	19	69	3.6
T-2	28.7	<0.1	<0.1	<10	-	-	23	82	3.5

Table 9-12. Surface water soluble cation and nutrient analyses for all sites (March 2013).

Site ID	Soluble cations				Nutrients			
	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
WC1-1	20.28	28.89	184.99	12.59	0.11	<0.01	0.02	0.006
WC1-2	19.94	28.85	179.67	12.38	0.11	<0.01	0.01	0.005
WP1-1	22.60	23.92	150.53	11.38	0.11	<0.01	0.03	0.013
WP1-2	20.30	26.22	163.63	11.95	0.10	<0.01	0.02	0.007
WJ1-1	20.83	26.50	165.40	12.31	0.10	<0.01	0.02	0.004
WJ1-2	20.71	28.29	178.09	12.58	0.11	<0.01	0.02	0.004
P1-1	15.20	12.87	68.78	8.47	0.10	<0.01	0.01	0.004
P1-2	15.28	8.62	48.99	5.28	0.10	<0.01	0.01	0.003
CP1-1	45.90	77.03	555.49	28.18	0.11	<0.01	0.04	0.006
CP1-2	28.41	50.03	350.90	19.65	0.11	<0.01	0.10	0.005
T-1	14.55	8.77	47.35	5.61	0.11	<0.01	0.05	0.002
T-2	17.45	9.95	53.50	6.09	0.11	<0.01	0.04	0.003

Table 9-13. Summary of surface water hydrochemical characteristics (mean values of duplicate/triplicated samples presented).

Parameter	Units	Waltowa			Poltalloch	Campbell Park	Tolderol
		Phragmites	*Cotula*	*Juncus*			
Temperature	°C	17.3	17.4	17.7	16.2	19.3	19.7
pH		8.40	8.65	8.65	8.25	8.57	8.53
Redox Potential (Eh)	mV	398	354	385	365	358	320
Electrical Conductivity (EC)	µS/cm	1225	1452	1282	557	3270	571
Dissolved Oxygen (DO)	% Sat.	65.0	73.0	68.1	68.2	81.7	72.5
Alkalinity	mmol/L	38.9	40.9	40.6	27.1	83.6	27.9
Ferrous Iron (Fe^{2+})	mg/L	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Ferric Iron(Fe^{3+})	mg/L	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Dissolved Sulfide (S^{2-})	µg/L	<10	<10	<10	<10	<10	<10
Total organic carbon (TOC)*	mg/L	17.7	16.2	17.9	16.9	34.4	12.5
Dissolved organic carbon (DOC)*	mg/L	9.0	11.0	10.3	8.9	15.9	8.3
Nitrate (NO_3^-)	mg/L N	0.10	0.11	0.11	0.10	0.11	0.11
Nitrite (NO_2^-)	mg/L N	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Ammonia (NH_3)	mg/L N	0.02	0.01	0.02	0.01	0.07	0.04
Orthophosphate (PO_4^{3-})	mg/L P	0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Chloride (Cl^-)	mg/L	256	308	287	87	822	75
Sulfate (SO_4^{2-})	mg/L	54	61	61	25	147	21
Sodium (Na^+)	mg/L	157	182	172	58.9	453	50.4
Potassium (K^+)	mg/L	11.7	12.5	12.4	6.9	23.9	5.9
Calcium (Ca^{2+})	mg/L	21.5	20.1	20.8	15.2	37.2	16.0
Magnesium (Mg^{2+})	mg/L	25.1	28.9	27.4	10.7	63.5	9.4

* TOC and DOC concentration determined on a single sample from each site

Table 9-14. Pore-water properties for Waltowa in March 2013.

Treatment	Layer	Depth (cm)	pH	Alkalinity (mmol/L)	Eh* (mV)	EC (µS/cm)	SO_4 (mg/L)	Cl (mg/L)	$Cl:SO_4$ ratio	Sulfide (µg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)
Established *Juncus*	1-1	0 - 2.5	8.16	57.9	250	2710	i.s.	i.s.	i.s.	78	0.18	<0.01
	1-2	2.5 - 5	8.22	111.7	246	4430	i.s.	i.s.	i.s.	109	0.05	<0.01
	1-3	5 - 10	7.99	49.7	313	3880	i.s.	i.s.	i.s.	i.s.	0.18	<0.01
	1-4	10 - 15	7.21	i.s.	262	5900	i.s.	i.s.	i.s.	647	0.45	3.32
	1-5	15 - 20	6.84	25.2	194	8350	i.s.	i.s.	i.s.	145	0.50	0.57
	1-6	20 - 30	6.79	35.6	152	11150	i.s.	i.s.	i.s.	173	14.23	14.62
	1-7	30 - 40	6.96	31.7	175	i.s.	i.s.	i.s.	i.s.	129	3.04	2.75
	2-1	0 - 2.5	8.13	71.3	270	3860	i.s.	i.s.	i.s.	69	0.24	<0.01
	2-2	2.5 - 5	8.28	i.s.	283	3700	i.s.	i.s.	i.s.	155	0.18	<0.01
	2-3	5 - 10	8.03	67.2	281	3510	i.s.	i.s.	i.s.	104	0.36	1.71
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	7.32	95.3	303	10210	i.s.	i.s.	i.s.	35	0.18	<0.01
	2-6	20 - 30	7.88	103.9	301	12620	i.s.	i.s.	i.s.	69	0.24	1.20
	2-7	30 - 40	8.14	163.9	298	16140	i.s.	i.s.	i.s.	285	0.09	<0.01
	3-1	0 - 2.5	8.11	63.0	268	2420	i.s.	i.s.	i.s.	36	0.20	0.38
	3-2	2.5 - 5	8.40	82.5	261	3560	i.s.	i.s.	i.s.	41	0.09	0.61
	3-3	5 - 10	8.23	63.4	281	3220	i.s.	i.s.	i.s.	181	0.09	1.80
	3-4	10 - 15	8.17	118.5	286	6190	i.s.	i.s.	i.s.	337	0.09	0.32
	3-5	15 - 20	7.92	109.0	298	8470	i.s.	i.s.	i.s.	155	1.07	2.85
	3-6	20 - 30	7.93	224.6	302	11880	i.s.	i.s.	i.s.	13	0.18	0.24
	3-7	30 - 40	7.63	383.8	299	16640	i.s.	i.s.	i.s.	119	0.97	1.36
Established *Cotula*	1-1	0 - 2.5	7.72	53.2	158	2625	96	720	7.5	36	0.18	0.82
	1-2	2.5 - 5	7.82	77.7	174	3660	86	1060	12.4	36	0.04	0.15
	1-3	5 - 10	7.15	81.8	143	4420	177	1196	6.8	21	1.50	1.56
	1-4	10 - 15	6.75	64.5	135	5800	495	1622	3.3	57	34.92	36.88
	1-5	15 - 20	6.64	33.6	157	6840	744	1806	2.4	52	39.27	43.11
	1-6	20 - 30	6.55	51.1	155	8950	1514	2471	1.6	62	119.92	112.25
	1-7	30 - 40	6.61	105.2	140	11700	2240	3289	1.5	145	119.92	119.28
	2-1	0 - 2.5	7.94	61.9	352	2080	i.s.	i.s.	i.s.	38	0.13	0.61
	2-2	2.5 - 5	8.22	77.5	345	3200	i.s.	i.s.	i.s.	342	0.21	0.11
	2-3	5 - 10	8.23	49.6	337	i.s.	i.s.	i.s.	i.s.	72	0.11	0.06
	2-4	10 - 15	7.80	94.3	341	4040	i.s.	i.s.	i.s.	129	0.10	0.13
	2-5	15 - 20	7.39	74.6	341	4430	i.s.	i.s.	i.s.	223	0.08	<0.01
	2-6	20 - 30	7.20	59.7	340	5840	i.s.	i.s.	i.s.	316	0.45	0.09
	2-7	30 - 40	6.79	146.0	157	9050	716	2881	4.0	79	21.03	23.17

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-14 (continued). Pore-water properties for Waltowa in March 2013.

Treatment	**Layer**	**Depth (cm)**	**pH**	**Alkalinity (mmol/L)**	**Eh* (mV)**	**EC (µS/cm)**	**SO_4 (mg/L)**	**Cl (mg/L)**	**Cl:SO_4 ratio**	**Sulfide (µg/L)**	**Fe^{2+} (mg/L)**	**Total Fe (mg/L)**
Established *Cotula*	3-1	0 - 2.5	7.91	68.1	168	1670	i.s.	i.s.	i.s.	36	0.27	0.02
	3-2	2.5 - 5	8.14	59.1	182	2200	i.s.	i.s.	i.s.	300	0.27	<0.01
	3-3	5 - 10	8.22	103.9	247	3300	i.s.	i.s.	i.s.	69	i.s.	i.s.
	3-4	10 - 15	7.53	35.3	250	i.s.	i.s.	i.s.	i.s.	i.s.	0.27	0.57
	3-5	15 - 20	7.54	45.8	276	4680	i.s.	i.s.	i.s.	i.s.	0.27	0.38
	3-6	20 - 30	6.70	56.9	137	7760	947	2188	2.3	186	39.13	37.89
	3-7	30 - 40	6.61	124.3	153	10870	1886	3505	1.9	254	77.50	76.54
Established *Phragmites*	1-1	0 - 2.5	7.45	325.4	238	6040	101	1886	18.8	192	0.56	0.87
	1-2	2.5 - 5	7.63	334.9	213	7500	62	2151	35.0	145	0.86	0.96
	1-3	5 - 10	7.52	250.1	236	7520	119	2971	25.1	161	0.27	0.27
	1-4	10 - 15	7.62	464.9	225	10330	284	3412	12.0	202	0.14	0.13
	1-5	15 - 20	7.39	247.2	158	12520	663	3947	6.0	i.s.	0.30	0.28
	1-6	20 - 30	7.16	213.8	114	14450	1286	4930	3.8	31	7.12	9.12
	1-7	30 - 40	7.12	138.1	111	18010	2288	5867	2.6	<10	18.44	20.13
	MBO	Surface	7.35	143.6	165	3300	36	1281	35.6	56	0.34	0.27
	2-1	0 - 2.5	7.41	340.4	193	7020	57	2042	35.8	124	0.51	0.51
	2-2	2.5 - 5	7.72	361.0	228	8350	83	2132	25.8	78	0.53	0.60
	2-3	5 - 10	7.91	270.8	258	7910	63	1266	20.1	104	0.13	0.21
	2-4	10 - 15	7.91	292.5	265	11280	i.s.	i.s.	i.s.	i.s.	0.16	2.31
	2-5	15 - 20	7.66	432.9	273	11440	330	4118	12.5	21	0.47	0.43
	2-6	20 - 30	7.42	371.7	209	14240	788	4583	5.8	16	0.25	0.68
	2-7	30 - 40	7.31	340.5	123	16830	1323	5724	4.3	109	0.21	0.36
	3-1	0 - 2.5	7.85	156.7	144	3610	140	933	6.7	223	0.39	0.30
	3-2	2.5 - 5	7.86	126.1	244	3700	107	950	8.9	181	0.39	0.36
	3-3	5 - 10	7.93	144.6	244	4350	i.s.	i.s.	i.s.	388	0.07	0.11
	3-4	10 - 15	7.58	246.9	257	7310	65	2163	33.5	67	0.14	0.08
	3-5	15 - 20	7.68	243.6	250	7460	78	2280	29.2	72	0.13	0.13
	3-6	20 - 30	7.78	282.7	258	10630	285	3190	11.2	72	0.43	0.42
	3-7	30 - 40	7.13	227.1	120	15560	1412	4953	3.5	21	8.56	8.32

