Lower Lakes Phase 1 Sulfate Reduction Monitoring Project

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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
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

This project focused on an ongoing assessment of bioremediation techniques, as the lakes re-filled, on sulfate reduction and associated processes in the acidified Lower Lakes' sediments that had been exposed during the drying event from 2007-2010. These assessments included examination of possible changes in acidity/alkalinity, sulfide contents and metal mobility consequent of these processes. In particular, Sullivan *et al.* (2011) examined these processes for the initial (i.e. up to 6 months) lake re-filling phase. This study complements this earlier study by providing an examination of these processes in the lake sediments at 19 months after lake re-filling. It should be noted that this is still considered to be in the early lake restoration phase.

The locations in the Lower Lakes (Waltowa, Poltalloch, Tolderol and Campbell Park) selected for this study each had a range of revegetation treatments (in terms of both the vegetation species and timing of plantings), as well as unvegetated control sites.

This report confirms many of the findings of Sullivan *et al.* (2011) that bioremediation of the exposed acidified lake sediments by revegetation produced substantial benefits in terms of reduced acidity of the surficial lake sediments due to the effects of vegetation. These benefits are likely to have accrued from a combination of vegetation associated processes including the provision of alkalinity from plant roots, the provision of alkalinity indirectly from sulfate reductive processes enabled by the provision of organic matter from the bioremediating vegetation, as well as from the vegetation minimising soil erosion and hence preventing the exposure of severely acidic subsoils that often occurred in unvegetated sites.

The possible hazards associated with a strategy of enhancing organic matter input into sediments to stimulate, post lake re-filling, sulfate reduction and the production of alkalinity appear to have been substantially avoided in the Lower Lakes wherever annual vegetation was too short to survive inundation. In the surficial lake sediments at these study areas there was a lack of accumulation of sulfide minerals (such as monosulfides and pyrite) and their associated hazards of acidification, metal and metalloid mobilisation, and deoxygenation.

However, when *Phragmites* – a species that survived lake re-filling and continues to grow vigorously when inundated - was used to bioremediate these sediments, the data in this study show considerable accumulation of both pyrite and monosulfide (as Monosulfidic Black Ooze (MBO)) in the uppermost sediment layers. These accumulated sulfides indicate that alkalinity has also been produced via sulfate reducing processes enabled by the ongoing production of organic matter by *Phragmites.* In addition, these uppermost sediments under *Phragmites* appear likely to act as sources of soluble phosphate to the overlying lake waters. This study strongly indicates a number of potentially important hazards would have arisen if *Phragmites* were to be used for bioremediation of exposed lake sediments: such hazards were avoided almost completely when inundation intolerant vegetation was used.

Of course, this study has only examined the early stages after refilling and further biogeochemical studies of the sediments are required in the near future to assess adequately the ongoing impact of lake refilling on the behavior of the formerly exposed and acidified lake sediments, especially in relation to the longer term accumulation of sulfide minerals, the production and mobility of metals (especially nickel and zinc) and nutrients as these sediments de-acidify and reduce further during inundation, and the effect of revegetation on these processes.

The key findings of this study are:

1) Considerable sulfate reduction was occurring during the March 2012 assessment only in the surface sediment layers where organic matter is continuing to be provided when the vegetation used for bioremediation are species that survived lake re-filling (i.e. *Phragmites*). There were clear on-going differences in the effectiveness of the bioremediation vegetation in driving this process. The annual plants and short perennial plants (relative to the inundation depth) produce appreciable amounts of organic matter but then die, however the tall perennial plants that survived inundation can continue to produce organic matter. This is important because the patterns of organic matter accumulation and production dictate the consequent patterns of sulfate reduction. Importantly, *Phragmites*, which successfully resisted prolonged inundation for at least 19 months when inundated beneath lake waters at least 1 metre in depth, is clearly continuing to supply organic matter to sediments long after inundation and hence is continuing to strongly drive sulfate reduction processes resulting in the accumulation of sulfides.

2) The March 2012 assessment clearly shows that the accumulation of appreciable quantities of pyrite (and hence the development of such a considerable potential sulfidic acidity hazard) was not observed, and given the lack of an organic matter supply, is unlikely to occur, when vegetation used for bioremediation is inundation intolerant and undergoes death during inundation.

   However, appreciable quantities of reduced inorganic sulfides (especially pyrite and monosulfides) were accumulating in surface sediment layers under the *Phragmites* treatment. As well as representing an appreciable amount of alkalinity produced in these sediments from sulfate reduction processes, this store of pyrite also represents an appreciable and likely growing potential sulfidic acidity hazard in the surface lake sediments under this bioremediation treatment. Similarly, the store of monosulfidic materials (i.e. Monosulfidic Black Oozes (MBOs)) under the *Phragmites* treatment also represents the development of associated acidification, metal mobilisation and deoxygenation hazards under this bioremediation treatment. Given their location in the surface layers of sediments when an inundation tolerant bioremediation species, in this case *Phragmites,* was used as for bioremediation, this potential sulfidic acidity hazard would be realised much earlier than would previously have been the case, should the Lower Lakes experience atmospheric exposure as was the case in the last drought.

3) Both the acidity and low pHs of the acidified acid sulfate sediment layers are continuing to be remediated by a number of processes and sources some consequent of the bioremediation, some not.

   The two main factors effecting on-going acidity remediation are:
   - the movement into the sediment of the alkalinity that is contained in the lake waters and;
   - the vegetation established during bioremediation when inundation tolerant (i.e. *Phragmites*) adding alkalinity indirectly to the soil via provision of organic matter and thus enabling sulfate reduction resulting in the accumulation of reduced inorganic sulfides (especially pyrite and monosulfides).

4) The data indicate appreciable increases both in ferrous iron (Fe^{2+}) concentrations in pore-waters and in the HCl-extractable zinc (Zn) concentrations in the sediments during the study period beneath both control and bioremediated sites.

5) The data indicate that, apart from under the *Phragmites*, there were few general trends in nutrient availability consequent of bioremediation at the March 2012 assessment. However, two strong trends in nutrient mobility were observed under the *Phragmites* with large decreases in ammonia concentrations in the pore-waters of the deeper sediment layers and greatly increased phosphate concentrations in the pore-waters of the surface sediments. It is likely that these sediments under *Phragmites* may be a source of soluble phosphate to the overlying lake waters. This could pose a risk to lake water quality but further information would be required to scale the hazard.

Recommendations

1) We recommend that future monitoring of the effects of bioremediation on the geochemistry of the lake sediments, by assessment programs similar to that used in this project, be undertaken to fully assess the possible effects in both the medium and long term of the various bioremediation techniques on the lake ecosystem.

2) We recommend that future monitoring of the pore-water nickel and zinc in the lake sediments as affected by bioremediation be undertaken to assess ongoing environmental risks posed by the presence of very high bio-accessible concentrations of these potentially-toxic trace metals.

3) We recommend that future monitoring of nutrients in the lake sediments as affected by bioremediation be undertaken to assess the ongoing environmental risks posed by the presence of an enhanced source of phosphate to the overlying lake waters provided by bioremediation using *Phragmites.*

4) The results of this study strongly indicate the need for a further detailed study on both:
   i. the effectiveness of the different vegetation types (especially differences between different annual vegetation species) and strategies used for bioremediation, and
   ii. the unbioremediated 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 collaborative studies of the sediments of the Lower Lakes and of the effects of bioremediation with the South Australian Environmental Protection Authority (EPA) and Department of Environment and Natural Resources (DENR) (Sullivan *et al*. 2010a, 2011) have highlighted the hazard of acid sulfate soils and their potential to impact on ecological processes. 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.

The most recent of these studies (Sullivan *et al.* 2011) examined several key locations around the Lower Lakes 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 this study 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 that occurred under unvegetated sites.

At the same time, the study by Sullivan *et al*. (2011) 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 6 months). 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.

It was recognised in this study by Sullivan *et al*. (2011) that 6 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) study 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 continue to be avoided.

The methodology followed in this study continues the general assessment and analytical strategy used in Sullivan *et al.* (2011). Following this methodology allows maximum benefit in terms of assessing temporal trends by 'building onto' the existing knowledge of the biogeochemistry of these sediments. One deviation from the methodology of Sullivan *et al.* (2011) is that the sampling and analysis of sediment inundated in the laboratory post sampling was not required given that the lakes have refilled.

Accordingly the 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 18 months 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 it allows ready quantification of the acidity hazard, necessary for the rational determination of liming rates and for verification of management practices (Ahern *et al.* 2004).

3.1.5 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:

$$CH_3COO^- + SO_4^{2-} + H^+ \rightarrow H_2S + 2HCO_3^- \quad [3.1]$$

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. 3.1. 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:

$$H_2S + Fe^{2+} \rightarrow FeS + 2H^+ \quad [3.2]$$

The initial FeS phase to form by reaction between H_2S and Fe^{2+} (Eq. 3.2) 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:

$$H_2S + 2FeOOH + 2H^+ \rightarrow S_8 + 2Fe^{2+} + 3H_2O \quad [3.3]$$

The Fe^{2+} produced via this reaction may then feed into the reaction described by Eq. 3.2 thus also resulting in mackinawite formation. This overall process, termed "sulfidisation" can be represented as:

$$3H_2S + 2FeOOH \rightarrow S_8 + FeS + 4H_2O \quad [3.4]$$

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

$$4FeS + 0.5O_2 + 2H^+ \rightarrow Fe_3S_4 + Fe^{2+} + H_2O \quad [3.5]$$

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:

$$Fe_3S_4 + 2H^+ \rightarrow FeS_2 + Fe^{2+} + H_2 \quad [3.6]$$

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: $FeS + S_n^{2-} \rightarrow FeS_2 + S_{n-1}^{2-}$ [3.7]

Hydrogen sulfide pathway: $FeS + H_2S \rightarrow FeS_2 + H_2$ [3.8]

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 – *frambois*). 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).

3.1.6 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:

$$FeS_2 + 3.75O_2 + 3.5H_2O \rightarrow Fe(OH)_3 + 4H^+ + 2SO_4^{2-} \quad [3.9]$$

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 (an 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).

3.1.7 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.

3.1.8 Hazards from acid sulfate soils

3.1.8.1 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.

3.1.8.2 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 elsewhere in this report and 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.

3.1.8.3 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 soil occurs in low-lying floodplain environments and therefore, is 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.

3.1.8.4 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 estuarine 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 estuarine 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 $$FeS \begin{cases} Fe^{2+} + 0.5O_2 + 1.5\,H_2O \rightarrow 2H^+ + FeOOH \\ S^{2-} + 0.5O_2 + 2H^+ \rightarrow H_2O + 0.125S_8 \end{cases} \quad [3.10]$$

Step 2 $$0.125S_8 + 1.5\,O_2 + H_2O \rightarrow SO_4^{2-} + 2H^+ \quad [3.11]$$

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). However, the extent and monosulfide concentration of MBOs in the Lower Lakes sediments has not yet been systematically examined. 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.

3.1.8.5 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 3.1, 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.

3.1.8.6 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. Of particular interest are the simpler interventions such as ridging and furrowing. This remediation involves the forming of ridges and furrows using cultivation, and especially when combined with a mulch layer (e.g. straw), has proven very effective in facilitating the establishment of vegetation. Ridges and furrows establish different micro-habitats, with the water-tolerant species occupying the wetter furrows (Rosicky *et al.* 2006). A similar approach for food crop production on acid sulfate soils has been used by farmers in South-East Asia for decades (Dent 1986).

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 fresh water 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).

3.1.9 Inundation of acid sulfate soils

Inundation with freshwater 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), Sullivan *et al.* (2010a) confirmed that the availability of organic carbon in the Lower Lakes environment was a limiting factor, which supported the approach undertaken by the South Australian government. The bioremediation of Lower Lakes sites via enhancing organic carbon availability was supported as feasible management option.

Sullivan *et al*. (2011) examined the effects of various bioremediation options carried out by the Department for Water, Environment and Natural Resources 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 this study 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 from plant roots, provision of organic carbon for sulfate reducing bacteria and the role of vegetation minimising soil erosion and hence preventing further exposure of severely acidic subsoils that occurred under unvegetated sites.

At the same time, the study by Sullivan *et al*. (2011) 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 6 months). 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.

It is recognized (e.g. Sullivan *et al*. 2011; Moreno-Mateos *et al.* 2012) that 6 months of re-inundation is too short a time to adequately assess the longer term on the biogeochemistry. This project aimed to monitor the biogeochemical state (with respect to sulfate reduction and associated processes) of the Lower Lake sediments approximately 18 months after lake refilling: more than 12 months after the last detailed monitoring of these sediments (Sullivan *et al.* 2011). This project builds on the results of the Sullivan *et al*. (2011) study 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 continue to be avoided.

The methodology followed in this study continues the general assessment and analytical strategy used in Sullivan *et al.* (2011). Following this methodology allows maximum benefit in terms of assessing temporal trends by 'building onto' the existing knowledge of the biogeochemistry of these sediments. One deviation from the methodology of Sullivan *et al.* (2011) is that the sampling and analysis of sediment cores inundated in the laboratory post sampling was not required given that the lakes have refilled.

3.3 Sampling strategy

The sampling strategy undertaken in both the previous study by Sullivan *et al.* (2011) and this study addresses contemporary conditions in the lakes and assesses sulfate reduction and alkalinity generation in the subsurface sediments arising from leaching of soluble organic matter - derived from bioremediation - into the subsoil.

In this study sediments were collected from the same four study areas as sampled by Sullivan *et al.* (2011). 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 previous study by Sullivan *et al.* (2011) a total of nine treatment sites were examined in detail between May 2010 and February 2011. In this study eight of these sites were re-examined in March 2012. Only one site at the Poltalloch study area was sampled in this study as this treatment site was

essentially duplicated in the earlier study. A summary of the treatments examined in this study are presented in Table 3-1.

Table 3-1. Summary of the treatments examined at each study area in the Lower Lakes (March 2012).

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

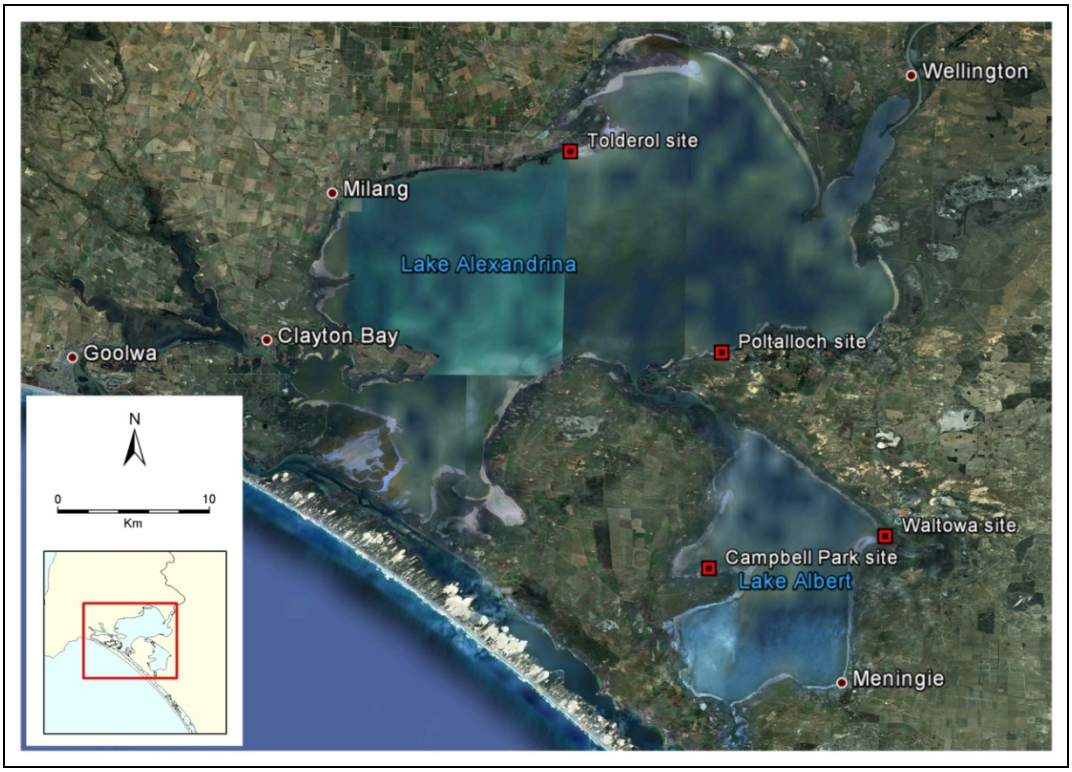

Figure 3-1. Map showing study areas in the Lower Lakes (Source: Google Maps).

3.4 Lower Lakes site locations and characteristics

Maps showing the sampling locations in each study area and selected photographs are presented in Sections 3.4.1 to 3.4.4. Bathymetry maps of each study area are also presented in Appendix 7 (Figures 9-113 to 9-116). Historical water level and salinity data for Lake Alexandrina and Lake Albert is also included in Appendix 7 (Figures 9-117 and 9-118).

3.4.1 Waltowa, east Lake Albert study area characteristics

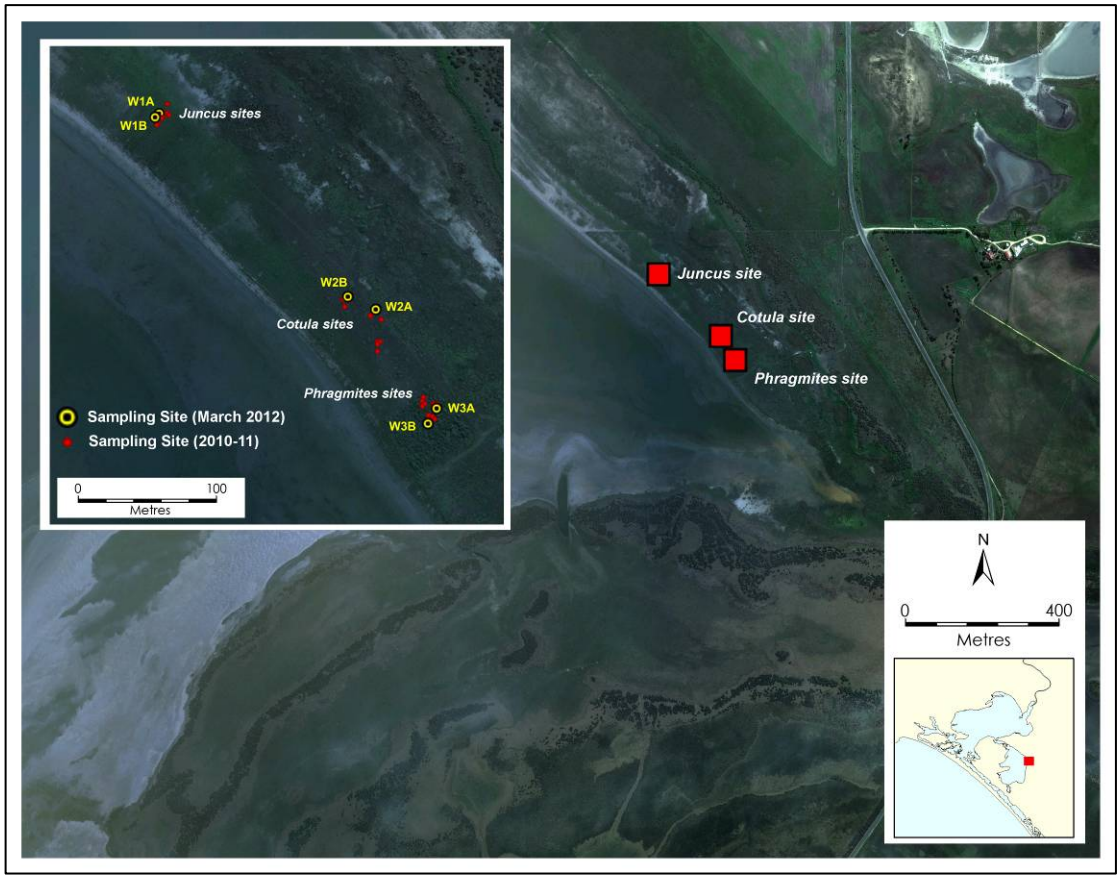

Figure 3-2. Waltowa sampling locations (Source: Google Maps).

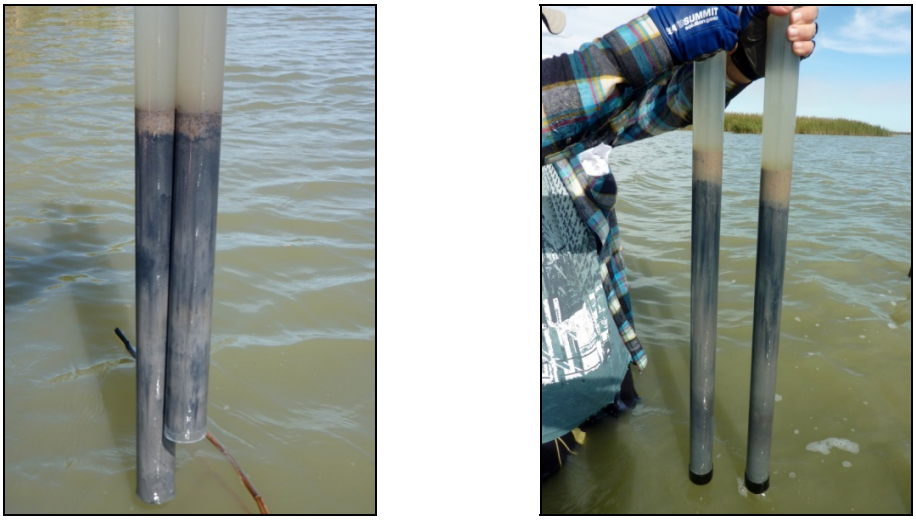

Figure 3-3. Sediment cores collected from the *Phragmites* site (left) (No MBO accumulation at this replicate site) and *Cotula* site (right) at Waltowa in March 2012. Profile descriptions are presented in Appendix 1.

Figure 3-4. Sediment cores collected from the *Juncus* site at Waltowa in March 2012. Profile descriptions are presented in Appendix 1.

Figure 3-5. Comparison of the sediment cores collected from the *Phragmites* site (left core) with MBO accumulation and *Cotula* site (right core). Profile descriptions are presented in Appendix 1.

3.4.2 Poltalloch, east Lake Alexandrina study area characteristics

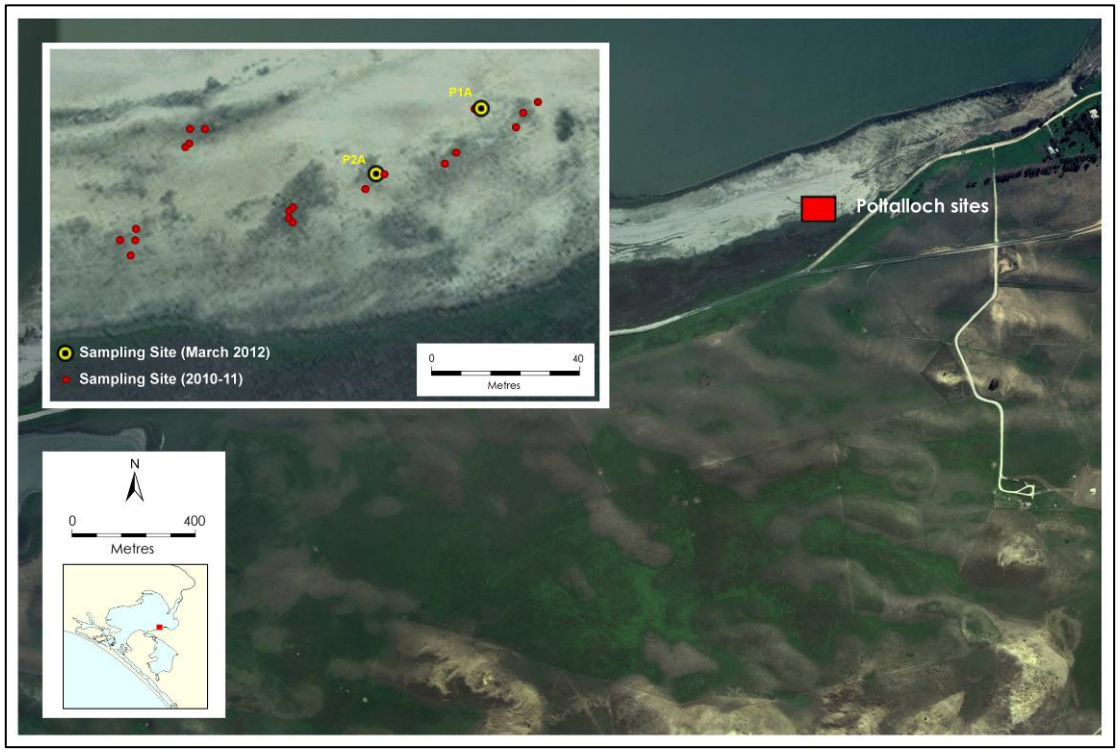

Figure 3-6. Poltalloch sampling locations (Source: Google Maps).

3.4.3 Tolderol, west Lake Alexandrina study area characteristics

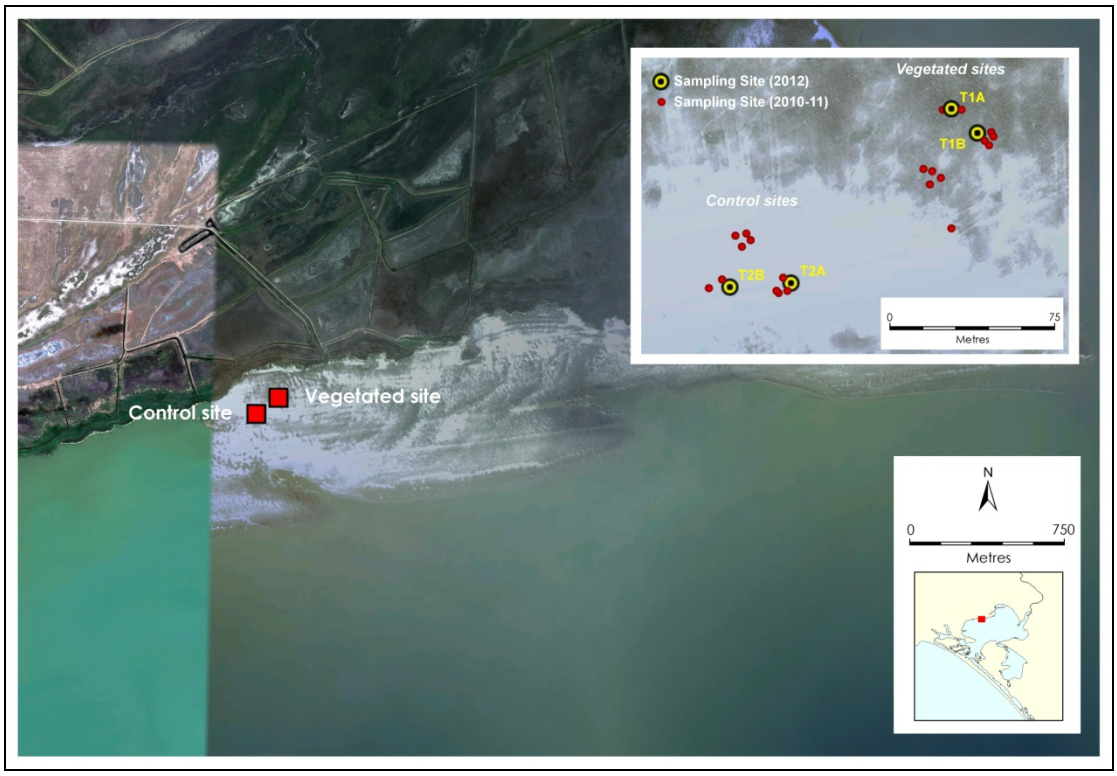

Figure 3-7. Tolderol sampling locations (Source: Google Maps).

Figure 3-8. Sampling at Tolderol in March 2012.

Figure 3-9. Sediment core collected from the scald site (left) and iron segregation (right) at Tolderol in March 2012. Profile descriptions are presented in Appendix 1.

Figure 3-10. Iron segregations (left) and iron/jarosite (right) in the sediment core collected from the scald site at Tolderol in March 2012. Profile descriptions are presented in Appendix 1.

Figure 3-11. Sediment cores collected from the vegetated (*Juncus* in Bevy rye) site at Tolderol in March 2012. Profile descriptions are presented in Appendix 1.

Figure 3-12. Comparison of the sediment cores collected from the vegetated (left core) and scald (right core) sites. Profile descriptions are presented in Appendix 1.

3.4.4 Campbell Park, west Lake Albert study area characteristics

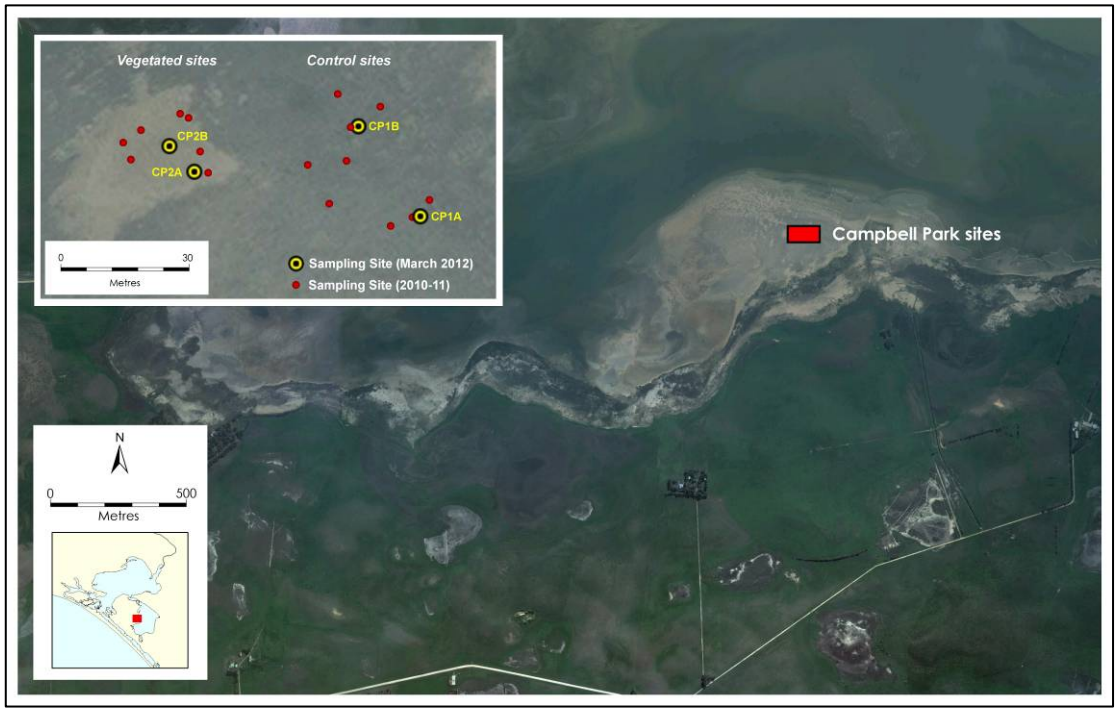

Figure 3-13. Campbell Park sampling locations (Source: Google Maps).

4.0 Materials and methods

The methodology followed in this study continues the general assessment and analytical strategy used in Sullivan *et al.* (2011). Following this methodology allows maximum benefit in terms of assessing temporal trends by 'building onto' the existing knowledge of the biogeochemistry of these sediments. One deviation from the methodology of Sullivan *et al.* (2011) is that the sampling and analysis of sediment cores inundated in the laboratory post sampling was not required given that the lakes have refilled.

4.1 Field sampling of soils

Field sampling at the four Lower Lakes study areas was undertaken between 29th and 31st March 2012. In the previous study by Sullivan *et al.* (2011), field sampling at the same study areas was undertaken before seeding/planting in May 2010, and then undertaken on three separate occasions (i.e. August 2010, November 2010 and February 2011). A summary of the sampling dates for this and the previous study are presented below in Table 4-1.

Table 4-1. Sampling dates for the field sulfate rate assessment and soil profile sampling (May 2010 – March 2012).

Season (Date)	Field Sulfate 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)	✓	✓

The sampling dates in the earlier 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 duplicate intact sediment cores were collected using a 5 cm diameter push-tube coring device from two replicate sampling sites from each treatment/location to a depth of 40 cm. Each core was collected within 4 m of the initial site sampled in the previous study (Sullivan *et al.* 2011) to ensure that the detection of any changes in soil properties since the last sampling time was optimised. A surficial monosulfidic black ooze (MBO) was observed 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).

Figure 4-1. Sediment sampling at Tolderol (March 2012).

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
- 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
- Sulfate reduction rates

The sediment moisture content was determined by weight loss due to drying at 105°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). The methodology followed in the determination of the sulfate reduction rates is summarised in Section 4.2.3.

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 Sulfate reduction analyses

In-situ SO_4^{2-}-reduction rates (SRR) were determined following the same methodology previously used in Sullivan *et al.* (2011). Sulfate reduction rates were determined using a radiotracer ($^{35}SO_4^{2-}$) incubation approach (Fossing and Jørgensen 1989) in which short-term products of sulfate reduction (i.e. iron-monosulfides, elemental sulfur and pyrite) were also investigated. Sediment profiles were collected using a 5 cm diameter push-tube coring device. The rate of sulfate reduction was measured at the surface in 2.5 cm increments (i.e. 0-2.5 cm, 2.5-5.0 cm), then in 5 cm increments to 20 cm, and 10 cm increments from 20 cm to 40 cm. Four replicate soil sub-samples for each soil layer were collected using 3 mL polypropylene syringes (with the distal end removed). After collection, each soil sample was immediately sealed within the 3 mL syringe using Parafilm and was subsequently injected with 100 kBq of carrier-free $^{35}SO_4^{2-}$. Three replicates from each depth interval were incubated at ambient temperature for 24 hours. These incubations were terminated by freezing the sealed syringes. In addition to the triplicate 24 hour incubations, a single replicate for selected soil samples also served as a time zero blank (i.e. this sample was frozen immediately after injection of $^{35}SO_4^{2-}$).

The RIS speciation of the radiolabelled samples was determined by selective, sequential extraction of iron-monosulfides, elemental sulfur (S(0)) and pyrite (Burton *et al.* 2007, 2009). Iron-monosulfides, defined operationally as AVS, were extracted by shaking (150 rpm) ~ 0.5 grams of sediment with 10 mL of 6.0 M HCl/0.1 M ascorbic acid in gas-tight 55 cm^3 polypropylene reactors for 18 hours (Burton *et al.* 2007). The use of ascorbic acid during this extraction prevents interferences from Fe(III) minerals, which can otherwise lead to S(0) formation (Hsieh *et al.* 2002). The evolved hydrogen sulfide ($H_2S_{(g)}$) was trapped in 7 mL of 3% zinc acetate in 2.0 M sodium hydroxide (NaOH), and subsequently quantified via iodometric titration. Elemental S was then extracted from the AVS-extracted sample by shaking the sediment with 10 mL of methanol for 16 hours. An aliquot of the methanol extract was analysed for S(0) by HPLC with a Dionex UltiMate 3000 system. Residual S(0) was then removed from the sediment sample by three rinses with 25 mL of acetone, and a final rinse with 20 mL of ethanol. Each rinse involved 5 to 10 minutes of shaking, with the sediment and acetone/ethanol phases separated between rinses by centrifugation at 4000 rpm for 10 minutes. Pyritic sulfur in the residual AVS- and S(0)-extracted sediment was then quantified as chromium reducible sulfur (CRS) using the method of Burton *et al*. (2008b).

The incorporation of ^{35}S into each of the three RIS fractions was determined by liquid-scintillation counting using a Perkin-Elmer microbeta counter (with Perkin-Elmer UltimaGold scintillation fluid). The SO_4^{2-}-reduction rate was determined by the sum of ^{35}S incorporated into AVS, S(0) and CRS according to:

$$SRR = \frac{a-b}{A}\left[SO_4^{2-}\right]\frac{1}{d} \bullet 1.06 \text{ nmol/g/day}$$

Where *a* is the radioactivity of the individual RIS species per mass of soil subjected to the incubation, *b* is the mean radioactivity of the time zero blanks, *A* is the radioactivity of the added $^{35}SO_4^{2-}$ per mass of soil, $[SO_4^{2-}]$ is the sulfate concentration per mass of soil (nmol/g), *d* is the incubation time in days, and 1.06 is the isotopic fractionation factor.

It is important to note that using ^{35}S incubations to quantify the importance of short-term sulfate reduction biomineralisation products is complicated by possible isotopic exchange of ^{35}S amongst separate RIS species. More specifically, absolute quantitative distinction between the in-situ formation rates of S(0) and AVS may be unreliable due to partial isotopic exchange of ^{35}S (Fossing *et al.* 1992). On the other hand, it is well established that isotopic exchange of ^{35}S does not occur between pyrite and other RIS species over a 24 hour period (Fossing *et al.* 1992). Therefore, differential incorporation of ^{35}S into the AVS versus S(0) pools must be interpreted cautiously, whereas ^{35}S incorporation into CRS can be soundly interpreted as real short-term pyrite formation (Burton *et al.* 2011).

All sulfate reduction data are presented in Appendix 3 (Tables 9-10 to 9-25).

4.2.4 Pore-water analyses

Pore-waters were extracted after centrifuging the soil samples at 3,500 rpm for 15 minutes. The parameters measured on the pore-water collected included:

- Redox potential (Eh)
- pH
- Electrical conductivity (EC)
- Alkalinity
- Dissolved sulfide
- Total dissolved iron (Fe^{3+} + Fe^{2+})
- Soluble chloride and sulfate
- Soluble cations (Ca, Mg, Na, K)
- Nutrients (orthophosphate, nitrate, nitrite, and ammonia)

Redox potential, pH and electrical conductivity were immediately measured on unfiltered pore-water 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.

Alkalinity, dissolved sulfide and total iron (Fe^{2+} + Fe^{3+}) were fixed immediately after sampling. The total iron trap was made up from a phenanthroline solution with an ammonium acetate buffer and 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 alkalinity, dissolved sulfide and iron concentrations 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).

All pore-water data are presented in Appendix 4 (Tables 9-26 to 9-33) and Appendix 5 (Figures 9-1 to 9-64).

4.2.5 Expression of results

The means (Av.) and the standard deviations for triplicates (± SD) 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 ±10% for all analyses.

5.0 Results

5.1 General sediment condition

5.1.1 Waltowa

5.1.1.1 $pH_{(1:1, soil:water)}$ and TAA

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 had a pH of ~4 (Figures 5-1 – 5-3). Upon near lake filling in August 2010 the pHs of these soil layers dropped considerably probably due to acidity exchange from the soil from 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 the pH of the sediment had increased by ~ 0.5 of a unit since last sampled in February 2011.