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-15. Pore-water soluble cation and nutrient analyses for Waltowa in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
Established *Juncus*	1-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
Established *Cotula*	1-1	0 - 2.5	54	61	434	34	<0.01	0.01	7.96	0.02
	1-2	2.5 - 5	69	88	637	38	0.04	0.02	3.58	0.02
	1-3	5 - 10	71	93	721	39	0.14	<0.01	4.82	<0.01
	1-4	10 - 15	105	134	996	43	0.05	0.05	7.60	0.04
	1-5	15 - 20	127	160	1112	60	0.74	0.26	11.08	0.22
	1-6	20 - 30	241	276	1520	58	0.04	0.03	14.27	0.02
	1-7	30 - 40	410	433	1971	63	0.07	0.03	17.20	0.02
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-7	30 - 40	247	303	1562	46	0.21	0.04	10.33	0.03

i.s. Insufficient sample for analysis

Table 9-15 (continued). Pore-water soluble cation and nutrient analyses for Waltowa in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
Established *Cotula*	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-6	20 - 30	205	232	1249	47	1.15	0.23	10.80	0.28
	3-7	30 - 40	413	459	1942	54	0.24	0.01	15.51	0.01
Established *Phragmites*	1-1	0 - 2.5	154	191	1141	61	<0.01	<0.01	11.56	1.68
	1-2	2.5 - 5	98	208	1289	63	i.s.	i.s.	i.s.	i.s.
	1-3	5 - 10	125	236	1680	59	1.06	0.04	1.53	0.07
	1-4	10 - 15	105	310	1945	46	0.44	0.03	1.68	0.02
	1-5	15 - 20	225	330	2339	59	0.40	0.02	2.46	0.02
	1-6	20 - 30	312	456	2797	74	0.07	0.02	3.49	0.02
	1-7	30 - 40	509	612	3325	65	0.30	0.04	1.34	0.04
	MBO	Surface	90	112	693	91	0.11	0.03	12.11	0.08
	2-1	0 - 2.5	77	200	1230	36	<0.01	0.03	6.75	0.12
	2-2	2.5 - 5	110	197	1270	40	0.03	0.04	3.98	0.02
	2-3	5 - 10	38	113	739	25	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	86	374	2404	43	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	203	432	2662	47	0.05	<0.01	0.59	<0.01
	2-7	30 - 40	288	562	3201	57	0.07	<0.01	0.75	<0.01
	3-1	0 - 2.5	91	96	598	46	<0.01	0.03	9.36	0.13
	3-2	2.5 - 5	72	88	638	45	0.26	<0.01	3.15	0.05
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	92	190	1334	35	0.04	0.01	9.91	0.01
	3-5	15 - 20	78	191	1375	43	0.03	0.01	19.38	0.02
	3-6	20 - 30	86	262	1854	53	0.16	0.03	6.86	0.04
	3-7	30 - 40	373	515	2867	52	0.06	0.03	0.60	0.03

i.s. Insufficient sample for analysis

Table 9-16. Pore-water properties for Poltalloch in March 2013.

Treatment	Layer	Depth (cm)	pH	Alkalinity (mmol/L)	Eh* (mV)	EC (µS/cm)	SO_4 (mg/L)	Cl (mg/L)	Cl:SO_4 ratio	Sulfide (µg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)
2009 plantings of Bevy rye	1-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-2	2.5 - 5	8.20	25.5	334	530	i.s.	i.s.	i.s.	233	0.18	0.19
	1-3	5 - 10	7.85	16.5	302	540	i.s.	i.s.	i.s.	30	0.10	0.46
	1-4	10 - 15	7.81	i.s.	327	550	i.s.	i.s.	i.s.	104	1.43	2.37
	1-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	8.03	43.9	333	1680	i.s.	i.s.	i.s.	24	0.05	<0.01
	1-7	30 - 40	7.90	4.6	351	5100	i.s.	i.s.	i.s.	36	0.08	0.06
	1-8	40 - 50	7.84	85.6	336	15200	i.s.	i.s.	i.s.	282	0.36	<0.01
	1-9	50 -60	7.94	117.4	326	17610	1323	3025	2.3	56	0.06	<0.01
	2-1	0 - 2.5	8.26	12.5	339	560	i.s.	i.s.	i.s.	483	0.18	1.28
	2-2	2.5 - 5	7.96	17.8	349	500	i.s.	i.s.	i.s.	104	0.27	<0.01
	2-3	5 - 10	7.67	14.2	333	930	i.s.	i.s.	i.s.	26	0.36	<0.01
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	7.31	14.2	228	1280	i.s.	i.s.	i.s.	17	0.45	2.41
	2-6	20 - 30	6.52	i.s.	214	4130	i.s.	i.s.	i.s.	155	11.81	14.24
	2-7	30 - 40	6.65	i.s.	184	5910	i.s.	i.s.	i.s.	337	40.99	41.40
	2-8	40 - 50	6.88	35.3	178	11550	i.s.	i.s.	i.s.	<10	6.89	14.24
	2-9	50 -60	7.72	63.0	178	15610	i.s.	i.s.	i.s.	<10	0.45	<0.01
	3-1	0 - 2.5	7.79	28.3	368	610	i.s.	i.s.	i.s.	958	0.09	2.56
	3-2	2.5 - 5	7.53	52.7	376	520	i.s.	i.s.	i.s.	375	0.45	0.57
	3-3	5 - 10	7.59	15.4	362	510	i.s.	i.s.	i.s.	621	0.30	0.51
	3-4	10 - 15	7.59	14.2	365	470	i.s.	i.s.	i.s.	26	0.09	1.61
	3-5	15 - 20	7.55	8.9	350	600	i.s.	i.s.	i.s.	362	0.27	0.57
	3-6	20 - 30	7.29	14.2	351	1370	i.s.	i.s.	i.s.	52	0.54	<0.01
	3-7	30 - 40	6.81	i.s.	285	2970	i.s.	i.s.	i.s.	i.s.	1.07	0.47
	3-8	40 - 50	6.45	3.8	225	6960	1094	1973	1.8	11	5.33	4.79
	3-9	50 -60	7.53	99.5	164	14720	2199	4329	2.0	<10	0.49	0.32

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-17. Pore-water soluble cation and nutrient analyses for Poltalloch in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
2009 plantings of Bevy rye	1-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-8	40 - 50	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-9	50 -60	332	248	1681	74	i.s.	i.s.	i.s.	i.s.
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-8	40 - 50	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-9	50 -60	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-8	40 - 50	226	136	1158	144	0.28	0.04	2.43	0.04
	3-9	50 -60	459	347	2474	135	0.37	<0.01	1.04	0.02

i.s. Insufficient sample for analysis

Table 9-18. Pore-water properties for Tolderol in March 2013.

Treatment	Layer	Depth (cm)	pH	Alkalinity (mmol/L)	Eh* (mV)	EC (µS/cm)	SO_4 (mg/L)	Cl (mg/L)	Cl:SO_4 ratio	Sulfide (µg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)
2010 planted *Juncus* into 2009 plantings of Bevy rye	1-1	0 - 2.5	7.92	39.6	305	840	18	62	3.4	17	0.12	0.06
	1-2	2.5 - 5	7.93	i.s.	301	380	i.s.	i.s.	i.s.	<10	<0.01	<0.01
	1-3	5 - 10	8.04	i.s.	301	880	i.s.	i.s.	i.s.	56	0.18	<0.01
	1-4	10 - 15	8.05	i.s.	299	1790	i.s.	i.s.	i.s.	113	0.36	1.61
	1-5	15 - 20	7.93	i.s.	303	1960	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	7.28	i.s.	291	2280	i.s.	i.s.	i.s.	<10	i.s.	i.s.
	1-7	30 - 40	6.01	i.s.	247	6060	i.s.	i.s.	i.s.	28	61.30	62.68
	1-8	40 - 50	5.76	i.s.	262	7600	i.s.	i.s.	i.s.	<10	157.15	173.98
	1-9	50 -60	5.17	i.s.	307	7840	i.s.	i.s.	i.s.	<10	152.14	183.28
	2-1	0 - 2.5	7.42	i.s.	223	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	7.67	i.s.	253	500	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	6.95	i.s.	282	1080	i.s.	i.s.	i.s.	<10	0.36	2.47
	2-4	10 - 15	6.43	i.s.	237	1570	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	5.21	i.s.	296	1370	i.s.	i.s.	i.s.	<10	82.34	99.71
	2-6	20 - 30	5.43	i.s.	283	2200	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-7	30 - 40	5.06	3.3	295	7680	1190	814	0.7	11	262.22	287.74
	2-8	40 - 50	5.73	4.0	222	8040	2286	1763	0.8	23	118.13	117.76
	2-9	50 -60	5.97	i.s.	213	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-1	0 - 2.5	8.00	38.4	229	610	i.s.	i.s.	i.s.	28	0.12	0.35
	3-2	2.5 - 5	7.96	28.3	324	370	i.s.	i.s.	i.s.	84	0.36	<0.01
	3-3	5 - 10	7.78	i.s.	328	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	9.50
	3-4	10 - 15	7.10	i.s.	329	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-5	15 - 20	6.38	i.s.	324	i.s.	i.s.	i.s.	i.s.	<10	i.s.	i.s.
	3-6	20 - 30	5.68	1.8	274	i.s.	i.s.	i.s.	i.s.	338	11.10	8.74
	3-7	30 - 40	4.95	i.s.	272	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-8	40 - 50	4.44	i.s.	363	i.s.	i.s.	i.s.	i.s.	141	147.67	132.00
	3-9	50 -60	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-18 (continued). Pore-water properties for Tolderol in March 2013.

Treatment	Layer	Depth (cm)	pH	Alkalinity (mmol/L)	Eh* (mV)	EC (µS/cm)	SO_4 (mg/L)	Cl (mg/L)	Cl:SO_4 ratio	Sulfide (µg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)
Scald (no bioremediation)	1-1	0 - 2.5	7.81	i.s.	303	520	i.s.	i.s.	i.s.	i.s.	0.72	<0.01
	1-2	2.5 - 5	7.29	i.s.	308	420	i.s.	i.s.	i.s.	26	0.18	1.71
	1-3	5 - 10	5.74	i.s.	266	i.s.	i.s.	i.s.	i.s.	52	17.72	9.02
	1-4	10 - 15	6.00	i.s.	266	i.s.	i.s.	i.s.	i.s.	52	10.38	11.78
	1-5	15 - 20	5.75	i.s.	271	i.s.	i.s.	i.s.	i.s.	52	51.55	119.47
	1-6	20 - 30	5.42	i.s.	296	i.s.	i.s.	i.s.	i.s.	52	89.50	92.69
	1-7	30 - 40	4.42	i.s.	385	i.s.	i.s.	i.s.	i.s.	78	151.78	165.24
	1-8	40 - 50	4.04	i.s.	440	i.s.	i.s.	i.s.	i.s.	84	314.66	308.83
	2-1	0 - 2.5	6.68	10.1	327	700	i.s.	i.s.	i.s.	90	0.08	<0.01
	2-2	2.5 - 5	6.05	i.s.	282	830	i.s.	i.s.	i.s.	181	44.57	50.52
	2-3	5 - 10	5.74	2.8	269	1900	i.s.	i.s.	i.s.	192	249.15	246.91
	2-4	10 - 15	5.04	2.1	317	2860	1371	271	0.2	119	416.33	435.32
	2-5	15 - 20	4.11	1.2	415	3510	932	176	0.2	145	502.60	523.26
	2-6	20 - 30	3.77	1.0	447	4770	2582	537	0.2	285	549.50	619.17
	2-7	30 - 40	3.26	i.s.	464	3780	i.s.	i.s.	i.s.	233	340.98	412.15
	2-8	40 - 50	3.83	0.5	454	6960	3657	886	0.2	73	375.52	378.72
	2-9	50 -60	4.01	2.8	441	7340	3692	975	0.3	<10	254.17	285.84
	3-1	0 - 2.5	7.74	22.9	333	590	i.s.	i.s.	i.s.	39	0.13	<0.01
	3-2	2.5 - 5	7.41	7.1	261	490	i.s.	i.s.	i.s.	104	0.36	1.52
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	6.77	i.s.	229	850	i.s.	i.s.	i.s.	i.s.	1.16	1.20
	3-5	15 - 20	6.28	i.s.	222	1010	i.s.	i.s.	i.s.	300	21.57	19.94
	3-6	20 - 30	6.33	3.6	270	1120	i.s.	i.s.	i.s.	28	12.89	10.07
	3-7	30 - 40	6.94	3.6	253	2250	i.s.	i.s.	i.s.	56	i.s.	7.98
	3-8	40 - 50	7.40	119.9	123	5210	2250	606	0.3	<10	1.60	1.58
	3-9	50 -60	7.77	130.0	241	4940	i.s.	i.s.	i.s.	<10	0.11	<0.01

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-19. Pore-water soluble cation and nutrient analyses for Tolderol in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
2010 planted *Juncus* into 2009 plantings of Bevy rye	1-1	0 - 2.5	18	10	50	10	0.58	0.04	0.45	0.04
	1-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-8	40 - 50	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-9	50 -60	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-7	30 - 40	105	174	567	56	0.16	0.07	5.91	0.03
	2-8	40 - 50	219	338	1095	98	0.19	0.05	2.91	0.02
	2-9	50 -60	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-8	40 - 50	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-9	50 -60	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.

i.s. Insufficient sample for analysis

Table 9-19 (continued). Pore-water soluble cation and nutrient analyses for Tolderol in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
Scald (no bioremediation)	1-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-8	40 - 50	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	61	83	221	45	0.04	0.11	4.53	0.04
	2-5	15 - 20	46	67	159	26	0.58	0.15	7.36	0.09
	2-6	20 - 30	149	221	451	52	0.04	0.08	8.63	0.03
	2-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-8	40 - 50	279	393	651	99	0.29	0.14	8.22	0.06
	2-9	50 -60	331	449	729	82	0.08	0.04	7.23	0.02
	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-4	10 - 15	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-5	15 - 20	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-7	30 - 40	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-8	40 - 50	543	203	483	47	0.16	0.03	2.88	0.02
	3-9	50 -60	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.

i.s. Insufficient sample for analysis

Table 9-20. Pore-water properties for Campbell Park in March 2013.

Treatment	Layer	Depth (cm)	pH	Alkalinity (mmol/L)	Eh* (mV)	EC (µS/cm)	SO_4 (mg/L)	Cl (mg/L)	Cl:SO_4 ratio	Sulfide (µg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)
Control (no bioremediation)	1-1	0 - 2.5	7.32	83.0	184	4010	108	530	4.9	68	2.04	2.05
	1-2	2.5 - 5	6.82	52.6	149	4300	195	587	3.0	11	15.00	19.94
	1-3	5 - 10	6.60	35.1	155	5370	680	1365	2.0	45	133.88	163.34
	1-4	10 - 15	6.21	23.8	180	7020	1548	1771	1.1	130	442.11	423.54
	1-5	15 - 20	6.11	12.0	190	7980	2343	2016	0.9	225	693.59	703.69
	1-6	20 - 30	i.s.	3.3	373	10170	3117	2890	0.9	84	773.24	809.10
	1-7	30 - 40	4.38	1.9	377	11090	3443	3920	1.1	56	448.37	501.42
	1-8	40 - 50	5.65	10.0	267	11290	2367	3436	1.5	<10	198.68	185.18
	1-9	50 -60	6.66	61.4	157	11560	1985	3863	1.9	<10	34.90	38.94
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	6.66	9.2	171	4430	i.s.	i.s.	i.s.	13	35.80	46.53
	2-3	5 - 10	6.21	7.0	182	5540	1524	1420	0.9	68	328.45	378.91
	2-4	10 - 15	5.96	6.3	211	6970	2787	1593	0.6	146	782.19	852.79
	2-5	15 - 20	4.37	4.5	389	7310	1370	676	0.5	158	783.98	757.82
	2-6	20 - 30	3.86	1.0	451	8640	1976	1005	0.5	113	745.49	687.55
	2-7	30 - 40	4.06	2.4	432	8770	3449	2174	0.6	68	484.17	520.41
	2-8	40 - 50	5.07	4.0	343	8690	3005	2425	0.8	<10	218.37	243.11
	2-9	50 -60	6.80	34.0	167	8600	2316	2461	1.1	<10	28.64	24.22
	3-1	0 - 2.5	7.98	35.3	179	i.s.	i.s.	i.s.	i.s.	77	<0.01	0.95
	3-2	2.5 - 5	7.68	68.1	200	4100	i.s.	i.s.	i.s.	23	0.31	0.24
	3-3	5 - 10	5.97	3.2	207	5060	612	646	1.1	11	207.63	238.36
	3-4	10 - 15	5.73	7.7	232	7020	2846	1611	0.6	203	759.81	784.41
	3-5	15 - 20	4.26	4.9	379	8090	3734	1845	0.5	282	934.33	1085.45
	3-6	20 - 30	3.91	2.6	426	8910	3845	2069	0.5	146	807.24	870.83
	3-7	30 - 40	3.84	1.4	442	9510	3858	2442	0.6	90	594.25	653.36
	3-8	40 - 50	4.68	3.3	386	9060	3755	2980	0.8	34	258.64	310.54

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-20 (continued). Pore-water properties for Campbell Park in March 2013.

Treatment	Layer	Depth (cm)	pH	Alkalinity (mmol/L)	Eh* (mV)	EC (µS/cm)	SO_4 (mg/L)	Cl (mg/L)	Cl:SO_4 ratio	Sulfide (µg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)
2010 seeded with Bevy rye and *Puccinellia*	1-1	0 - 2.5	7.84	64.5	211	3770	i.s.	i.s.	i.s.	11	0.45	0.76
	1-2	2.5 - 5	7.59	54.4	196	3970	i.s.	i.s.	i.s.	<10	1.16	1.76
	1-3	5 - 10	6.23	17.8	210	4910	i.s.	i.s.	i.s.	38	118.67	110.16
	1-4	10 - 15	5.94	12.5	209	7000	2249	1555	0.7	124	474.32	511.86
	1-5	15 - 20	5.20	4.5	300	8800	1382	760	0.6	130	607.67	648.61
	1-6	20 - 30	4.12	3.1	406	9410	3497	2065	0.6	434	512.81	519.46
	1-7	30 - 40	4.22	4.2	410	9890	3735	2580	0.7	56	310.55	364.67
	1-8	40 - 50	3.95	4.0	431	11890	5292	3773	0.7	51	416.15	415.00
	1-9	50 -60	7.19	54.7	146	9600	2705	2810	1.0	11	2.33	2.55
	2-1	0 - 2.5	8.06	58.4	368	3550	i.s.	i.s.	i.s.	28	i.s.	1.66
	2-2	2.5 - 5	7.99	i.s.	357	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	6.85	i.s.	208	i.s.	i.s.	i.s.	i.s.	i.s.	1.43	1.71
	2-4	10 - 15	6.15	7.9	210	6960	1208	1162	1.0	51	211.21	251.66
	2-5	15 - 20	5.68	8.6	222	9110	2814	1848	0.7	124	537.86	536.55
	2-6	20 - 30	4.51	6.3	333	11540	4157	2403	0.6	169	795.61	813.85
	2-7	30 - 40	3.98	4.9	433	12380	4448	2748	0.6	73	551.29	572.64
	2-8	40 - 50	5.72	8.2	272	9550	3240	2780	0.9	17	92.18	80.72
	2-9	50 -60	4.91	5.9	337	10960	3753	3352	0.9	62	148.56	179.48
	3-1	0 - 2.5	7.79	68.6	354	3350	i.s.	i.s.	i.s.	11	0.11	<0.01
	3-2	2.5 - 5	7.98	45.8	347	3310	i.s.	i.s.	i.s.	75	i.s.	3.55
	3-3	5 - 10	6.63	38.8	158	4900	779	1286	1.7	28	41.35	54.70
	3-4	10 - 15	6.22	7.5	212	6430	1569	1533	1.0	23	130.22	142.45
	3-5	15 - 20	6.02	11.6	205	9010	2726	1809	0.7	141	476.11	528.01
	3-6	20 - 30	4.47	5.9	380	9460	i.s.	i.s.	i.s.	118	516.39	523.26
	3-7	30 - 40	4.20	4.7	410	10250	3765	2488	0.7	68	388.41	415.95
	3-8	40 - 50	4.62	5.4	661	10170	3402	2754	0.8	28	179.88	168.09
	3-9	50 -60	6.66	35.1	468	9670	3119	3139	1.0	23	15.04	21.27

* Eh measurements are presented versus the standard hydrogen electrode
i.s. Insufficient sample for analysis

Table 9-21. Pore-water soluble cation and nutrient analyses for Campbell Park in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
Control (no bioremediation)	1-1	0 - 2.5	40	52	341	25	i.s.	i.s.	i.s.	i.s.
	1-2	2.5 - 5	47	57	381	29	0.29	0.02	1.94	<0.01
	1-3	5 - 10	95	133	795	62	0.12	0.02	4.73	<0.01
	1-4	10 - 15	132	199	1029	89	<0.01	0.16	7.14	0.07
	1-5	15 - 20	162	254	1152	102	<0.01	0.25	7.74	0.10
	1-6	20 - 30	276	447	1751	113	<0.01	0.22	8.03	0.03
	1-7	30 - 40	399	582	2218	115	0.04	0.11	7.49	0.03
	1-8	40 - 50	324	458	1707	93	0.06	0.07	5.13	0.02
	1-9	50 -60	350	489	1897	97	0.13	0.02	4.14	0.02
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	128	192	873	78	0.04	0.11	4.74	0.05
	2-4	10 - 15	167	270	1017	116	<0.01	0.22	7.45	0.05
	2-5	15 - 20	83	137	448	46	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	142	227	659	54	i.s.	i.s.	i.s.	i.s.
	2-7	30 - 40	316	489	1349	82	0.04	0.07	7.88	0.02
	2-8	40 - 50	317	469	1307	81	0.04	0.08	5.26	0.02
	2-9	50 -60	305	436	1330	81	0.10	0.02	4.21	0.02
	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	56	82	414	49	0.63	0.09	7.00	0.04
	3-4	10 - 15	161	275	1025	112	0.04	0.30	10.67	0.14
	3-5	15 - 20	209	365	1197	103	0.02	0.09	12.35	0.04
	3-6	20 - 30	270	442	1358	90	0.04	0.07	11.98	0.04
	3-7	30 - 40	317	502	1543	83	<0.01	0.15	11.50	0.05
	3-8	40 - 50	370	559	1727	92	0.04	0.08	7.74	0.02

i.s. Insufficient sample for analysis

Table 9-21 (continued). Pore-water soluble cation and nutrient analyses for Campbell Park in March 2013.

Treatment	Layer	Depth (cm)	Soluble cations				Nutrients			
			Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Nitrate (mg/L N)	Nitrite (mg/L N)	Ammonia (mg/L N)	Orthophosphate (mg/L P)
2010 seeded with Bevy rye and *Puccinellia*	1-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	1-4	10 - 15	177	272	1029	87	0.10	0.15	8.37	0.04
	1-5	15 - 20	107	158	534	50	i.s.	i.s.	i.s.	i.s.
	1-6	20 - 30	299	443	1475	99	0.04	0.09	8.90	0.03
	1-7	30 - 40	373	559	1766	93	0.04	0.07	11.93	0.02
	1-8	40 - 50	449	705	2233	112	0.03	0.06	12.36	0.03
	1-9	50 -60	332	485	1653	109	0.11	<0.01	6.88	0.01
	2-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-3	5 - 10	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	2-4	10 - 15	115	180	772	81	i.s.	i.s.	i.s.	i.s.
	2-5	15 - 20	221	355	1275	114	i.s.	i.s.	i.s.	i.s.
	2-6	20 - 30	306	511	1648	98	0.01	0.10	13.81	0.04
	2-7	30 - 40	376	596	1865	83	0.03	0.07	12.83	0.03
	2-8	40 - 50	345	517	1646	100	0.07	0.03	8.18	0.01
	2-9	50 -60	395	597	1913	105	0.05	0.05	9.76	0.01
	3-1	0 - 2.5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-2	2.5 - 5	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-3	5 - 10	102	151	726	50	0.53	0.04	4.72	0.03
	3-4	10 - 15	150	227	896	82	0.15	0.06	8.43	0.03
	3-5	15 - 20	207	334	1078	91	0.10	0.04	2.38	0.02
	3-6	20 - 30	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.	i.s.
	3-7	30 - 40	316	495	1542	89	0.05	0.12	12.38	0.03
	3-8	40 - 50	334	530	1642	89	0.06	0.05	9.70	0.02
	3-9	50 -60	366	548	1797	107	0.10	0.02	8.17	0.03

i.s. Insufficient sample for analysis

.

APPENDIX 4. Additional plots

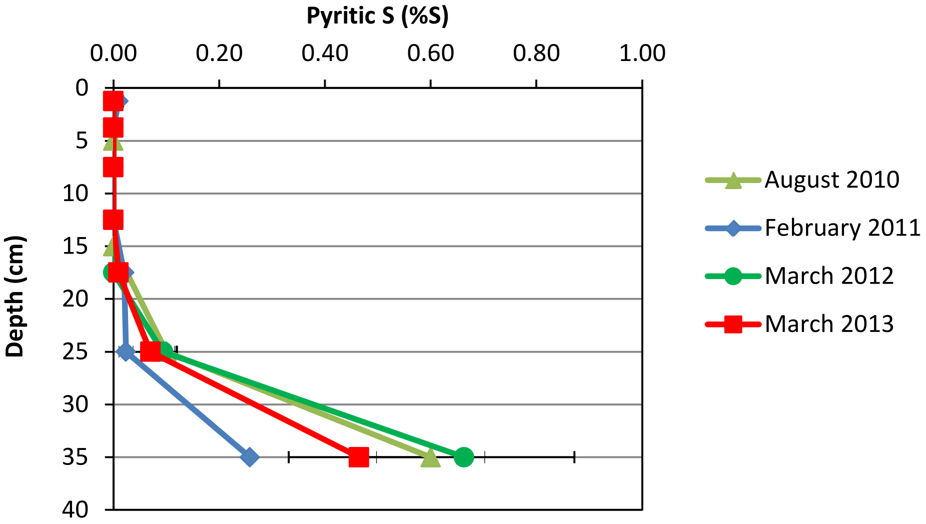


Figure 9-1. Campbell Park pyritic sulfur dynamics at the control site (August 2010 - March 2013).