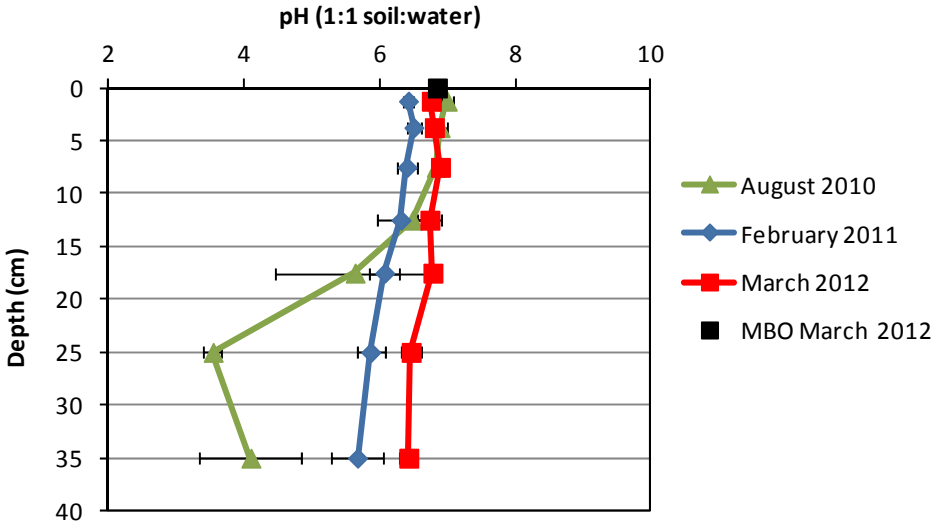

Figure 5-1. Waltowa field pH dynamics at the established *Phragmites* site (August 2010, February 2011 and March 2012).

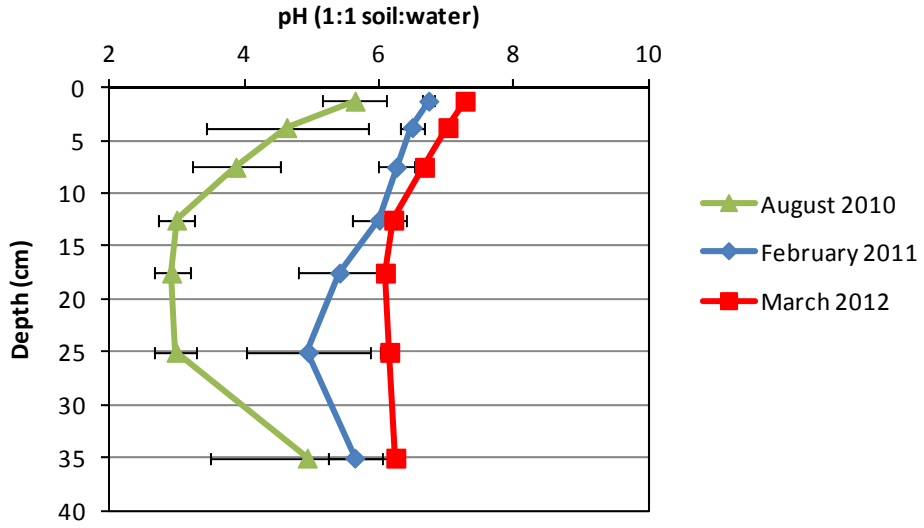

Figure 5-2. Waltowa field pH dynamics at the established *Cotula* site (August 2010, February 2011 and March 2012).

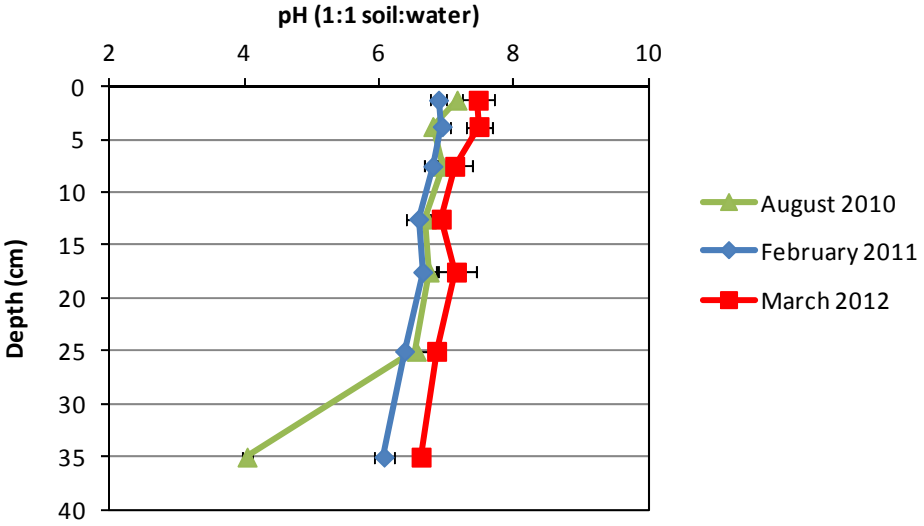

Figure 5-3. Waltowa field pH dynamics at the established *Juncus* site (August 2010, February 2011 and March 2012).

The TAAs (Figures 5-4 – 5-6) 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 decreased further since last sampled in February 2011.

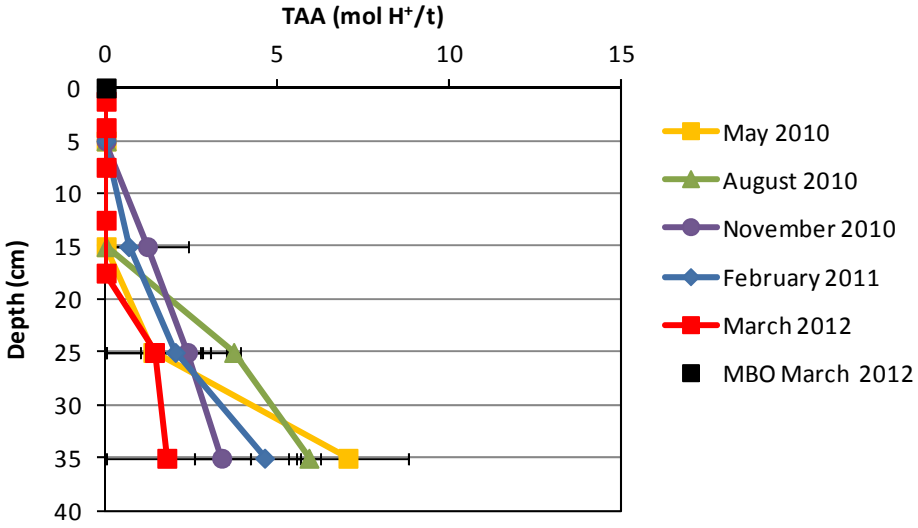

Figure 5-4. Waltowa TAA dynamics at the established *Phragmites* site (May 2010 – March 2012).

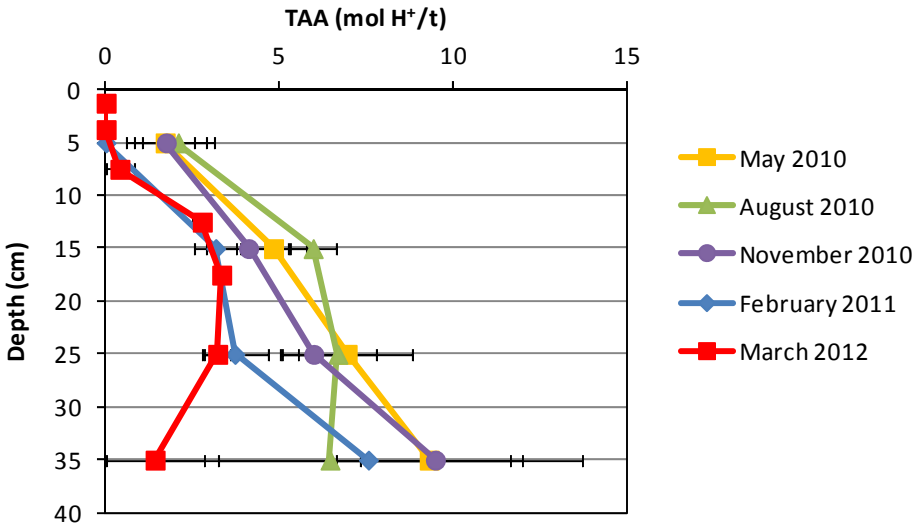

Figure 5-5. Waltowa TAA dynamics at the established *Cotula* site (May 2010 – March 2012).

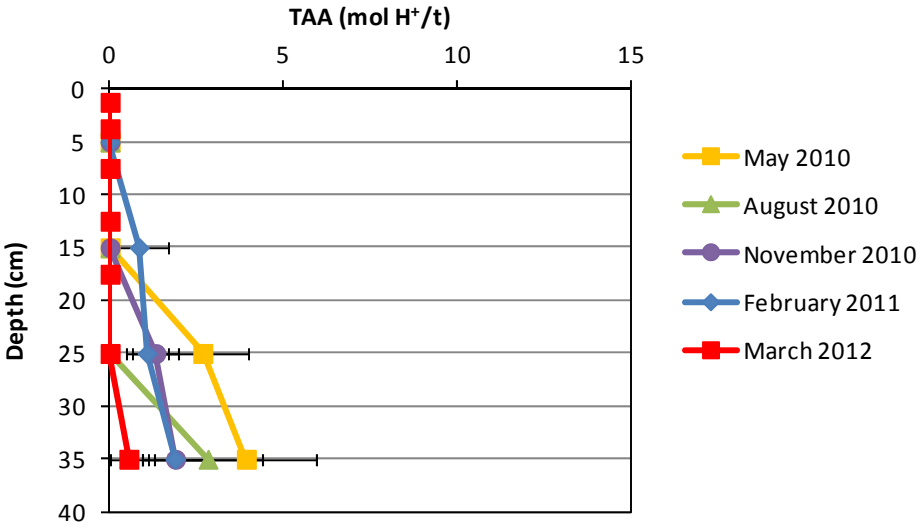

Figure 5-6. Waltowa TAA dynamics at the established *Juncus* site (May 2010 – March 2012).

5.1.1.2 Redox potential (Eh)

All sites initially (i.e. in May 2010) had oxic conditions (Figures 5-7 – 5-9), but during the inundation process increasingly reductive conditions developed initially most strongly in the subsurface layers (August 2010) but then in the whole profile down to 40 cm during the inundations. In the surface sediments the Ehs were ~ 0 mV during the February 2011 sampling in all treatment sites.

Since last sampled in February 2011 the < 20 cm deep sediments under all treatments at this study area had further decreased. However, the Eh in the surface soil layers at both the *Cotula* and *Juncus* sites had increased slightly, whereas the very low Eh in the surface soil layers at the *Phragmites* sites had been maintained during this period.

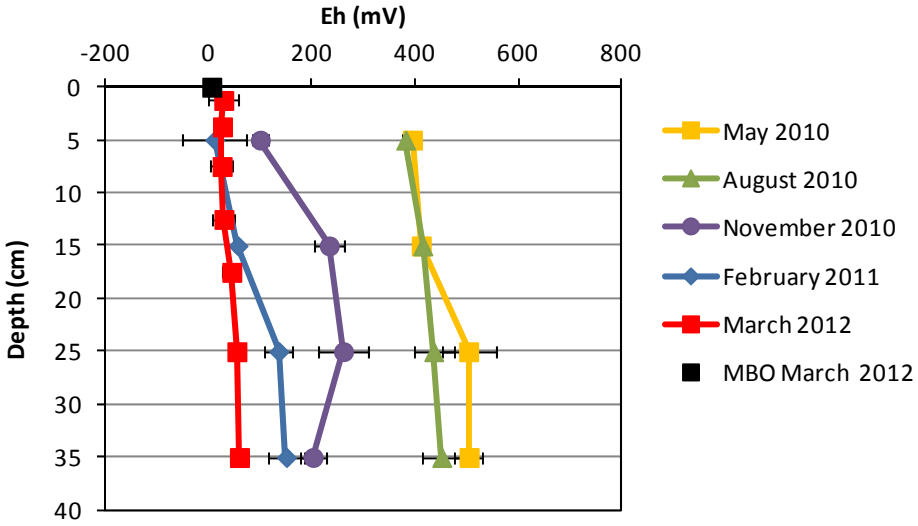

Figure 5-7. Waltowa field Eh dynamics at the established *Phragmites* site (May 2010 – March 2012).

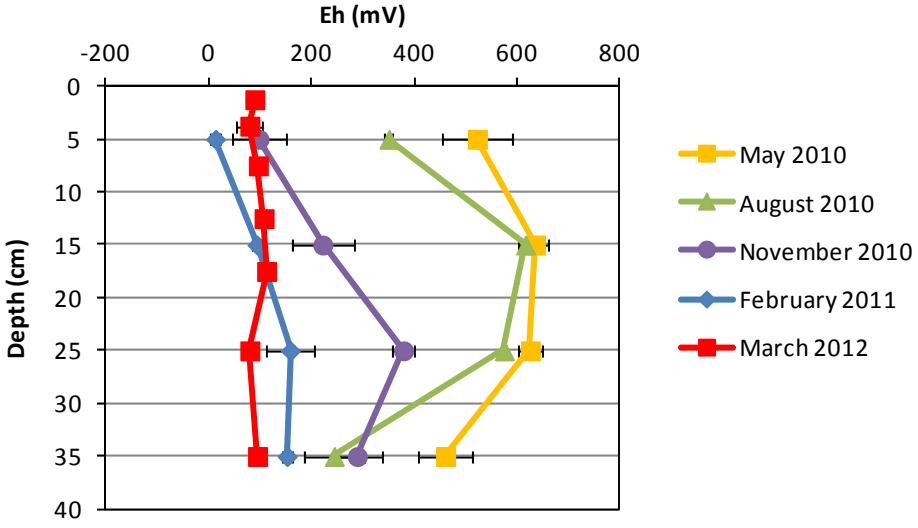

Figure 5-8. Waltowa field Eh dynamics at the established *Cotula* site (May 2010 – March 2012).

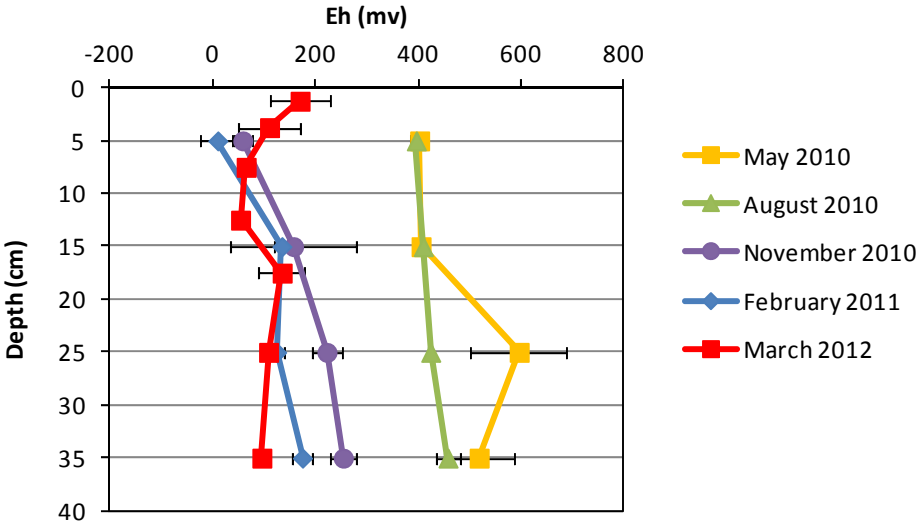

Figure 5-9. Waltowa field Eh dynamics at the established *Juncus* site (May 2010 – March 2012).

5.1.1.3 Electrical conductivity (EC)

The salinity (i.e. EC) in the sediments has decreased only slightly from before inundation to after prolonged inundation. As shown in Figures 5-10 – 5-12 the salinity in all treatments was between 700 and 1500 µS cm^{-1} in the surface layers down to ~30 cm depth but increased to up to 3,000 µS cm^{-1} in the 30-40 cm layer.

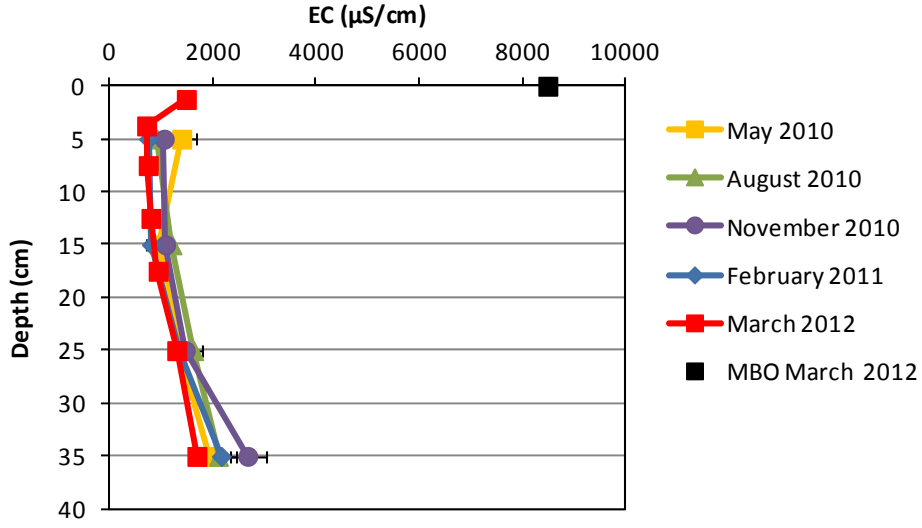

Figure 5-10. Waltowa EC dynamics at the established *Phragmites* site (May 2010 – March 2012).

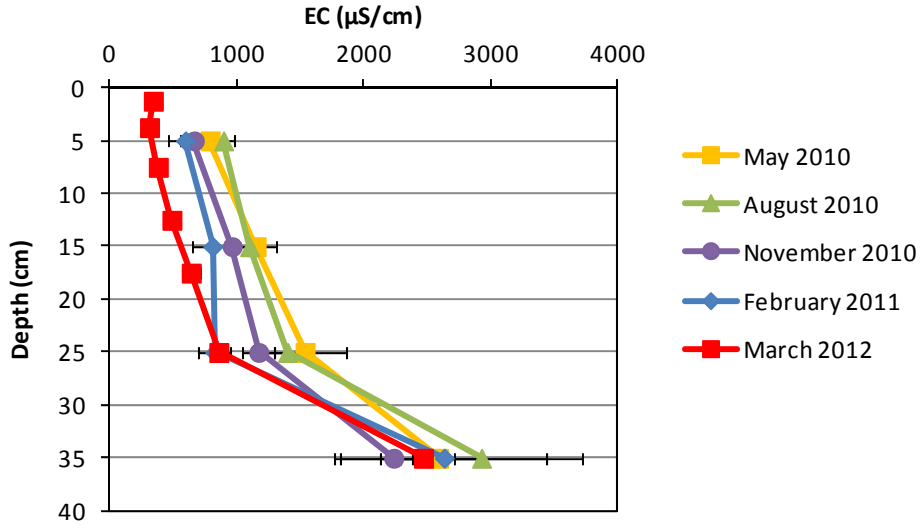

Figure 5-11. Waltowa EC dynamics at the established *Cotula* site (May 2010 – March 2012).

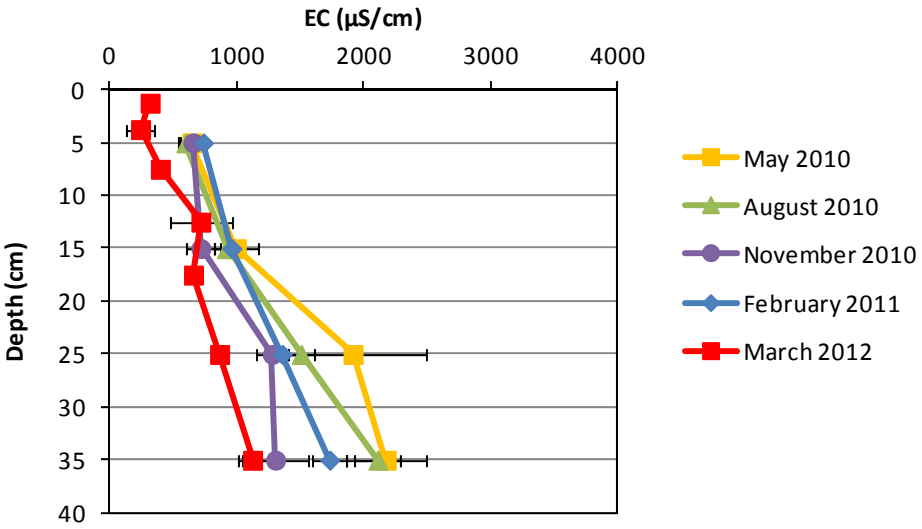

Figure 5-12. Waltowa EC dynamics at the established *Juncus* site (May 2010 – March 2012).

5.1.1.4 Chromium reducible sulfur (CRS)

The pyritic sulfur content in the top 30 cm of sediment was very low in all sites prior to inundation (Figures 5-13 – 5-15). The deeper soil materials contained pyrite at all sites. There was evidence of accumulation of appreciable concentrations 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.

The concentration of Acid Volatile Sulfur (i.e. monosulfides) has trended in parallel with the pyrite concentrations (Figures 5-16 – 5-18). 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.

Elemental sulfur (Figure 5-19) was still present in the surficial layers at each site but in very low concentrations.

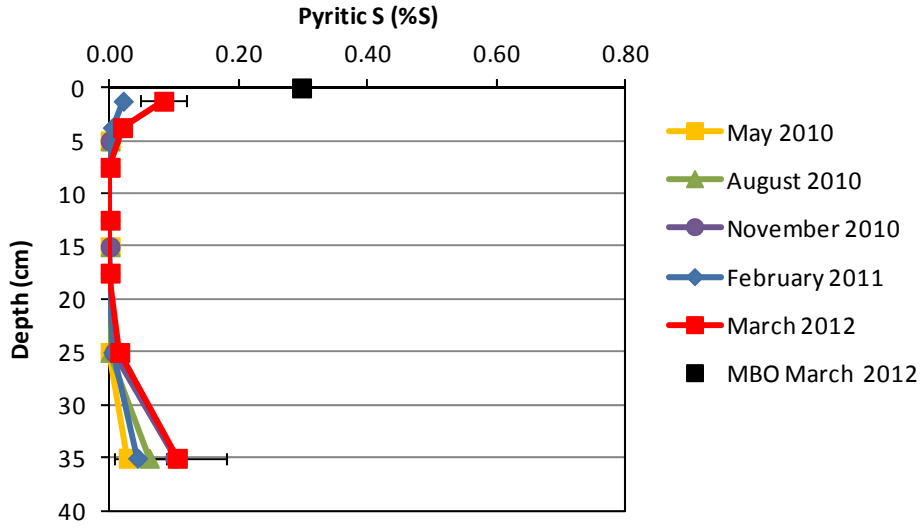

Figure 5-13. Waltowa pyritic sulfur dynamics at the established *Phragmites* site (May 2010 – March 2012).

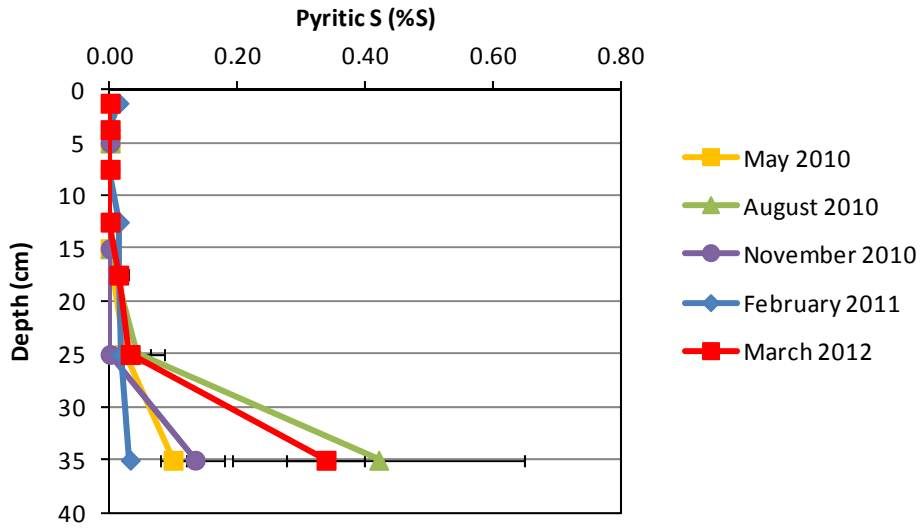

Figure 5-14. Waltowa pyritic sulfur dynamics at the established *Cotula* site (May 2010 – March 2012).

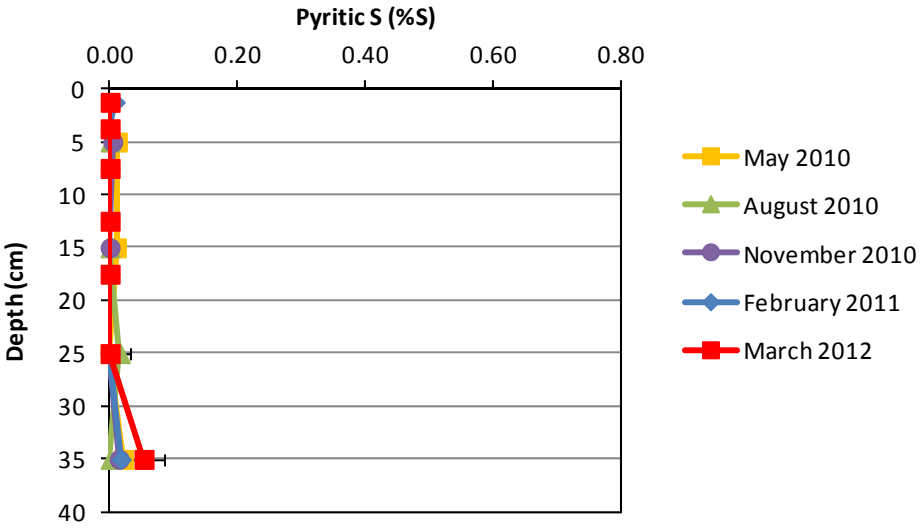

Figure 5-15. Waltowa pyritic sulfur dynamics at the established *Juncus* site (May 2010 – March 2012).

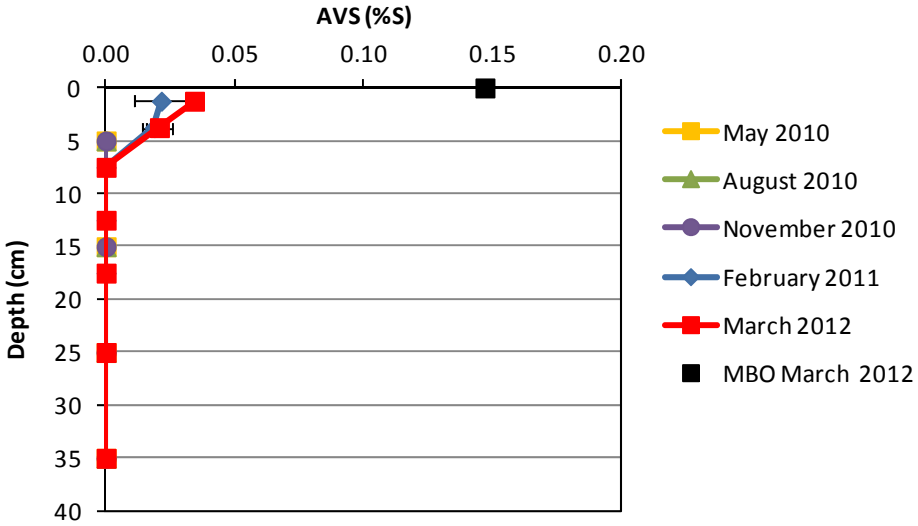

Figure 5-16. Waltowa AVS dynamics at the established *Phragmites* site (May 2010 – March 2012).

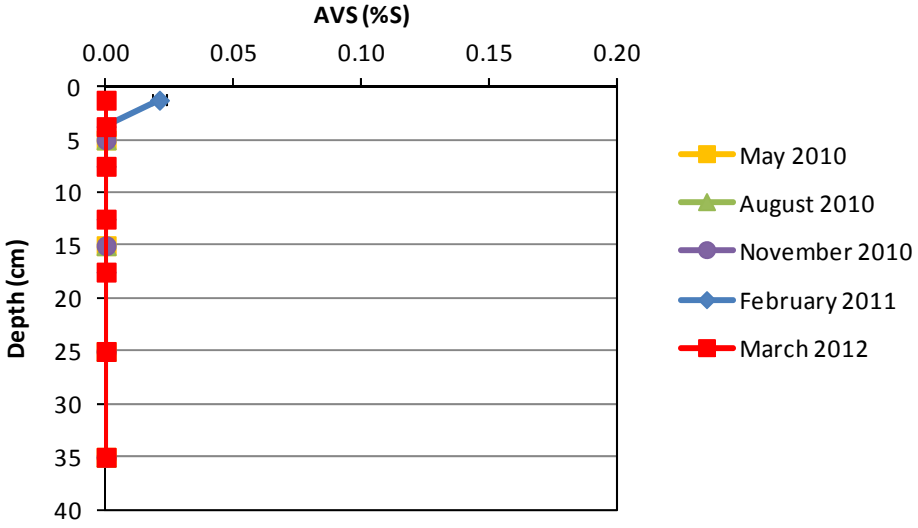

Figure 5-17. Waltowa AVS dynamics at the established *Cotula* site (May 2010 – March 2012).

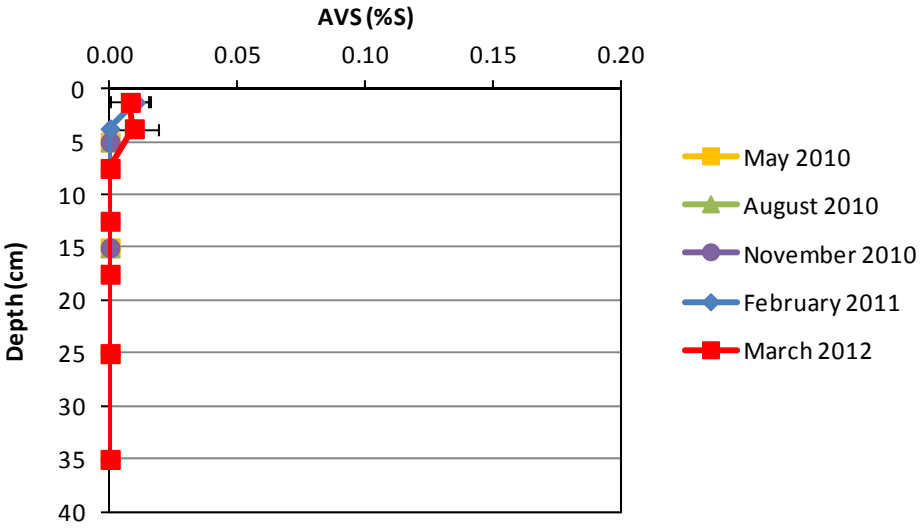

Figure 5-18. Waltowa AVS dynamics at the established *Juncus* site (May 2010 – March 2012).

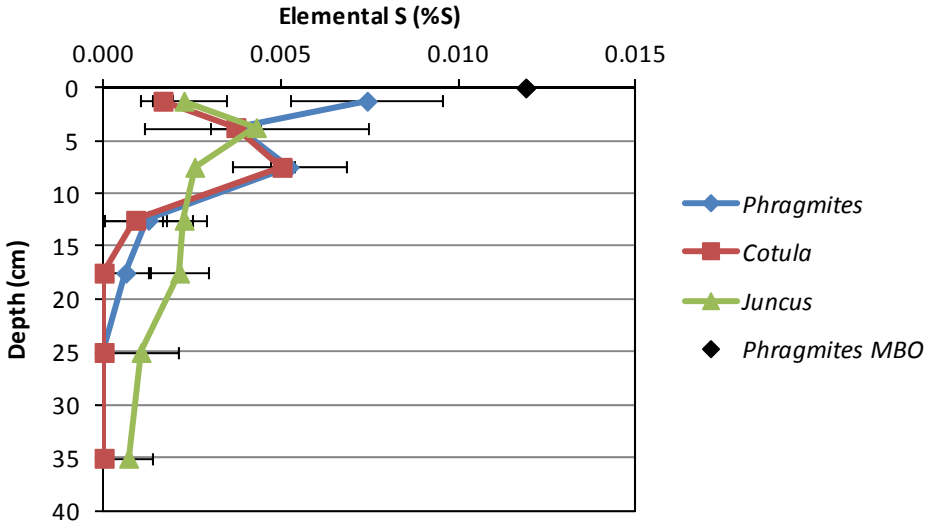

Figure 5-19. Elemental sulfur characteristics at the Waltowa study area (March 2012).

5.1.1.5 Total organic carbon (TOC) and hydrolysable carbon

The total organic carbon and hydrolysable carbon contents measured at the three Waltowa sites between August 2010 and March 2012 are shown below in Figures 5-20 – 5-25.

It is apparent from the data that both the total organic carbon and hydrolysable carbon that had accumulated in the surface sediment layers under both the *Cotula* and *Juncus* treatments prior to inundation in August 2010 have been exhausted by March 2012. In contrast, although having declined in concentration, there was still appreciable total organic carbon and hydrolysable carbon in the uppermost surface sediment layers under the *Phragmites* treatment. The carbon contents in the Waltowa study area are discussed in further detail in Section 5.3.1

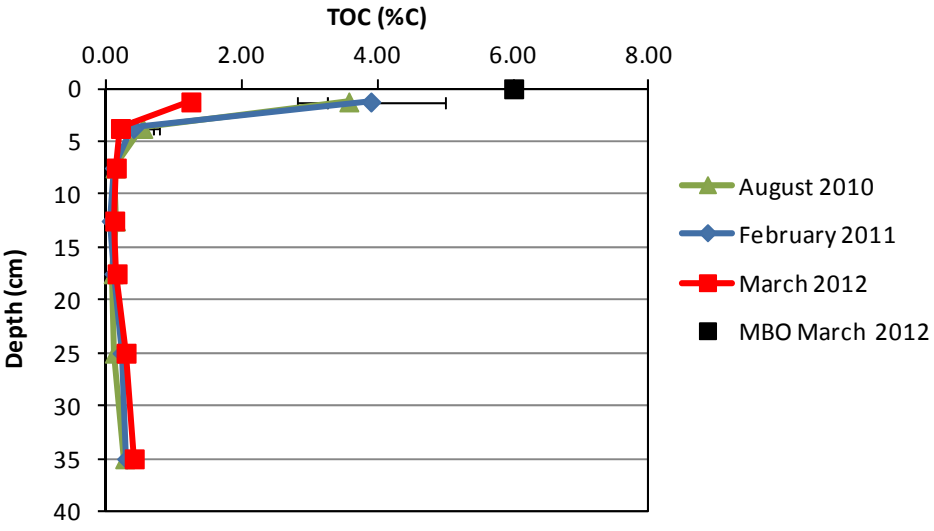

Figure 5-20. Waltowa TOC at *Phragmites* site (August 2010, February 2011 and March 2012).

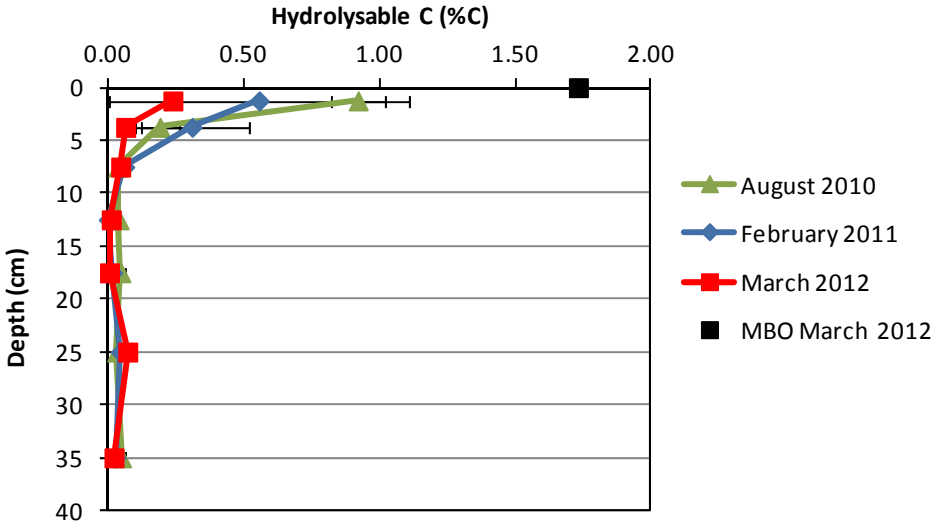

Figure 5-21. Waltowa hydrolysable C at *Phragmites* site (August 2010, February 2011 and March 2012).

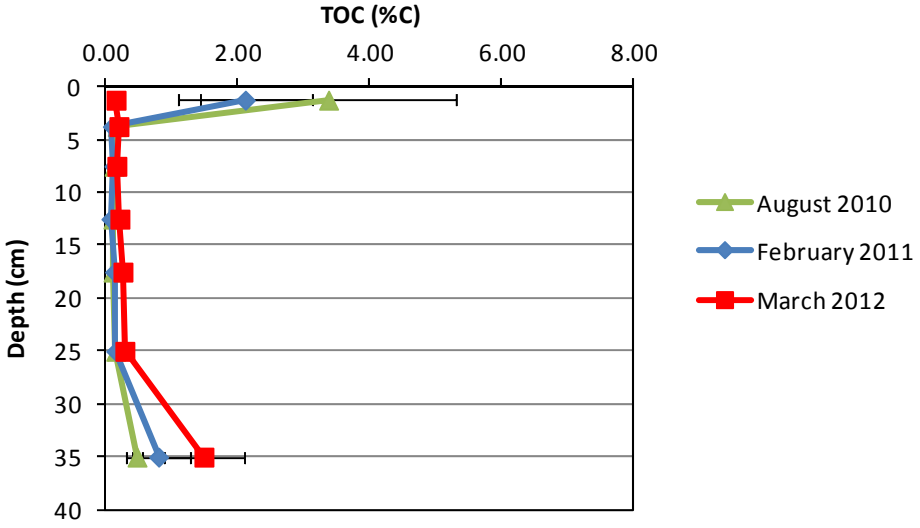

Figure 5-22. Waltowa TOC at *Cotula* site (August 2010, February 2011 and March 2012).