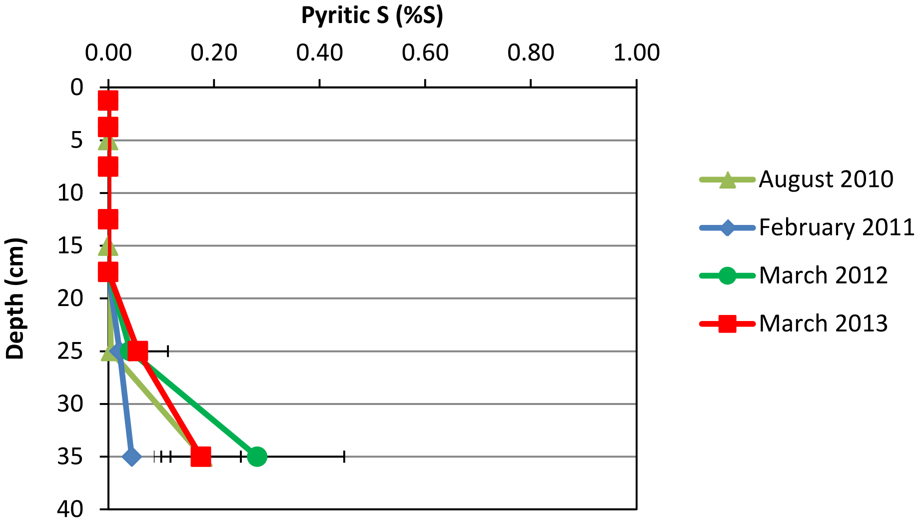


Figure 9-2. Campbell Park pyritic sulfur dynamics at the Bevy rye/*Puccinellia* site (August 2010 - March 2013).

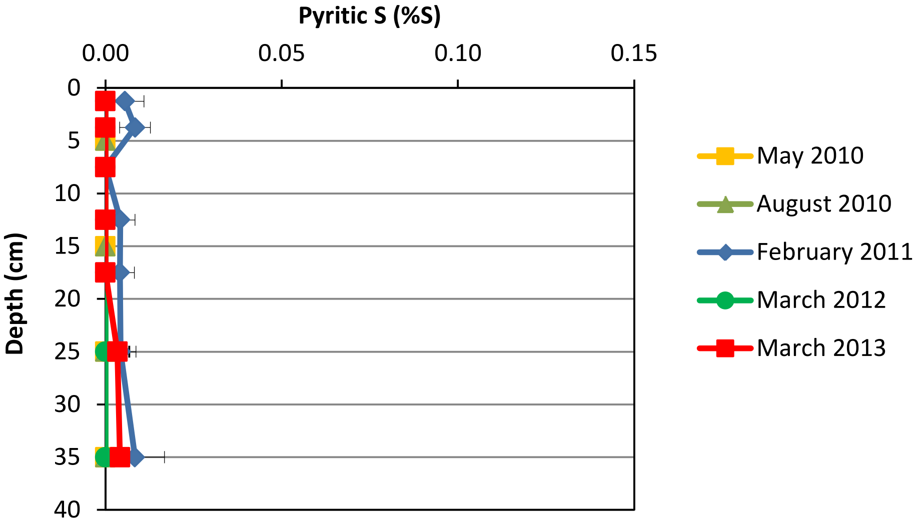


Figure 9-3. Tolderol pyritic sulfur dynamics at the control site (May 2010 – March 2013).

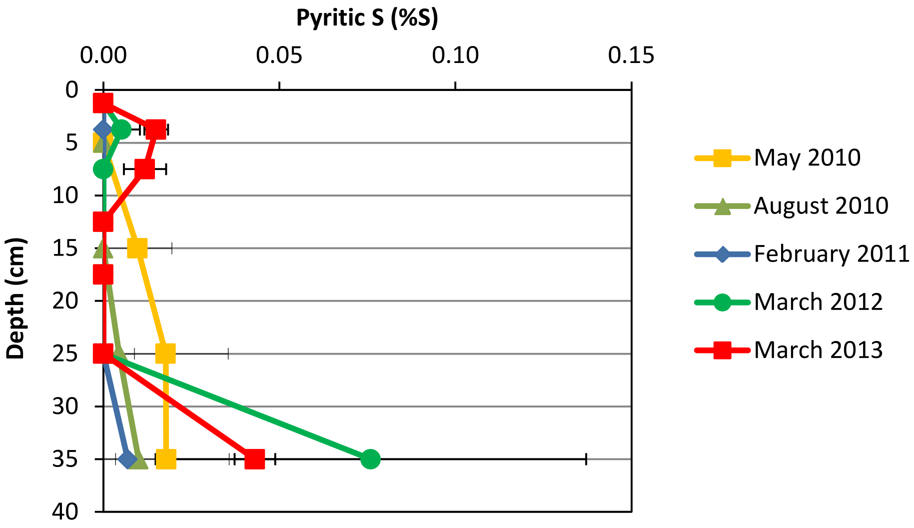


Figure 9-4. Tolderol pyritic sulfur dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2013).

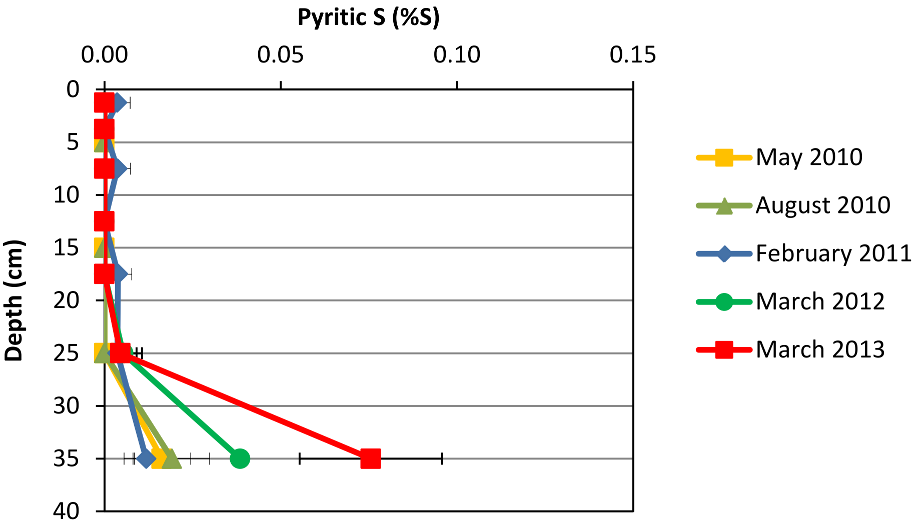


Figure 9-5. Poltalloch pyritic sulfur dynamics in the surface soil (0-40 cm) at the Bevy rye site (May 2010 – March 2013).

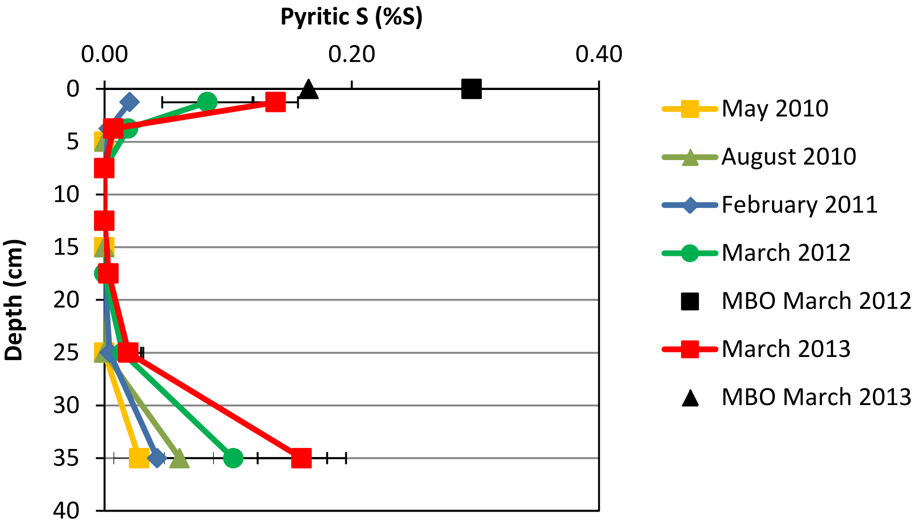


Figure 9-6. Waltowa pyritic sulfur dynamics at the established *Phragmites* site (May 2010 – March 2013).

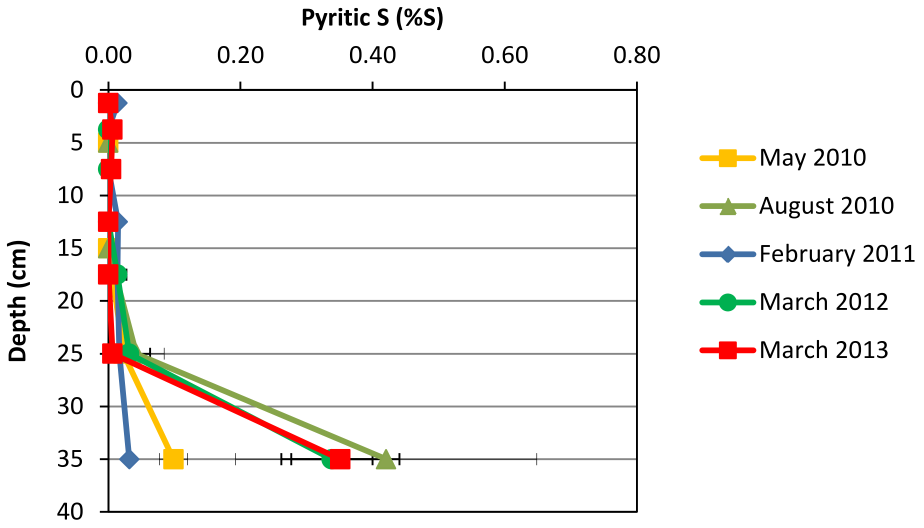


Figure 9-7. Waltowa pyritic sulfur dynamics at the established *Cotula* site (May 2010 – March 2013).

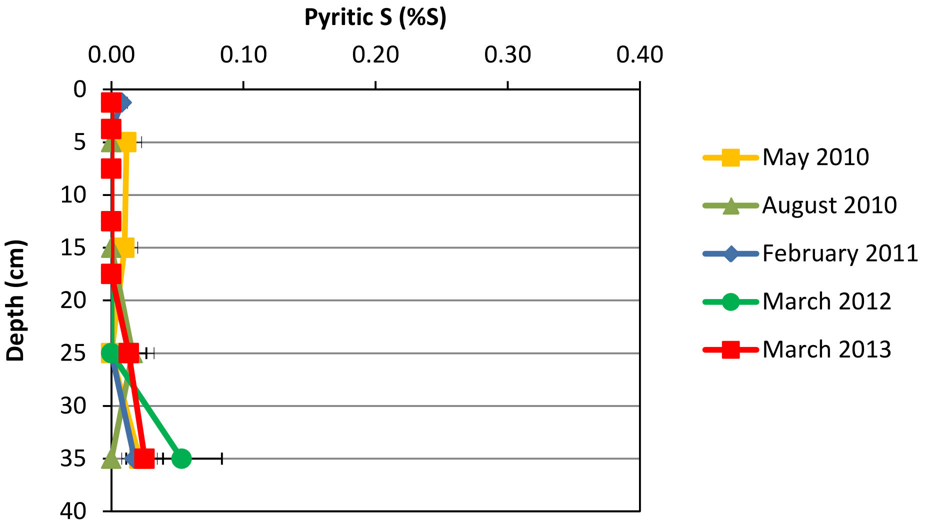


Figure 9-8. Waltowa pyritic sulfur dynamics at the established *Juncus* site (May 2010 – March 2013).