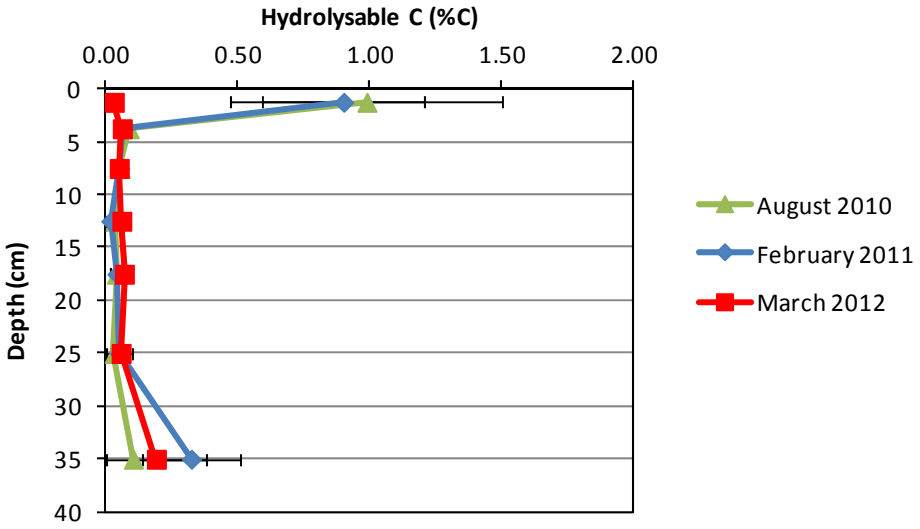

Figure 5-23. Waltowa hydrolysable C at *Cotula* site (August 2010, February 2011 and March 2012).

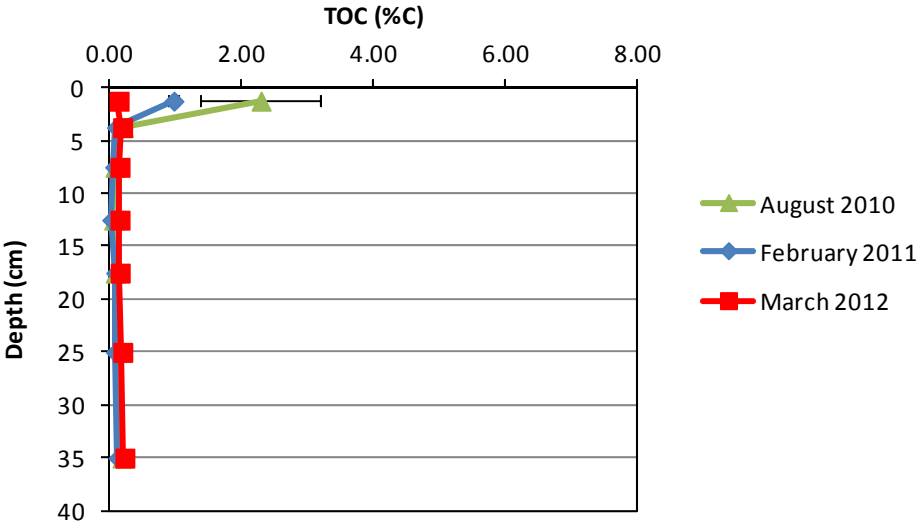

Figure 5-24. Waltowa TOC at *Juncus* site (August 2010, February 2011 and March 2012).

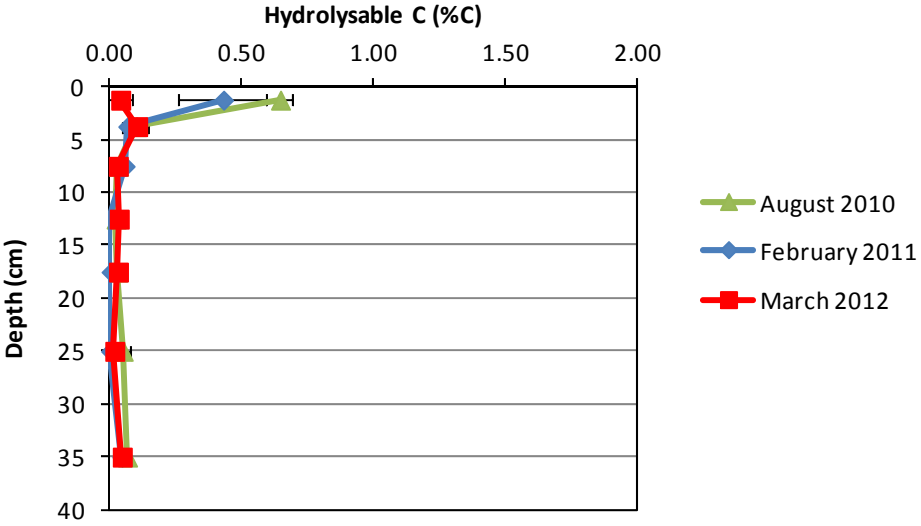

Figure 5-25. Waltowa hydrolysable C at *Juncus* site (August 2010, February 2011 and March 2012).

5.1.2 Poltalloch

5.1.2.1 $pH_{(1:1, soil:water)}$ and TAA

Since re-inundation in August 2010, the pHs of the upper 20 cm of sediment has increased slightly from a pH of 6.5 - 7 to a pH of 7 -7.3 (Figure 5-26). Although the pHs at this study area were < 4 in the 20 – 30 cm layer prior to inundation and after 6 months of inundation, the subsequent 13 months of inundation have resulted in substantial pH increases in these layers to between pH 5.5 -6.

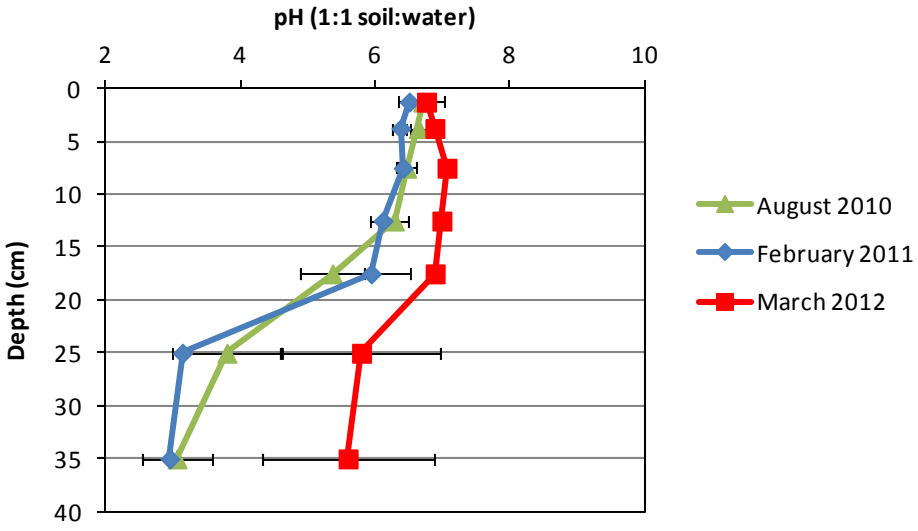

Figure 5-26. Poltalloch field pH dynamics at the Bevy rye site (August 2010, February 2011 and March 2012).

The TAAs (Figure 5-27) 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, during, and post inundation.

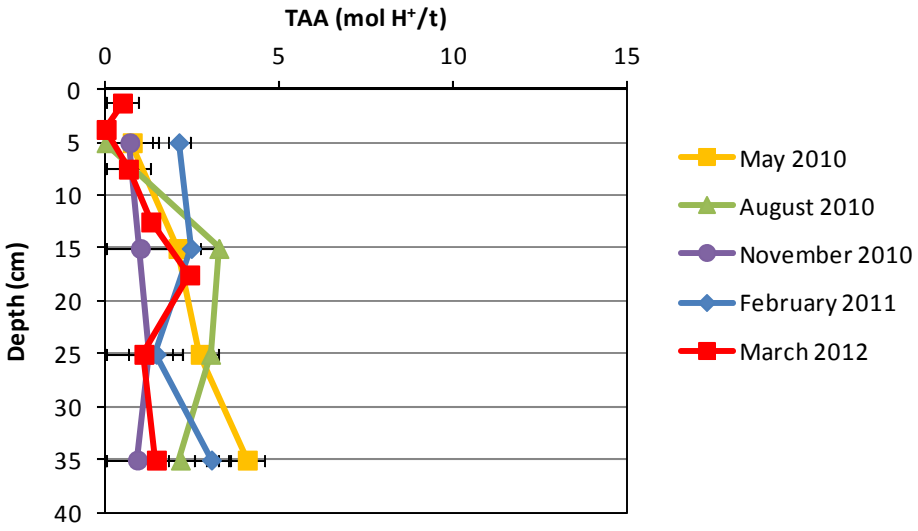

Figure 5-27. Poltalloch TAA dynamics at the Bevy rye site (May 2010 – March 2012).

5.1.2.2 Redox potential (Eh)

Initially (i.e. in May 2010) the site had oxic conditions (Figure 5-28), but during the inundation process increasingly reductive conditions developed throughout the whole profile down to 40 cm and have further decreased slightly during the prolonged inundation.

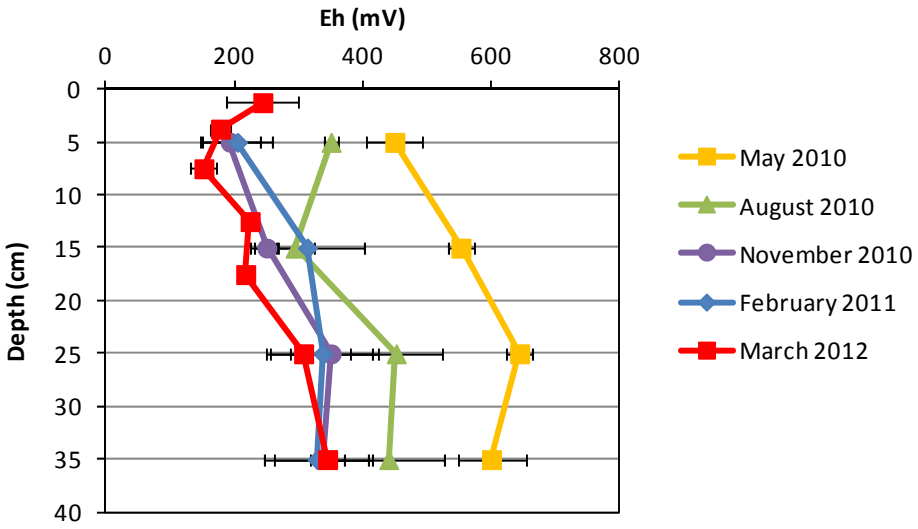

Figure 5-28. Poltalloch field Eh dynamics at the Bevy rye site (May 2010 – March 2012).

5.1.2.3 Electrical conductivity (EC)

The salinity (i.e. EC) did not change appreciably from before inundation to after prolonged inundation. As shown in Figure 5-29 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 2012 sampling the EC in the lowest layer was ~700 µS cm^{-1} at 40 cm.

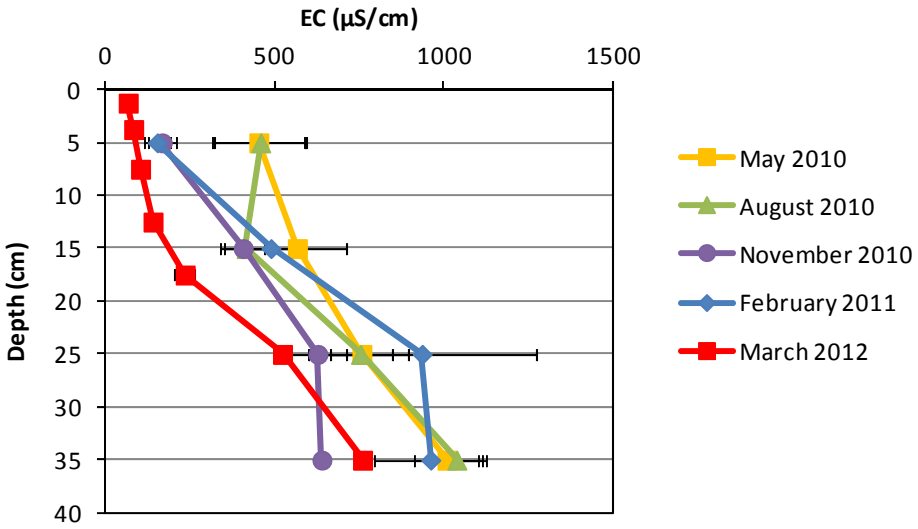

Figure 5-29. Poltalloch EC dynamics at the Bevy rye site (May 2010 – March 2012).

5.1.2.4 Chromium reducible sulfur (CRS)

The pyritic sulfur contents were very low (i.e. < 0.02% S) in the upper 30 cm (Figure 5-30) prior to inundation. There were appreciable concentrations of residual reduced inorganic sulfides (i.e. up to 0.06% S as pyritic sulfur) in the 30 – 40 cm depth sediment layer.

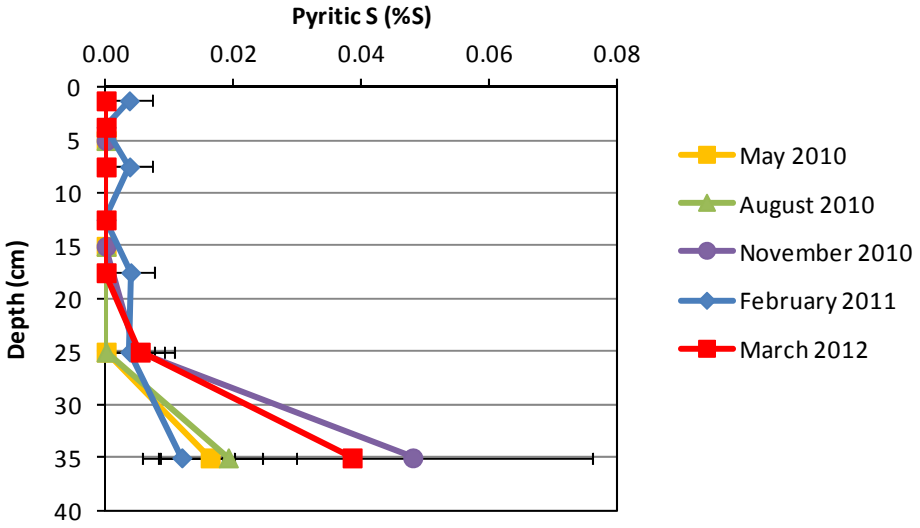

Figure 5-30. Poltalloch pyritic sulfur dynamics in the surface soil (0-40 cm) at the Bevy rye site (May 2010 – March 2012).

5.1.2.5 Total organic carbon (TOC) and hydrolysable carbon

The total organic carbon and hydrolysable carbon contents measured at the Poltalloch site between August 2010 and March 2012 are shown below in Figures 5-31 – 5-32. The carbon contents at the Poltalloch sites are discussed in detail in Section 5.3.2.

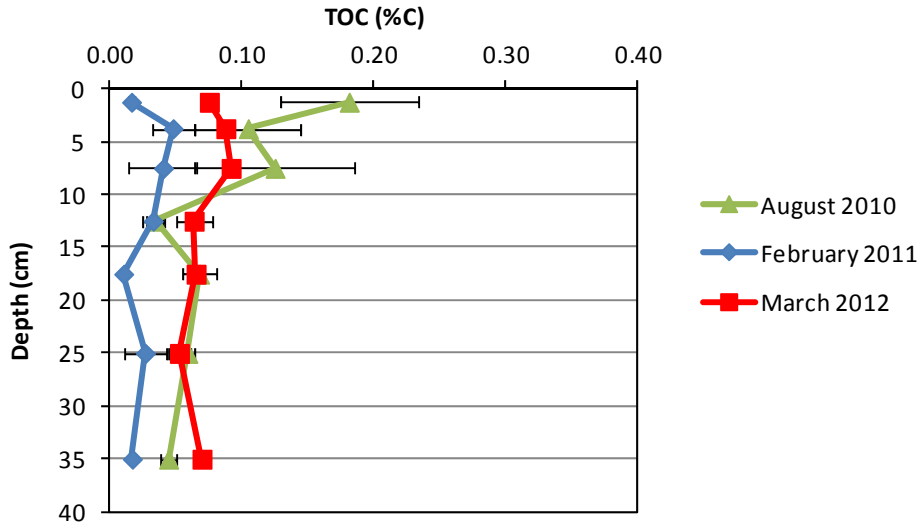

Figure 5-31. Poltalloch TOC at the *Juncus* plantings in Bevy rye site (August 2010, February 2011 and March 2012).

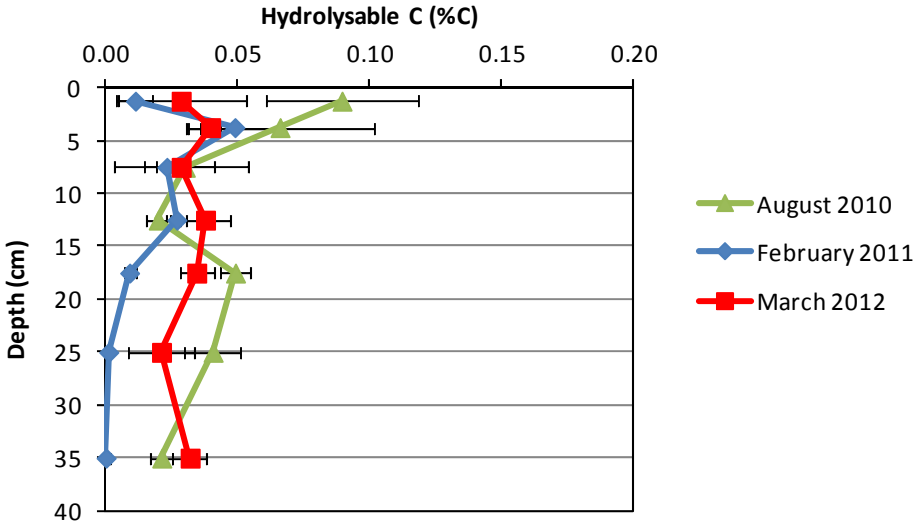

Figure 5-32. Poltalloch hydrolysable C at the *Juncus* plantings in Bevy rye site (August 2010, February 2011 and March 2012).

5.1.3 Tolderol

5.1.3.1 $pH_{(1:1, soil:water)}$ and TAA

The control site (from 0 - 40 cm) and the Bevy rye site (only 10 - 40 cm) initially were acidic (pH < ~4) prior to inundation (Figures 5-33 – 5-34). Upon lake filling in August 2010 the pHs of the surface soil layers down to 40 cm depth in the control site further acidified considerably likely due to acidity exchange from the soil from the inundating waters. This acidification upon inundation effect was confined to the 20 – 50 cm layer in the Bevy rye site.

During prolonged inundation of both sites since August 2010 the pHs of sediments under both sites increased although this was initially most pronounced in the surficial sediment layers under the Bevy rye site in comparison with the control site.

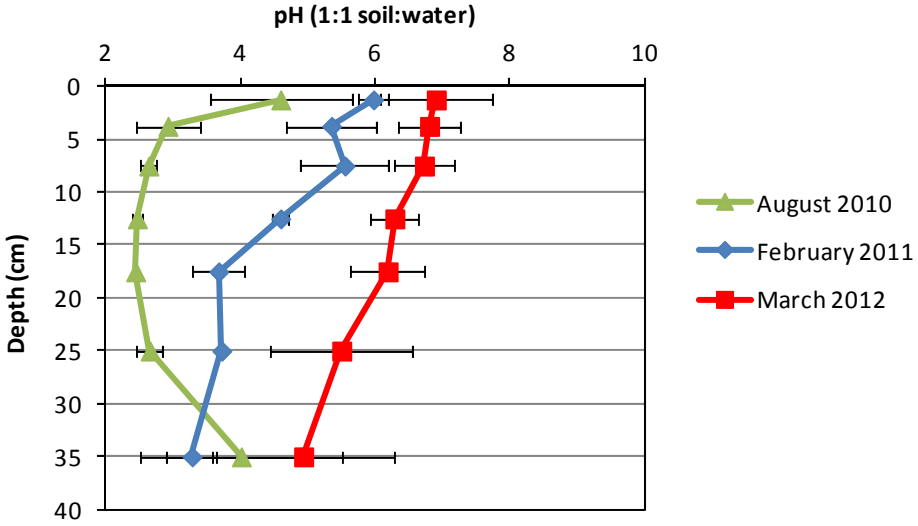

Figure 5-33. Tolderol field pH dynamics at the control site (August 2010, February 2011 and March 2012).

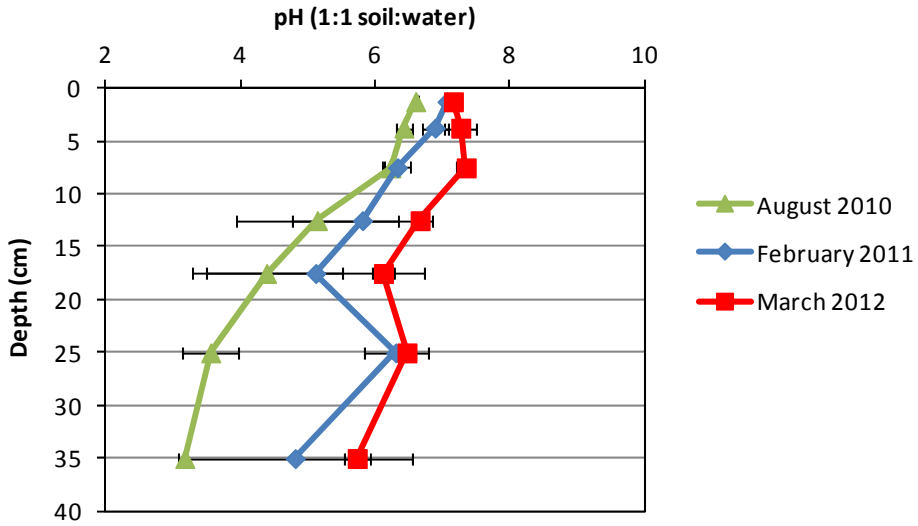

Figure 5-34. Tolderol field pH dynamics at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

The TAAs (Figures 5-35 – 5-36) 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 exposure in all soil layers of each site.

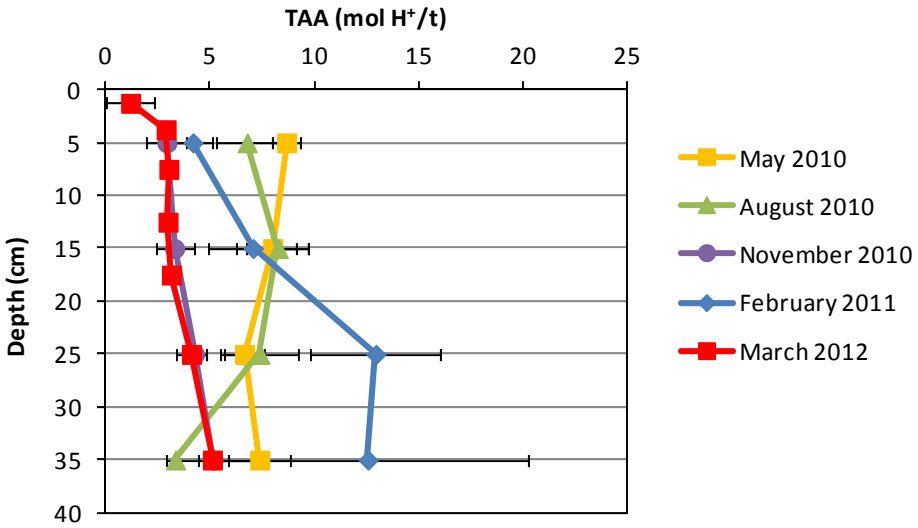

Figure 5-35. Tolderol TAA dynamics at the control site (May 2010 – March 2012).

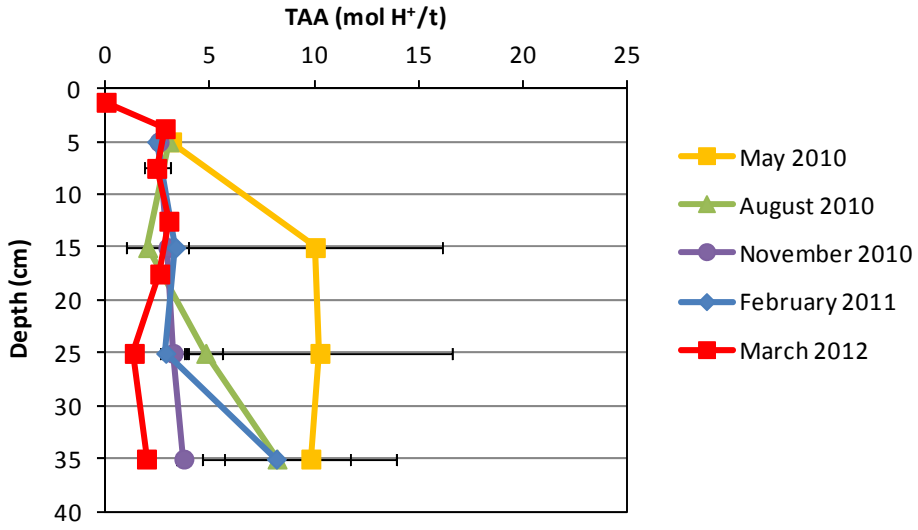

Figure 5-36. Tolderol TAA dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

5.1.3.2 Redox potential (Eh)

All treatments initially (i.e. in May 2010) had oxic conditions (Figures 5-37 – 5-38), but during the inundation increasingly reductive conditions have developed throughout the whole profile down to 40 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) and more intensively (i.e. down to 200 mV cf. 300 mV in the 0 – 10 cm layer) 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.

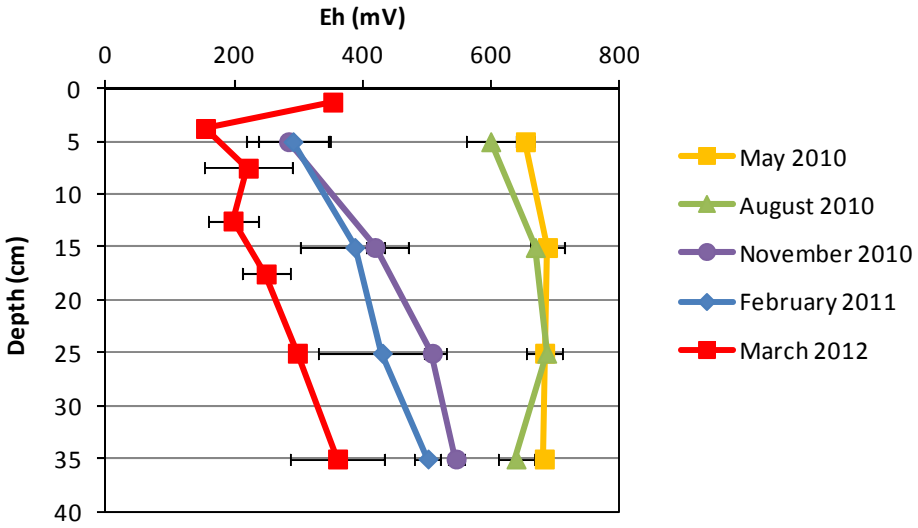

Figure 5-37. Tolderol field Eh dynamics at the control site (May 2010 – March 2012).

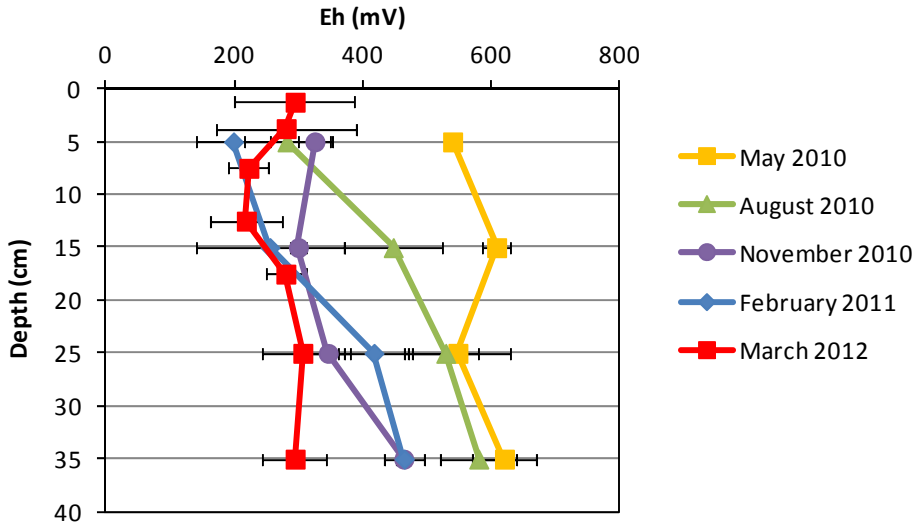

Figure 5-38. Tolderol field Eh dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

5.1.3.3 Electrical conductivity (EC)

As shown in Figures 5-39 – 5-40 the salinity (i.e. EC) decreased appreciably over the 13 months to March 2012 in the sediment layers under both treatments.

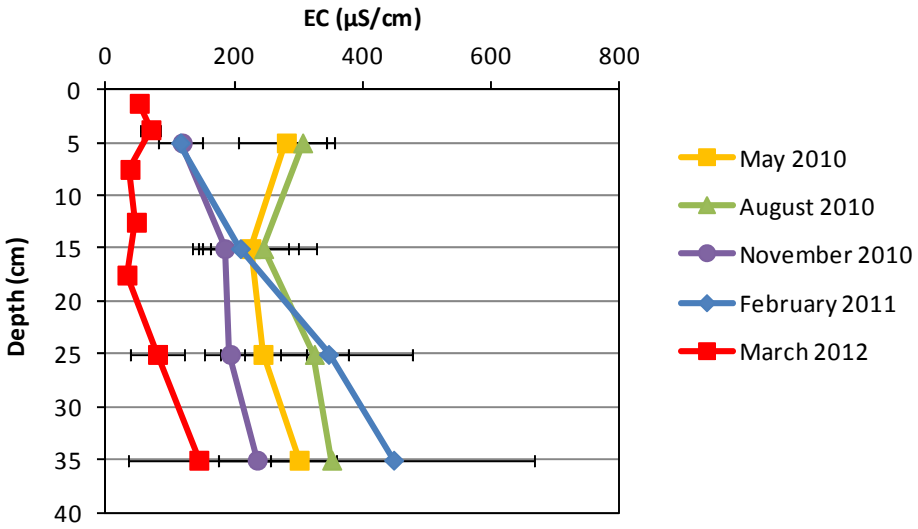

Figure 5-39. Tolderol EC dynamics at the control site (May 2010 – March 2012).

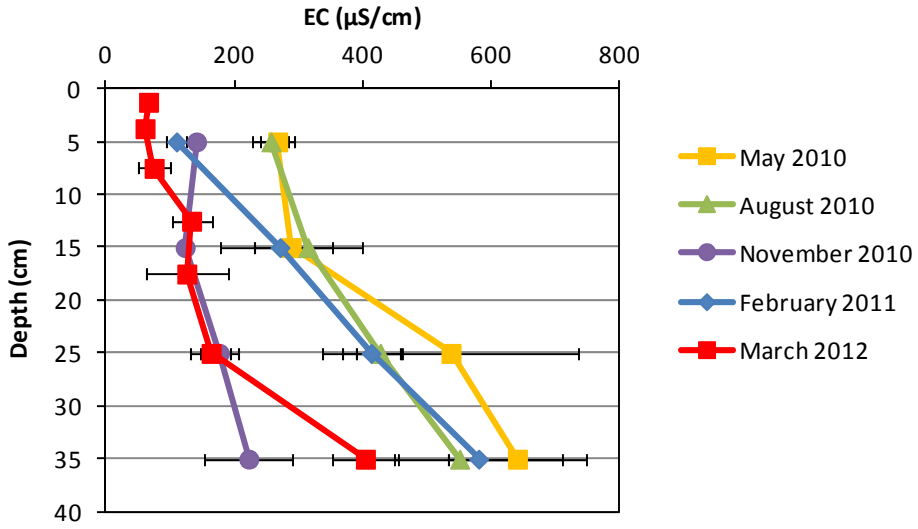

Figure 5-40. Tolderol EC dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

5.1.3.4 Chromium reducible sulfur (CRS)

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 5-41 – 5-42). These have remained low during the inundation period to date, despite a slight accumulation at the February 2011 assessment and decrease from then at the March 2012 assessment. 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.

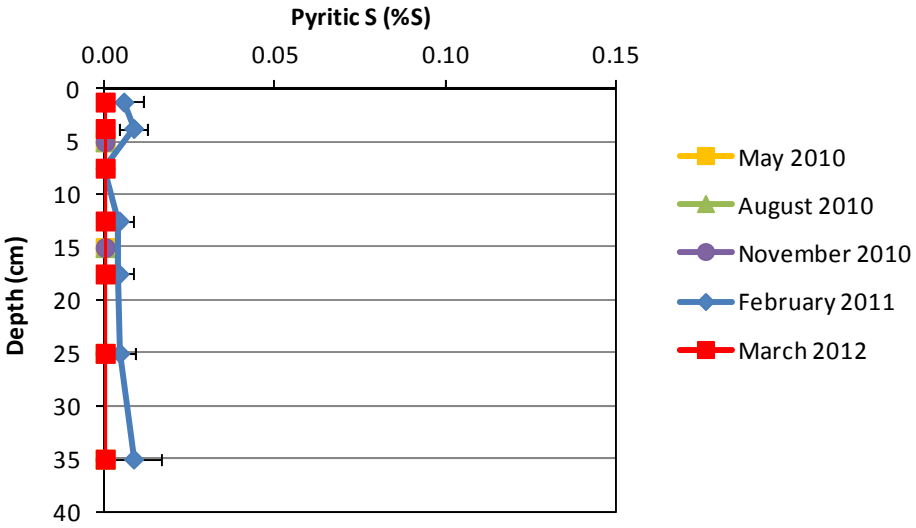

Figure 5-41. Tolderol pyritic sulfur dynamics at the control site (May 2010 – March 2012).

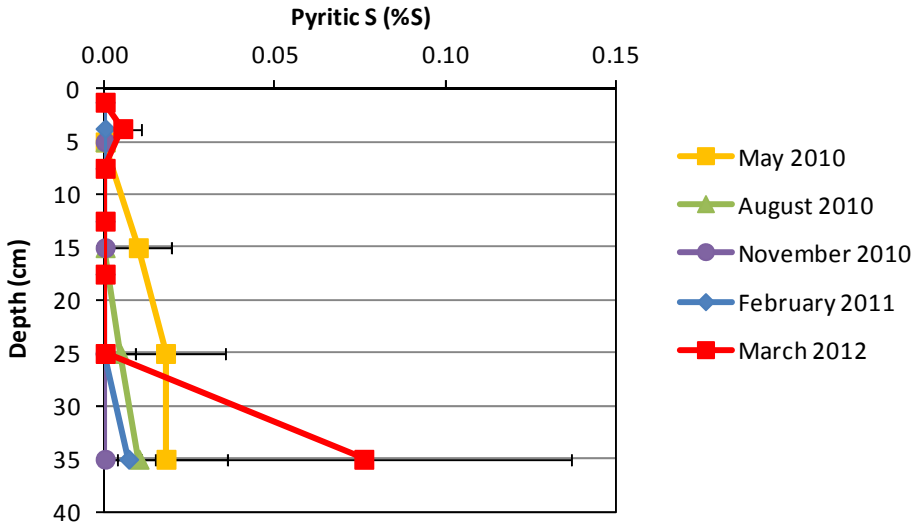

Figure 5-42. Tolderol pyritic sulfur dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

5.1.3.5 Total organic carbon (TOC) and hydrolysable carbon

The total organic carbon and hydrolysable carbon contents measured at the two Tolderol sites between August 2010 and March 2012 are shown below in Figures 5-43 – 5-46. The carbon contents in the Tolderol study area are discussed in detail in Section 5.3.3.

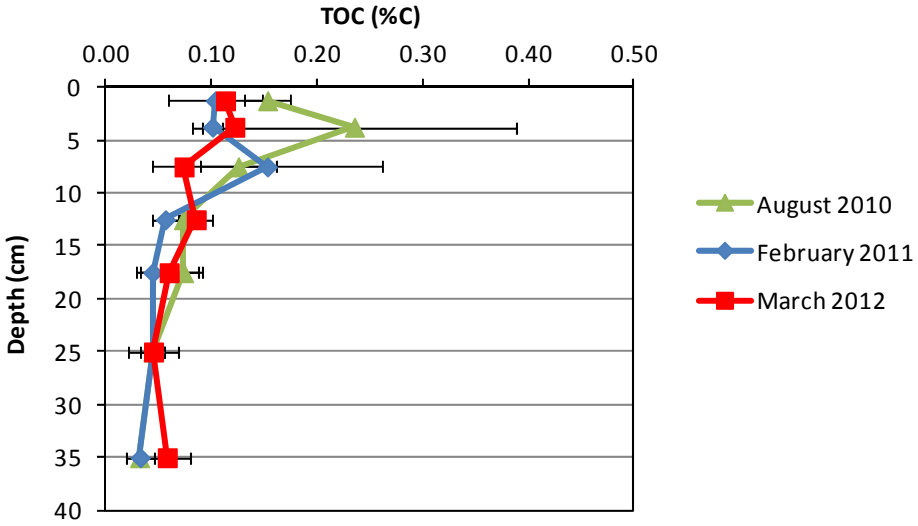

Figure 5-43. Tolderol TOC at the control site (August 2010, February 2011 and March 2012).

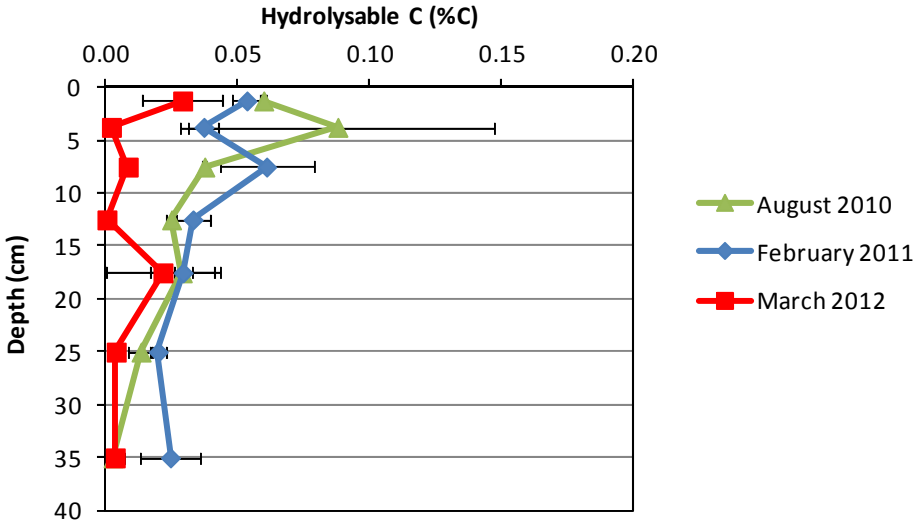

Figure 5-44. Tolderol hydrolysable C at the control site (August 2010, February 2011 and March 2012).

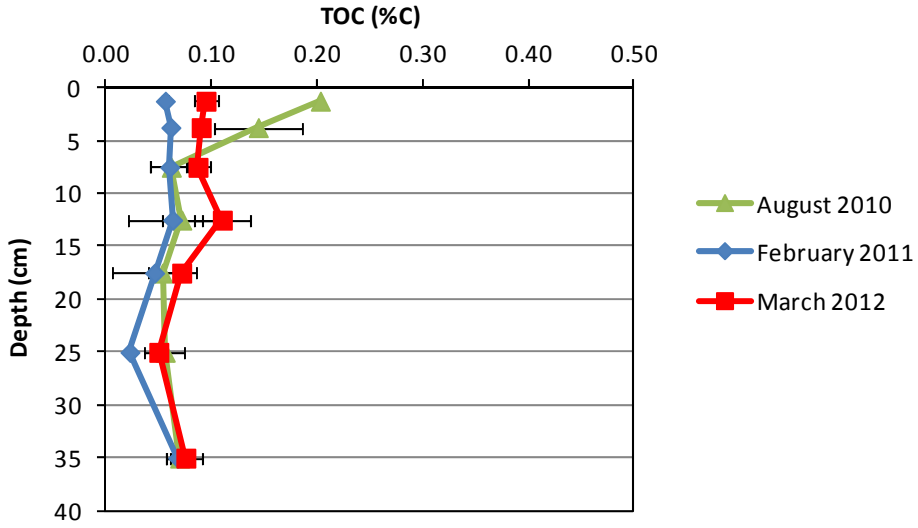

Figure 5-45. Tolderol TOC at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

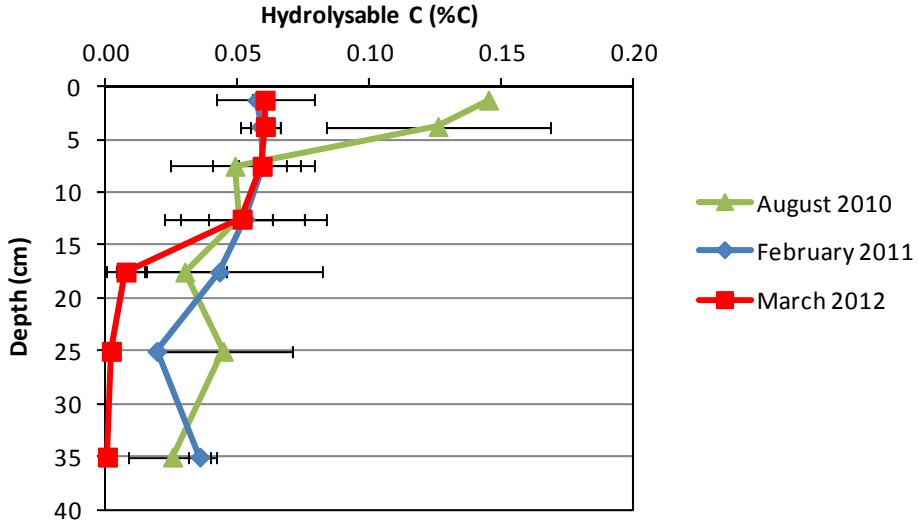

Figure 5-46. Tolderol hydrolysable C at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

5.1.4 Campbell Park

5.1.4.1 $pH_{(1:1, soil:water)}$ and TAA

Both sites initially had acidic soil layers prior to the inundation that took place after the August 2010 sampling (Figures 5-47 – 5-48). For the control site this acidic layer was severely acidic pH < 3 down to 30 cm depth where as 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 ~6.5.

Prior to inundation the surficial layer of the vegetated treatment were elevated relative to the control treatment. As these treatments were not able to be sampled separately prior to the establishment of the vegetation is not possible to ascribe this pH difference 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 in the case of the control sites.

The pH of the surface layer of the control site increased after inundation to a pH of 6.5. The pH of the subsoil layers have continued to increase over the inundation period with the greatest increases occurring closest to the surface.

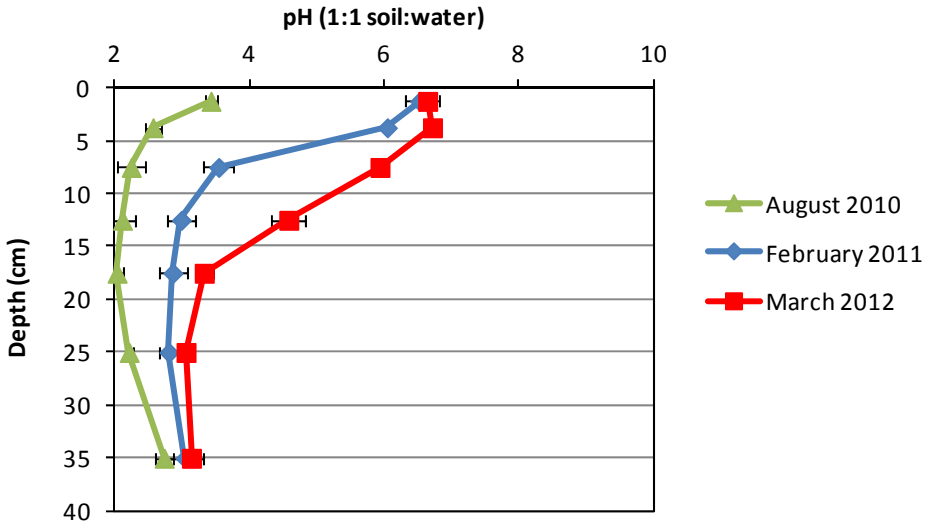

Figure 5-47. Campbell Park field pH dynamics at the control site (August 2010, February 2011 and March 2012).

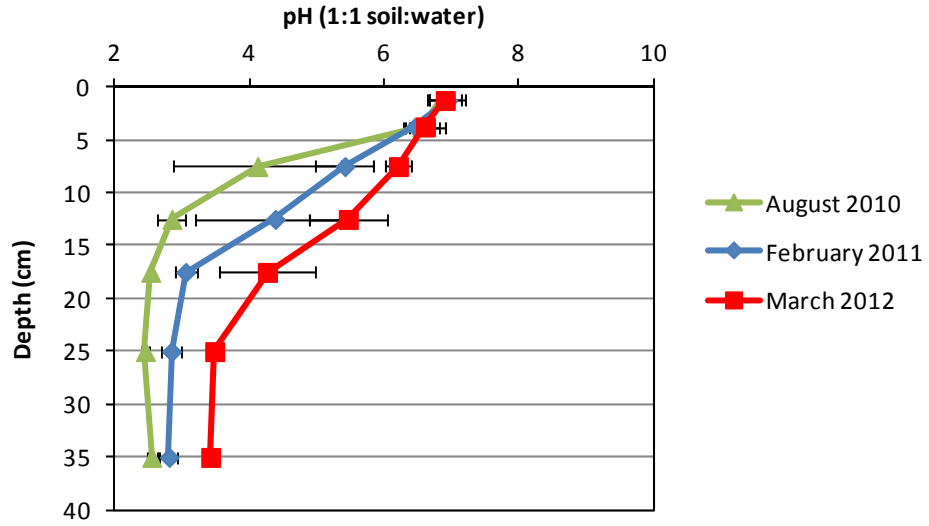

Figure 5-48. Campbell Park field pH dynamics at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

The TAAs (Figures 5-49 – 5-50) 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 and other iron oxides and this may account for the much higher TAAs found at Campbell Park compared to the other experimental locations in this study.

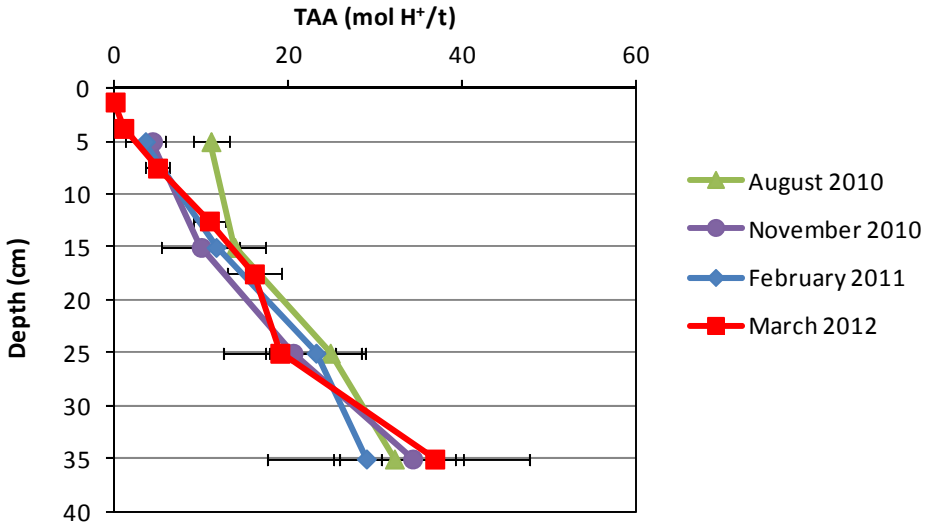

Figure 5-49. Campbell Park TAA dynamics at the control site (August 2010 – March 2012).

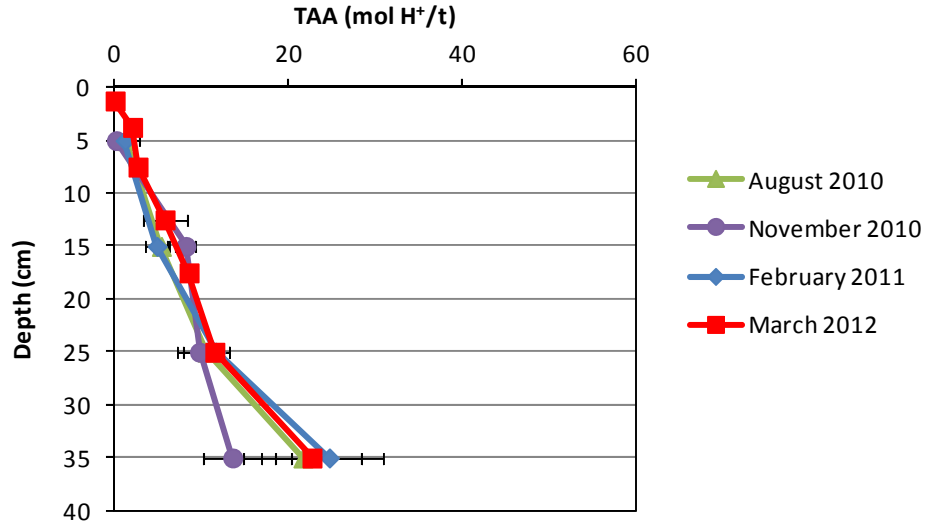

Figure 5-50. Campbell Park TAA dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

5.1.4.2 Redox potential (Eh)

All sites initially (i.e. in August 2010) had oxic conditions of from 400 – 700 mV in the top 40 cm sandy-textured layers (Figures 5-51 – 5-52), 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 ~100 - 150 mV at the March 2012 sampling.

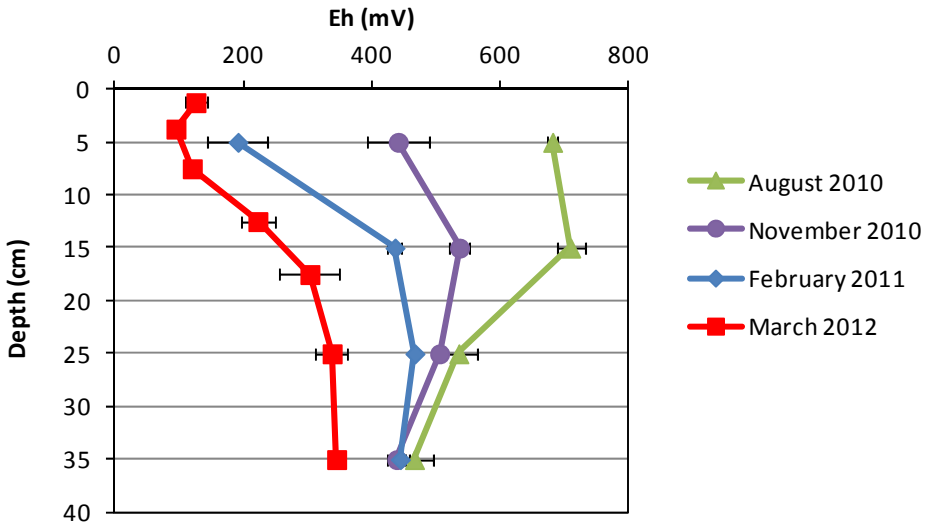

Figure 5-51. Campbell Park field Eh dynamics at the control site (August 2010 – March 2012).

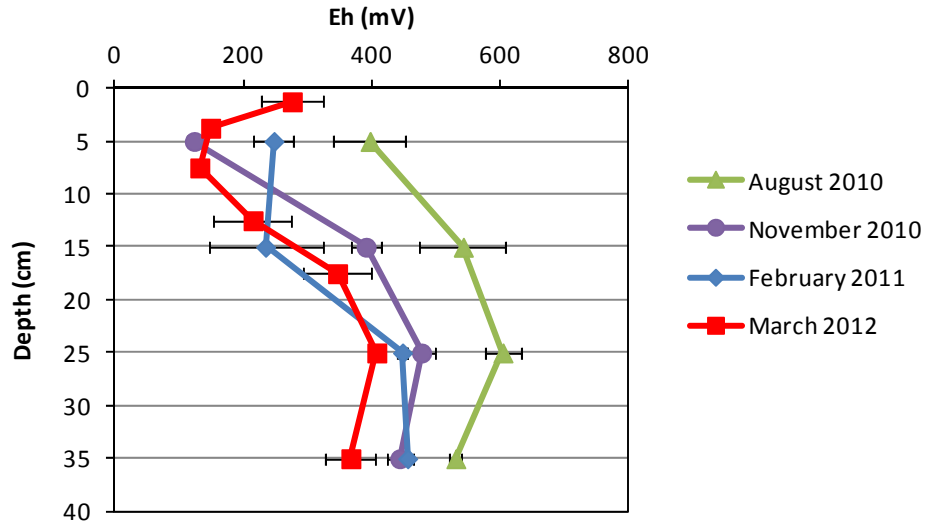

Figure 5-52. Campbell Park field Eh dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

5.1.4.3 Electrical conductivity (EC)

The salinity (i.e. EC) has continued to decrease during inundation as shown in Figures 5-53 – 5-54. The salinity in the sediments under both treatments gradually increased with depth from ~ 300 – 400 µS cm^{-1} to nearly 2,000 µS cm^{-1}.

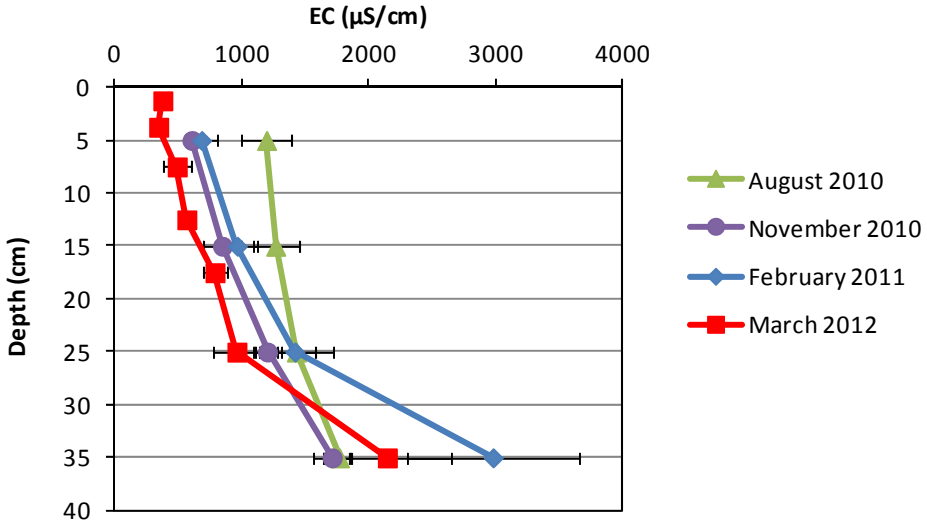

Figure 5-53. Campbell Park EC dynamics at the control site (August 2010 – March 2012).

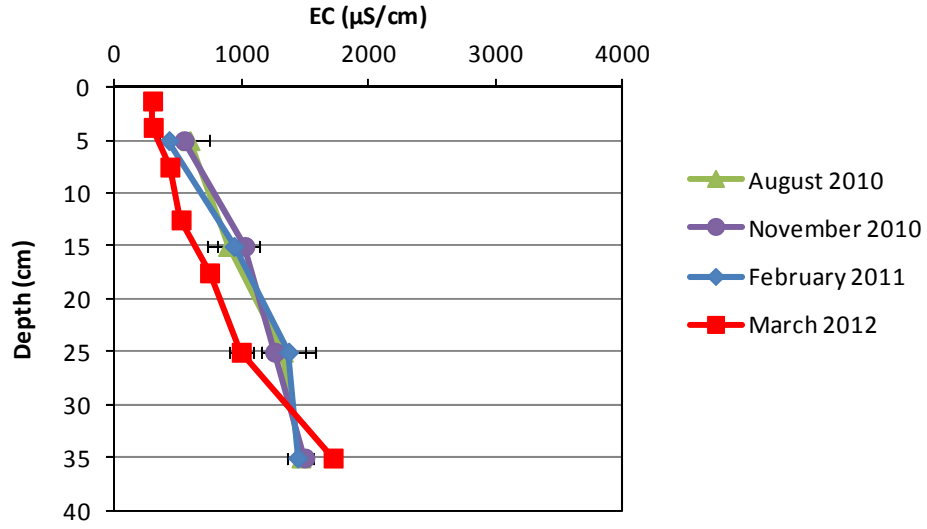

Figure 5-54. Campbell Park EC dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

5.1.4.4 Chromium reducible sulfur (CRS)

The pyritic sulfur contents were low (i.e. < 0.02% S) in the surficial layers (0 – 20 cm) at both sites (Figures 5-55 – 5-56) prior to inundation. The apparent accumulation of an appreciable concentration of reduced inorganic sulfides (i.e. up to 0.70% S as pyrite) in the 30 – 40 cm layer in the control 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. There was no evidence of the accumulation of reduced inorganic sulfides in the surficial sediments after inundation to date.

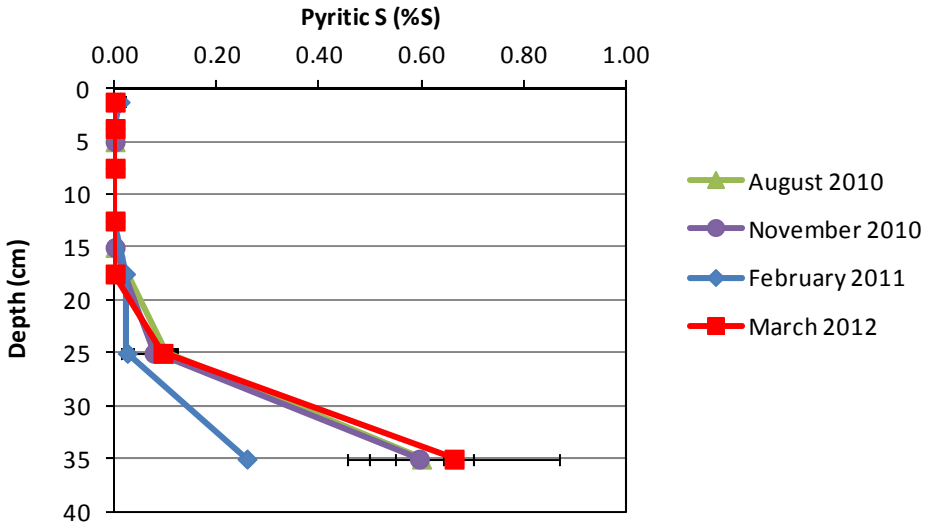

Figure 5-55. Campbell Park pyritic sulfur dynamics at the control site (August 2010 – March 2012).

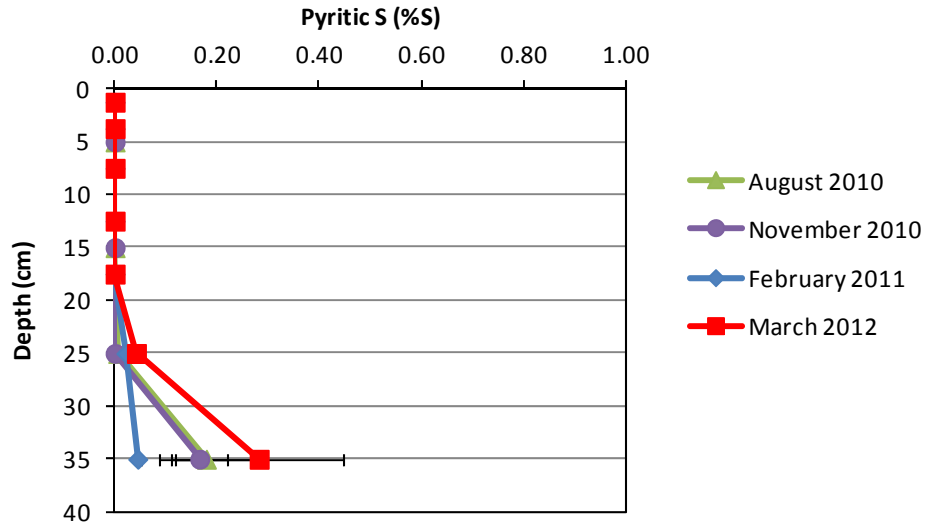

Figure 5-56. Campbell Park pyritic sulfur dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

5.1.4.5 Total organic carbon (TOC) and hydrolysable carbon

The total organic carbon and hydrolysable carbon contents measured at the two Campbell Park sites between August 2010 and March 2012 are shown below in Figures 5-57 – 5-60. The carbon contents in the Campbell Park study area are discussed in detail in Section 5.3.4.

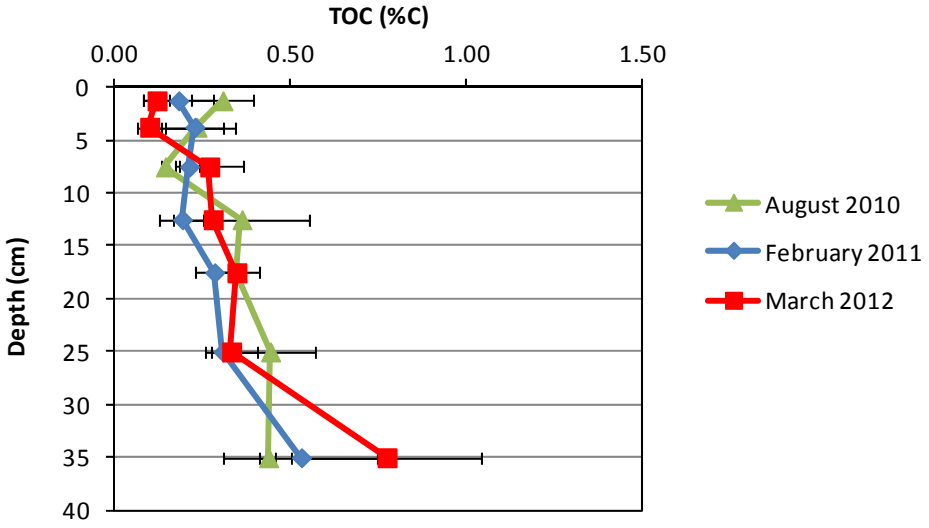

Figure 5-57. Campbell Park field TOC at the control site (August 2010, February 2011 and March 2012).

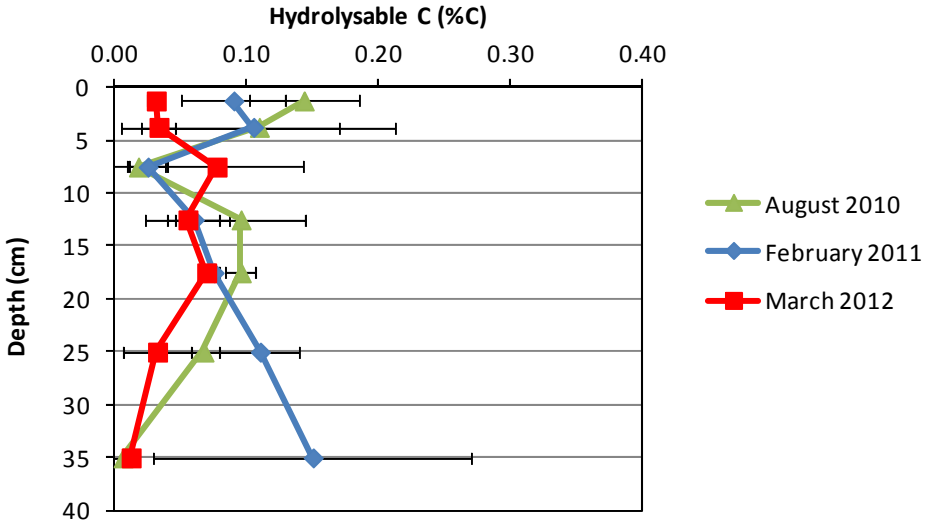

Figure 5-58. Campbell Park field hydrolysable C at the control site (August 2010, February 2011 and March 2012).

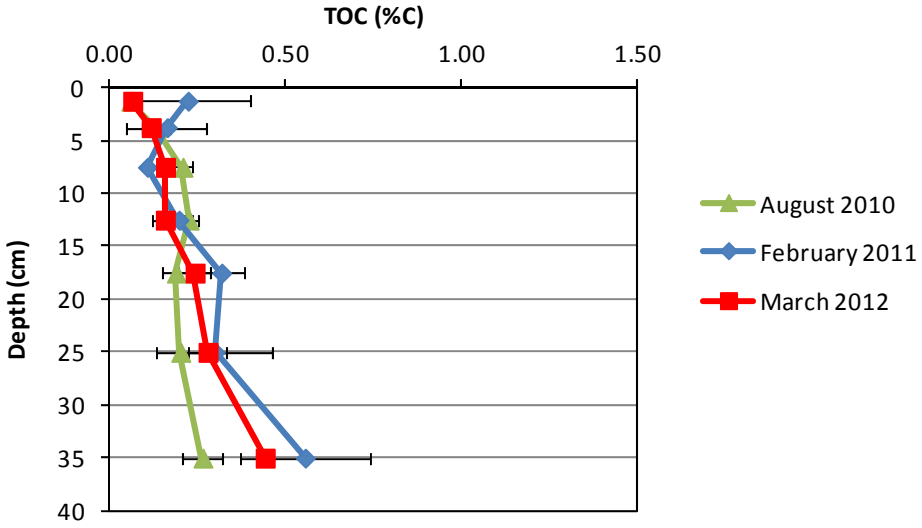

Figure 5-59. Campbell Park field TOC at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

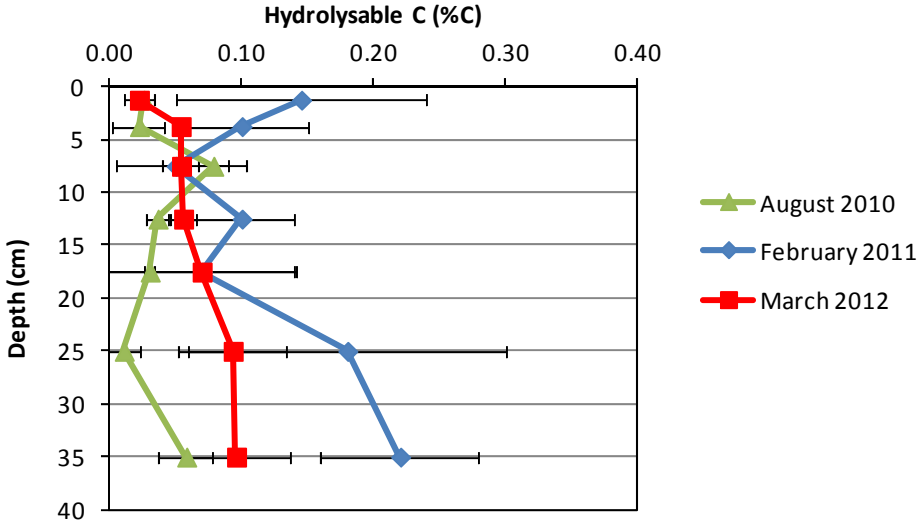

Figure 5-60. Campbell Park field hydrolysable C in at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

5.2 Pore-water properties

5.2.1 Waltowa

Figures 5-61 - 5-66 show that in general the concentrations of sulfate and chloride in the sediment under all vegetation types have decreased appreciably during the inundation period.

Figures 5-67 – 5-69 show that the Cl:SO_4 ratios, an indicator of sulfide oxidation or formation have only increased appreciably in the surficial layers under the *Phragmites* vegetation. In coastal settings such as these, appreciable decreases in the Cl:SO_4 ratio below 7 – 8 (the 'normal' ratio of coastal settings affected by seawater either tidally or as atmospheric deposition) can be used to indicate that sulfide oxidation (and hence the production of sulfate) has occurred, whereas appreciable increases in the Cl:SO_4 ratio above 7 – 8 indicate that sulfate is being lost at greater rates relative to chloride, one possible process responsible for this can be sulfide mineral formation.

The very low Cl:SO_4 ratios in most of these soils initially indicate sulfide oxidation has taken place, whereas the appreciable increase in the Cl:SO_4 ratio in the 0 - 20 cm layer under the *Phragmites* vegetation would indicate that appreciable sulfate reduction is taking place at Waltowa only in these sediments.

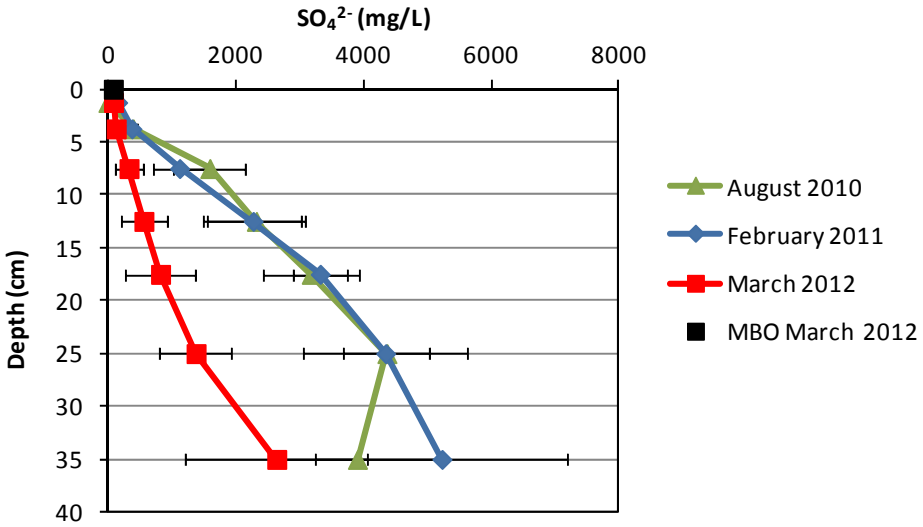

Figure 5-61. Waltowa pore-water sulfate concentrations at the established *Phragmites* site (August 2010, February 2011 and March 2012).

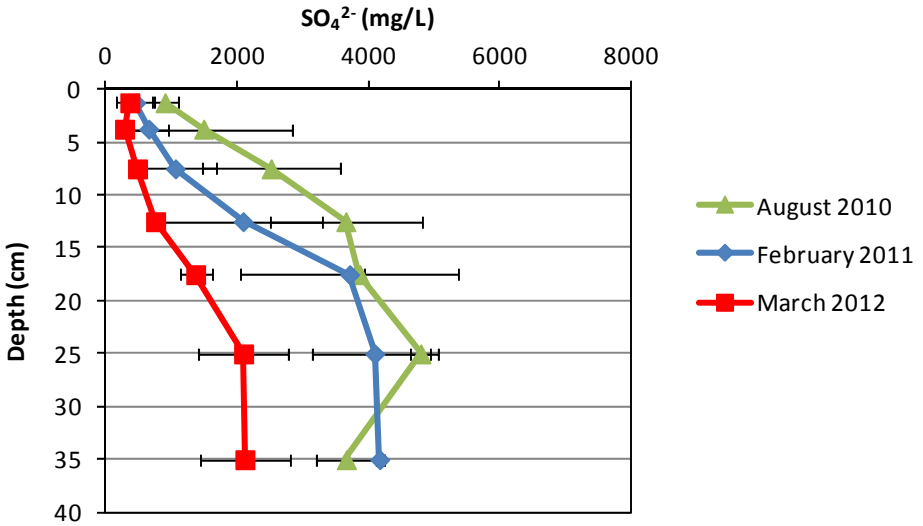

Figure 5-62. Waltowa pore-water sulfate concentrations at the established *Cotula* site (August 2010, February 2011 and March 2012).

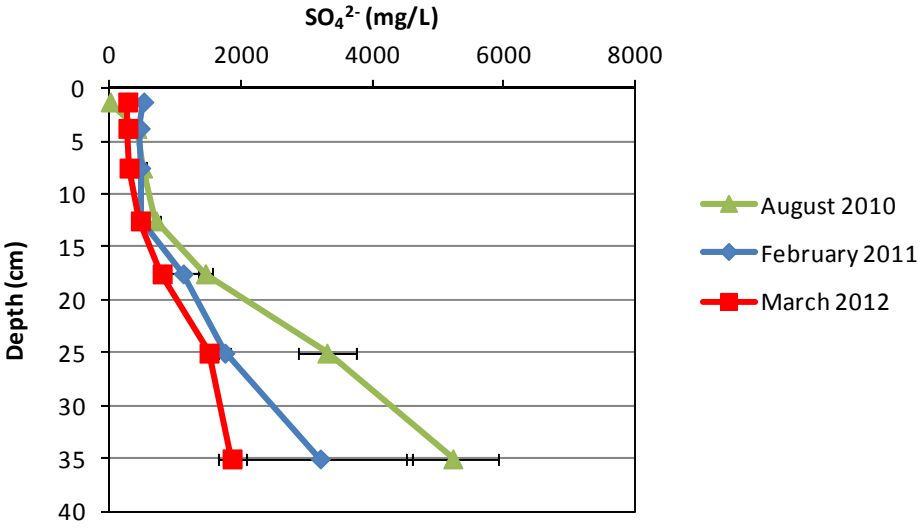

Figure 5-63. Waltowa pore-water sulfate concentrations at the established *Juncus* site (August 2010, February 2011 and March 2012).

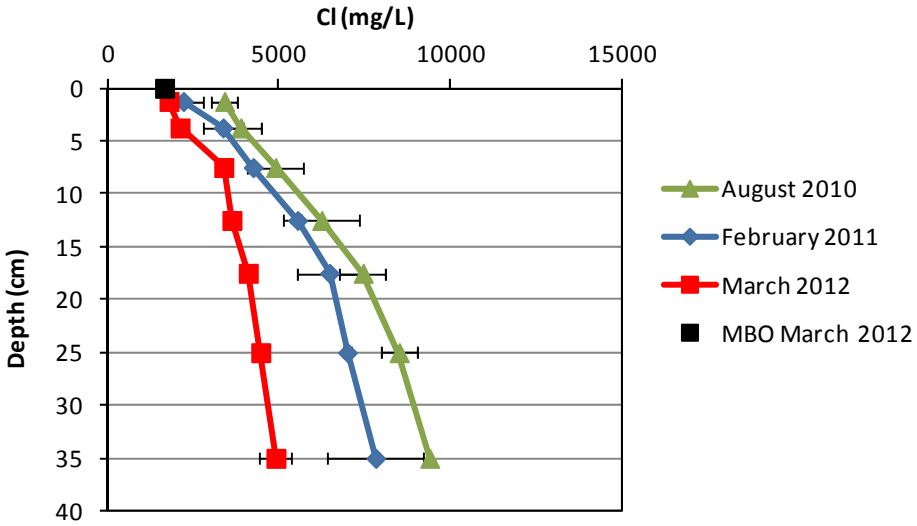

Figure 5-64. Waltowa pore-water chloride concentrations at the established *Phragmites* site (August 2010, February 2011 and March 2012).

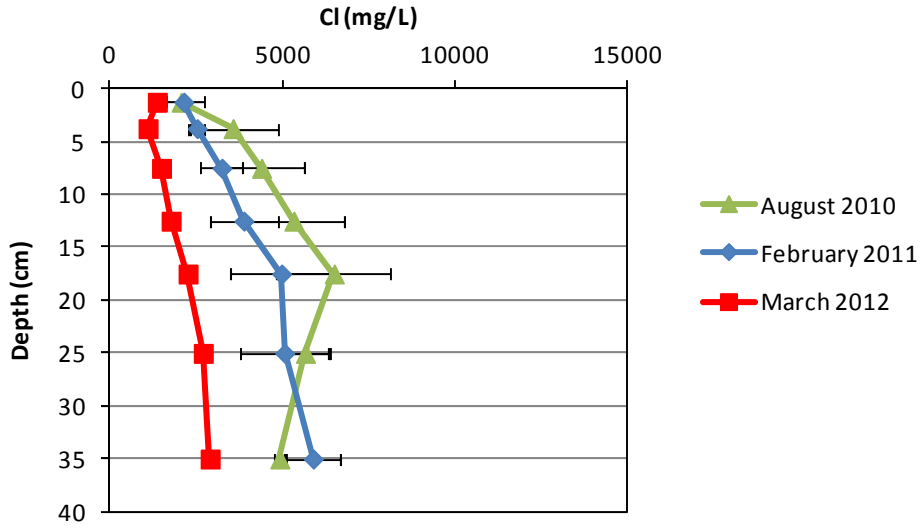

Figure 5-65. Waltowa pore-water chloride concentrations at the established *Cotula* site (August 2010, February 2011 and March 2012).

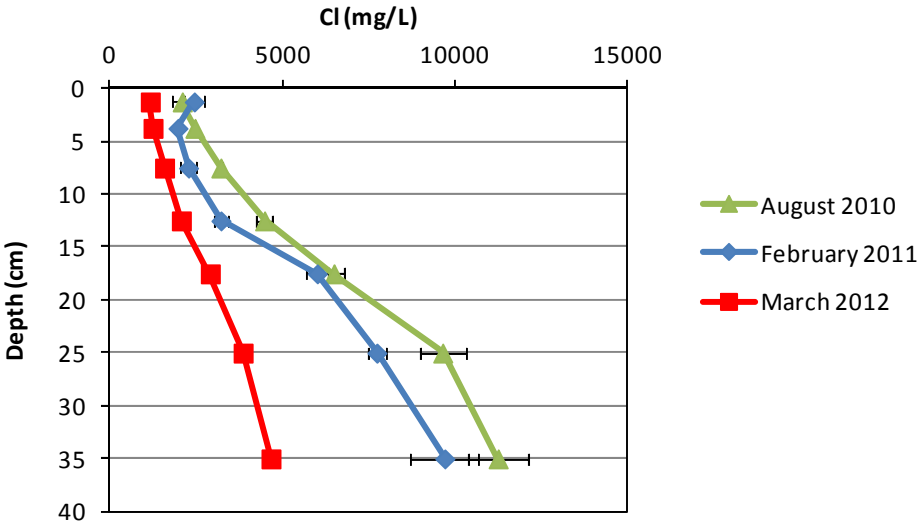

Figure 5-66. Waltowa pore-water chloride concentrations at the established *Juncus* site (August 2010, February 2011 and March 2012).