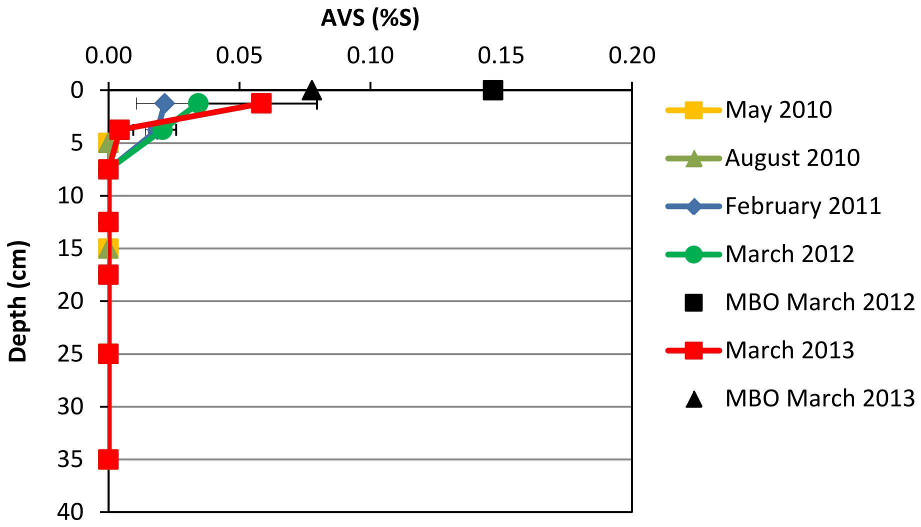


Figure 9-9. Waltowa AVS dynamics at the established *Phragmites* site (May 2010 – March 2013).

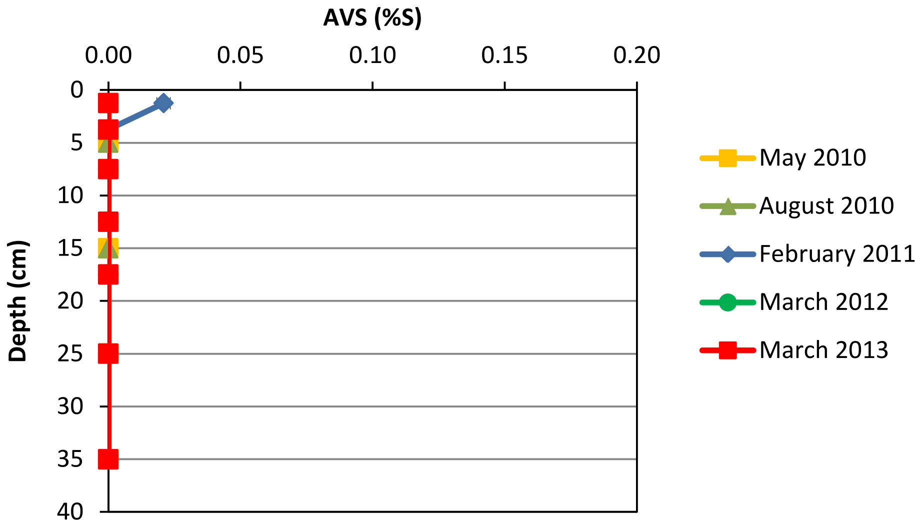


Figure 9-10. Waltowa AVS dynamics at the established *Cotula* site (May 2010 – March 2013).

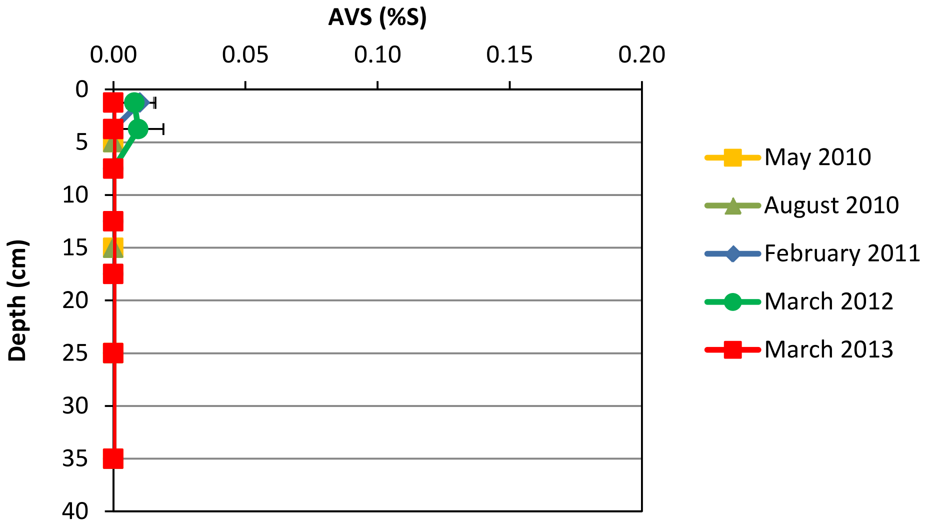


Figure 9-11. Waltowa AVS dynamics at the established *Juncus* site (May 2010 – March 2013).

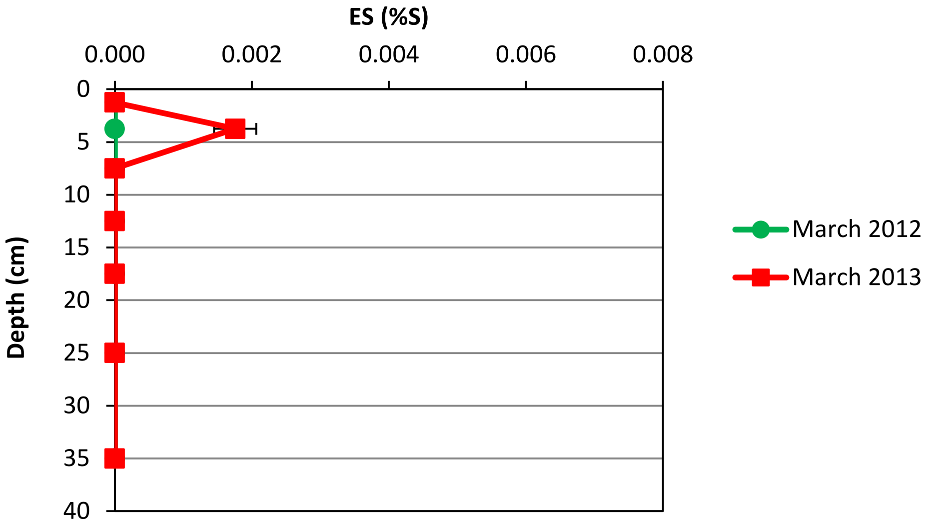


Figure 9-12. Campbell Park elemental sulfur dynamics at the control site (March 2012 – March 2013).

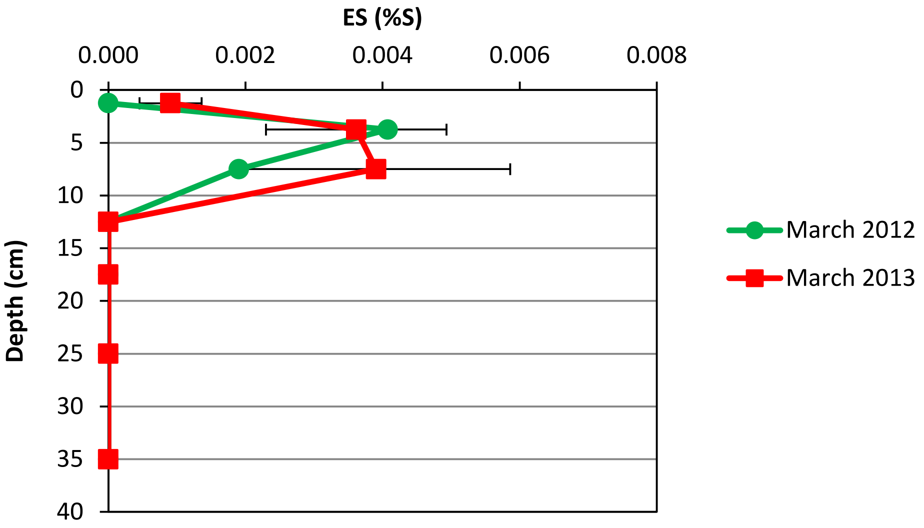


Figure 9-13. Campbell Park elemental sulfur dynamics at the Bevy rye/*Puccinellia* site (March 2012 – March 2013).

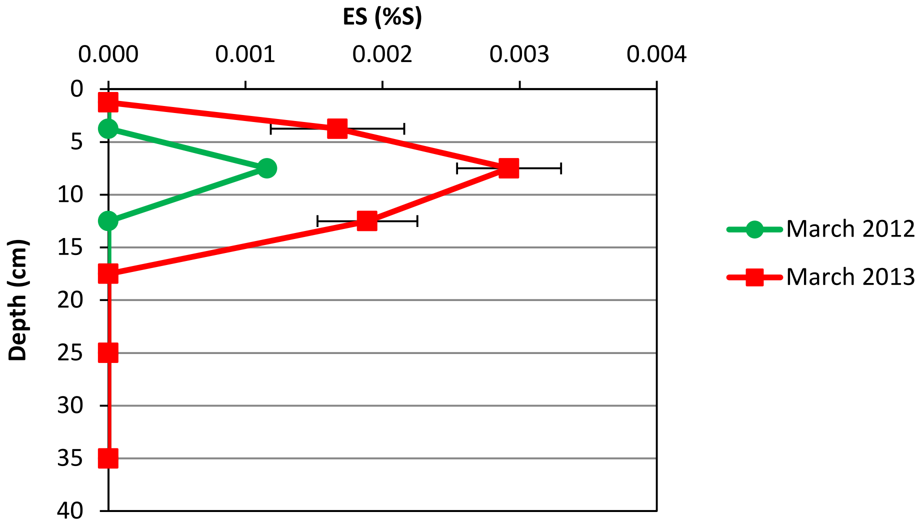


Figure 9-14. Poltalloch elemental sulfur dynamics in the surface soil (0-40 cm) at the Bevy rye site (March 2012 – March 2013).

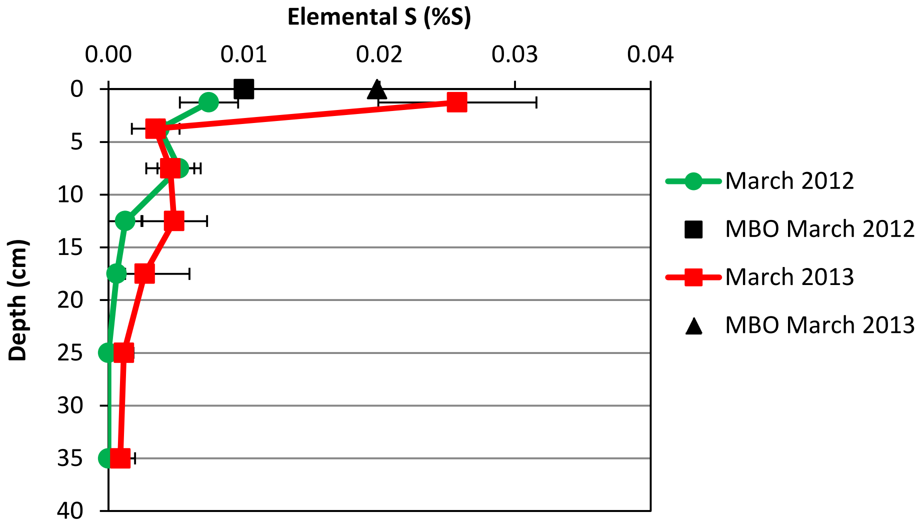


Figure 9-15. Waltowa elemental sulfur dynamics at the established *Phragmites* site (March 2012 – March 2013).

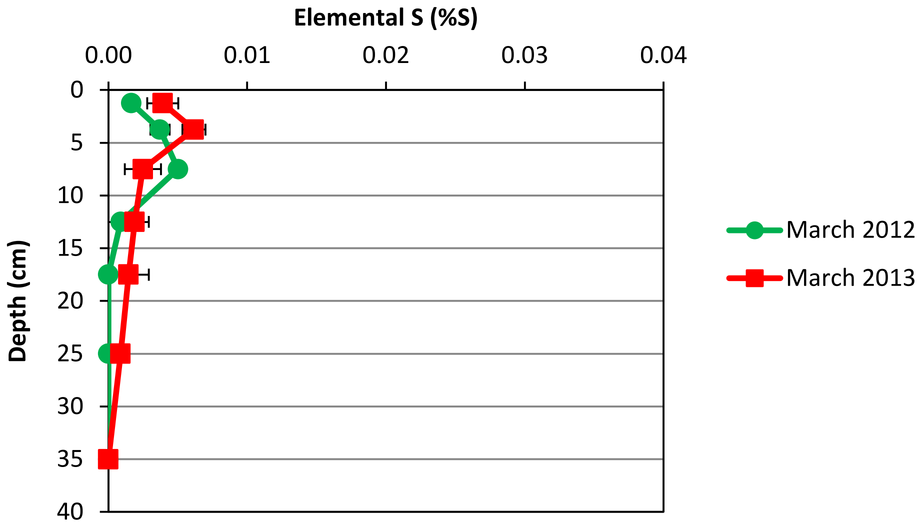


Figure 9-16. Waltowa elemental sulfur dynamics at the established *Cotula* site (March 2012 – March 2013).