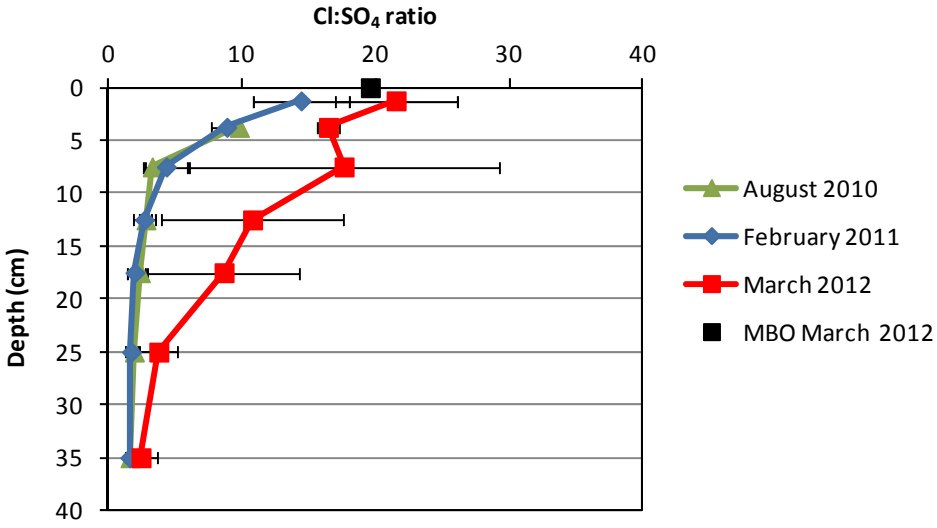

Figure 5-67. Waltowa pore-water chloride/sulfate ratios at the established *Phragmites* site (August 2010, February 2011 and March 2012).

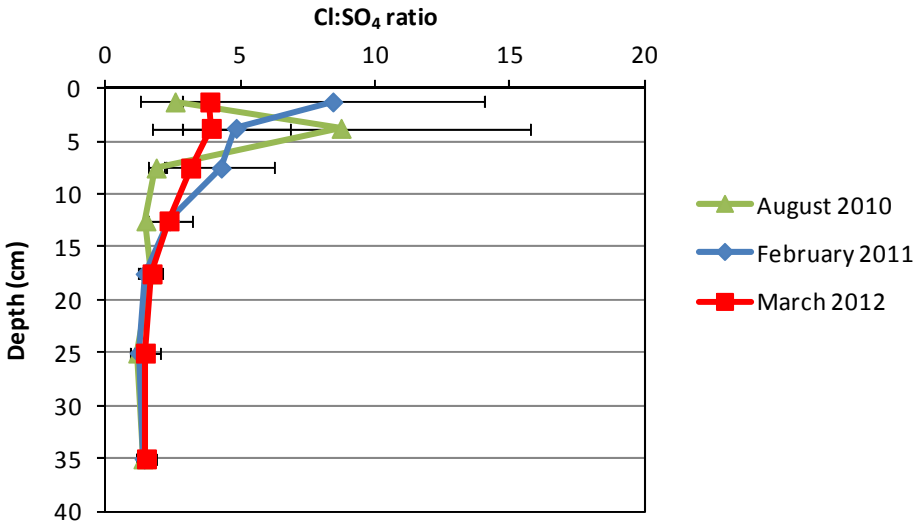

Figure 5-68. Waltowa pore-water chloride/sulfate ratios at the established *Cotula* site (August 2010, February 2011 and March 2012).

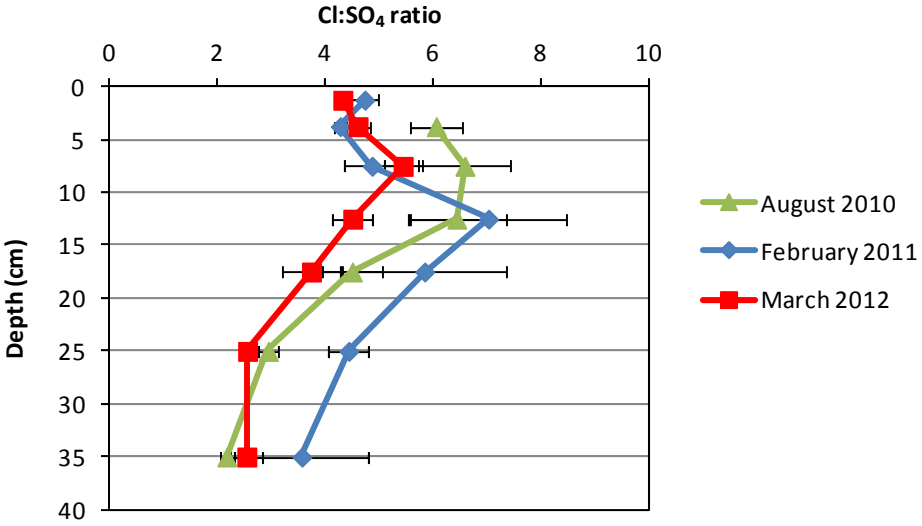

Figure 5-69. Waltowa pore-water chloride/sulfate ratios at the established *Juncus* site (August 2010, February 2011 and March 2012).

Graphs of all the pore-water data collected in March 2012 are presented in Appendix 5 (Figures 9-1 to 9-64).

5.2.2 Poltalloch

Figures 5-70 - 5-71 show that in general the concentrations of sulfate and chloride in the sediment have decreased appreciably at this site during the inundation period.

The very low Cl:SO_4 ratios in most of these soils (Figure 5-72) initially and post inundation indicate sulfide oxidation has taken place in these sediments and that negligible sulfate reduction has occurred. The initially very high Cl:SO_4 ratio in the uppermost layer at this site is likely the presence of chloride evaporite minerals that accumulated on the sediment surface during the lengthy period of sediment exposure resulting from the extended drought affecting this region prior to August 2010.

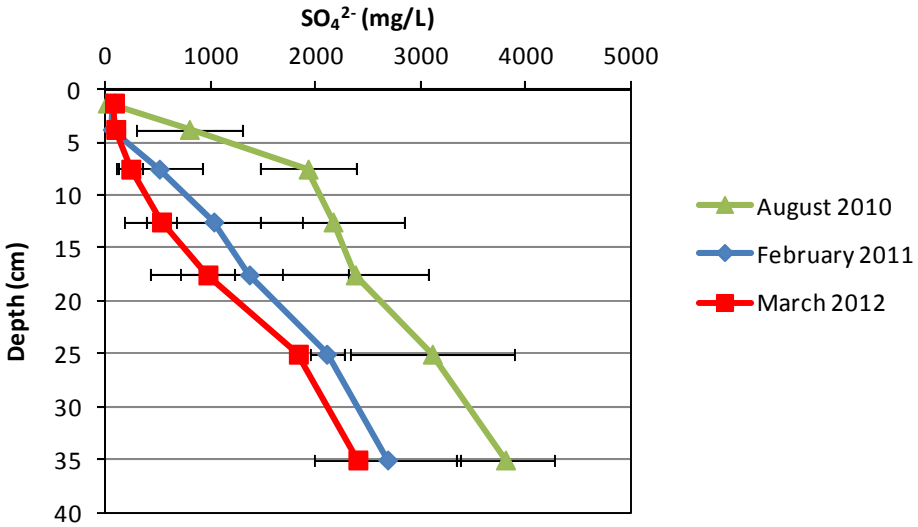

Figure 5-70. Poltalloch pore-water sulfate concentrations at the Bevy rye site (August 2010, February 2011 and March 2012).

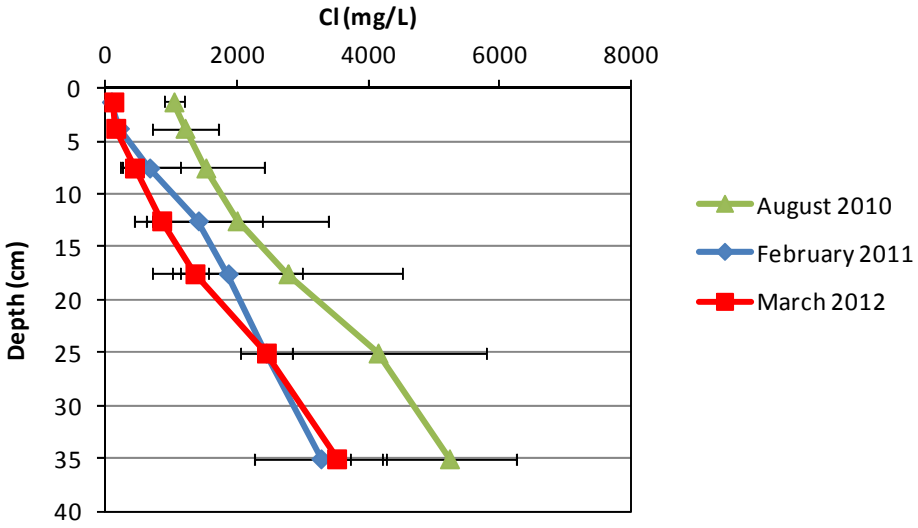

Figure 5-71. Poltalloch pore-water chloride concentrations at the Bevy rye site (August 2010, February 2011 and March 2012).

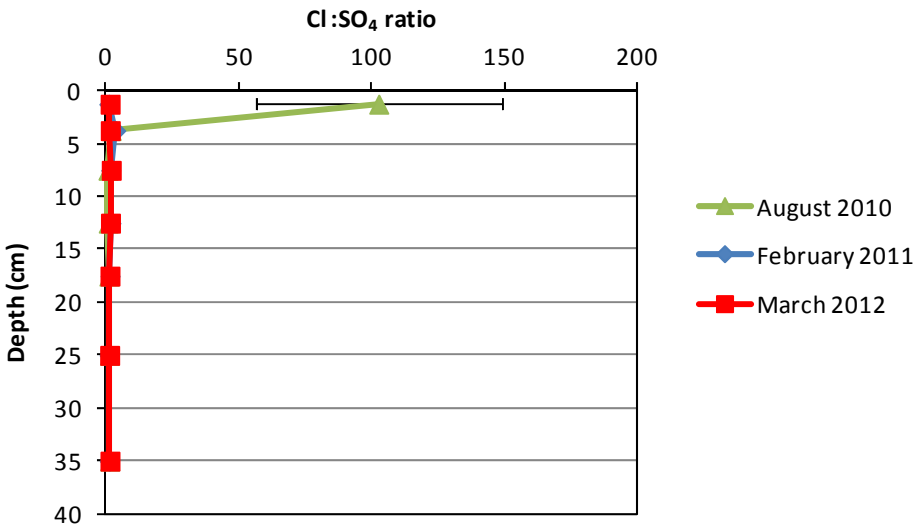

Figure 5-72. Poltalloch pore-water chloride/sulfate ratios at the Bevy rye site (August 2010, February 2011 and March 2012).

Graphs of all the pore-water data collected in March 2012 are presented in Appendix 5 (Figures 9-1 to 9-64).

5.2.3 Tolderol

Figures 5-73 - 5-76 show that in general the concentrations of sulfate and chloride in the sediment have decreased appreciably at this study area during the inundation period.

The very low Cl:SO_4 ratios in most of these soils (Figure 5-77 – 5-78) initially and post inundation indicate sulfide oxidation has taken place in these sediments and that negligible sulfate reduction has occurred. The initially high Cl:SO_4 ratio in the uppermost layer at the Bevy Rye site is likely the presence of chloride evaporite minerals that accumulated on the sediment surface during the lengthy period of sediment exposure resulting from the extended drought affecting this region prior to August 2010.

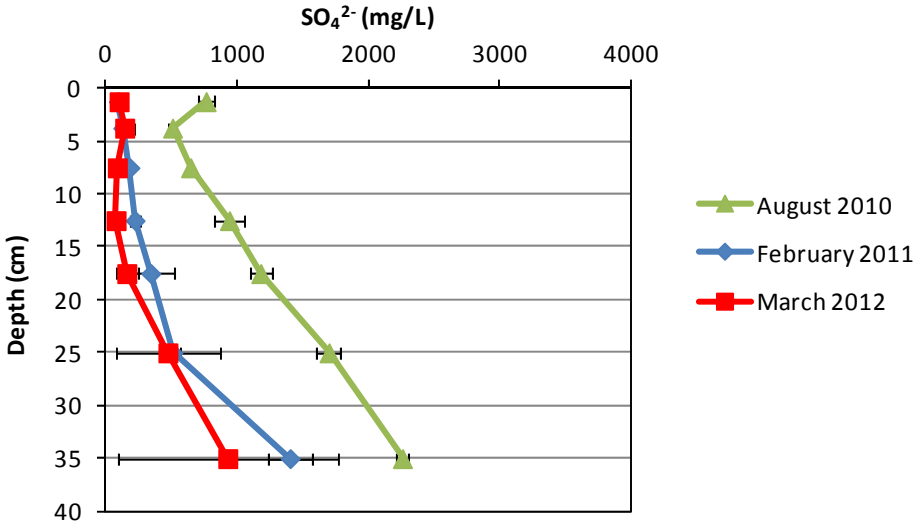

Figure 5-73. Tolderol pore-water sulfate concentrations at the control site (August 2010, February 2011 and March 2012).

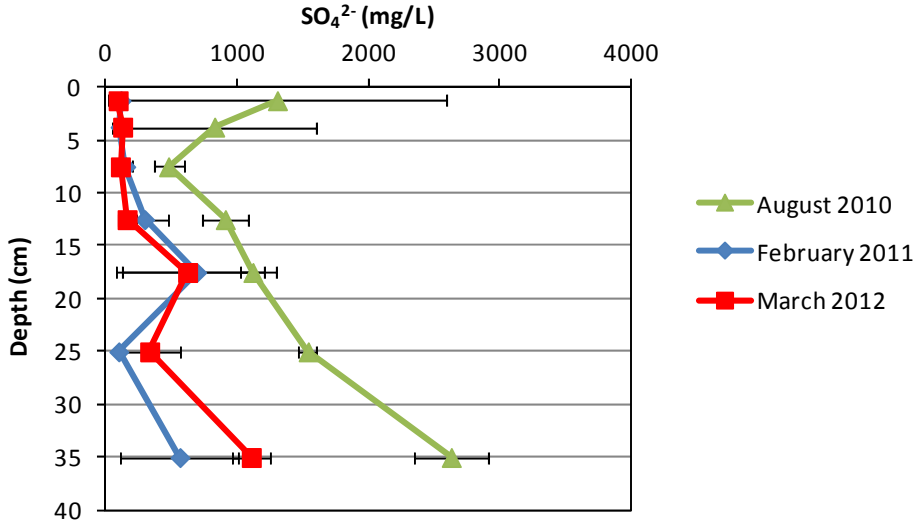

Figure 5-74. Tolderol pore-water sulfate concentrations at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

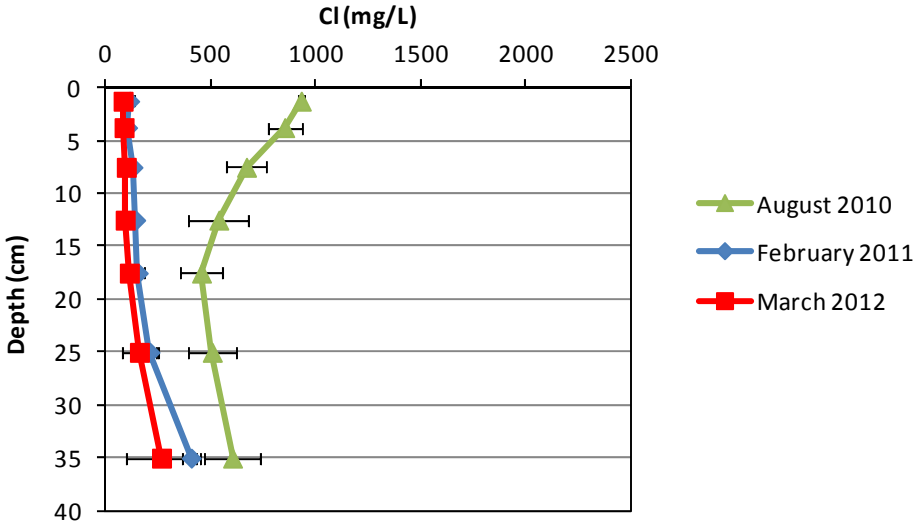

Figure 5-75. Tolderol pore-water chloride concentrations at the control site (August 2010, February 2011 and March 2012).

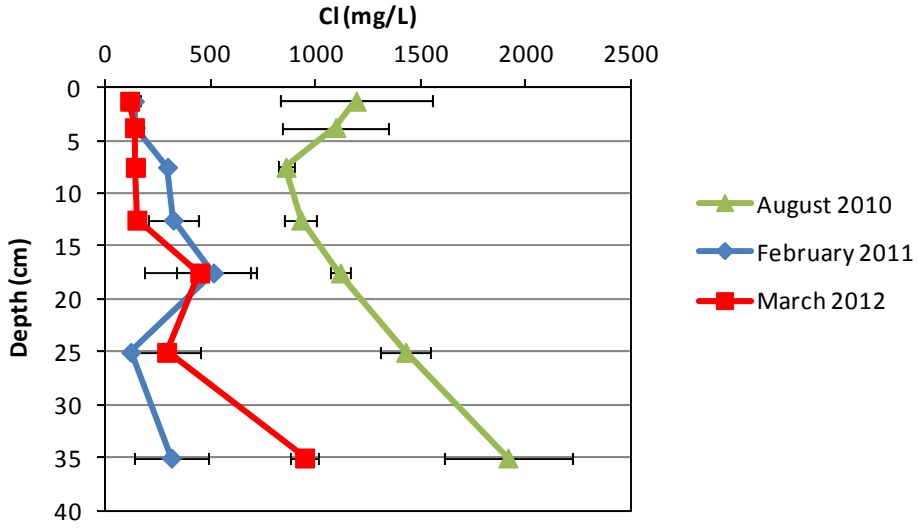

Figure 5-76. Tolderol pore-water chloride concentrations at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

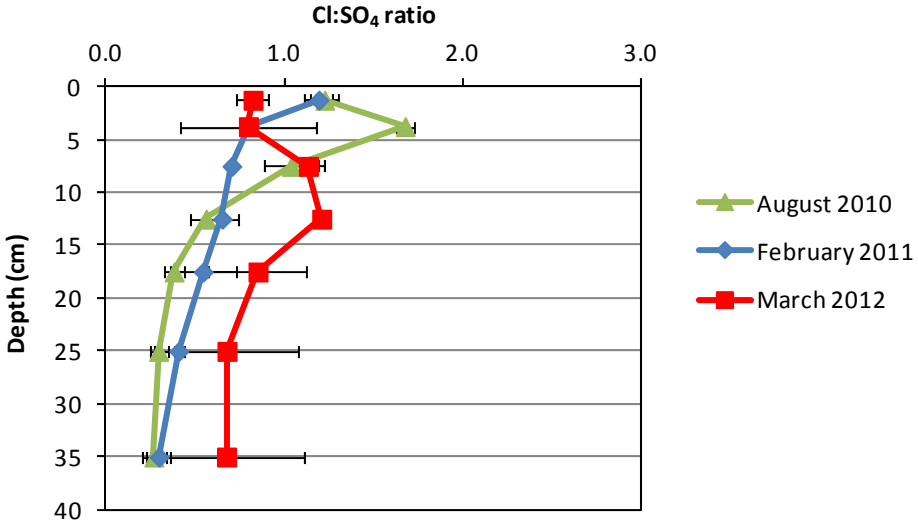

Figure 5-77. Tolderol pore-water chloride/sulfate ratios at the control site (August 2010, February 2011 and March 2012).

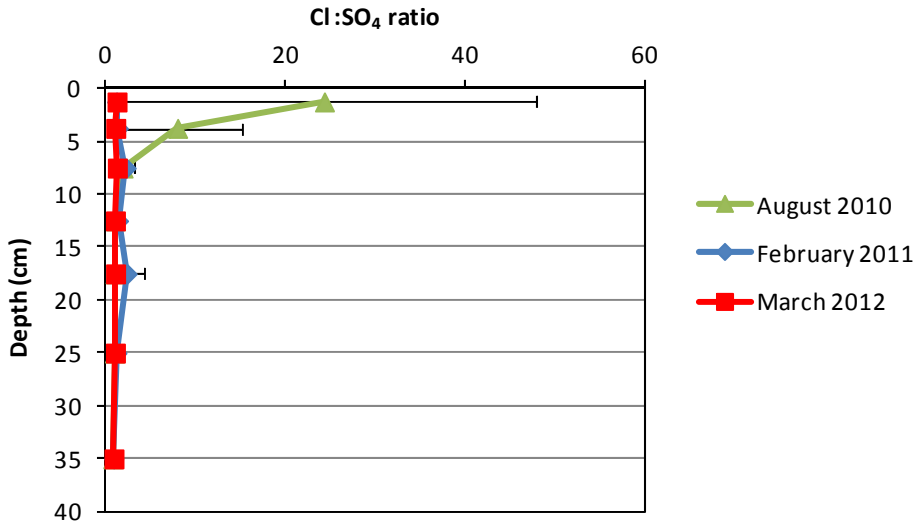

Figure 5-78. Tolderol pore-water chloride/sulfate ratios at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

Graphs of all the pore-water data collected in March 2012 are presented in Appendix 5 (Figures 9-1 to 9-64).

5.2.4 Campbell Park

Figures 5-79 - 5-82 show that in general the concentrations of sulfate and chloride in the sediment have decreased appreciably at this study area during the inundation period.

The very low Cl:SO_4 ratios in most of these soils (Figure 5-83 – 5-84) initially and post inundation indicate sulfide oxidation has taken place in these sediments. The appreciable increase in the Cl:SO_4 ratios in the uppermost sediment layers at both sites indicates that sulfate reduction has occurred in both sites since inundation.

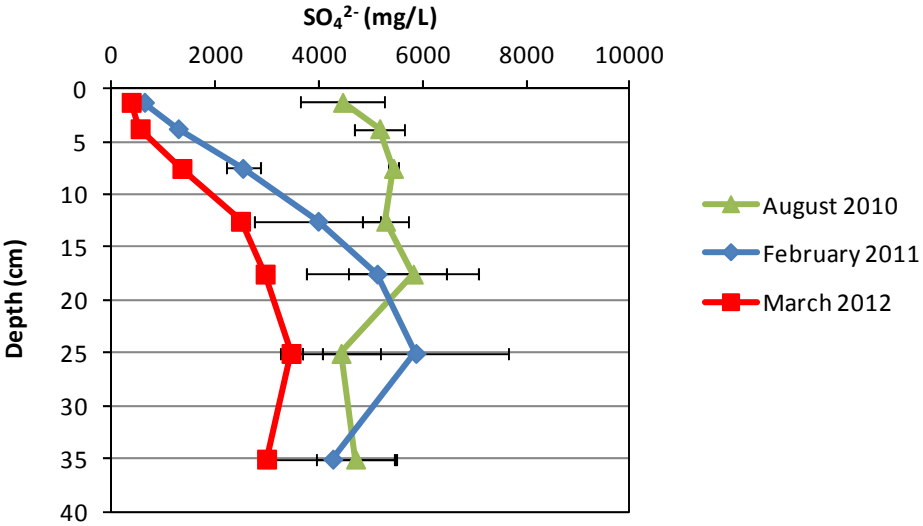

Figure 5-79. Campbell Park pore-water sulfate concentrations at the control site (August 2010, February 2011 and March 2012).

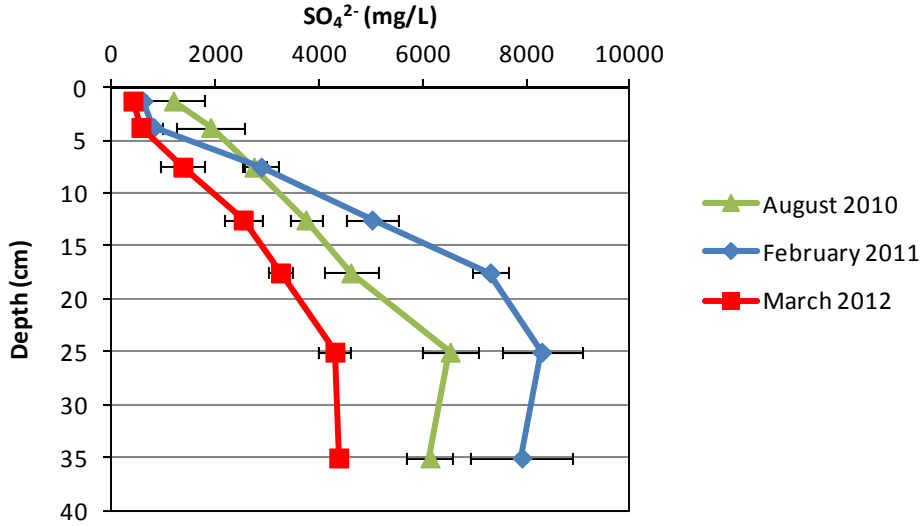

Figure 5-80. Campbell Park pore-water sulfate concentrations at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

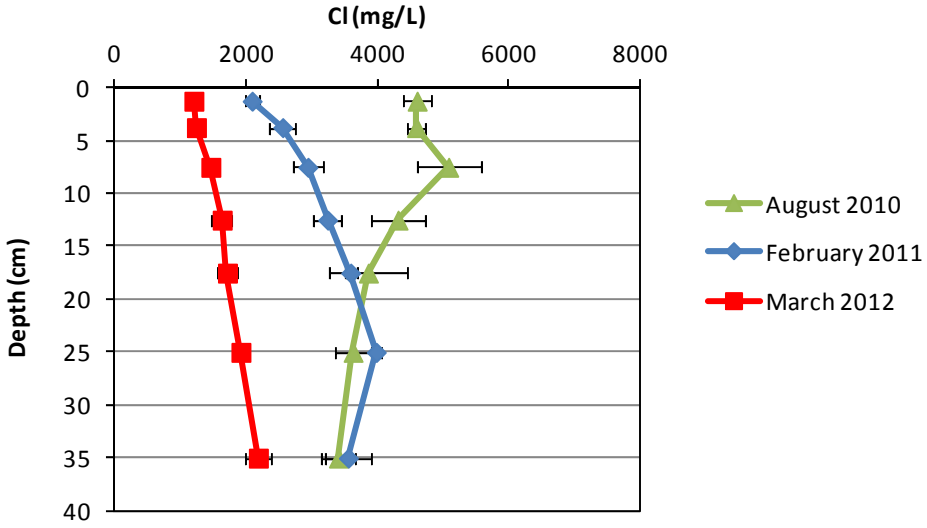

Figure 5-81. Campbell Park pore-water chloride concentrations at the control site (August 2010, February 2011 and March 2012).

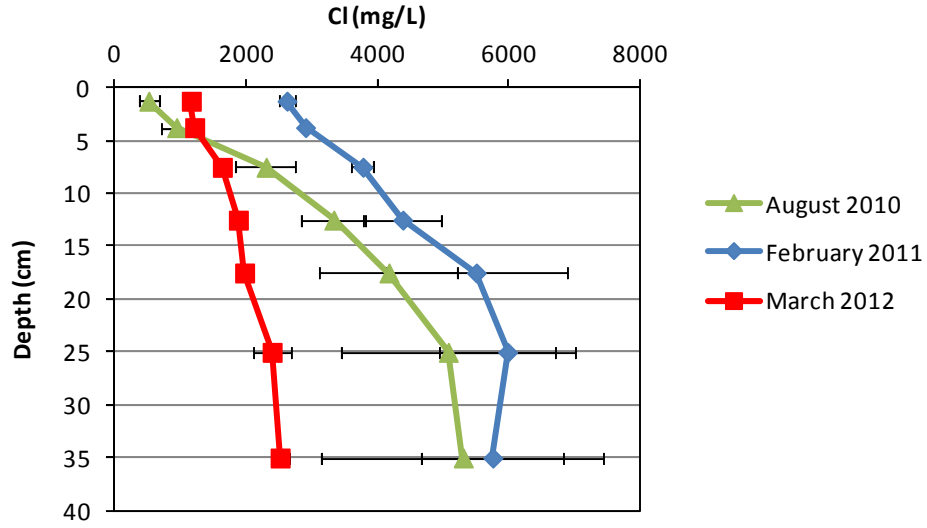

Figure 5-82. Campbell Park pore-water chloride concentrations at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

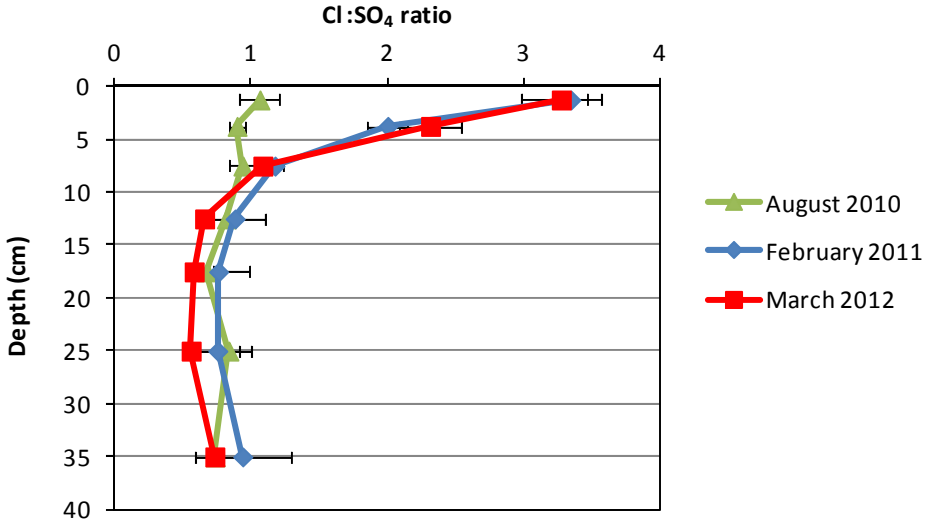

Figure 5-83. Campbell Park pore-water chloride/sulfate ratios at the control site (August 2010, February 2011 and March 2012).

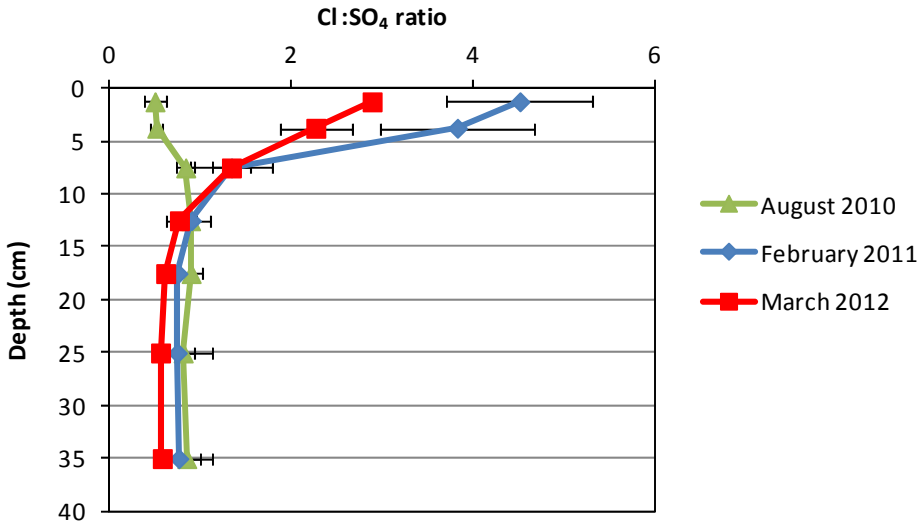

Figure 5-84. Campbell Park pore-water chloride/sulfate ratios at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

Graphs of all the pore-water data collected in March 2012 are presented in Appendix 5 (Figures 9-1 to 9-64).

5.3 Sulfate reduction rates

5.3.1 Waltowa sulfate reduction rates

The sulfate reduction rates measured at the three Waltowa sites between August 2010 and March 2012 are shown below in Figures 5-85, 5-86 and 5-87.

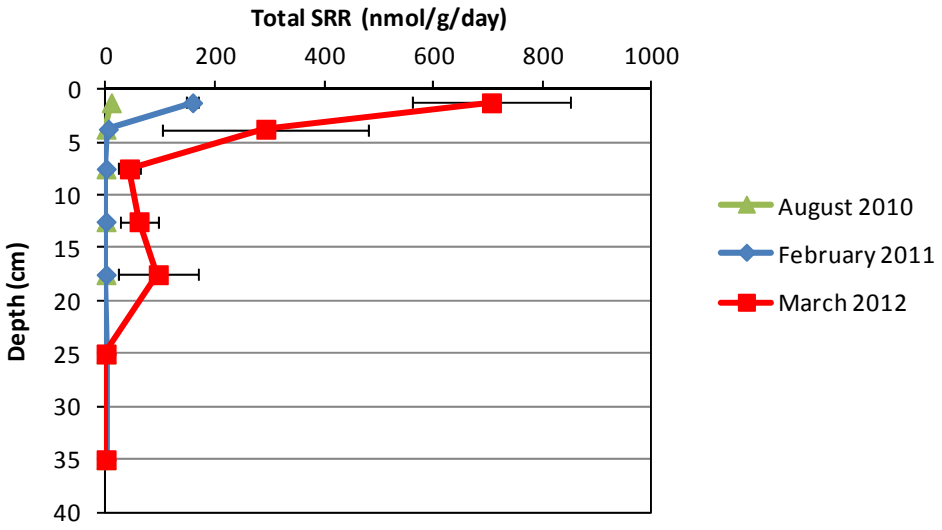

Figure 5-85. Waltowa sulfate reduction rates (nmol/g/day) at the established *Phragmites* site (August 2010, February 2011 and March 2012).

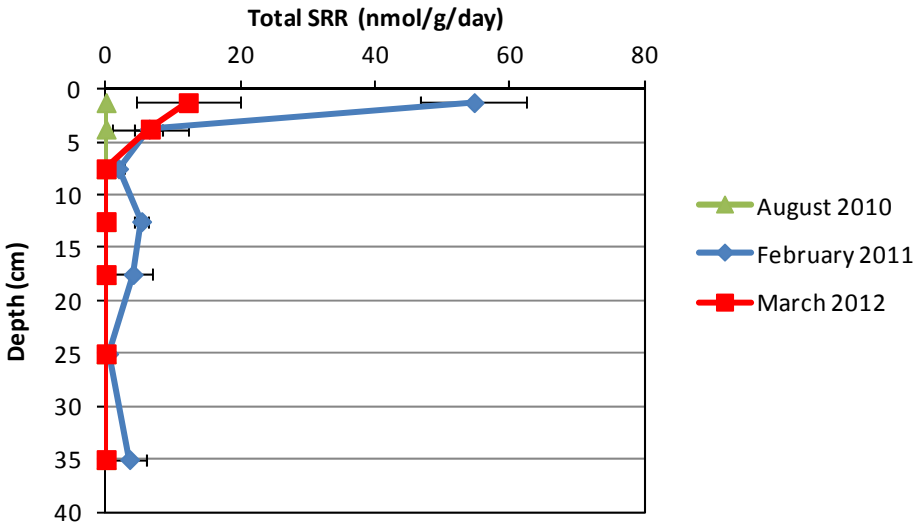

Figure 5-86. Waltowa sulfate reduction rates (nmol/g/day) at the established *Cotula* site (August 2010, February 2011 and March 2012).

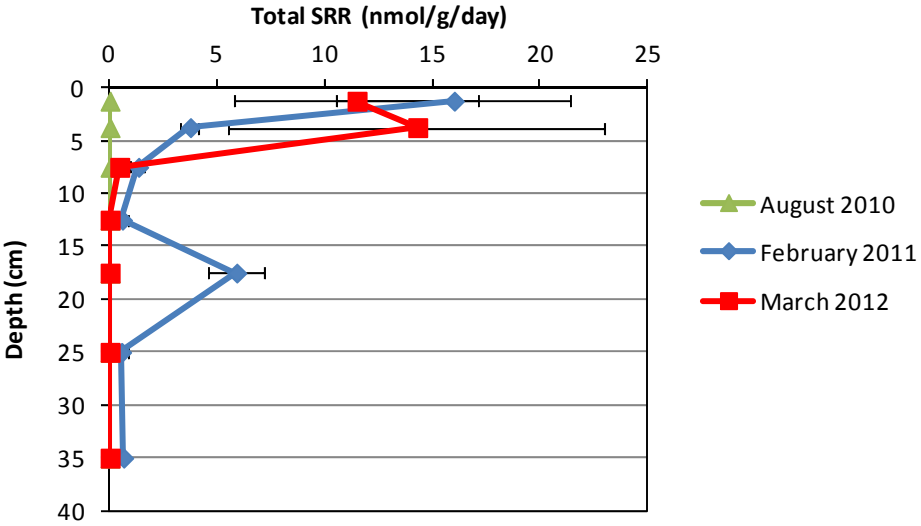

Figure 5-87. Waltowa sulfate reduction rates (nmol/g/day) at the established *Juncus* site in (August 2010, February 2011 and March 2012).

Figures 5-85 – 5-87 clearly show:

- Sulfate reduction was mainly limited to the 0 - 20 cm layers of the sediment at each treatment site.
- Sulfate reduction in the near inundating conditions in the August 2010 assessment only allowed limited sulfate reduction and only in the *Phragmites* treatment and only at relatively low rates (i.e. ~10 nmol g^{-1} day^{-1}).
- 6 months of inundation (February 2011 assessment) allowed appreciable sulfate reduction to occur in the 0 - 2.5 cm layer of all treatments, but with much higher rates in the *Phragmites* site (i.e. 170 nmol g^{-1} day^{-1}) than in the *Cotula* treatment (32 nmol g^{-1} day^{-1}) or the *Juncus* treatment (~15 nmol g^{-1} day^{-1}).
- 19 months of months of inundation (March 2012 assessment) allowed sulfate reduction to be further enhanced at the *Phragmites* site at rates of up to 700 nmol g^{-1} day^{-1} and within the 0 - 20 cm layer (Figure 5-85).
- In contrast 19 months of months of inundation resulted in further restricted sulfate reduction in the sediments under both the *Cotula* and *Juncus* treatments (~15 nmol g^{-1} day^{-1}) (Figures 5-86 – 5-87).
- Figure 5-88 shows the strong effect of the *Phragmites* vegetation on the rates of sulfate reduction in the sediments after 19 months of inundation.