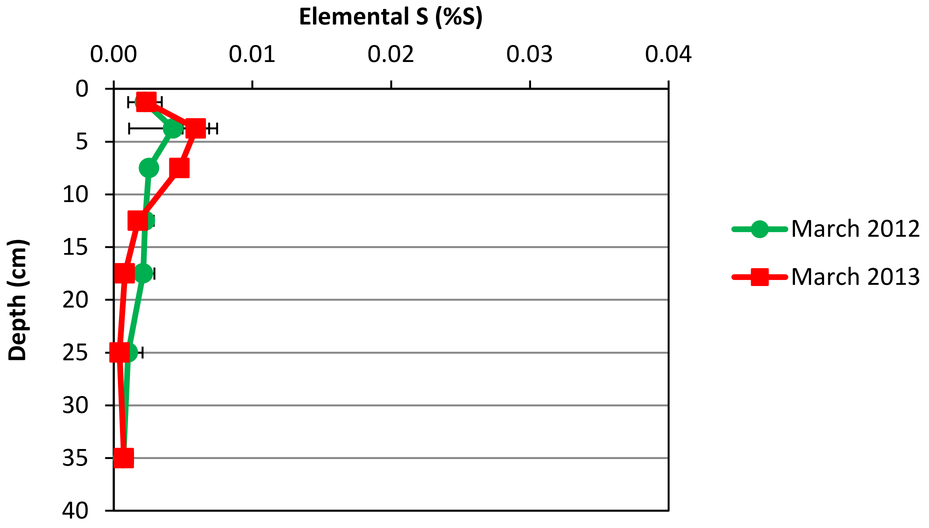


Figure 9-17. Waltowa elemental sulfur dynamics at the established *Juncus* site (March 2012 – March 2013).

APPENDIX 5. Vegetation sampling and results

Table 9-22. Lower Lakes vegetation sampling (March 2013).

Location	Date	GPS Co-ordinates Zone East, North.	Vegetation Types Collected	Sediment Details		
				Depth (cm)	**pH**	**Profile Remarks**
Waltowa	22/03/13	-	*Phragmites australis*, bull rush, *Schoenoplectus valaidus* and unidentified *juncus*			
Poltalloch	23/03/13	-	*Phragmites australis*			
Campbell Park	23/03/13	-	*Phragmites australis* and *Schoenoplectus valaidus*			
Tolderol	24/03/13	-	*Phragmites australis* from both the foredune and the inundated former scald			
Wellington Lodge	25/03/13	54H 0349373, 6078962	*Phragmites australis* and *Schoenoplectus valaidus*	2.5	7.50	
				10	7.15	
				20	7.10	
Meningie	25/03/13	Site 1 54H 0349333, 6049331	Couch, shoreline *Typha orientalis*, *Phragmites australis*, unidentified reed and *Schoenoplectus valaidus* (from shoreline and 50 m from shore)	0-10	7.46	0-10 cm: Dark medium sand.
				10-30	7.34	10-30 cm: Grey medium sand.
				30-50	7.12	30-50 cm: Grey silty sand.
				50-60	6.99	50-60 cm: Grey clay/silt.
		Site 2 54H 0347615, 6049138	*Phragmites australis* and sedge	0-10	5.98	0-10 cm: Organic sandy/clay silt.
Loveday Bay	25/03/13	54H 0326554, 6061602	*Phragmites australis*, *Schoenoplectus valaidus* and *Typha orientalis*	0-10	7.41	0-10 cm: Grey medium sand.
				10-20	7.30	10-20 cm: Black fine sand.
				20-30	7.14	20-30 cm: Light grey sand.
Naro Point	25/03/13	54H 0341693, 6063591	*Phragmites australis* (from shoreline and further in lake), *Schoenoplectus valaidus* and *Typha orientalis*	0-10	7.79	0-10 cm: Sand. Black organics.
				10-30	7.19	10-30 cm: Sand. Mottled iron.
				30-40	7.08	30-40 cm: Grey sand.
Boggy Lake	26/03/13	54H 0335035, 6089134	*Phragmites australis*, unidentified grass, *Typha orientalis* and unidentified rush			0-10 cm: Light clay.
				10-20	6.67	10-20 cm: Light clay.
Milang Jetty	26/03/13	54H 0316204, 6079852	*Phragmites australis*	0-5	7.90	0-5 cm: Red sand.
				5-15	6.49	5-15 cm: Grey sand.
				15-30	6.40	15-30 cm: Grey sand.
Currency Creek	26/03/13	54H 0300366, 6073032	*Phragmites australis* and *Schoenoplectus valaidus*	0-10	5.47	0-10 cm: Grey sand.
				10-20	4.61	10-20 cm: Grey clay.
				20-30	4.02	20-30 cm: Grey clay.
				30-40	4.50	30-40 cm: Grey silt/clay.
				40-50	5.47	40-50 cm: Grey clay.
Hunters Creek	26/03/13	Site 1 54H 0308268, 6065522	*Bolboschoenus*	0-10	6.65	
				10-20	6.91	
				20-30	6.76	
				30-40	6.87	
				40-50	6.96	
	26/03/13	Site 2 54H 0308828, 6066394	*Schoenoplectus valaidus*	0-10	6.78	0-10 cm: Grey clay.
				10-30	6.70	10-30 cm: Grey clay.
Finniss Creek	26/03/13	54H 0303078, 6079610	*Phragmites australis*, *Schoenoplectus valaidus* and *Typha orientalis*	0-10	6.47	0-10 cm: Black ooze.
				10-20	6.47	10-20 cm: Grey clay.
				20-30	6.48	20-30 cm: Grey clay.
				30-40	6.47	30-40 cm: Grey clay.

Table 9-23. Metal/metalloid content (mg/kg) of the vegetation (March 2013).

Site	Vegetation Type	Leaf/Stem/ Flower	Iron	Aluminium	Silver	Arsenic	Lead	Cadmium	Chromium	Copper	Manganese	Nickel	Selenium	Zinc
Boggy Lake	Unidentified rush	Stem	987	814	<1	<2	<1	<0.5	<2	2	27	39.0	0.3	4
Boggy Lake	Unidentified grass	Leaf	295	224	<1	<2	<1	<0.5	<2	5	446	21.0	0.7	10
Boggy Lake	Unidentified grass	Stem	186	226	<1	<2	<1	<0.5	<2	5	157	35.4	0.3	17
Boggy Lake	Phragmites	Leaf	201	117	<1	<2	<1	<0.5	3.6	4	105	20.6	0.2	14
Boggy Lake	Phragmites	Stem	73	110	<1	<2	<1	<0.5	2.4	3	51	26.3	0.1	8
Boggy Lake	Typha	Leaf	159	202	<1	<2	<1	<0.5	<2	3	794	6.8	0.7	9
Boggy Lake	Typha	Stem	118	163	<1	<2	<1	<0.5	<2	2	400	4.5	1.6	6
Campbell Park	Phragmites	Leaf	427	362	<1	<2	<1	<0.5	<2	8	156	64.3	1.2	28
Campbell Park	Phragmites	Stem	386	538	<1	<2	7.1	<0.5	<2	6	105	49.2	0.4	37
Campbell Park	Schoenoplectus	Stem	344	397	<1	<2	1.6	<0.5	<2	3	222	16.9	1.2	13
Currency Creek	Phragmites	Leaf	975	924	<1	<2	<1	<0.5	4.0	4	190	28.6	<0.1	28
Currency Creek	Phragmites	Stem	372	413	<1	<2	<1	<0.5	<2	3	64	14.5	0.2	58
Currency Creek	Schoenoplectus	Flower	823	1,053	<1	<2	<1	<0.5	<2	5	148	8.7	0.8	13
Currency Creek	Schoenoplectus	Stem	769	889	<1	<2	<1	<0.5	<2	2	183	8.4	1.3	5
Finniss Creek	Phragmites	Leaf	133	90	<1	<2	<1	<0.5	<2	5	194	6.7	0.9	23
Finniss Creek	Phragmites	Stem	85	99	<1	<2	<1	<0.5	<2	4	58	13.1	0.5	18
Finniss Creek	Schoenoplectus	Stem	360	318	<1	<2	<1	<0.5	2.8	3	443	18.4	1.5	10
Finniss Creek	Typha	Leaf	192	77	<1	<2	<1	<0.5	<2	3	403	4.7	2.2	16
Finniss Creek	Typha	Stem	278	148	<1	<2	<1	<0.5	<2	2	275	7.5	2.6	14
Hunters Creek (site 1)	Bolboschoenus	Leaf	133	150	<1	<2	<1	<0.5	<2	3	126	21.3	0.7	9
Hunters Creek (site 1)	Bolboschoenus	Stem	54	66	<1	<2	<1	<0.5	<2	2	208	6.5	0.6	5
Hunters Creek (site 2)	Schoenoplectus	Stem	139	171	<1	<2	<1	<0.5	<2	2	162	8.5	1.0	7
Hunters Creek (site 2)	Schoenoplectus	Flower	81	112	<1	<2	<1	<0.5	<2	4	93	173.1	1.1	10
Loveday Bay	Phragmites	Leaf	578	413	<1	<2	<1	<0.5	<2	4	204	16.2	0.3	20
Loveday Bay	Phragmites	Stem	255	343	<1	<2	<1	<0.5	<2	3	75	25.6	0.3	9
Loveday Bay	Schoenoplectus	Stem	551	505	<1	<2	<1	<0.5	<2	3	109	12.3	1.6	7
Loveday Bay	Typha	Leaf	601	648	<1	<2	<1	<0.5	<2	3	442	9.9	1.1	14
Loveday Bay	Typha	Stem	569	515	<1	<2	<1	<0.5	<2	2	202	7.5	1.7	10
Meningie (site 1)	Couch	Leaf	463	485	<1	<2	<1	<0.5	<2	9	62	35.1	0.4	40
Meningie (site 1)	Couch	Stem	245	166	<1	<2	<1	<0.5	<2	8	39	22.0	1.5	29

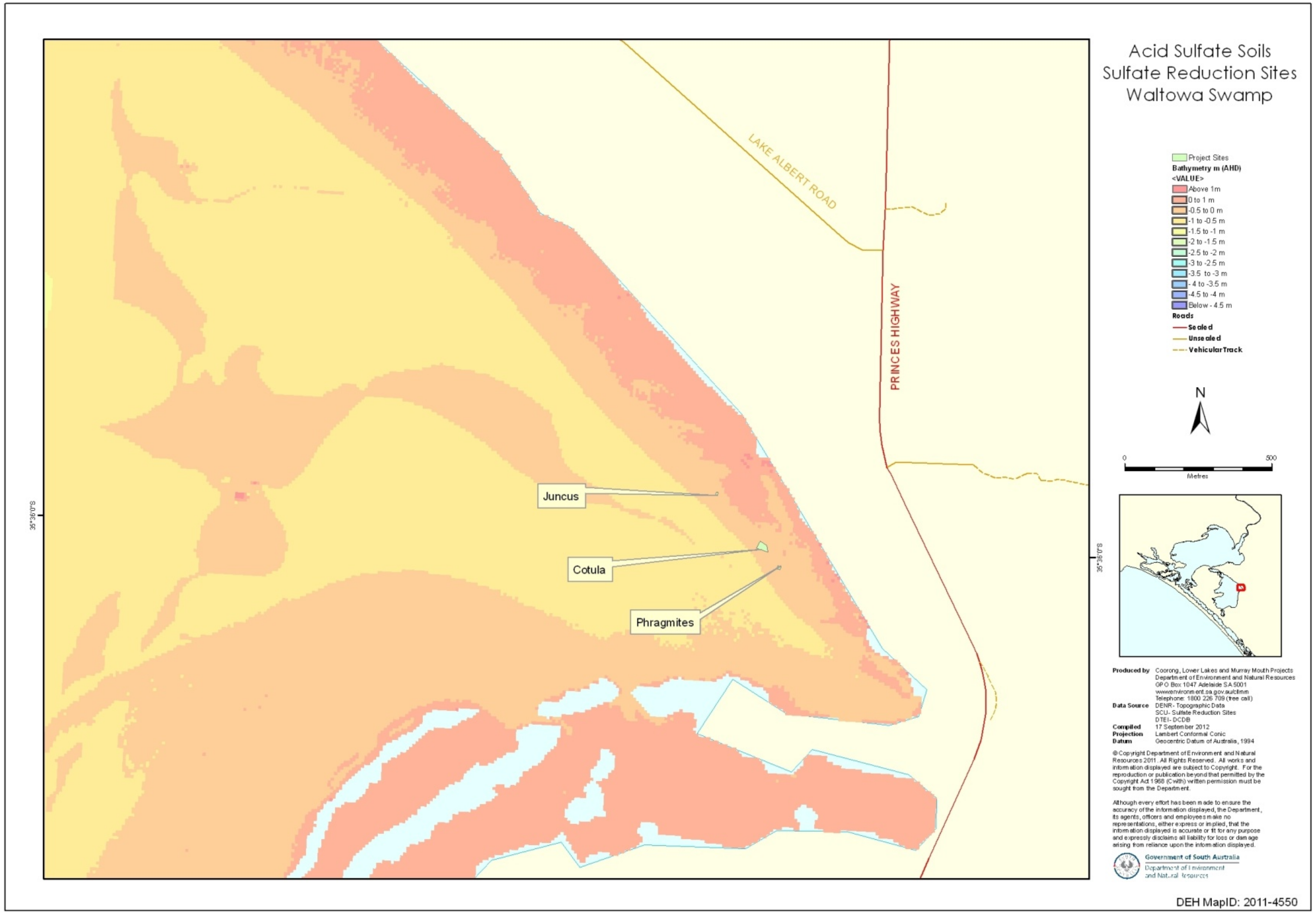
Table 9-23 (continued). Metal/metalloid content (mg/kg) of the vegetation (March 2013).

Site	Vegetation Type	Leaf/Stem/ Flower	Iron	Aluminium	Silver	Arsenic	Lead	Cadmium	Chromium	Copper	Manganese	Nickel	Selenium	Zinc
Meningie (site 1)	Phragmites	Leaf	368	328	<1	<2	<1	<0.5	<2	7	208	37.4	0.7	16
Meningie (site 1)	Phragmites	Stem	155	166	<1	<2	<1	<0.5	<2	4	85	41.3	0.6	7
Meningie (site 1)	Unidentified Reed	Stem	103	116	<1	<2	<1	<0.5	<2	2	176	6.4	0.6	4
Meningie (site 1)	Schoenoplectus (*50 m from shore*)	Stem	207	198	<1	<2	<1	<0.5	<2	4	137	26.9	4.4	6
Meningie (site 1)	Schoenoplectus (*shoreline*)	Leaf	260	212	<1	<2	<1	<0.5	9.0	5	1,885	574.4	1.2	14
Meningie (site 1)	Schoenoplectus (*shoreline*)	Stem	227	270	<1	<2	<1	<0.5	2.0	4	923	22.5	2.1	14
Meningie (site 1)	Typha (*shoreline*)	Flower	232	162	<1	<2	<1	<0.5	<2	7	77	31.9	3.4	32
Meningie (site 1)	Typha (*shoreline*)	Stem	207	219	<1	<2	<1	<0.5	<2	3	140	18.4	2.5	5
Meningie (site 2)	Phragmites	Flower	491	518	<1	<2	<1	<0.5	<2	5	144	20.2	0.9	44
Meningie (site 2)	Phragmites	Leaf	408	438	<1	<2	1.6	<0.5	<2	5	241	24.3	1.2	13
Meningie (site 2)	Phragmites	Stem	229	286	<1	<2	62.1	<0.5	<2	3	100	31.9	0.2	6
Meningie (site 2)	Sedge	Flower	1,245	706	<1	<2	<1	<0.5	10.5	9	270	79.9	0.6	25
Meningie (site 2)	Sedge	Leaf	667	474	<1	<2	<1	<0.5	<2	6	2,636	88.4	1.4	54
Meningie (site 2)	Sedge	Stem	546	457	<1	<2	<1	<0.5	<2	6	1,703	27.7	3.0	30
Milang	Phragmites	Leaf	607	665	<1	<2	<1	<0.5	<2	4	371	22.1	0.3	23
Milang	Phragmites	Stem	206	255	<1	<2	<1	<0.5	<2	2	79	23.6	<0.1	22
Naro Point	Phragmites	Leaf	931	1,083	<1	<2	<1	<0.5	<2	4	206	22.1	0.4	18
Naro Point	Phragmites	Stem	811	1,015	<1	<2	<1	<0.5	<2	3	69	20.4	0.6	24
Naro Point	Phragmites (*further in lake*)	Leaf	850	990	<1	<2	<1	<0.5	6.2	6	218	21.8	0.2	17
Naro Point	Phragmites (*further in lake*)	Stem	770	1,036	<1	<2	<1	<0.5	2.1	4	74	24.0	0.2	7
Naro Point	Schoenoplectus	Stem	505	727	<1	<2	<1	<0.5	<2	3	236	11.6	0.7	7
Naro Point	Typha	Leaf	363	333	<1	<2	<1	<0.5	<2	3	839	8.0	2.1	13
Naro Point	Typha	Stem	365	351	<1	<2	<1	<0.5	<2	2	398	7.1	2.9	10
Poltalloch	Phragmites	Leaf	375	320	<1	<2	<1	<0.5	<2	6	258	32.3	0.7	16
Poltalloch	Phragmites	Stem	303	327	<1	<2	118	<0.5	<2	4	86	41.8	0.2	6
Tolderol	Phragmites (*foredune*)	Leaf	341	136	<1	<2	<1	<0.5	<2	4	82	17.3	0.7	8
Tolderol	Phragmites (*foredune*)	Stem	110	94	<1	<2	<1	<0.5	<2	2	32	31.2	<0.1	3
Tolderol	Phragmites (*former scald*)	Leaf	1,299	1,152	<1	<2	<1	<0.5	7.9	8	100	49.8	0.8	17
Tolderol	Phragmites (*former scald*)	Stem	524	513	<1	<2	<1	<0.5	<2	3	40	26.8	<0.1	7

Table 9-23 (continued). Metal/metalloid content (mg/kg) of the vegetation (March 2013).

Site	Vegetation Type	Leaf/Stem/ Flower	Iron	Aluminium	Silver	Arsenic	Lead	Cadmium	Chromium	Copper	Manganese	Nickel	Selenium	Zinc
Waltowa	Bull Rush	Leaf	191	162	<1	<2	<1	<0.5	<2	6	465	29.7	1.8	18
Waltowa	Bull Rush	Stem	245	288	<1	<2	<1	<0.5	<2	4	240	24.5	1.5	10
Waltowa	Unidentified Juncus	Leaf	553	450	<1	<2	<1	<0.5	<2	5	59	37.1	2.2	16
Waltowa	Unidentified Juncus	Stem	256	267	<1	<2	<1	<0.5	<2	4	101	26.9	1.7	14
Waltowa	Phragmites	Leaf	455	412	<1	<2	<1	<0.5	<2	12	149	137.2	0.8	25
Waltowa	Phragmites	Stem	303	427	<1	<2	<1	<0.5	<2	13	67	135.3	0.1	12
Waltowa	Schoenoplectus	Stem	333	415	<1	<2	<1	<0.5	<2	4	275	29.9	1.4	5
Wellington Lodge	Phragmites	Leaf	241	223	<1	<2	<1	<0.5	<2	5	165	33.9	0.6	12
Wellington Lodge	Phragmites	Leaf	454	289	<1	<2	<1	<0.5	<2	3	110	14.3	0.3	14
Wellington Lodge	Phragmites	Leaf	487	432	<1	<2	<1	<0.5	<2	6	260	18.0	0.1	11
Wellington Lodge	Phragmites	Stem	464	475	<1	<2	<1	<0.5	<2	3	70	60.5	0.2	7
Wellington Lodge	Phragmites	Stem	412	489	<1	<2	<1	<0.5	<2	4	87	33.9	0.1	6
Wellington Lodge	Phragmites	Stem	204	241	<1	<2	<1	<0.5	<2	4	85	124.0	0.2	7
Wellington Lodge	Schoenoplectus	Flower	484	181	<1	<2	<1	<0.5	<2	7	69	5.6	2.3	29
Wellington Lodge	Schoenoplectus	Stem	306	329	<1	<2	<1	<0.5	<2	3	307	37.8	1.2	10
Wellington Lodge	Schoenoplectus	Stem	266	278	<1	<2	<1	<0.5	<2	3	254	9.9	0.7	7
Wellington Lodge	Schoenoplectus	Stem	451	246	<1	<2	<1	<0.5	<2	4	106	51.5	1.7	6

APPENDIX 6. Additional information



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SCU - Sulfate Reduction Sites
DTEI - DCDB
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Projection Lambert Conformal Conic
Datum Geocentric Datum of Australia, 1994

Government of South Australia
Department of Environment and Natural Resources

Figure 9-18. Bathymetry map for the Waltowa study area (Source: DEWNR).

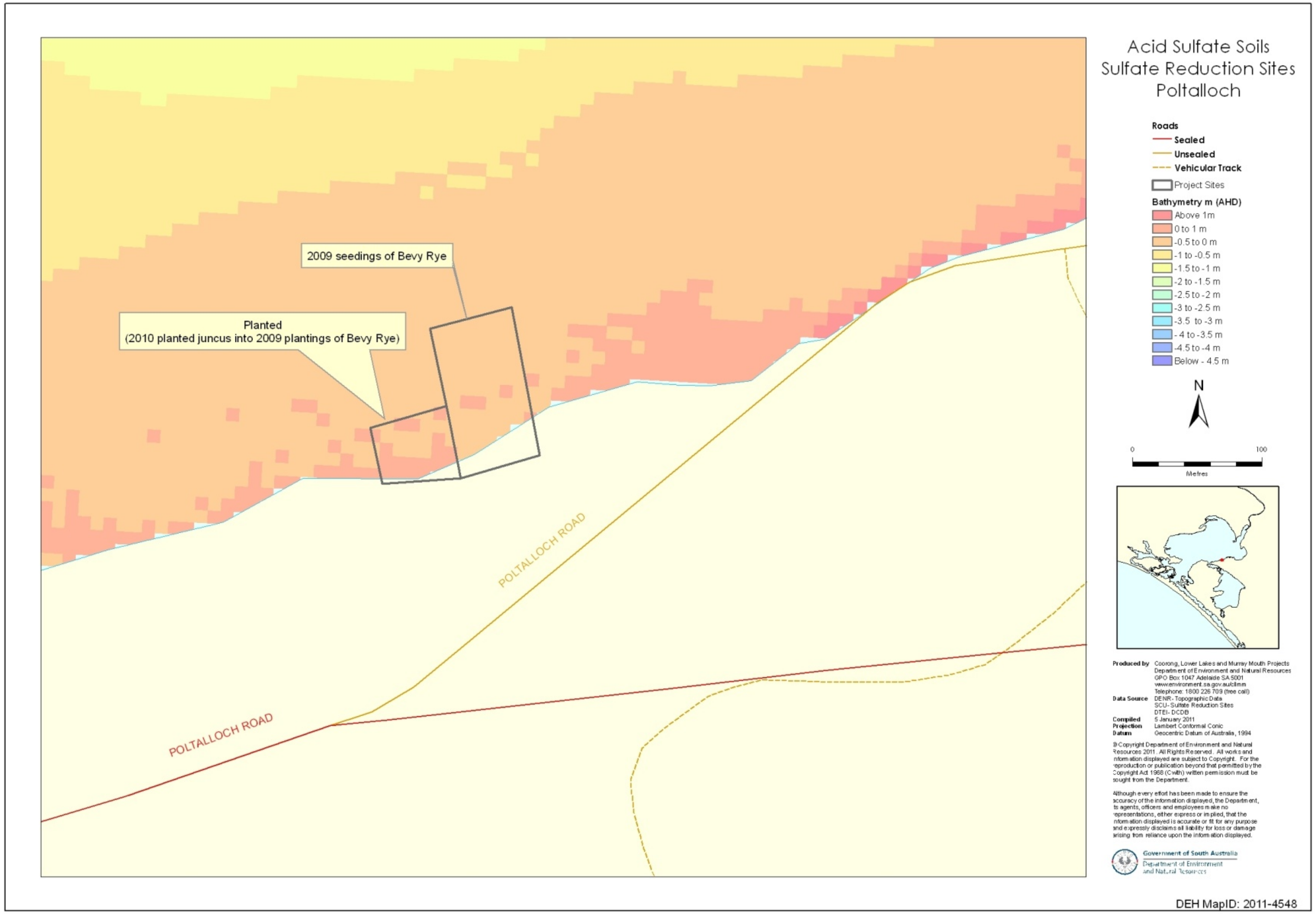

Figure 9-19. Bathymetry map for the Poltalloch study area (Source: DEWNR).