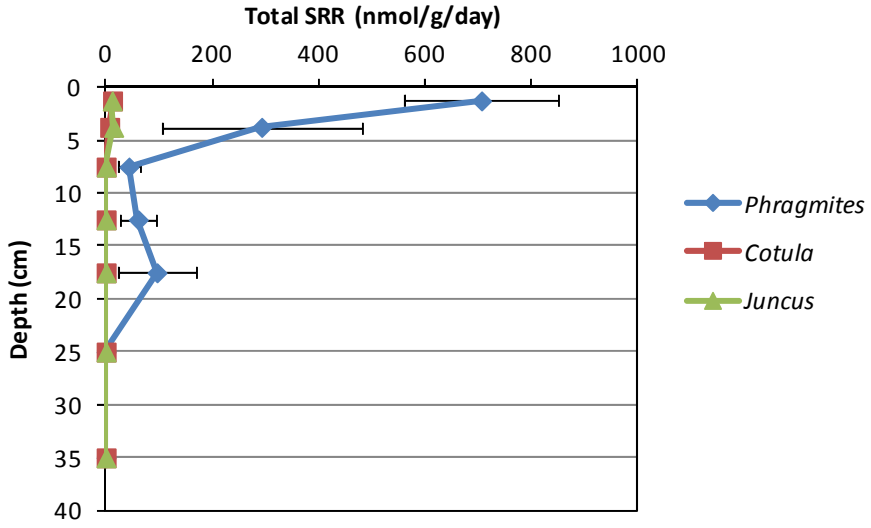

Figure 5-88. Waltowa sulfate reduction rates (nmol/g/day) at the established *Phragmites*, *Cotula* and *Juncus* sites in March 2012.

Figures 5-89 to 5-91 show that during the sulfate reduction assessment in March 2012 that the bulk of the sulfate that was reduced ended up in the form of elemental sulfur, although there was some production of monosulfides (as measured as Acid Volatile Sulfur (AVS)) and pyrite during the 24 hour assessment period.

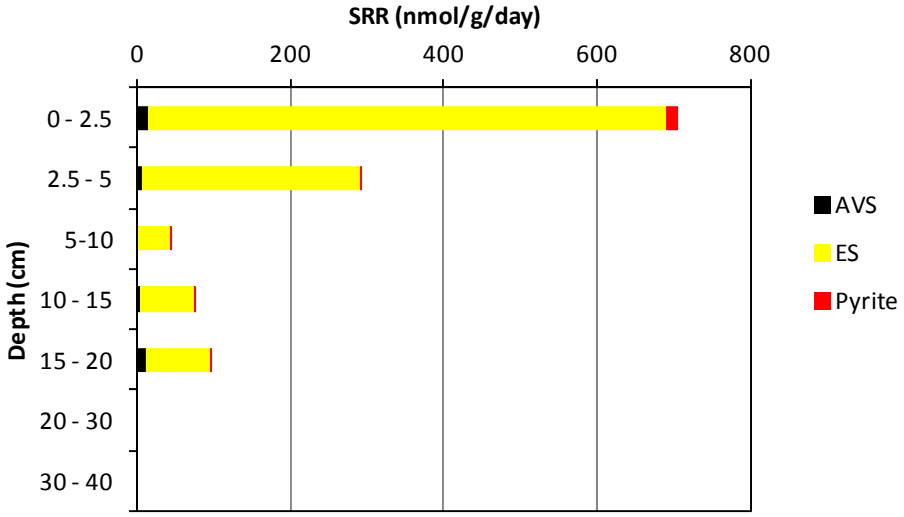

Figure 5-89. Products of sulfate reduction at the established *Phragmites* site, Waltowa (March 2012).

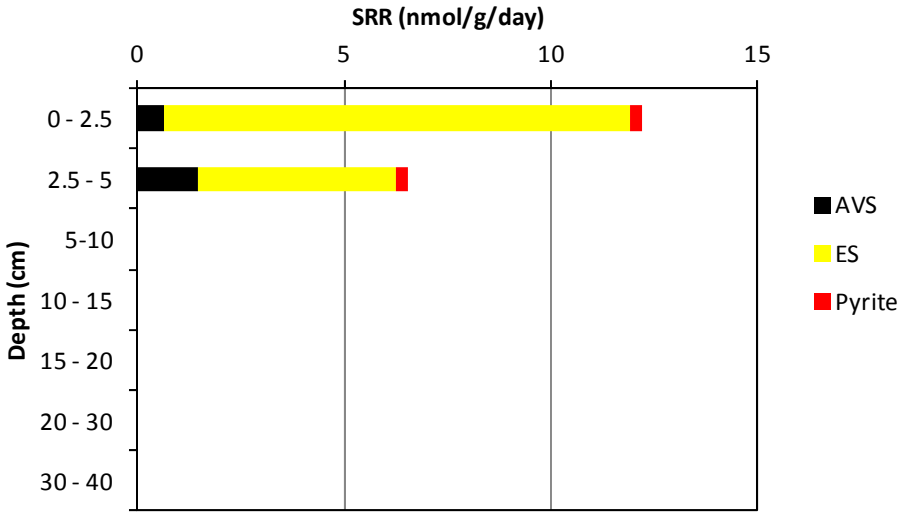

Figure 5-90. Products of sulfate reduction at the established *Cotula* site, Waltowa (March 2012).

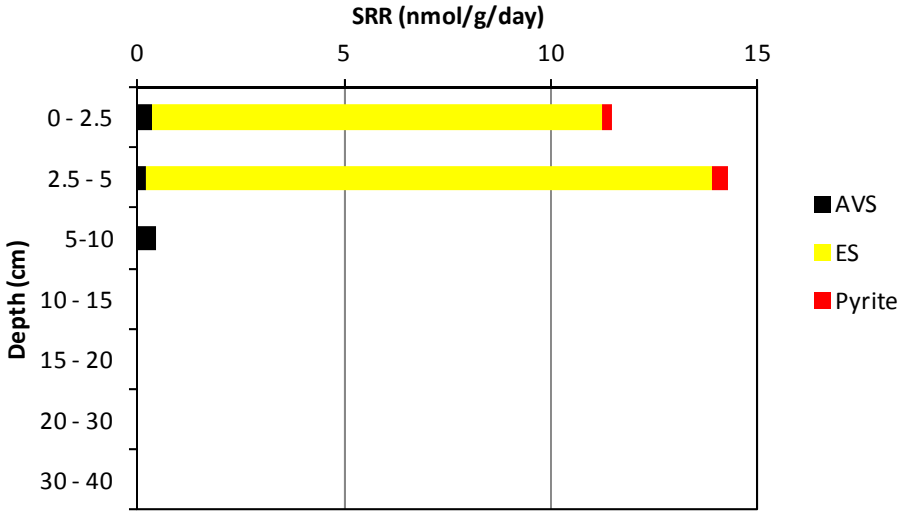

Figure 5-91. Products of sulfate reduction at the established *Juncus* site, Waltowa (March 2012).

The total organic matter contents at these sites (see Figures 5-20, 5-22 and 5-24 in Section 5.1.1.5) indicate that after inundation occurred at this location the concentration of organic matter in the surficial layers has continued to decrease substantially in the *Juncus* (from 2.3% TOC in August 2010 to 0.9% TOC in February 2011 to ~0.2% TOC in March 2012) and *Cotula* (from 3.5% TOC in August 2010to 2.1% TOC in February 2011 to ~0.2% TOC in March 2012)) treatments. Some of this decomposition was no doubt the result of via sulfate reduction as shown previously. In the *Phragmites* treatment, in contrast to the other two treatments, the presence of this vigorous growing vegetation instead of the completely inundated and decomposing *Cotula* and *Juncus* organic matter caused the TOC concentration to be essentially maintained in this surficial sediment layer with only a minor decrease of TOC from 3.9% TOC in August 2010 to 3.7% TOC in February 2011 and a further and more substantial decrease to 1.4% TOC in March 2012.

The hydrolysable C data for these three sites are also shown in Section 5.1.1.5 (Figures 5-21, 5-23 and 5-25) and parallels the amounts and trends for total organic matter. Hydrolysable carbon is regarded as available carbon and in decaying vegetation such as under recent flooded vegetation as in the experimental sites, is a balance between losses arising from processes such as sulfate reduction and gains due to lysis of vegetative material. Under these conditions both hydrolysable carbon and total organic carbon are desirable if sulfate reduction is desired as the hydrolysable carbon contents inform on the availability of organic matter for sulfate reduction, whereas the total organic carbon gives a better view on the net accumulation or decomposition of organic matter.

The pore-water properties in the inundation cores sampled from the *Phragmites* site (Figure 5-67) clearly indicate that sulfate was being consumed by sulfate reduction during the prolonged periods of the inundation. The increase in $Cl:SO_4$ ratio was more prominent in the *Phragmites* treatment than in the other two treatments in line with the ^{35}S-sulfate reduction data discussed previously. These data suggest that it is likely that at Waltowa for the *Phragmites* treatment that a ready source of sulfate could possibly constrain sulfate reduction in the future.

5.3.2 Sulfate reduction rates at Poltalloch

The sulfate reduction rates measured at the Poltalloch site between August 2010 and March 2012 are shown below in Figure 5-92.

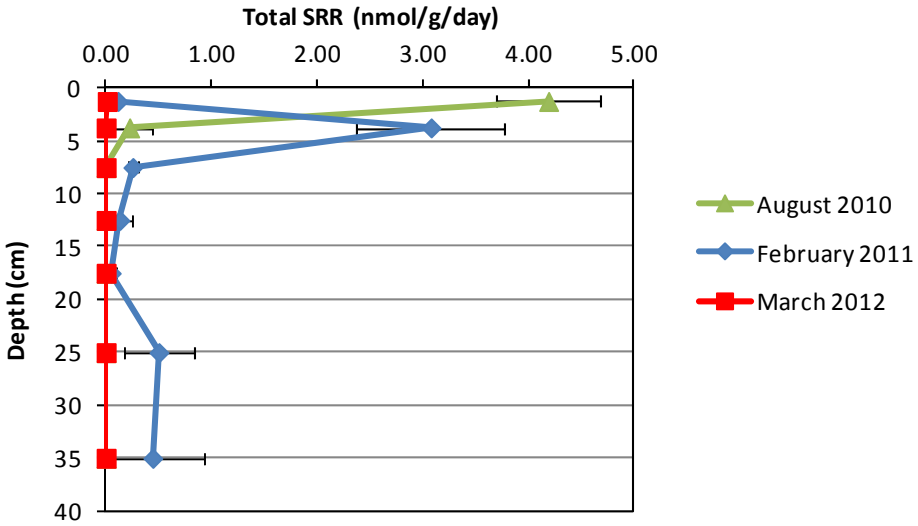

Figure 5-92. Poltalloch sulfate reduction rates (nmol/g/day) at the Bevy rye only site (August 2010, February 2011 and March 2012).

Figure 5-92 clearly show:

- Sulfate reduction was negligible during the March 2012 assessment period and declined considerably from even the very low rates of sulfate reduction observed during the two earlier assessment periods.
- The product of the sulfate reduction during the March 2012 assessment period was a very low rate of AVS production in the 0 – 2.5 cm layer

Figure 5-93 show that during the sulfate reduction period in March 2012 that the small amount of sulfate that was reduced ended up in the form of monosulfides (as measured as Acid Volatile Sulfur (AVS)).

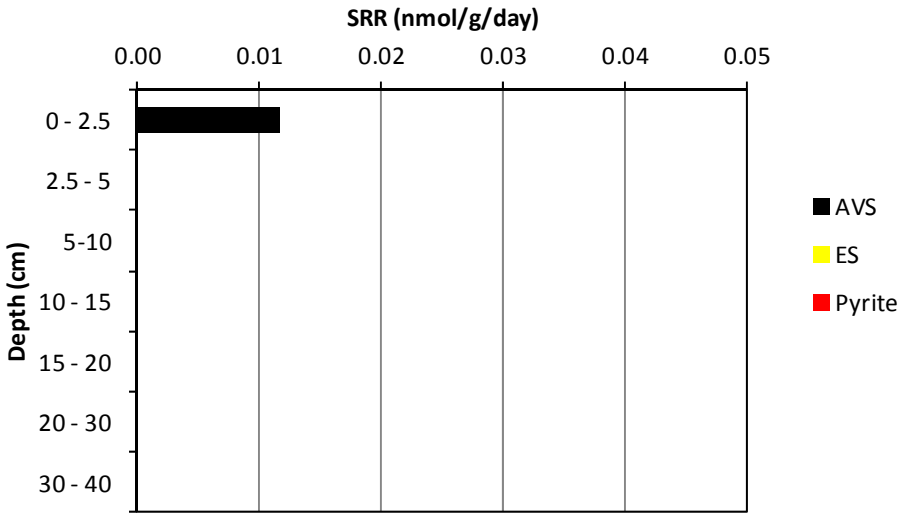

Figure 5-93. Products of sulfate reduction at the Bevy rye only site, Poltalloch (March 2012).

The total organic matter contents at this site (see Figures 5-31 and 5-32 in Section 5.1.2.5) indicate that after inundation occurred at this location the concentration of organic matter in the surficial layer decreased from a relatively low value of ~ 0.20% TOC and ~ 0.10% hydrolysable carbon in August, to 0.02 – 0.10 % TOC comprised mainly of hydrolysable carbon after prolonged inundation. Some of this organic matter decomposition was likely the result of sulfate reduction occurring in the first 6 months of inundation as shown previously.

It is most probable that the observed substantial decreases in sulfate reduction rates in February 2011 at this site compared to those observed in August 2010 are due to the near exhaustion of initially low organic matter content in these sediments, as a result of sulfate reduction since inundation in August 2010. In March 2012 the content of organic matter remained very low and apparently acted as the constraint on sulfate reduction at this site.

5.3.3 Sulfate reduction rates at Tolderol

The sulfate reduction rates measured at the two Tolderol sites August 2010 and March 2012 are shown below in Figures 5-94 and 5-95.

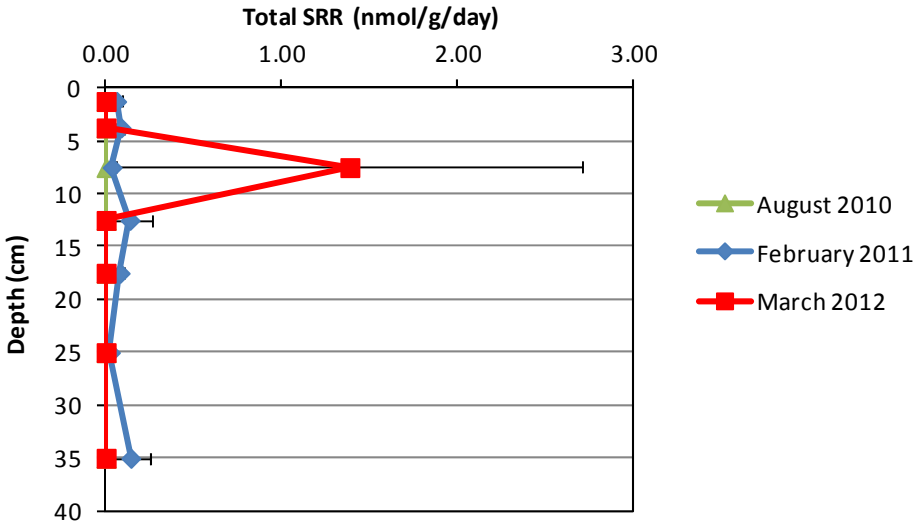

Figure 5-94. Tolderol sulfate reduction rates (nmol/g/day) at the control site (August 2010, February 2011 and March 2012).

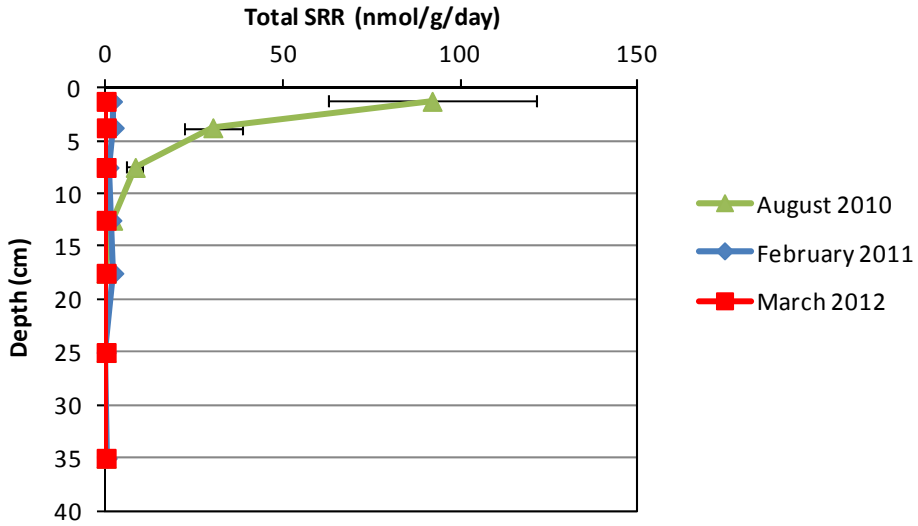

Figure 5-95. Tolderol sulfate reduction rates (nmol/g/day) at the *Juncus* in Bevy rye site (August 2010, February 2011 and March 2012).

Figures 5-94 and 5-95 clearly show:

- Sulfate reduction was negligible during the March 2012 assessment period. The effect of depletion of available organic matter on the sulfate rates can be readily observed by the decrease in sulfate reduction rates in the top 10 cm layers from August 2010 when organic matter was available to later sampling dates when organic matter had been depleted.

Figures 5-96 and 5-97 show that during the sulfate reduction period in March 2012 there was a very low rate of elemental sulfur and AVS production in the 5 – 10 cm layer at the control site.

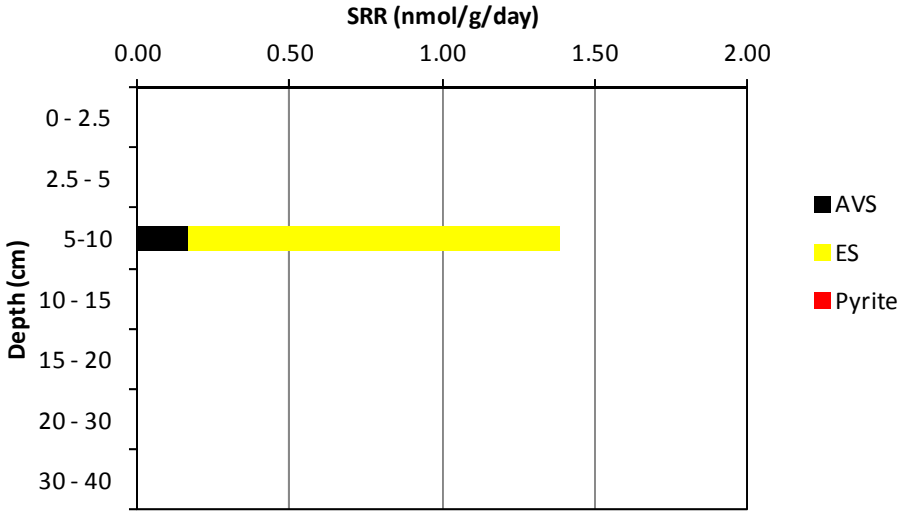

Figure 5-96. Products of sulfate reduction at the control site, Tolderol (March 2012).

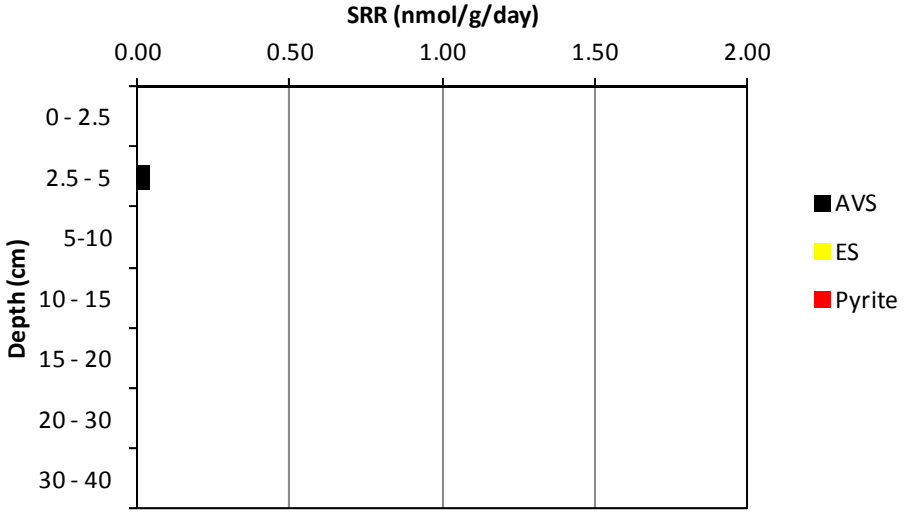

Figure 5-97. Products of sulfate reduction at the *Juncus* in Bevy rye site, Tolderol (March 2012).

The total organic matter contents at these sites (see Figures 5-43 and 5-45 in Section 5.1.3.5) indicate that after inundation occurred at this location the concentration of organic matter in the surficial layers decreased from a relatively low values of 0.15 - 0.20% TOC and ~ 0.10 – 0.20 % hydrolysable carbon in August 2010, to 0.02 – 0.10 % TOC comprised mainly of hydrolysable carbon after prolonged inundation. Some of this organic matter decomposition was likely the result of sulfate reduction occurring in the first 6 months of inundation as shown previously.

It is most probable that the observed substantial decreases in sulfate reduction rates in February 2011 at these sites compared to those observed in August 2010 are due to the near exhaustion of initially low organic matter content in these sediments, as a result of sulfate reduction since inundation in August 2010. In March 2012 the content of organic matter remained very low and apparently acted as the constraint on sulfate reduction at this study area.

5.3.4 Sulfate reduction rates at Campbell Park

The sulfate reduction rates measured at the two Campbell Park sites August 2010 and March 2012 are shown below in Figures 5-98 and 5-99.

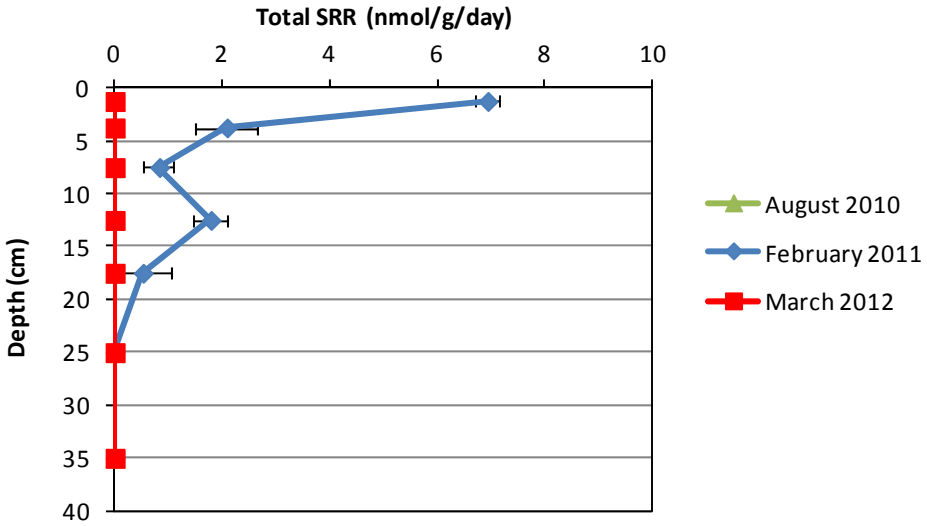

Figure 5-98. Campbell Park sulfate reduction rates (nmol/g/day) at the control site (August 2010, February 2011 and March 2012).

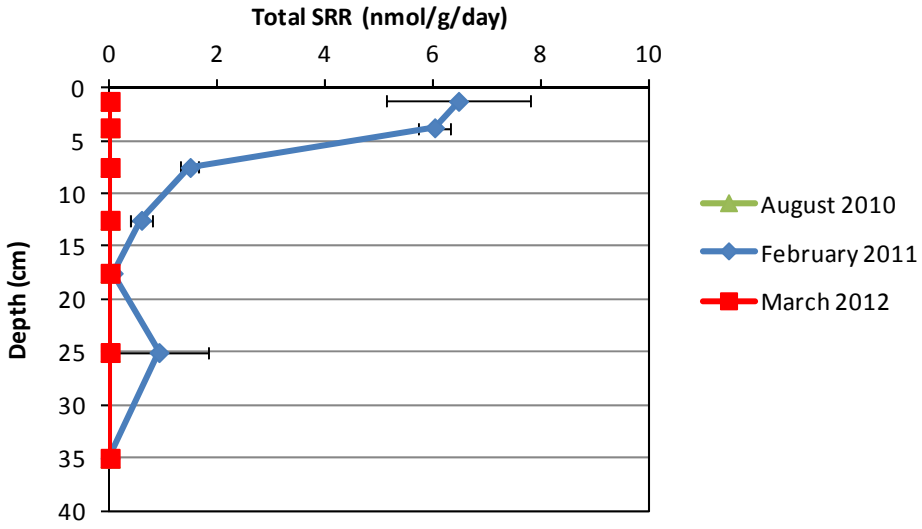

Figure 5-99. Campbell Park sulfate reduction rates (nmol/g/day) at the Bevy rye/*Puccinellia* site (August 2010, February 2011 and March 2012).

Figures 5-98 and 5-99 clearly show:

- Although very low rates were observed during the February 2011 assessment period there was no sulfate reduction apparent at either of the Campbell Park sites during the March 2012 assessment period.

The total organic matter contents at these sites (see Figures 5-57 and 5-59 in Section 5.1.4.5) indicate that after inundation occurred at this location the concentration of organic matter in the surficial layers decreased from a relatively low values of 0.15 - 0.20% TOC and ~ 0.10 – 0.20 % hydrolysable carbon in August 2010, to 0.06 – 0.10 % TOC comprised mainly of hydrolysable carbon after prolonged inundation. Some of this organic matter decomposition was likely the result of sulfate reduction occurring in the first 6 months of inundation as shown previously.

It is most probable that the observed substantial decreases in sulfate reduction rates in February 2011 at these sites compared to those observed in August 2010 are due to the near exhaustion of the initially low organic matter content in these sediments, as a result of sulfate reduction since inundation in August 2010. In March 2012 the content of organic matter remained very low and apparently acted as the constraint on sulfate reduction at this study area.

5.4 Discussion

5.4.1 Remediation of acidified sediment layers

The data from the March 2012 assessment clearly support the findings of Sullivan *et al.* (2011) showing that both the acidity and low pHs of many of the acidified acid sulfate sediment layers at the study areas are being remediated by a number of processes and sources that deliver organic carbon and alkalinity to these layers. These include:

- Movement of the alkalinity that is contained in the lake waters - and derived from the River Murray water - entering the sediment profile via either convective or diffusive processes. The unvegetated control sites at both Tolderol and Campbell Park where the other likely sources of alkalinity addressed in this study are either absent or negligible, are instructive in assessing the magnitude of the contribution of this source of alkalinity in the remediation of acidified acid sulfate sediments. Figures 5-33 and 5-47 both indicate that the diffusion of the substantial alkalinity in the lake waters is capable of causing appreciable increases in sediment pH down to 30 cm depth within a few months. For example, at Tolderol control site the pH of the 0 - 10 cm layer initially 2.4 prior to inundation (May 2010) and 3.0 after inundation (August 2010), rose to a pH of 5.7 by November 2010. Similarly, at Campbell Park control site the pH of the 0 - 10 cm layer initially 3.7 prior to inundation (August 2010), rose to a pH of 4.3 by November 2010 and then to 6.4 by February 2011. The data from this study confirms these trends with continued increases in pH evident at the March 2012 assessment at both of these sites especially in the deeper sediment layers. This data showing the neutralising trends in both of the unvegetated and formerly strongly acidified sites strongly suggests that even left unvegetated, these formerly degraded lake sediments will slowly remediate via the movement of alkalinity into the sediments if surface water is present.

- Alkalinity from added solid phase liming materials. Ultra-fine Robe lime was added to the surface layers at only two of the treatment sites; the *Juncus* and *Phragmites* treatments at Waltowa. The pH data from field sampling (Sullivan *et al.* 2011) clearly showed that this application resulted in elevated pHs in the top 0 – 20 cm sediment layers with pHs between 7 and 8.8 pre-inundation, much higher that the pHs of between 4.7 and 6.7 observed at this time in the unlimed *Cotula* treatment and Waltowa. However, the effect of such liming materials are often localised and this is exemplified by subsurface sediment layers at both the *Juncus* and *Phragmites* treatments at Waltowa having pHs < 5 during the study period.

- Alkalinity already existing in the sediments (i.e. the sediment's Acid Neutralising Capacity (ANC)). As may be expected from recently severely acidified surface soil layers the ANC contents of the 0 - 40 cm surficial layers were negligible apart from those two treatments discussed above that had received additions of Aglime and the sediments as Poltalloch down to 20 cm depth (where there were only minor sporadic occurrences of $CaCO_3$). Similarly, the Titratable Actual Alkalinity values of these surficial sediment layers were also generally very low. The data in Sullivan *et al. (*2011) indicated that the ANC of the exposed acidified lake sediments was negligible (unless Aglime had been applied) and unable to supply alkalinity to remediate acidity upon lake refilling.

- Alkalinity derived from sulfate reduction. The data in Sullivan *et al. (*2011) and in this study clearly show that sulfate reduction has taken place where organic materials have been added to the lake sediments by revegetation. Whether this process results in the net production of alkalinity is anymore than only minor or ephemeral quantities will be discussed in a later section (see Section 5.4.2).

Sullivan *et al.* (2011) clearly showed that there are two main constraints against sulfate reduction in the formerly exposed lake sediments:

1. Lack of organic matter, and
2. Severely acidified (i.e. pH < 4) sediment layers.

The severe acidification constraint was discussed at length in Sullivan *et al.* (2011) and only the organic matter constraint will be addressed here.

Organic matter
Organic matter in the lake sediments (as noted previously in Sullivan *et al.* 2010a & 2011) is very low. The data in this study demonstrates that the organic matter content of these sediments has been greatly increased by the bioremediation program using the establishment of vegetation. However, different vegetation produces organic matter in different amounts and over different time periods.

For example, annual plants like Bevy rye produce appreciable amounts of vegetation relatively quickly but then die, leaving dead dry straw-like residue. In contrast perennial plants like *Phragmites* and *Juncus* continue to produce organic matter: it is clear from the Waltowa study area that *Phragmites* could successfully resist prolonged flooding to greater inundation depths than the *Juncus* species planted at this location.

These different patterns of organic matter production and capacity clearly affected sulfate reduction rates. This is best shown by both the TOC and sulfate reduction data from the Waltowa study area.

Waltowa: The organic matter supply differed markedly between the three treatments (*Phragmites, Cotula and Juncus*) at Waltowa. The *Phragmites* treatment continued to produce abundant organic matter and thus maintained high TOC levels even during the March 2012 assessment (Figure 5-20), whereas the production of organic matter in both the *Cotula* and *Juncus* treatments, ceased after substantial inundation (and certainly by the February 2011 and March 2012 assessments) resulting in TOC depletions in the uppermost sediment layers (Figures 5-22 and 5-24). The comparative sulfate reduction data for this study area (Figure 5-88) indicates that the ability of *Phragmites* to continue to supply the surficial 0 – 20 cm sediment layer with organic matter even after relatively deep inundation has greatly enhanced the rates of sulfate reduction as compared to the two other vegetation types that had completely submerged, died and undergone decomposition soon after inundation in August 2010.

5.4.2 The nature of sulfur cycling and organic matter decomposition during the initial inundation of the Lower Lakes sediments

It is clear that the provision of organic matter in these sediments by revegetation has generally resulted in sulfate reduction to take place. The nature of the sulfate reduction process was dependent on the supply of organic matter that varied according to the type of vegetation used for bioremediation including such factors as whether the vegetation is perennial or annual, and importantly, whether the vegetation can maintain viability and productivity after inundation.

However, the data also indicate that the alkalinity supplied to the sediments by sulfate reduction during the assessment period generally occurs via a sulfur cycling process that will be only ephemeral and relatively minor. The data at all locations where sulfate reduction occurred at appreciable rates, indicate a common process of sulfate reduction in these sediments upon bioremediation by revegetation as described in Figure 5-100 below:

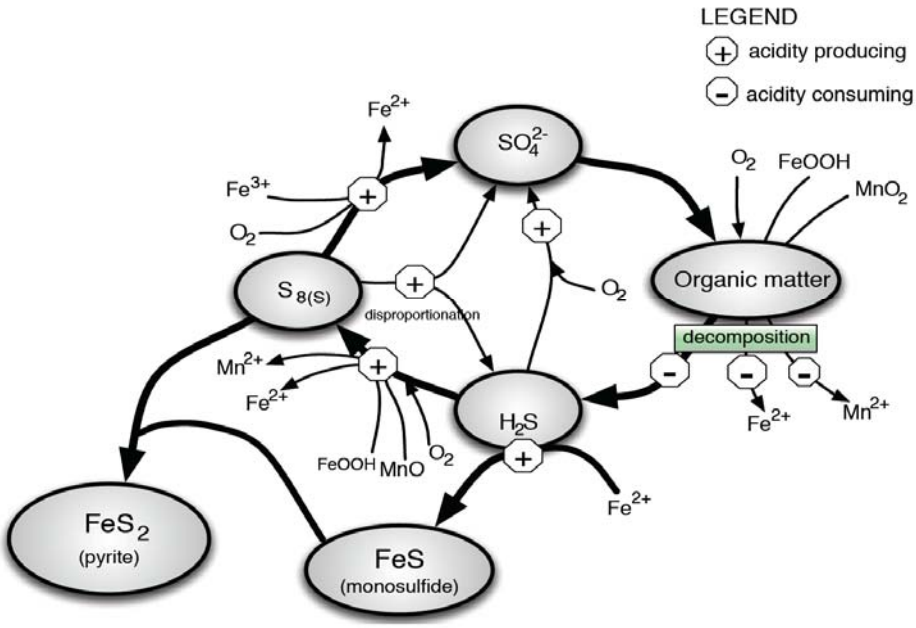

Figure 5-100. Conceptual diagram of sulfur cycle operating in the upper layers of the bioremediated inundated Lower Lakes sediments.

This conceptual diagram shows that sulfate is reduced during organic matter decomposition (often in microsites around the roots of the plants) used for bioremediation. The sulfide (e.g. H_2S) released from this process is mainly converted to elemental sulfur ($S_8^o{}_{(s)}$). This could be by either chemical means (reaction with O_2, or manganese and iron oxides) as per the equations below:

$$O_2 + 2H_2S \rightarrow \frac{1}{4} S_8^0{}_{(S)} + 2H_2O \quad [5.1]$$

$$2Fe(OH)_{3(s)} + H_2S + 4H^+ \rightarrow \frac{1}{8} S_8^0{}_{(S)} + 2Fe^{2+} + 6H_2O \quad [5.2]$$

$$MnO_{2(s)} + H_2S + 4H^+ \rightarrow \frac{1}{8} S_8^0{}_{(S)} + Mn^{2+} + 2H_2O \quad [5.3]$$

The data at some of the sites showing intense sulfate reduction during the February 2011 assessment (e.g. Tolderol vegetated site in the uppermost soil layer) did show gradual removal of manganese and iron upon prolonged inundation as well as greatly increased Fe^{2+} contents in the surfical sediment pore-waters after inundation (Sullivan *et al.* 2011). These findings are consistent with the process described in Equations 5.2 and 5.3 above operating. The observed decreases in both HCl-extractable manganese and iron such as observed in both Sullivan *et al.* (2011) and this study and the observed increases in Fe^{2+} concentrations in the sediment pore-waters may also have occurred as a result of manganese oxides and iron oxides acting as electron acceptors during the decomposition of organic matter.

Another possible formation pathway of elemental sulfur in this system is via bacterial oxidation of the sulfide produced by sulfate reduction (e.g. Elsgaard and Jørgensen 1992).

It is important that there was formation and accumulation at the March 2012 assessment of appreciable quantities of both iron monosulfides and pyrite in the surficial sediment layers when organic matter is non-limiting to continued sulfate reduction (i.e. under the *Phragmites* site at Waltowa). Elemental sulfur was the most common product forming during the sulfate reduction rate determinations. However, in the sediments under the *Phragmites* site at Waltowa the contents of elemental sulfur (Figure 5-19) were minimal compared to the contents of both pyrite and AVS (Figures 5-13 and 5-16, respectively).

One typical mechanism for this transformation is as below (e.g. Berner, 1984):

$$FeS + S_8^0 \rightarrow FeS_2 \quad [5.4]$$

Other pathways to pyrite formation involve the reaction of monosulfide with the dissolution products of elemental sulfur, especially polysulfides (Rickard, 1997; Rickard and Luther, 1997), such as:

$$FeS + S_n^{2-} \rightarrow FeS_2 + S_{n-1}^{2-} \quad [5.5]$$

Elemental sulfur can also oxidise back to sulfate soon after formation using O_2 or Fe^{3+} (Burton *et al.* 2006a) as below:

$$\frac{1}{8}S_8^o{}_{(s)} + \frac{3}{2}O_2 + H_2O \rightarrow SO_4^{2-} + 2H^+ \quad [5.6]$$

$$\frac{1}{8}S_8^o{}_{(s)} + 6Fe^{3+} + 4\,H_2O \rightarrow SO_4^{2-} + 6Fe^{2+} + 8H^+ \quad [5.7]$$

Elemental sulfur can also undergo bacterial disproportionation (Thamdrup *et al.* 1993) as below:

$$\frac{1}{2}S_8^o{}_{(s)} + 4H_2O \rightarrow 3H_2S + SO_4^{2-} + 2H^+ \quad [5.8]$$

The sulfate reduction process as a result of bioremediation using vegetation on the exposed sediments of the Lower Lakes will be an acidity-neutral process unless potential acidity is stored in sulfides, or acidity is lost from the system (e.g. elemental sulfur gets entrained in overlying lake waters and oxidises there), or acidity from elemental sulfur oxidation escapes from sediment into lake waters. The data from this study indicate that appreciable alkalinity arising from sulfate reduction in these systems and stored as pyrite and monosulfides is occurring only under the *Phragmites* site as Waltowa (where organic matter is being produced in the sediments 19 months post-inundation). Such alkalinity production was minimal for all of the other sites be they bioremediated or not, during the initial period (~ 19 months) of inundation.

The formation of an MBO layer on the sediments under the *Phragmites* site at Waltowa and within the top 5 cm of these sediments was observed during the 2012 assessment indicating that appreciable monosulfides as well as pyritic sulfur are accumulating in and over the sediments at this location under *Phragmites*.

Although the avoidance of the accumulation of appreciable quantities of elemental sulfur, monosulfides and disulfides in the surficial lake sediments after bioremediation – apart from under the *Phragmites* vegetation - results in negligible net production of alkalinity in the upper lake sediments, this may be regarded as a positive outcome as this situation also results in the avoidance of some key possible future hazards that were important foci of this study. These are:

1. there was no appreciable development of potential sulfidic acidity in these surface layers over the duration of the study – apart from under the *Phragmites* vegetation. Thus during any future drying events if lake levels are lowered sufficiently to expose sediments, the hazard of rapid surface acidification that may have been present if for example pyrite had accumulated in these layers, should not be realised. However, pyrite has begun to accumulate in appreciable quantities after even 19 months of inundation under the *Phragmites*. This represents a potential acidification hazard due to sulfide accumulation that may continue to develop at longer inundation durations than those able to be observed in this study, especially in areas where lake vegetation such as *Phragmites* continues to supply organic matter that can fuel further sulfate reduction in such sediments.