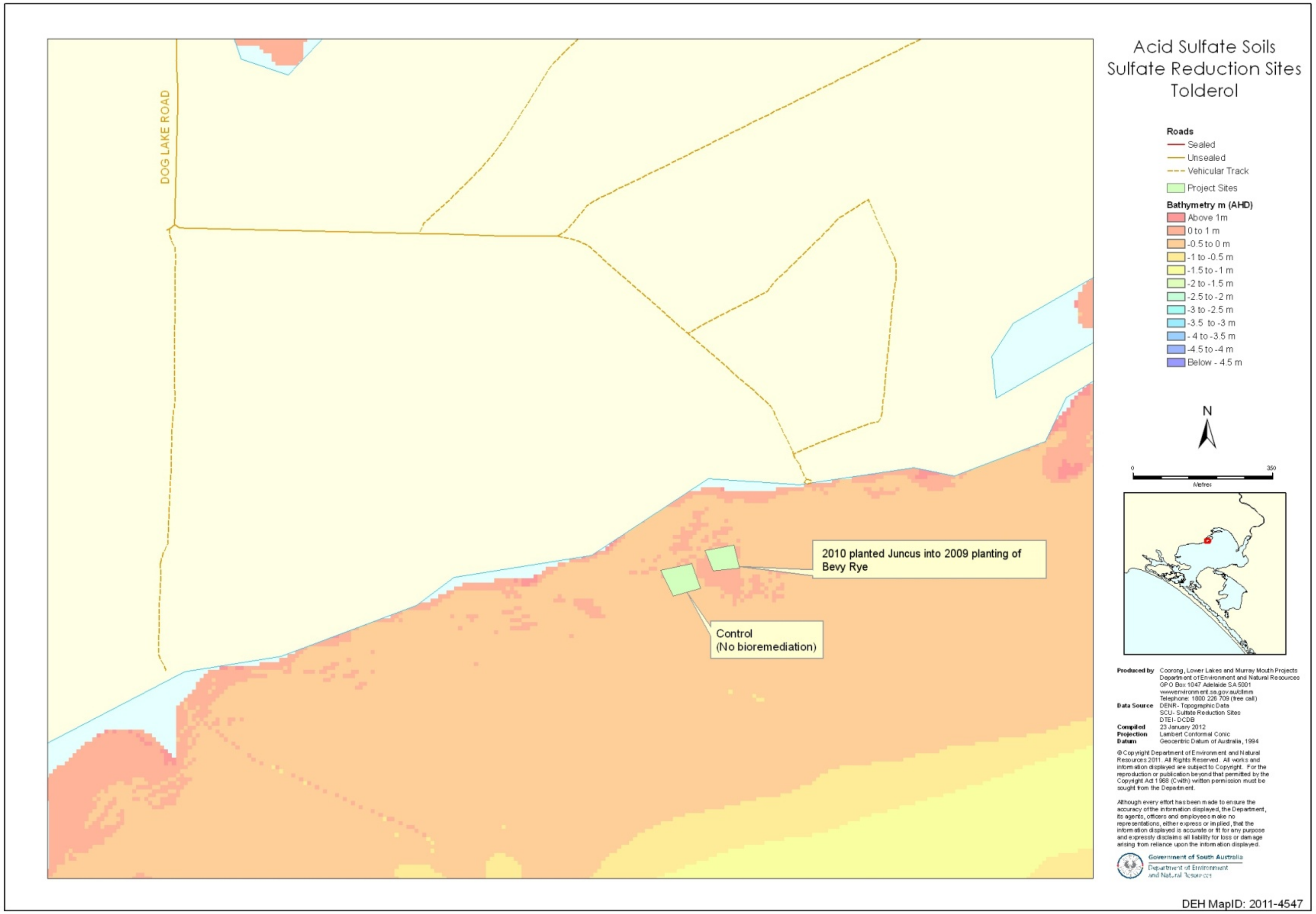

Figure 9-20. Bathymetry map for the Tolderol study area (Source: DEWNR).

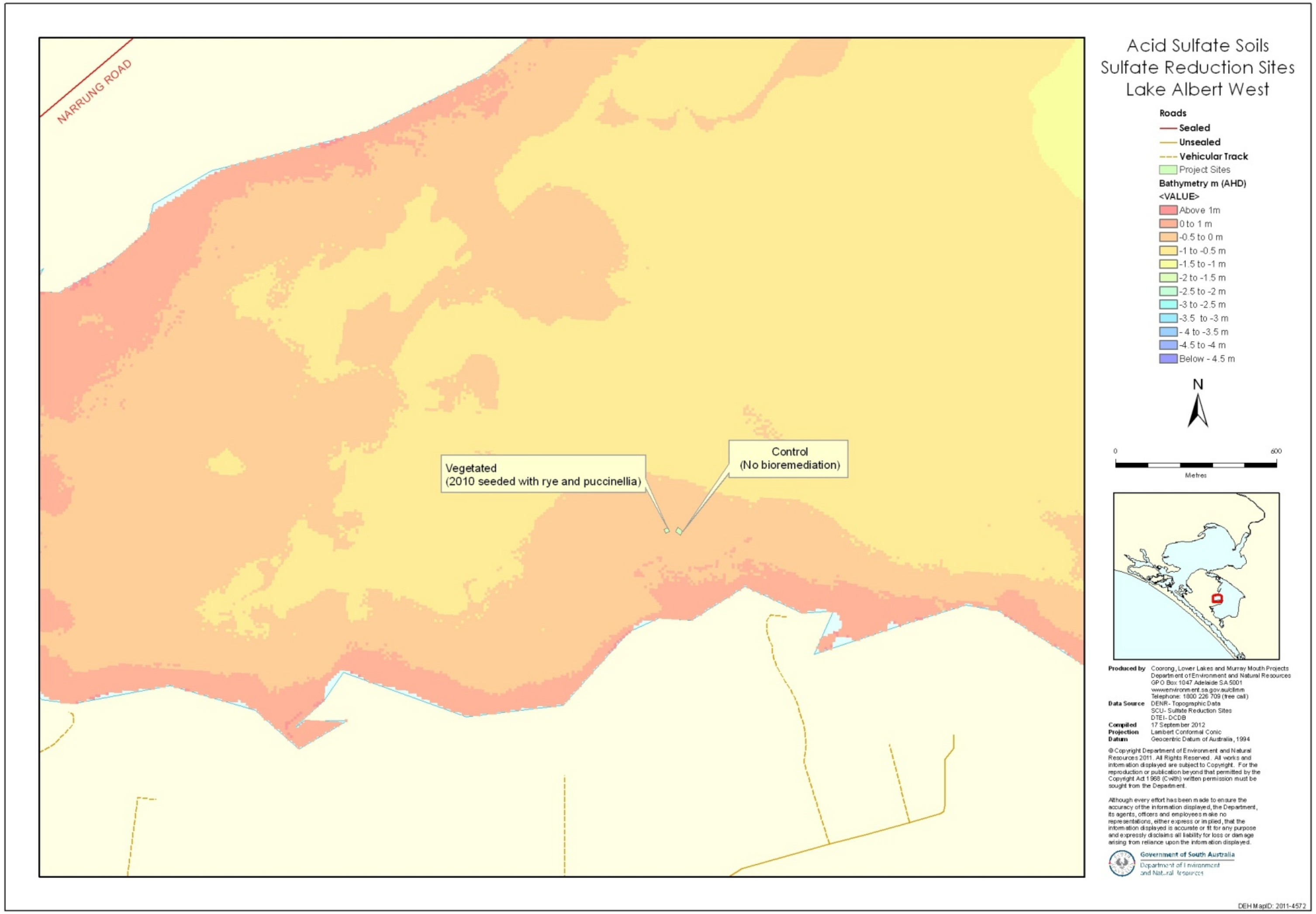

Figure 9-21. Bathymetry map for the Campbell Park study area (Source: DEWNR).

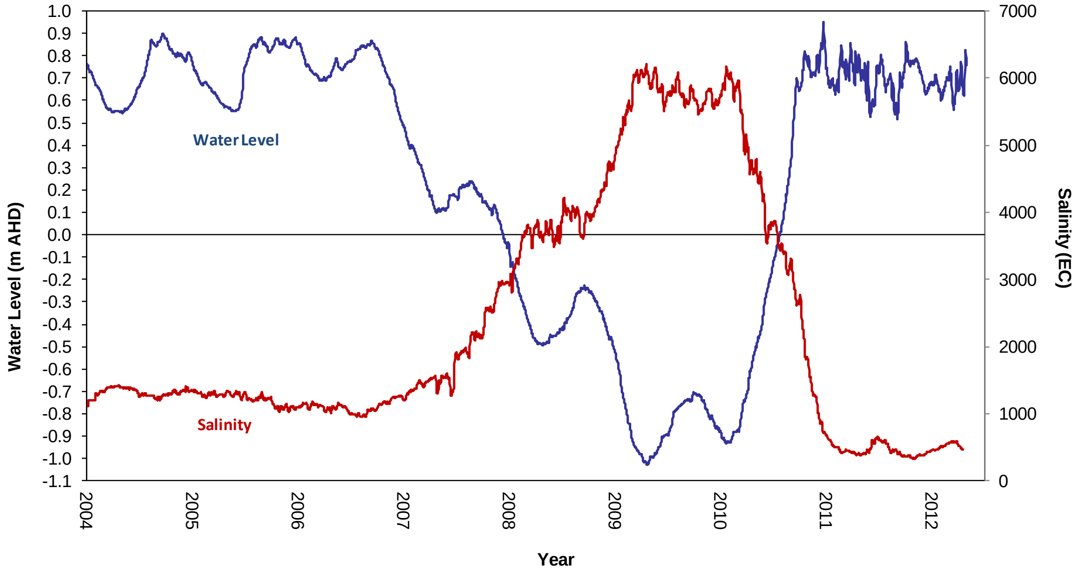

Figure 9-22. Lake Alexandrina historical water level and salinity data (Source: DEWNR).

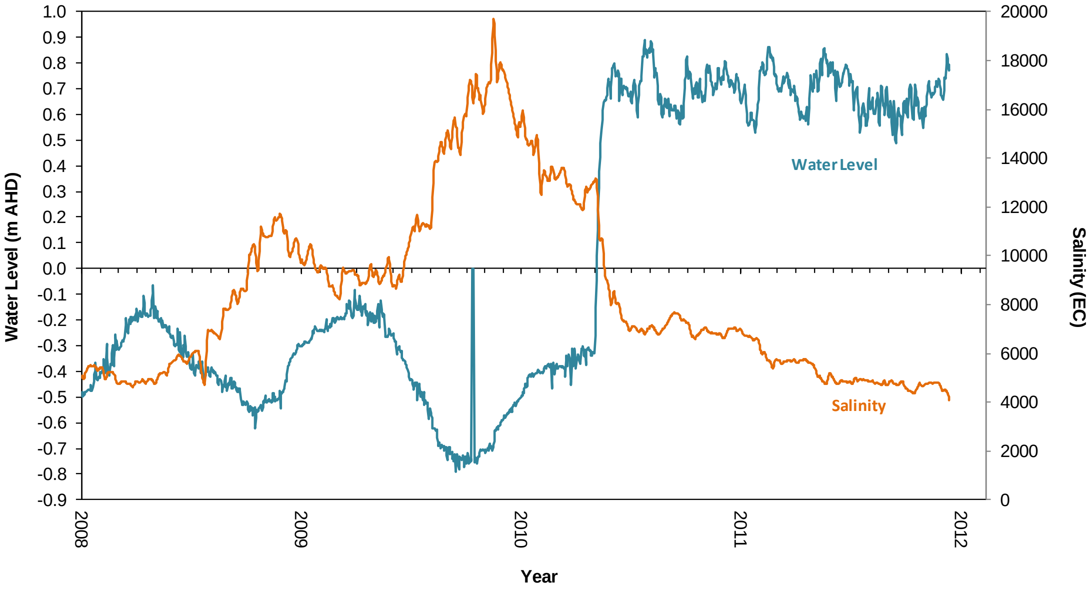

Figure 9-23. Lake Albert historical water level and salinity data (Source: DEWNR).

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