2. there was no appreciable development of monosulfides in these surface layers – again apart from under the *Phragmites* vegetation - over the duration of the study. Thus the hazards of both deoxygenation and metal accumulation (as monosulfides) in the sediment surface layers did not develop to any appreciable extent during the short to medium (i.e. ~ 0.2 to 2 years) period of this study apart from under *Phragmites*. Such hazards may continue to develop at longer inundation durations than those able to be observed in this study, in areas where lake vegetation such as *Phragmites* continues to supply organic matter that can fuel further sulfate reduction in these sediments.

5.4.3 Metal and metalloid dynamics in the sediments resulting from bioremediation

The mobility of metals is likely to be affected by numerous factors and processes operating in the bioremediated sediments. These include the effects on metal mobility of the increased pH of the sediments (often after the initial decrease in pH due to exchange of acidity) most likely (as discussed previously) arising mainly from movement of alkalinity downward from inundating lake waters.

Another process that often affects metal mobility in sediments experiencing appreciable sulfate reduction is the sequestration of metals in the low-solubility sulfides that can accumulate under such conditions. However as discussed previously, the accumulation of both monosulfides and disulfides (e.g. pyrite) in these sediments was – apart from under the *Phragmites* vegetation - minimal due to the nature of the sulfur cycling (see Figure 5-100). Thus the sequestration within sulfides of appreciable quantities of metals or metalloids was not likely during the initial inundation of these lake sediments apart from under the *Phragmites* vegetation.

In addition, and as shown in Figure 5-100, the processes of sulfur cycling and organic matter decomposition can independently impact on the mobility of metal oxides especially iron oxides and oxyhydroxides and manganese oxides. As these two phases are, of course, comprised of metals and are known for their ability to adsorb a wide range of metals it is likely that the bioremediation may have affected the mobility of a range of metals in these sediments.

Indeed, the data from the March 2012 assessment continue to show appreciable systematic changes in Fe mobility following the initial inundation of these lake sediments. Iron concentrations have generally appreciably increased after inundation (e.g. Figure 5-101). Differences in the mobility of Fe and other metals may become more pronounced during longer durations of inundation as the sediments sweep through a wider range of biogeochemical regimes than occurred during the initial inundation of these lake sediments, especially where living bioremediation vegetation that survived the inundation of the lakes continue to provide organic matter to drive these geochemical regime changes. In addition it is likely that the general changes in Eh (towards more reducing) and pH (from acidic to neutral) of the sediments have begun to drive the reductive dissolution of iron minerals such as jarosite as is indicated by the distinctive iron macromorphology exhibited in Figures 3-9 and 3-10.

The HCl-extractable Zn concentrations generally increased in the sediments (see Figures 9-105 – 9-112, Appendix 6) presumably as a result of the increased Fe mineral dissolution that has resulted in the increased in Fe concentrations in these sediments. There were no consistent observable changes in HCl-extractable As, Cr, Cu, Ni or Pb during the inundation of these lake sediments to

date, however, it should be noted that this study did not replicate the February 2011 study examining pore-water metal concentrations in these sediments.

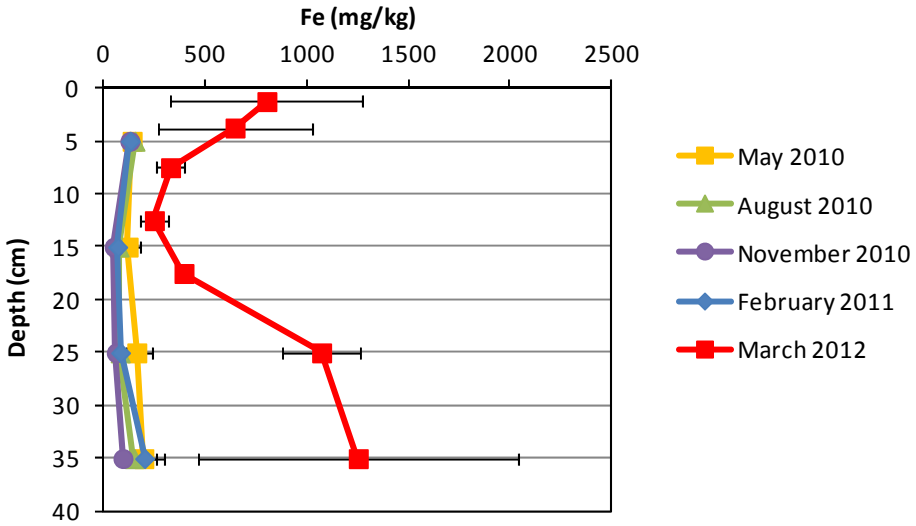

Figure 5-101. Tolderol total iron dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

5.4.4 Nutrient dynamics in the sediments resulting from bioremediation

The production and consumption of nutrients in sediments is very likely to be affected by numerous factors and processes operating in the bioremediated sediments. These include the mineralisation of the nutrients contained in the organic matter provided by the bioremediating vegetation.

The decomposition of organic matter in freshwater environments can occur under either aerobic or anaerobic conditions. Under aerobic conditions oxygen is used to facilitate decomposition whereas under anaerobic conditions the lack of oxygen forces bacteria to use other terminal electron acceptors. Nitrates are a nutrient that can be used for this process in place of oxygen and are energetically favoured relative to the use of sulfate for this purpose, but generally if sulfate is present in adequate concentration it fulfils this role leading to sulfate reduction (sulfides are inhibitory to nitrification (Joye and Hollibaugh 1995)).

If sulfate is not present in adequate amounts then carbon in organic matter can be used to facilitate decomposition of organic matter (albeit at very slow rates) and methane will be produced (i.e. methanogenesis). In marine water affected sediments the terminal electron acceptor is predominantly sulfur, whereas in freshwater sediments it is carbon (Harris 1999).

The presence of sulfate in appreciable concentrations can greatly increase the rate of organic matter decomposition and nutrient mineralisation in sediments (Jørgenson 1982). Caraco *et al.* (1989) proposed that phosphorus release from freshwater sediments correlates with the sulfate concentration of the overlying water. The work of Lamers and co-workers (e.g. Lamers *et al.* 2002) confirm this for sulfate-polluted freshwater wetlands. Therefore, it was proposed by Sullivan *et al.* (2011) that it was likely that continued enhanced sulfate reduction under the living lake vegetations such as *Phragmites* may also increase the release of phosphates (and other nutrients) from sediments.

The nutrient data gained from Sullivan *et al.* (2011) showed few general trends during inundation through to February 2011, apart from a considerable decrease in ammonium during that period. It was noted in that study that such changes may become more pronounced during later stages of inundation as the sediments sweep through a wider range of biogeochemical regimes than occurred during the initial inundation of these lake sediments especially where living bioremediation vegetation that survived the inundation of the lakes continue to provide organic matter to drive these geochemical regime changes.

This study shows that such changes in nutrients have happened at the longer durations of inundation at the Waltowa study area, where large decreases in ammonia concentrations in the pore-waters of the deeper (i.e. 20 – 40 cm) sediment layers has occurred under the *Phragmites* cf. under the *Cotula*

and *Juncus* (Figure 5-102) presumably due to uptake by the living *Phragmites*. No such systematic changes in ammonia occurred under the other study areas.

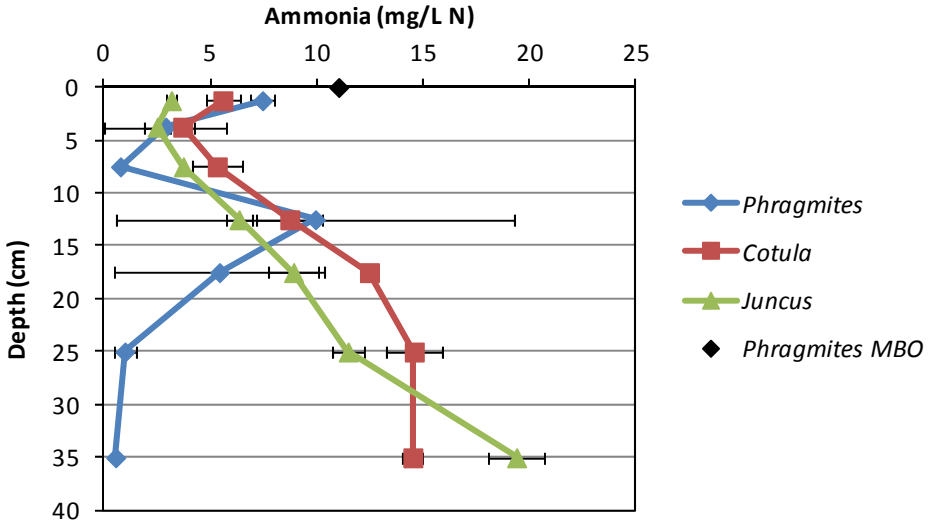

Figure 5-102. Pore-water ammonia characteristics at the Waltowa study area (March 2012).

Similarly the *Phragmites* has greatly increased the concentration of orthophosphate in the pore-waters of the surface sediment layers (i.e. 0 – 5 cm) cf. under the *Cotula* and *Juncus* in the Waltowa study area (Figure 5-103); presumably due to enhanced mineralisation of phosphate from accumulated organic matter consequent of sulfate reduction. The considerable decrease in TOC in these uppermost sediment layers at this site from the February 2011 assessment to the March 2012 assessment (Figure 5-20) supports this proposition. It is likely that these *Phragmites* sediments may be a source of phosphate to the overlying lake waters.

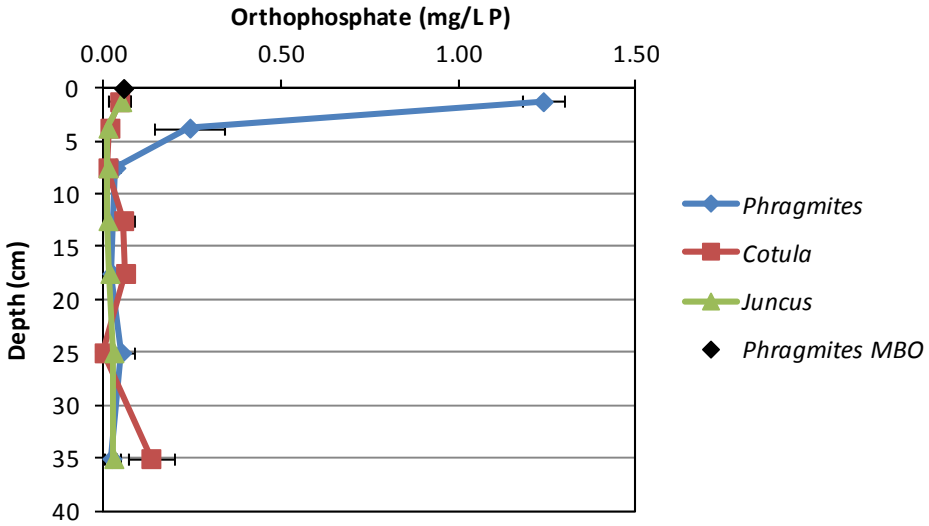

Figure 5-103. Pore-water orthophosphate characteristics at the Waltowa study area (March 2012).

6.0 Conclusions

The key findings of this study are:

1) Considerable sulfate reduction was occurring during the March 2012 assessment only in the surface sediment layers where organic matter is continuing to be provided when the vegetation used for bioremediation are species that survived lake re-filling (i.e. *Phragmites*). There were clear on-going differences in the effectiveness of the bioremediation vegetation in driving this process. Whereas the annual plants and short perennial plants (relative to the inundation depth) produce appreciable amounts of organic matter but then die, tall perennial plants that survive inundation can continue to produce organic matter. This is important because in sediments such as these where the availability of organic matter is the main constraint against sulfate reduction, the patterns of organic matter accumulation and production dictate the consequent patterns of sulfate reduction. Importantly, *Phragmites,* which successfully resisted prolonged inundation, is clearly continuing to supply organic matter to sediments long after inundation and hence is continuing to strongly drive sulfate reduction processes.

2) The March 2012 assessment clearly shows that appreciable quantities of reduced inorganic sulfides (especially pyrite and monosulfides) were accumulating in surface sediment layers under the *Phragmites* treatment. As well as representing an appreciable amount of alkalinity produced in these sediments from sulfate reduction processes, this store of pyrite also represents an appreciable and likely growing potential sulfidic acidity hazard in the surface lake sediments under this bioremediation treatment. Similarly, the store of monosulfidic materials (i.e. Monosulfidic Black Oozes (MBOs)) under the *Phragmites* treatment also represents the development of associated acidification, metal mobilisation and deoxygenation hazards under this bioremediation treatment.

   Given their location in the surface layers of sediments when an inundation tolerant bioremediation species, in this case *Phragmites,* was used as for bioremediation, this potential sulfidic acidity hazard would be realised much earlier than would previously have been the case, should the Lower Lakes experience atmospheric exposure as was the case in the last drought.

   The accumulation of appreciable quantities of pyrite (and hence the development of such a considerable potential sulfidic acidity hazard) was not observed, and given the lack of an organic matter supply, is unlikely to occur when vegetation used for bioremediation is inundation intolerant and undergoes death during inundation.

3) Both the acidity and low pHs of the acidified acid sulfate sediment layers are continuing to be remediated by a number of processes and sources some consequent of the bioremediation, some not.

   The two main factors effecting on-going acidity remediation are:
   - the movement into the sediment of the alkalinity that is contained in the lake waters and;
   - the vegetation established during bioremediation when inundation tolerant adding alkalinity indirectly to the soil via provision of organic matter and thus enabling sulfate reduction resulting in the accumulation of reduced inorganic sulfides (especially pyrite and monosulfides).

4) The data indicate appreciable increases both in ferrous iron (Fe^{2+}) concentrations in pore-waters and in the HCl-extractable zinc (Zn) concentrations in the sediments during the study period.

5) The data indicate that, apart from under the *Phragmites*, there were few general trends in nutrient availability consequent of bioremediation at the March 2012 assessment. However, two strong trends in nutrient mobility were observed under the *Phragmites* with large decreases in ammonia concentrations in the pore-waters of the deeper (i.e. 20 – 40 cm) sediment layers and greatly increased phosphate concentrations in the pore-waters of the surface sediment likely due to enhanced mineralisation of phosphate from accumulated organic matter consequent of sulfate reduction. It is likely that these sediments under *Phragmites* may be a source of phosphate to the overlying lake waters.

7.0 Recommendations

1) Southern Cross University is aware that this study has focused on the geochemical processes and changes that have occurred in the relatively early stages of lake re-filling and likely ecological and biogeochemical restoration. This is unlikely to be a major limitation for the examination of sulfate-reduction associated processes for those bioremediated sites where the organic matter produced by the bioremediation vegetation was largely removed or exhausted during this and the February 2011 assessment. However it is an important consideration when assessing the possible long-term effects of bioremediation using vegetation that is still producing organic materials or where there was (at the end of our study) still a relative abundance of organic matter to drive further changes to the biogeochemistry of the lake sediments. It is also an important consideration when assessing the possible long-term effects of de-acidification of the lake sediments on metal and metalloid mobilisation as non-sulfate reduction associated alkalinity drives sediments from being strongly acidic to neutral to alkaline.

   Differences in geochemical behaviour to those observed in this study may develop during later stages of inundation as the sediments sweep through a wider range of biogeochemical regimes than occurred initially, especially but not only, where living bioremediation vegetation that survived the inundation of the lakes continue to provide organic matter to drive such geochemical regime changes. Such changes could result in the development of the hazards that can be associated with sulfate reduction such as the accumulation of sulfides such as monosulfides and pyrite, hazards that were avoided during the initial 6 month period of lake infilling but have started to develop after another 13 months.

   It is our recommendation that future monitoring of the effects of bioremediation on the geochemistry of the lake sediments, by assessment programs similar to that used in this project, be undertaken to fully assess both the medium and long term effects of the various bioremediation techniques on the lake ecosystem.

2) The March 2012 results showing appreciable build up of HCl-extractable zinc: when taken in conjunction with the results of the February 2011 assessment also indicate that attention needs to be paid to the mobility of metals in these sediments especially nickel and zinc. In a lake setting, including sites treated by bioremediation techniques, there are a number of important scenarios where subsurface bio-available trace metals could enter the surface aquatic ecosystem. This includes ingestion by burrowing benthic organisms, translocation into plants via roots (this is an especially important consideration for lake sediment bioremediation via revegetation) and direct ingestion by foraging animals (e.g. birds and fish). As such, the fate and possible mobility of subsurface pore-water nickel and zinc at these sites requires consideration from both a geochemical perspective (i.e. developing the knowledge required to predict how pore-water nickel and zinc concentrations will change into the future) and an ecological perspective (i.e. examining nickel and zinc uptake in potentially exposed organisms).

   It is our recommendation that future monitoring of the pore-water nickel and zinc in the lake sediments as affected by bioremediation is required in order to assess ongoing environmental risks posed by the presence of very high bio-accessible concentrations of these two potentially-toxic trace metals (nickel and zinc).

3) The March 2012 results show a strong trend in nutrient mobility under the *Phragmites* with large increases in the concentrations of phosphate in the pore-waters of the surface sediments indicating that it is likely that these *Phragmites* sediments may be a source of phosphate to the overlying lake waters.

   It is our recommendation that future monitoring of nutrients in the lake sediments as affected by bioremediation is required in order to assess the ongoing environmental risks posed by the presence of an enhanced source of phosphate to the overlying lake waters provided by bioremediation using *Phragmites.*

4) The results also confirms the results of Sullivan *et al.* (2011) that the different vegetation types used for bioremediation had very different organic matter production characteristics and that these differences markedly affected the sediment's geochemical behaviour during the first 19 months of lake re-filling. The results of this study strongly indicate the need for a further detailed study on both:
   i. the effectiveness of the different vegetation types and strategies used for bioremediation, and
   ii. the unbioremediated lake sediment behaviour.

   Such 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 ecological restoration. It is our recommendation that such a study be undertaken.

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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	29/03/12	W1A	54H 0352058, 6059358	0-2.5	7.71	110	*Juncus* site down to 40 cm.
					2.5-5	7.68	171	
					5-10	7.38	64	Top ~5 cm white/beige wave-sorted sand beneath grey and beige mottled sand.
					10-15	6.91	42	
					15-20	7.43	90	0-30 cm: beige sand with iron segregations.
					20-30	6.97	107	30-40 cm: grey sand.
					30-40	6.62	93	40-70 cm: grey clay.
		29/03/12	W1B	54H 0352055, 6059355	0-2.5	7.23	229	*Juncus* site down to 40 cm.
					2.5-5	7.28	49	
					5-10	6.86	65	
					10-15	6.93	65	
					15-20	6.86	179	
					20-30	6.73	109	
					30-40	6.62	97	
	Cotula bioremediation	29/03/12	W2A	54H 0352244, 6059193	0-2.5	7.36	102	*Cotula* site down to 40 cm.
					2.5-5	7.13	55	
					5-10	6.70	81	0-30 cm: beige sand with iron segregations.
					10-15	6.24	109	30-40 cm: grey sand.
					15-20	6.16	97	40-70 cm: grey clay.
					20-30	6.29	83	
					30-40	6.18	94	
		29/03/12	W2B	54H 0352221, 6059203	0-2.5	7.19	76	*Cotula* site down to 40 cm.
					2.5-5	6.91	104	
					5-10	6.64	111	
					10-15	6.19	105	
					15-20	6.02	127	
					20-30	6.02	76	
					30-40	6.31	95	
	Phragmites bioremediation	29/03/12	W3A	54H 0352293, 6059114	0-2.5	6.75	-1	*Phragmites* site down to 40 cm.
					2.5-5	6.77	41	
					5-10	6.78	46	0-30 cm: beige sand with iron segregations.
					10-15	6.57	50	30-40 cm: grey sand.
					15-20	6.69	61	
					20-30	6.31	61	
					30-40	6.29	42	
		29/03/12	W3B	54H 0352286, 6059102	0	6.85	5	*Phragmites* site down to 40 cm.
					0-2.5	6.77	57	
					2.5-5	6.84	10	A ~12 cm deep (variable) MBO observed at the surface at this site.
					5-10	7.01	4	
					10-15	6.91	7	
					15-20	6.88	26	
					20-30	6.62	46	
					30-40	6.57	75	

* 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	30/03/12	P1A	54H 0341295, 6070677	0-2.5	6.48	299	Bevy Rye site down to 40 cm.
					2.5-5	6.81	162	
					5-10	6.95	131	0-3 cm: wave washed beige sand.
					10-15	6.87	231	3-11 cm: dark grey sand.
					15-20	6.79	210	11-32 cm: light grey sand with frequent orange segregations.
					20-30	4.61	359	>32 cm: grey sand.
					30-40	4.32	372	
		30/03/12	P1B	54H 0341267, 6070659	0-2.5	7.04	188	Bevy Rye site down to 40 cm.
					2.5-5	6.96	194	
					5-10	7.18	173	0-3 cm: wave washed beige sand.
					10-15	7.11	217	3-11 cm: dark grey sand.
					15-20	6.99	224	11-32 cm: light grey sand with frequent orange segregations.
					20-30	6.98	255	>32 cm: grey sand.
					30-40	6.87	317	
Campbell Park	Scald (no bioremediation)	30/03/12	CP1A	54H 0340786, 6056740	0-2.5	6.81	109	Scald (no bioremediation) site down to 40 cm.
					2.5-5	6.60	89	Jarosite still around 15-30 cm layer.
					5-10	5.99	114	
					10-15	4.32	196	0-2 cm: wave washed sand.
					15-20	3.45	256	2-10 cm: grey sand.
					20-30	3.11	313	10-30 cm: light grey sand with jarosite.
					30-40	3.21	344	30-60 cm: blue grey clay.
		30/03/12	CP1B	54H 0340772, 6056761	0-2.5	6.48	142	Scald (no bioremediation) site down to 40 cm.
					2.5-5	6.83	100	
					5-10	5.88	126	
					10-15	4.84	248	
					15-20	3.20	350	
					20-30	3.00	362	
					30-40	3.06	345	
	2010 seeded with Bevy rye and *Puccinellia* bioremediation	30/03/12	CP2A	54H 0340734, 6056750	0-2.5	6.67	323	Bevy rye and *Puccinellia* bioremediation site down to 40 cm.
					2.5-5	6.31	139	
					5-10	6.02	141	
					10-15	4.88	275	
					15-20	3.54	400	
					20-30	3.40	420	
					30-40	3.37	328	
		30/03/12	CP2B	54H 0340728, 6056756	0-2.5	7.14	228	Bevy rye and *Puccinellia* bioremediation site down to 40 cm.
					2.5-5	6.91	158	
					5-10	6.40	122	
					10-15	6.05	154	
					15-20	4.99	292	
					20-30	3.56	394	
					30-40	3.47	405	

* 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	31/03/12	T1A	54H 0331148, 6083496	0-2.5	7.17	388	*Juncus* into plantings of Bevy Rye bioremediation site down to 40 cm.
					2.5-5	7.50	389	
					5-10	7.48	254	0-30 cm: beige sand with lots of iron segregations at 20-50 cm, iron band at 30 cm.
					10-15	6.71	163	30-45 cm: beige sand with jarosite band at 40-45 cm.
					15-20	6.27	311	45-60 cm: grey sand with some iron segregations.
					20-30	6.51	243	60-80 cm: grey sand.
					30-40	5.92	245	
		31/03/12	T1B	54H 0331160, 6083485	0-2.5	7.15	201	*Juncus* into plantings of Bevy Rye bioremediation site down to 40 cm.
					2.5-5	7.04	172	
					5-10	7.22	191	
					10-15	6.62	274	
					15-20	5.97	249	
					20-30	6.43	370	
					30-40	5.54	344	
	Scald (no bioremediation)	31/03/12	T2A	54H 0331075, 6083416	0-2.5	6.07	355	Scald (no bioremediation) site down to 40 cm.
					2.5-5	6.34	149	
					5-10	6.28	154	Iron-rich crust on some scald surface layers.
					10-15	5.94	160	Jarosite still around 25 cm and lower layers.
					15-20	5.64	213	
					20-30	4.45	304	0-40 cm: beige sand with very occasional jarosite in roots.
					30-40	3.58	433	40-50 cm: beige sand with abundant jarosite in roots.
								50-60 cm: dark grey sandy clay with abundant jarosite in roots.
								60-80 cm: dark grey sandy clay but no jarosite.
		31/03/12	T2B	54H 0331047, 6083414	0-2.5	7.74	351	Scald (no bioremediation) site down to 40 cm.
					2.5-5	7.27	161	
					5-10	7.18	289	Iron-rich crust on some scald surface layers.
					10-15	6.64	236	Jarosite still around 25 cm and lower layers.
					15-20	6.74	286	
					20-30	6.55	292	
					30-40	6.29	288	

* 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 2012.

Profile ID* (Site Code, Core)	Depth Range (cm)	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)	Hydrolysable C (%C)	Total Organic C (%C)
W 1A	0-2.5	19.42	9.10	350	9.26	0.00	0.11	11.44	0.00	<0.01	0.001	<0.01	-14.64	0.14	0.003	0.08	0.13
W 1A	2.5-5	17.75	9.48	129	9.23	0.00	0.21	8.70	0.00	<0.01	0.001	<0.01	-27.72	0.13	0.005	0.06	0.13
W 1A	5-10	19.71	9.37	416	9.56	0.00	0.19	16.74	0.00	<0.01	0.003	<0.01	-24.86	0.19	0.006	0.06	0.17
W 1A	10-15	19.82	9.24	962	9.43	0.00	0.34	17.14	0.00	<0.01	0.003	<0.01	-45.34	0.24	0.012	0.05	0.19
W 1A	15-20	19.52	9.14	693	9.35	0.00	0.29	13.87	0.00	<0.01	0.001	<0.01	-38.75	0.22	0.014	0.04	0.18
W 1A	20-30	19.21	8.77	829	7.78	0.00	0.07	3.94	0.00	<0.01	0.002	<0.01	-9.64	0.18	0.010	0.01	0.18
W 1A	30-40	20.02	7.27	1243	6.43	1.08	0.00	0.00	0.00	0.08	0.001	<0.01	53.21	0.20	0.012	0.04	0.21
W 1B	0-2.5	19.14	9.31	282	9.43	0.00	0.22	12.43	0.00	<0.01	0.003	0.02	-19.37	0.17	0.006	<0.01	0.14
W 1B	2.5-5	20.65	9.33	351	9.68	0.00	0.32	26.64	0.00	<0.01	0.007	0.02	-31.14	0.26	0.013	0.15	0.26
W 1B	5-10	19.67	8.98	378	7.90	0.00	0.18	3.74	0.00	<0.01	0.003	<0.01	-23.41	0.14	0.006	<0.01	0.13
W 1B	10-15	19.81	9.19	470	8.52	0.00	0.11	6.22	0.00	<0.01	0.002	<0.01	-14.94	0.14	0.008	0.02	0.12
W 1B	15-20	19.01	9.03	622	8.57	0.00	0.10	5.92	0.00	<0.01	0.003	<0.01	-13.83	0.13	0.007	0.02	0.13
W 1B	20-30	20.32	8.53	897	7.60	0.00	0.06	4.22	0.00	<0.01	<0.001	<0.01	-7.55	0.21	0.014	0.02	0.21
W 1B	30-40	19.33	7.82	1004	6.85	0.00	0.03	2.87	0.00	0.02	<0.001	<0.01	10.33	0.26	0.022	0.05	0.24
W 2A	0-2.5	19.50	9.24	379	8.88	0.00	0.20	6.97	0.00	<0.01	0.002	<0.01	-26.56	0.16	0.011	0.07	0.14
W 2A	2.5-5	20.89	8.44	346	6.97	0.00	0.08	3.77	0.00	<0.01	0.004	<0.01	-10.37	0.23	0.014	0.06	0.22
W 2A	5-10	19.95	7.67	403	6.72	0.00	0.22	4.28	0.00	<0.01	0.005	<0.01	-29.92	0.25	0.026	0.02	0.14
W 2A	10-15	20.68	7.18	534	6.43	2.51	0.00	0.00	0.00	<0.01	0.002	<0.01	2.51	0.22	0.019	0.03	0.20
W 2A	15-20	22.90	6.47	705	5.97	3.25	0.00	0.00	0.00	0.03	<0.001	<0.01	20.27	0.32	0.029	0.04	0.32
W 2A	20-30	27.14	6.51	894	5.92	3.57	0.00	0.00	0.00	0.06	<0.001	<0.01	42.87	0.40	0.042	0.04	0.34
W 2A	30-40	48.59	6.46	2560	6.43	2.82	0.00	0.00	0.00	0.40	<0.001	<0.01	252.15	2.09	0.226	0.38	2.08
W 2B	0-2.5	19.37	9.33	303	8.82	0.00	0.12	8.70	0.00	<0.01	0.001	<0.01	-16.37	0.16	0.012	<0.01	0.16
W 2B	2.5-5	18.62	8.57	276	7.26	0.00	0.09	3.34	0.00	<0.01	0.003	<0.01	-12.20	0.19	0.018	0.07	0.18
W 2B	5-10	19.74	7.91	356	6.49	0.79	0.00	0.00	0.00	<0.01	0.005	<0.01	0.79	0.19	0.018	0.08	0.18
W 2B	10-15	19.97	7.25	440	5.98	3.03	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.03	0.22	0.016	0.09	0.22
W 2B	15-20	20.49	6.72	575	5.58	3.41	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.41	0.21	0.018	0.10	0.19
W 2B	20-30	21.22	6.54	819	5.79	2.86	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.86	0.23	0.023	0.07	0.23
W 2B	30-40	37.12	7.45	2380	7.17	0.00	0.16	9.87	0.00	0.28	<0.001	<0.01	150.88	0.88	0.096	<0.01	0.88

** 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)	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)	Hydrolysable C (%C)	Total Organic C (%C)
W 3A	0-2.5	48.90	9.13	1436	9.29	0.00	9.05	29.41	0.00	0.12	0.010	0.04	-1108.7	2.35	0.122	0.23	1.35
W 3A	2.5-5	23.73	9.50	705	9.71	0.00	1.31	94.02	0.00	0.02	0.004	0.02	-149.92	0.35	0.020	0.08	0.23
W 3A	5-10	20.50	9.08	752	9.00	0.00	0.13	8.69	0.00	<0.01	0.007	<0.01	-17.63	0.21	0.023	0.05	0.18
W 3A	10-15	19.92	8.17	820	7.54	0.00	0.11	4.10	0.00	<0.01	<0.001	<0.01	-14.09	0.15	0.014	0.02	0.15
W 3A	15-20	19.95	7.37	986	6.80	0.00	0.06	3.87	0.00	<0.01	<0.001	<0.01	-7.91	0.16	0.021	0.01	0.16
W 3A	20-30	23.89	6.85	1319	6.10	2.82	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.82	0.29	0.026	0.06	0.29
W 3A	30-40	25.50	6.60	1781	6.11	3.53	0.00	0.00	0.00	0.03	<0.001	<0.01	21.34	0.41	0.039	0.04	0.41
W 3B	MBO	90.82	8.04	8490	8.32	0.00	3.43	75.46	0.00	0.30	0.012	0.15	-179.77	6.46	0.626	1.73	6.00
W 3B	0-2.5	50.35	8.87	1524	9.20	0.00	2.06	111.60	0.00	0.05	0.005	0.03	-225.29	1.36	0.100	0.25	1.15
W 3B	2.5-5	24.00	9.22	714	9.69	0.00	0.65	43.28	0.00	0.01	0.004	0.03	-61.37	0.28	0.019	0.05	0.19
W 3B	5-10	20.97	9.12	725	8.88	0.00	0.18	6.21	0.00	<0.01	0.004	<0.01	-24.22	0.16	0.011	0.04	0.12
W 3B	10-15	19.54	8.96	765	8.31	0.00	0.07	4.36	0.00	<0.01	0.002	<0.01	-9.50	0.18	0.018	0.01	0.11
W 3B	15-20	20.55	8.62	880	8.12	0.00	0.09	5.19	0.00	<0.01	0.001	<0.01	-12.36	0.18	0.014	<0.01	0.17
W 3B	20-30	23.59	7.73	1266	7.22	0.00	0.04	3.81	0.00	0.03	<0.001	<0.01	12.74	0.30	0.032	0.09	0.30
W 3B	30-40	27.06	7.33	1583	6.92	0.00	0.11	3.86	0.00	0.18	<0.001	<0.01	97.20	0.42	0.044	0.01	0.42

** 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 2012.

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
W 1A	0-2.5	590	71	0.05	0.54	0.64	<0.01	0.17	0.91	11.89	0.41	0.03	1.48
W 1A	2.5-5	683	66	0.02	0.57	0.59	<0.01	0.23	0.68	9.21	0.35	<0.01	2.36
W 1A	5-10	563	97	0.01	0.14	1.05	<0.01	0.18	0.93	9.75	0.53	<0.01	2.56
W 1A	10-15	721	139	0.02	0.39	1.19	<0.01	0.15	1.35	18.54	0.87	<0.01	4.13
W 1A	15-20	698	138	0.01	0.41	1.19	<0.01	0.14	1.29	17.10	1.07	<0.01	2.37
W 1A	20-30	529	117	0.01	0.29	0.87	<0.01	0.18	1.22	11.01	0.55	0.01	1.53
W 1A	30-40	416	145	0.01	0.71	0.98	<0.01	0.14	1.46	11.13	0.87	0.03	1.69
W 1B	0-2.5	673	71	0.01	0.49	0.57	<0.01	0.20	0.63	10.88	0.33	<0.01	2.41
W 1B	2.5-5	1251	135	0.01	0.36	0.93	<0.01	0.47	1.35	18.26	0.77	<0.01	1.90
W 1B	5-10	1145	138	0.01	<0.01	1.24	<0.01	0.50	1.05	14.58	0.82	0.04	1.88
W 1B	10-15	830	103	0.11	0.19	0.99	<0.01	0.32	0.94	12.95	0.65	0.05	1.13
W 1B	15-20	569	91	0.05	0.20	1.12	<0.01	0.16	0.83	11.55	0.59	0.02	0.99
W 1B	20-30	989	199	0.02	0.42	1.46	<0.01	0.28	1.81	21.43	1.21	0.05	1.51
W 1B	30-40	464	130	0.02	0.71	0.89	<0.01	0.16	1.56	12.92	0.38	0.01	1.12
W 2A	0-2.5	574	71	0.01	0.39	0.61	<0.01	0.19	0.71	7.82	0.30	<0.01	2.48
W 2A	2.5-5	553	115	0.01	0.40	0.92	<0.01	0.14	1.28	4.20	0.35	0.01	1.21
W 2A	5-10	2020	183	0.01	0.34	0.86	<0.01	1.39	1.51	21.32	0.78	<0.01	1.33
W 2A	10-15	1960	230	0.01	0.36	1.19	<0.01	1.34	1.74	24.13	0.95	<0.01	2.28
W 2A	15-20	1189	166	0.01	0.87	1.55	<0.01	0.54	2.15	16.29	0.73	0.03	1.23
W 2A	20-30	565	206	0.01	0.74	1.42	0.01	0.15	2.13	14.67	0.79	0.02	2.73
W 2A	30-40	3433	620	0.05	2.67	3.53	0.02	1.77	9.80	84.78	4.81	0.12	7.31
W 2B	0-2.5	1349	106	0.02	0.51	0.65	<0.01	0.71	1.05	17.09	0.62	0.03	1.86
W 2B	2.5-5	566	119	0.05	0.44	1.11	<0.01	0.16	1.18	5.87	0.61	0.01	1.24
W 2B	5-10	644	120	0.04	0.41	1.04	<0.01	0.17	1.37	6.28	0.43	0.03	1.03
W 2B	10-15	618	108	0.02	0.50	0.69	<0.01	0.26	1.06	7.71	0.48	0.01	1.20
W 2B	15-20	497	106	0.03	0.65	0.78	<0.01	0.16	1.27	7.81	0.37	<0.01	1.19
W 2B	20-30	574	154	0.02	0.55	0.98	0.01	0.16	1.34	10.65	0.57	0.06	1.39
W 2B	30-40	2328	481	0.02	1.50	2.48	0.01	0.92	4.92	51.21	3.17	0.08	3.79

** 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
W 3A	0-2.5	2577	495	0.01	1.37	2.82	0.02	1.12	4.35	96.87	2.77	0.12	5.17
W 3A	2.5-5	1633	161	0.01	0.41	1.61	<0.01	0.81	1.41	25.38	0.90	0.09	1.57
W 3A	5-10	711	116	0.15	0.20	1.33	<0.01	0.26	1.25	8.43	0.52	0.06	1.16
W 3A	10-15	2235	207	0.02	0.47	1.33	<0.01	1.67	1.72	26.02	1.62	0.02	1.33
W 3A	15-20	1883	193	0.01	0.44	1.14	<0.01	1.18	1.86	23.68	1.32	0.01	2.99
W 3A	20-30	816	150	0.03	0.83	1.40	<0.01	0.31	1.69	13.99	0.80	0.03	1.21
W 3A	30-40	1442	324	0.02	0.68	1.70	0.01	0.57	2.87	26.57	1.41	0.05	2.21
W 3B	MBO	4434	2007	0.04	4.58	9.28	0.05	0.55	17.71	225.24	7.85	0.48	19.63
W 3B	0-2.5	3294	484	0.02	1.19	2.58	0.01	1.78	4.36	75.44	2.60	0.09	3.85
W 3B	2.5-5	1251	149	0.01	0.37	1.24	<0.01	0.47	1.25	22.58	0.71	<0.01	2.92
W 3B	5-10	658	109	0.02	0.26	1.24	<0.01	0.19	0.97	8.07	0.40	0.06	1.50
W 3B	10-15	646	116	0.03	0.34	1.55	<0.01	0.14	1.00	10.22	0.53	0.05	1.55
W 3B	15-20	1293	205	0.01	0.65	1.62	<0.01	0.41	1.68	18.26	0.99	0.04	1.46
W 3B	20-30	676	165	0.16	1.11	1.40	<0.01	0.16	2.15	13.93	0.81	0.08	1.60
W 3B	30-40	1496	374	0.02	1.33	1.99	0.01	0.67	3.07	34.66	1.45	0.19	2.47

** 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 2012.

Profile ID* (Site Code, Core)	Depth Range (cm)	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)	Hydrolysable C (%C)	Total Organic C (%C)
P1A	0-2.5	18.48	7.76	60.8	6.48	0.94	0.00	0.00	0.00	<0.01	<0.001	<0.01	0.94	0.09	0.007	0.05	0.08
P1A	2.5-5	17.98	8.11	75.9	6.84	0.00	0.00	3.56	0.00	<0.01	<0.001	<0.01	0.00	0.11	0.009	0.04	0.09
P1A	5-10	19.27	7.17	90.7	5.92	1.29	0.00	0.00	0.00	<0.01	0.001	<0.01	1.29	0.12	0.008	<0.01	0.09
P1A	10-15	18.06	6.61	128	5.70	1.46	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.46	0.05	0.002	0.03	0.05
P1A	15-20	17.75	6.40	205	5.77	2.23	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.23	0.05	0.003	0.03	0.06
P1A	20-30	17.52	5.33	534	5.10	2.18	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.18	0.06	0.006	0.03	0.06
P1A	30-40	17.85	4.64	738	4.91	2.89	0.00	0.00	0.00	0.04	<0.001	<0.01	27.48	0.09	0.007	0.03	0.07
P1B	0-2.5	18.66	7.63	69.7	6.88	0.00	0.09	3.27	0.00	<0.01	<0.001	<0.01	-11.34	0.07	0.011	<0.01	0.07
P1B	2.5-5	18.26	9.30	87.0	8.98	0.00	0.19	7.47	0.00	<0.01	<0.001	<0.01	-25.93	0.08	0.010	0.04	0.08
P1B	5-10	18.27	8.77	113	7.25	0.00	0.00	3.23	0.00	<0.01	<0.001	<0.01	0.00	0.10	0.014	0.05	0.09
P1B	10-15	18.41	7.72	150	6.36	1.11	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.11	0.07	0.013	0.05	0.08
P1B	15-20	17.78	7.29	263	6.21	2.58	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.58	0.06	0.009	0.04	0.07
P1B	20-30	18.61	7.73	509	6.71	0.00	0.12	3.03	0.00	0.01	<0.001	<0.01	-9.25	0.06	0.010	0.01	0.04
P1B	30-40	18.24	8.84	778	9.24	0.00	0.07	10.70	0.00	0.04	<0.001	<0.01	14.45	0.07	0.011	0.04	0.07

** 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 2012.

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
P1A	0-2.5	381	60	0.03	0.37	0.37	<0.01	0.19	0.31	9.82	0.27	<0.01	2.90
P1A	2.5-5	377	89	0.04	0.06	0.43	<0.01	0.19	1.37	4.13	0.53	0.05	1.16
P1A	5-10	432	75	0.03	0.32	0.51	<0.01	0.19	0.76	4.33	0.29	0.03	0.73
P1A	10-15	262	37	0.02	<0.01	0.29	<0.01	0.21	0.29	3.23	0.17	0.03	0.49
P1A	15-20	794	67	0.01	0.69	0.28	<0.01	0.48	0.44	9.33	0.36	0.01	0.52
P1A	20-30	255	49	0.01	0.28	0.22	<0.01	0.19	0.40	4.59	0.24	0.01	0.49
P1A	30-40	246	76	0.02	0.74	0.35	<0.01	0.18	0.34	7.25	0.58	0.04	1.13
P1B	0-2.5	960	69	0.01	0.41	0.35	<0.01	0.48	0.48	20.78	0.45	<0.01	3.35
P1B	2.5-5	440	61	0.14	0.19	0.35	<0.01	0.19	0.39	9.04	0.30	0.03	1.49
P1B	5-10	397	70	0.05	0.38	0.42	<0.01	0.17	0.45	7.43	0.31	0.07	0.67
P1B	10-15	340	58	0.03	0.55	0.34	<0.01	0.21	0.39	7.07	0.29	0.03	0.56
P1B	15-20	327	43	0.02	1.21	0.28	<0.01	0.18	0.34	6.83	0.20	0.02	0.46
P1B	20-30	396	49	0.02	0.83	0.31	<0.01	0.19	0.39	11.85	0.45	<0.01	0.61
P1B	30-40	251	78	0.01	0.56	0.35	<0.01	0.19	0.30	14.90	0.51	<0.01	0.72

** 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 2012.

Profile ID* (Site Code, Core)	Depth Range (cm)	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)	Hydrolysable C (%C)	Total Organic C (%C)
T 1A	0-2.5	21.41	8.33	55.5	6.63	0.00	0.12	2.42	0.00	<0.01	<0.001	<0.01	-15.62	0.10	0.016	0.08	0.11
T 1A	2.5-5	18.92	7.51	47.9	6.17	3.01	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.01	0.10	0.015	0.07	0.09
T 1A	5-10	18.99	7.04	49.4	6.07	1.79	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.79	0.09	0.013	0.05	0.08
T 1A	10-15	20.39	7.32	102	6.08	3.25	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.25	0.10	0.011	0.03	0.08
T 1A	15-20	20.72	7.06	189	6.11	2.29	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.29	0.09	0.010	0.02	0.07
T 1A	20-30	20.81	7.15	194	6.20	1.09	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.09	0.07	0.012	<0.01	0.05
T 1A	30-40	21.01	5.41	454	5.13	2.18	0.00	0.00	0.00	0.01	<0.001	<0.01	11.41	0.08	0.010	<0.01	0.06
T 1B	0-2.5	21.01	7.72	76.4	6.67	0.00	0.11	2.88	0.00	<0.01	<0.001	<0.01	-14.94	0.10	0.012	0.04	0.08
T 1B	2.5-5	19.67	7.30	73.4	6.27	2.66	0.00	0.00	0.00	0.01	<0.001	<0.01	9.16	0.10	0.013	0.05	0.09
T 1B	5-10	19.90	6.91	100	6.07	3.07	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.07	0.10	0.015	0.07	0.10
T 1B	10-15	19.86	6.56	164	5.81	2.78	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.78	0.14	0.017	0.08	0.14
T 1B	15-20	20.00	7.04	61.9	6.05	2.86	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.86	0.07	0.007	<0.01	0.07
T 1B	20-30	20.37	6.54	133	6.05	1.61	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.61	0.07	0.009	<0.01	0.05
T 1B	30-40	22.26	6.35	353	6.07	1.70	0.00	0.00	0.00	0.14	<0.001	<0.01	87.25	0.10	0.012	<0.01	0.09
T 2A	0-2.5	18.98	6.89	46.7	6.24	2.35	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.35	0.12	0.015	0.01	0.12
T 2A	2.5-5	22.73	6.68	84.9	6.03	3.16	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.16	0.14	0.023	<0.01	0.12
T 2A	5-10	19.42	6.05	35.1	5.46	3.17	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.17	0.08	0.010	0.01	0.08
T 2A	10-15	19.25	6.07	50.8	5.62	3.17	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.17	0.10	0.014	<0.01	0.10
T 2A	15-20	19.29	5.68	35.5	4.97	3.40	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.40	0.09	0.014	0.04	0.09
T 2A	20-30	19.93	4.83	122	4.61	4.78	0.00	0.00	0.00	<0.01	<0.001	<0.01	4.78	0.09	0.009	0.01	0.07
T 2A	30-40	20.92	4.36	255	4.51	7.28	0.00	0.00	0.00	<0.01	<0.001	<0.01	7.28	0.08	0.010	0.01	0.08
T 2B	0-2.5	20.79	7.62	56.5	6.61	0.00	0.11	1.56	0.00	<0.01	<0.001	<0.01	-15.02	0.12	0.013	0.04	0.11
T 2B	2.5-5	21.27	6.82	54.5	6.00	2.60	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.60	0.14	0.013	<0.01	0.12
T 2B	5-10	20.19	6.51	39.1	5.67	2.84	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.84	0.09	0.010	0.01	0.07
T 2B	10-15	20.20	6.24	43.2	5.90	2.78	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.78	0.08	0.009	<0.01	0.07
T 2B	15-20	19.11	5.90	29.3	5.21	2.88	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.88	0.06	0.010	<0.01	0.03
T 2B	20-30	20.40	6.15	38.1	5.66	3.41	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.41	0.09	0.011	<0.01	0.02
T 2B	30-40	20.61	5.83	33.3	5.06	2.93	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.93	0.06	0.009	<0.01	0.04

** 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 2012.

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
T 1A	0-2.5	330	82	0.01	0.09	0.47	<0.01	0.21	0.47	7.20	0.48	<0.01	5.62
T 1A	2.5-5	268	85	0.01	<0.01	0.50	<0.01	0.19	0.54	3.90	0.47	<0.01	1.24
T 1A	5-10	262	71	0.01	<0.01	0.36	<0.01	0.21	0.45	4.60	0.33	<0.01	0.69
T 1A	10-15	181	108	0.01	<0.01	0.35	<0.01	0.18	0.52	5.13	0.36	<0.01	1.58
T 1A	15-20	361	92	0.13	<0.01	0.27	<0.01	0.25	0.42	5.91	0.24	0.03	0.81
T 1A	20-30	1268	142	0.01	<0.01	0.32	<0.01	0.91	0.79	15.78	0.57	0.03	1.20
T 1A	30-40	460	81	0.02	0.30	0.29	<0.01	0.24	0.58	6.56	0.39	<0.01	1.01
T 1B	0-2.5	1274	102	0.01	0.16	0.54	<0.01	0.90	0.72	20.87	0.60	<0.01	4.64
T 1B	2.5-5	1022	119	0.01	0.05	0.52	<0.01	0.70	0.75	14.35	0.54	0.05	1.36
T 1B	5-10	393	82	0.01	0.05	0.45	<0.01	0.26	0.53	5.97	0.46	0.03	0.93
T 1B	10-15	315	108	0.01	0.11	0.60	<0.01	0.17	0.85	6.28	0.39	<0.01	1.07
T 1B	15-20	426	83	0.01	0.09	0.29	<0.01	0.26	0.59	6.19	0.25	0.04	0.86
T 1B	20-30	875	115	0.01	0.14	0.28	<0.01	0.57	0.64	11.27	0.39	<0.01	1.10
T 1B	30-40	2044	199	0.01	0.16	0.45	<0.01	1.62	1.29	24.55	1.03	0.01	3.69
T 2A	0-2.5	1377	109	0.12	0.50	0.54	<0.01	0.49	0.68	12.28	0.50	0.02	1.55
T 2A	2.5-5	1616	182	0.02	0.23	2.60	<0.01	1.15	1.17	19.10	1.00	<0.01	4.27
T 2A	5-10	458	88	0.03	0.44	0.51	<0.01	0.22	0.59	5.21	0.27	<0.01	1.35
T 2A	10-15	1688	214	0.01	0.49	0.68	<0.01	0.97	0.97	20.29	0.65	0.02	1.39
T 2A	15-20	479	100	0.02	0.21	0.47	<0.01	0.20	0.60	5.56	0.19	0.02	0.69
T 2A	20-30	572	105	0.03	0.37	0.57	<0.01	0.20	0.61	6.20	0.24	0.02	0.90
T 2A	30-40	578	116	0.03	0.44	0.35	<0.01	0.19	0.71	7.77	0.24	0.03	0.80
T 2B	0-2.5	1217	105	0.01	0.29	0.59	<0.01	0.56	0.71	14.76	0.48	<0.01	5.94
T 2B	2.5-5	801	146	0.01	0.46	0.65	<0.01	0.43	0.91	8.50	0.37	<0.01	1.48
T 2B	5-10	449	84	0.01	0.17	0.52	<0.01	0.22	0.47	4.49	0.25	0.03	0.79
T 2B	10-15	14632	1251	0.03	3.07	5.27	0.01	9.15	6.12	166.64	6.12	0.18	6.84
T 2B	15-20	4274	664	0.02	2.48	2.81	<0.01	1.96	3.84	53.73	1.84	0.11	3.42
T 2B	20-30	9867	1045	0.02	2.41	2.41	<0.01	6.20	4.95	122.27	3.99	0.13	3.28
T 2B	30-40	5182	666	0.02	1.72	3.36	<0.01	1.74	2.26	43.36	1.50	0.06	3.76

** 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)

Table 9-8. Characteristics of the Campbell Park soil materials, March 2012.

Profile ID* (Site Code, Core)	Depth Range (cm)	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)	Hydrolysable C (%C)	Total Organic C (%C)
CP 1A	0-2.5	22.77	8.96	421	9.26	0.00	0.24	18.23	0.00	<0.01	<0.001	<0.01	-32.41	0.22	0.027	0.04	0.16
CP 1A	2.5-5	18.90	7.85	332	6.47	1.99	0.00	0.00	0.00	<0.01	<0.001	<0.01	1.99	0.15	0.020	0.05	0.13
CP 1A	5-10	24.86	4.84	597	4.51	6.37	0.00	0.00	0.00	<0.01	<0.001	<0.01	6.37	0.40	0.035	0.14	0.36
CP 1A	10-15	24.15	4.72	574	4.54	12.63	0.00	0.00	0.00	<0.01	<0.001	<0.01	12.63	0.34	0.029	0.02	0.29
CP 1A	15-20	27.96	4.26	878	4.19	19.08	0.00	0.00	17.00	<0.01	<0.001	<0.01	36.08	0.42	0.043	0.07	0.41
CP 1A	20-30	23.38	4.10	782	4.20	12.50	0.00	0.00	3.00	0.07	<0.001	<0.01	57.72	0.28	0.025	0.06	0.26
CP 1A	30-40	38.54	4.30	1642	4.31	25.94	0.00	0.00	14.00	0.46	<0.001	<0.01	323.77	0.55	0.055	0.01	0.50
CP 1B	0-2.5	19.17	9.15	331	9.15	0.00	0.08	10.43	0.00	<0.01	<0.001	<0.01	-10.84	0.13	0.010	0.02	0.08
CP 1B	2.5-5	23.27	9.15	355	8.70	0.00	0.02	7.10	0.00	<0.01	<0.001	<0.01	-2.81	0.09	0.006	0.02	0.06
CP 1B	5-10	20.13	5.26	383	4.94	3.41	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.41	0.20	0.018	0.01	0.17
CP 1B	10-15	22.57	4.24	557	4.22	9.09	0.00	0.00	22.00	<0.01	<0.001	<0.01	31.09	0.27	0.026	0.09	0.26
CP 1B	15-20	23.41	4.17	698	4.24	12.99	0.00	0.00	33.00	<0.01	<0.001	<0.01	45.99	0.30	0.029	0.07	0.28
CP 1B	20-30	31.25	4.14	1138	4.15	25.45	0.00	0.00	5.00	0.12	<0.001	<0.01	105.20	0.43	0.037	0.01	0.40
CP 1B	30-40	52.35	4.11	2650	4.07	47.66	0.00	0.00	23.00	0.87	<0.001	<0.01	614.41	1.07	0.107	0.01	1.04
CP 2A	0-2.5	18.34	9.05	301	8.94	0.00	0.15	8.88	0.00	<0.01	<0.001	<0.01	-19.33	0.10	0.008	0.03	0.06
CP 2A	2.5-5	18.87	7.62	291	6.26	2.94	0.00	0.00	0.00	<0.01	<0.001	<0.01	2.94	0.15	0.010	0.05	0.14
CP 2A	5-10	19.42	5.57	439	5.21	3.18	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.18	0.19	0.011	<0.01	0.16
CP 2A	10-15	21.06	4.43	510	4.38	8.33	0.00	0.00	23.00	<0.01	<0.001	<0.01	31.33	0.20	0.015	0.07	0.19
CP 2A	15-20	22.71	4.53	691	4.49	8.92	0.00	0.00	16.00	<0.01	<0.001	<0.01	24.92	0.21	0.020	<0.01	0.20
CP 2A	20-30	23.48	4.05	894	4.22	11.30	0.00	0.00	6.00	0.04	<0.001	<0.01	39.98	0.26	0.022	0.05	0.22
CP 2A	30-40	34.95	4.05	1741	4.22	24.97	0.00	0.00	12.00	0.45	<0.001	<0.01	315.53	0.46	0.039	0.05	0.44
CP 2B	0-2.5	16.72	8.37	291	7.69	0.00	0.04	3.80	0.00	<0.01	<0.001	<0.01	-5.81	0.11	0.007	0.01	0.06
CP 2B	2.5-5	17.95	7.67	310	6.48	1.10	0.00	0.00	0.00	<0.01	0.004	<0.01	1.10	0.12	0.007	0.05	0.09
CP 2B	5-10	20.48	6.79	424	5.98	2.09	0.00	0.00	0.00	<0.01	0.002	<0.01	2.09	0.17	0.011	0.10	0.15
CP 2B	10-15	19.23	5.61	530	5.11	3.20	0.00	0.00	0.00	<0.01	<0.001	<0.01	3.20	0.17	0.006	0.05	0.12
CP 2B	15-20	21.99	4.59	798	4.47	8.13	0.00	0.00	16.00	<0.01	<0.001	<0.01	24.13	0.30	0.028	0.14	0.28
CP 2B	20-30	24.46	4.12	1095	4.24	11.68	0.00	0.00	15.00	0.05	<0.001	<0.01	56.08	0.36	0.042	0.13	0.33
CP 2B	30-40	34.78	4.17	1697	4.19	20.37	0.00	0.00	17.00	0.12	<0.001	<0.01	110.66	0.44	0.052	0.14	0.44

** 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 2012.

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
CP 1A	0-2.5	1308	124	0.03	0.61	0.81	<0.01	0.53	1.42	18.18	0.69	0.10	2.78
CP 1A	2.5-5	608	91	0.05	0.26	0.59	<0.01	0.25	0.99	6.18	0.46	0.04	1.44
CP 1A	5-10	775	165	0.05	0.60	1.00	<0.01	0.19	1.68	8.26	0.59	0.06	1.92
CP 1A	10-15	1329	256	0.03	0.28	0.57	<0.01	0.71	1.76	17.80	0.80	<0.01	1.53
CP 1A	15-20	1338	331	0.02	0.39	0.52	<0.01	0.50	1.74	22.61	0.72	<0.01	2.43
CP 1A	20-30	624	205	0.03	0.29	0.50	<0.01	0.22	1.77	12.17	0.73	0.01	1.43
CP 1A	30-40	1653	633	0.12	1.34	2.32	0.01	0.91	4.41	43.38	2.38	0.12	4.15
CP 1B	0-2.5	791	67	0.04	0.51	0.59	<0.01	0.25	0.67	12.71	0.42	<0.01	0.99
CP 1B	2.5-5	1029	80	0.02	0.44	0.53	<0.01	0.46	0.72	11.63	0.52	<0.01	0.85
CP 1B	5-10	763	118	0.02	0.28	0.53	<0.01	0.36	1.95	9.05	0.48	<0.01	1.65
CP 1B	10-15	622	145	0.04	0.01	0.51	<0.01	0.17	1.22	8.11	0.41	<0.01	0.98
CP 1B	15-20	679	194	0.03	0.12	0.45	<0.01	0.21	1.37	12.43	0.45	0.01	1.38
CP 1B	20-30	1248	454	0.03	0.59	0.79	<0.01	0.48	2.74	27.63	0.89	0.01	2.31
CP 1B	30-40	1781	1292	0.03	1.56	4.48	0.01	0.56	8.55	71.35	3.94	0.16	6.78
CP 2A	0-2.5	622	67	0.01	0.31	0.50	<0.01	0.22	0.67	11.13	0.45	<0.01	0.78
CP 2A	2.5-5	581	104	0.01	0.30	0.60	<0.01	0.24	0.98	6.94	0.50	<0.01	2.88
CP 2A	5-10	1076	160	0.12	0.26	0.65	<0.01	0.59	1.24	13.96	0.79	0.03	1.45
CP 2A	10-15	517	115	0.07	0.31	0.37	<0.01	0.16	0.95	7.24	0.34	<0.01	3.25
CP 2A	15-20	935	184	0.04	0.45	0.29	<0.01	0.45	1.09	15.73	0.48	0.08	1.45
CP 2A	20-30	410	170	0.03	0.09	0.36	<0.01	0.11	1.97	11.03	0.43	<0.01	1.40
CP 2A	30-40	837	523	0.02	0.94	1.79	0.01	0.26	2.85	32.26	2.41	0.08	4.33
CP 2B	0-2.5	434	67	0.01	0.30	0.40	<0.01	0.18	0.48	6.57	0.27	<0.01	2.20
CP 2B	2.5-5	1063	112	0.01	0.10	0.61	<0.01	0.47	0.96	10.34	0.50	<0.01	1.00
CP 2B	5-10	828	96	0.01	0.38	0.72	<0.01	0.35	0.99	9.42	0.56	<0.01	1.50
CP 2B	10-15	610	103	0.01	0.15	0.50	<0.01	0.31	0.87	9.02	0.58	<0.01	1.27
CP 2B	15-20	585	152	0.02	0.16	0.56	<0.01	0.20	1.26	10.29	0.43	0.01	1.28
CP 2B	20-30	597	193	0.17	0.07	0.36	<0.01	0.18	2.16	13.48	0.54	<0.01	1.19
CP 2B	30-40	837	429	0.04	0.24	0.75	<0.01	0.30	2.90	25.91	0.98	0.01	2.46

** 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. Data for sulfate reduction rate samples

Table 9-10. Mean total sulfate reduction rates for Waltowa in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Established *Cotula*	1A	0 - 2.5	41.050	31.788	0.000	24.279	21.530
	2A	2.5 - 5	1.002	0.000	1.206	0.736	0.645
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.422	0.000	0.000	0.141	0.244
	2B	2.5 - 5	34.321	1.659	1.165	12.381	19.001
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Juncus*	1A	0 - 2.5	21.344	14.191	33.248	22.928	9.627
	2A	2.5 - 5	19.662	11.299	54.733	28.565	23.045
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	1.739	0.937	0.892	0.870
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Phragmites*	1A	0 - 2.5	1118.890	960.591	673.004	917.495	226.045
	2A	2.5 - 5	1224.204	102.524	196.512	507.747	622.248
	3A	5 - 10	4.612	0.000	2.383	2.332	2.307
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	896.281	304.726	279.578	493.528	349.020
	2B	2.5 - 5	87.276	22.568	121.059	76.968	50.048
	3B	5 - 10	100.566	101.193	48.796	83.518	30.072
	4B	10 - 15	n.a.	141.277	161.160	151.219	14.059
	5B	15 - 20	43.518	461.699	69.086	191.434	234.405
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-11. Mean AVS sulfate reduction rates for Waltowa in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Established *Cotula*	1A	0 - 2.5	3.784	0.000	0.000	1.261	2.185
	2A	2.5 - 5	1.002	0.000	1.206	0.736	0.645
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	4.909	1.659	0.000	2.189	2.497
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Juncus*	1A	0 - 2.5	1.965	0.000	0.000	0.655	1.134
	2A	2.5 - 5	0.447	0.000	0.676	0.375	0.344
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	1.739	0.937	0.892	0.870
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Phragmites*	1A	0 - 2.5	20.011	15.428	3.441	12.960	8.556
	2A	2.5 - 5	14.856	2.319	0.876	6.017	7.689
	3A	5 - 10	4.612	0.000	2.383	2.332	2.307
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	22.886	7.809	10.034	13.577	8.139
	2B	2.5 - 5	5.940	0.000	3.451	3.130	2.983
	3B	5 - 10	1.782	0.000	1.297	1.027	0.921
	4B	10 - 15	n.a.	9.140	4.128	6.634	3.543
	5B	15 - 20	4.255	52.023	3.132	19.803	27.909
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-12. Mean S^0 sulfate reduction rates for Waltowa in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Established *Cotula*	1A	0 - 2.5	36.534	31.081	0.000	22.538	19.708
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	28.846	0.000	0.000	9.615	16.654
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Juncus*	1A	0 - 2.5	19.047	13.888	32.671	21.869	9.704
	2A	2.5 - 5	18.828	10.876	52.757	27.487	22.243
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Phragmites*	1A	0 - 2.5	1092.260	931.399	660.804	894.821	218.042
	2A	2.5 - 5	1200.026	98.143	193.905	497.358	610.409
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	829.801	283.746	260.582	458.043	322.160
	2B	2.5 - 5	78.483	21.895	114.989	71.789	46.907
	3B	5 - 10	96.943	97.890	45.747	80.193	29.835
	4B	10 - 15	n.a.	128.563	153.800	141.182	17.846
	5B	15 - 20	39.263	406.555	65.954	170.591	204.787
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-13. Mean pyrite sulfate reduction rates for Waltowa in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Established *Cotula*	1A	0 - 2.5	0.732	0.707	0.000	0.480	0.416
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.422	0.000	0.000	0.141	0.244
	2B	2.5 - 5	0.566	0.000	1.165	0.577	0.583
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Juncus*	1A	0 - 2.5	0.332	0.302	0.577	0.404	0.151
	2A	2.5 - 5	0.388	0.423	1.300	0.704	0.517
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
Established *Phragmites*	1A	0 - 2.5	6.619	13.765	8.759	9.714	3.667
	2A	2.5 - 5	9.323	2.062	1.731	4.372	4.291
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	43.593	13.170	8.963	21.909	18.897
	2B	2.5 - 5	2.853	0.673	2.620	2.049	1.197
	3B	5 - 10	1.841	3.303	1.752	2.299	0.871
	4B	10 - 15	n.a.	3.575	3.231	3.403	0.243
	5B	15 - 20	0.000	3.120	0.000	1.040	1.802
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-14. Mean total sulfate reduction rates for Poltalloch in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.070	0.000	0.023	0.041
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-15. Mean AVS sulfate reduction rates for Poltalloch in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.070	0.000	0.023
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-16. Mean S^0 sulfate reduction rates for Poltalloch in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-17. Mean pyrite sulfate reduction rates for Poltalloch in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-18. Mean total sulfate reduction rates for Tolderol in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	7.979	0.324	0.000	2.768	4.516
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 planted *Juncus* into 2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.249	0.000	0.083	0.144
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-19. Mean AVS sulfate reduction rates for Tolderol in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.662	0.324	0.000	0.328	0.331
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 planted *Juncus* into 2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.249	0.000	0.083	0.144
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-20. Mean S^0 sulfate reduction rates for Tolderol in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	7.317	0.000	0.000	2.439	4.225
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 planted *Juncus* into 2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-21. Mean pyrite sulfate reduction rates for Tolderol in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 planted *Juncus* into 2009 plantings of Bevy rye	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-22. Mean total sulfate reduction rates for Campbell Park in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 seeded with Bevy rye and *Puccinellia*	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-23. Mean AVS sulfate reduction rates for Campbell Park in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 seeded with Bevy rye and *Puccinellia*	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-24. Mean S^0 sulfate reduction rates for Campbell Park March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 seeded with Bevy rye and *Puccinellia*	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

Table 9-25. Mean pyrite sulfate reduction rates for Campbell Park in March 2012 (in units of nmol/g/day).

Treatment	Sample	Depth (cm)	Rep 1	Rep 2	Rep 3	Av.	S.D.
Scald (no bioremediation)	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000
2010 seeded with Bevy rye and *Puccinellia*	1A	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2A	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3A	5 - 10	0.000	0.000	0.000	0.000	0.000
	4A	10 - 15	0.000	0.000	0.000	0.000	0.000
	5A	15 - 20	0.000	0.000	0.000	0.000	0.000
	6A	20 - 30	0.000	0.000	0.000	0.000	0.000
	7A	30 - 40	0.000	0.000	0.000	0.000	0.000
	1B	0 - 2.5	0.000	0.000	0.000	0.000	0.000
	2B	2.5 - 5	0.000	0.000	0.000	0.000	0.000
	3B	5 - 10	0.000	0.000	0.000	0.000	0.000
	4B	10 - 15	0.000	0.000	0.000	0.000	0.000
	5B	15 - 20	0.000	0.000	0.000	0.000	0.000
	6B	20 - 30	0.000	0.000	0.000	0.000	0.000
	7B	30 - 40	0.000	0.000	0.000	0.000	0.000

Note: Values shown as 0.000 are less than the method detection limit.

APPENDIX 4. Pore-water characteristics

Table 9-26. Pore-water properties for Waltowa in March 2012.

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)	Total Fe (mg/L)
Established *Cotula*	1A	0 - 2.5	7.48	34.52	308	4220	455	1621	3.6	39	<0.01
	2A	2.5 - 5	7.78	29.76	301	4550	270	1103	4.1	69	0.05
	3A	5 - 10	6.95	19.62	196	5650	559	1652	3.0	36	0.10
	4A	10 - 15	5.94	15.61	164	6550	773	1855	2.4	313	1.98
	5A	15 - 20	6.29	19.62	173	7780	1119	2268	2.0	40	0.82
	6A	20 - 30	6.71	38.08	128	9380	1410	2806	2.0	43	1.21
	7A	30 - 40	6.67	42.81	116	9340	1434	2716	1.9	93	3.06
	1B	0 - 2.5	7.71	34.86	209	3650	268	1111	4.1	52	0.01
	2B	2.5 - 5	7.97	34.37	293	3770	293	1094	3.7	98	0.01
	3B	5 - 10	7.22	14.98	168	4910	399	1327	3.3	38	0.06
	4B	10 - 15	6.69	12.76	148	6130	738	1682	2.3	38	1.25
	5B	15 - 20	6.35	10.25	162	8600	1611	2245	1.4	235	6.99
	6B	20 - 30	6.19	10.06	210	10660	2780	2588	0.9	43	4.12
	7B	30 - 40	5.95	19.28	162	11810	2799	3080	1.1	90	4.69
Established *Juncus*	1A	0 - 2.5	8.06	30.10	363	4730	268	1196	4.5	86	0.02
	2A	2.5 - 5	8.20	27.24	339	4350	254	1224	4.8	132	<0.01
	3A	5 - 10	8.21	37.24	352	6240	285	1446	5.1	128	0.01
	4A	10 - 15	7.91	33.21	326	7870	421	2052	4.9	460	<0.01
	5A	15 - 20	7.72	69.74	344	9210	673	2889	4.3	68	<0.01
	6A	20 - 30	6.92	29.33	214	11080	1490	3663	2.5	23	0.13
	7A	30 - 40	7.26	26.90	254	11980	2069	4643	2.2	319	0.09
	1B	0 - 2.5	8.18	34.06	268	4080	271	1132	4.2	80	<0.01
	2B	2.5 - 5	8.21	31.56	286	4770	288	1263	4.4	45	<0.01
	3B	5 - 10	8.04	33.06	290	5280	295	1709	5.8	45	0.01
	4B	10 - 15	7.96	31.10	302	6460	502	2078	4.1	40	<0.01
	5B	15 - 20	7.77	34.17	317	11010	913	2912	3.2	40	<0.01
	6B	20 - 30	7.20	42.38	306	14380	1526	4041	2.6	48	<0.01
	7B	30 - 40	7.24	27.90	309	16250	1649	4676	2.8	120	<0.01

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-36 (continued). Pore-water properties for Waltowa in March 2012.

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)	Total Fe (mg/L)
Established *Phragmites*	1A	0 - 2.5	7.10	110.30	141	6310	113	1926	17.0	528	0.01
	2A	2.5 - 5	7.36	126.30	189	7940	148	2309	15.6	271	0.02
	3A	5 - 10	7.73	104.93	204	8310	544	3309	6.1	152	0.01
	4A	10 - 15	7.19	92.57	132	10820	922	3676	4.0	64	0.11
	5A	15 - 20	7.21	58.17	246	8940	1359	4081	3.0	29	0.19
	6A	20 - 30	6.82	43.67	140	12740	1940	4655	2.4	43	1.81
	7A	30 - 40	6.24	24.50	197	15490	4059	5368	1.3	57	11.34
	MBO	Surface	7.72	78.60	354	5300	84	1652	19.6	64	0.03
	1B	0 - 2.5	7.31	102.40	216	5530	63	1643	26.1	286	0.03
	2B	2.5 - 5	7.38	113.48	183	6270	109	1897	17.4	140	0.02
	3B	5 - 10	7.60	139.68	227	9950	118	3448	29.2	60	0.03
	4B	10 - 15	7.44	177.17	155	10710	202	3556	17.6	79	0.08
	5B	15 - 20	7.57	154.07	174	11400	286	4108	14.3	60	0.02
	6B	20 - 30	7.61	125.08	219	12050	818	4259	5.2	16	<0.01
	7B	30 - 40	7.43	112.57	224	12360	1224	4436	3.6	52	0.01

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-27. Pore-water soluble cation and nutrient analyses for Waltowa in March 2012.

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*	1A	0 - 2.5	102	201	1106	52	0.14	0.04	6.37	0.01
	2A	2.5 - 5	56	120	823	51	0.08	0.02	3.16	0.01
	3A	5 - 10	101	198	1181	61	0.14	0.01	6.47	0.01
	4A	10 - 15	132	239	1309	55	0.12	0.08	10.30	0.09
	5A	15 - 20	198	339	1532	55	0.11	0.03	12.91	0.04
	6A	20 - 30	259	429	1882	56	0.09	0.02	15.86	<0.01
	7A	30 - 40	274	414	1867	56	0.20	0.11	14.98	0.20
	1B	0 - 2.5	64	132	843	51	0.10	0.02	4.80	0.08
	2B	2.5 - 5	61	133	841	53	0.29	0.02	4.22	0.02
	3B	5 - 10	66	134	1003	51	0.12	0.02	4.17	0.01
	4B	10 - 15	110	195	1218	53	0.47	0.04	7.15	0.02
	5B	15 - 20	224	345	1634	71	0.64	0.08	12.01	0.08
	6B	20 - 30	522	471	1786	95	0.32	0.03	13.28	<0.01
	7B	30 - 40	451	577	2079	77	0.11	0.04	14.01	0.07
Established *Juncus*	1A	0 - 2.5	59	139	892	54	0.23	0.03	3.36	0.08
	2A	2.5 - 5	58	133	923	55	0.07	<0.01	1.93	0.02
	3A	5 - 10	62	157	1078	56	0.12	0.02	3.74	0.02
	4A	10 - 15	83	203	1430	67	<0.01	0.02	6.94	0.02
	5A	15 - 20	161	330	1972	72	0.03	0.01	10.10	0.02
	6A	20 - 30	287	463	2439	74	0.06	0.02	12.21	0.03
	7A	30 - 40	373	603	2947	90	0.11	0.02	18.05	0.02
	1B	0 - 2.5	55	131	849	49	0.19	0.03	2.97	0.02
	2B	2.5 - 5	61	147	938	53	0.30	0.06	3.07	<0.01
	3B	5 - 10	87	172	1209	57	0.02	0.01	3.71	<0.01
	4B	10 - 15	121	227	1455	60	<0.01	0.02	5.73	<0.01
	5B	15 - 20	200	360	1907	65	0.01	<0.01	7.70	0.01
	6B	20 - 30	282	526	2586	69	0.08	0.02	10.73	0.02
	7B	30 - 40	351	600	2792	90	0.12	0.02	20.70	0.03

Table 9-37 (continued). Pore-water soluble cation and nutrient analyses for Waltowa in March 2012.

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 *Phragmites*	1A	0 - 2.5	106	241	1445	54	<0.01	0.03	6.91	1.30
	2A	2.5 - 5	124	282	1708	80	0.29	0.09	0.06	0.34
	3A	5 - 10	155	395	2232	57	0.25	0.02	0.84	0.02
	4A	10 - 15	131	454	2468	65	0.13	0.02	0.56	0.01
	5A	15 - 20	251	489	2752	111	0.11	0.02	0.46	0.02
	6A	20 - 30	355	560	3020	68	0.15	0.07	0.48	0.09
	7A	30 - 40	741	831	3500	134	0.17	0.03	0.51	0.04
	MBO	Surface	84	194	1275	53	0.09	0.02	11.02	0.06
	1B	0 - 2.5	111	211	1229	42	0.02	0.01	7.98	1.18
	2B	2.5 - 5	117	234	1428	59	0.03	0.02	5.73	0.14
	3B	5 - 10	44	380	2361	182	0.12	0.03	0.70	0.04
	4B	10 - 15	66	450	2414	207	0.04	0.02	19.29	0.04
	5B	15 - 20	90	488	2771	122	0.04	0.01	10.38	0.02
	6B	20 - 30	149	516	2847	76	0.06	0.01	1.47	0.01
	7B	30 - 40	172	544	2878	193	0.07	<0.01	0.58	<0.01

Table 9-28. Pore-water properties for Poltalloch in March 2012.

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)	Total Fe (mg/L)
2009 plantings of Bevy rye	1A	0 - 2.5	7.80	9.67	252	464	92	126	1.4	45	0.01
	2A	2.5 - 5	7.69	14.16	256	637	90	149	1.7	10	<0.01
	3A	5 - 10	7.11	6.53	277	2130	350	611	1.7	23	<0.01
	4A	10 - 15	6.51	3.11	227	2960	673	976	1.5	21	0.35
	5A	15 - 20	4.94	2.43	319	5650	1224	1567	1.3	20	5.99
	6A	20 - 30	3.87	3.54	416	6110	1838	2453	1.3	16	8.35
	7A	30 - 40	4.11	3.49	418	8540	2333	3715	1.6	12	9.06
	1B	0 - 2.5	7.62	14.70	330	454	66	111	1.7	16	0.01
	2B	2.5 - 5	7.17	13.45	314	485	94	143	1.5	31	0.03
	3B	5 - 10	7.77	18.56	320	835	122	254	2.1	14	0.05
	4B	10 - 15	7.83	14.71	322	2317	382	718	1.9	52	0.02
	5B	15 - 20	7.85	14.47	334	3390	717	1143	1.6	3	<0.01
	6B	20 - 30	7.64	17.52	347	7590	1822	2433	1.3	113	0.05
	7B	30 - 40	7.92	24.12	343	8520	2463	3309	1.3	n.a.	0.03

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-29. Pore-water soluble cation and nutrient analyses for Poltalloch in March 2012.

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	1A	0 - 2.5	17	12	71	26	0.06	0.01	2.34	0.03
	2A	2.5 - 5	29	16	97	15	0.11	0.01	1.90	<0.01
	3A	5 - 10	84	57	449	68	0.06	0.03	2.26	0.03
	4A	10 - 15	145	111	734	87	0.18	0.01	2.05	0.01
	5A	15 - 20	248	190	1154	107	0.32	0.03	2.34	0.01
	6A	20 - 30	370	293	1762	75	0.31	0.03	2.23	0.01
	7A	30 - 40	431	418	2572	181	0.35	0.02	2.19	0.01
	1B	0 - 2.5	18	15	70	13	0.14	0.04	3.78	0.04
	2B	2.5 - 5	22	13	96	16	0.10	0.01	2.46	0.03
	3B	5 - 10	36	25	202	39	0.12	<0.01	2.51	<0.01
	4B	10 - 15	98	73	541	86	0.06	0.01	2.13	<0.01
	5B	15 - 20	181	137	881	56	0.02	0.02	2.52	<0.01
	6B	20 - 30	406	322	1802	94	0.06	0.02	1.58	<0.01
	7B	30 - 40	533	448	2358	128	0.07	0.01	1.24	<0.01

Table 9-30. Pore-water properties for Tolderol in March 2012.

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)	Total Fe (mg/L)
2010 planted *Juncus* into 2009 plantings of Bevy rye	1A	0 - 2.5	8.01	12.48	342	503	97	122	1.3	26	0.04
	2A	2.5 - 5	8.12	17.60	341	697	123	139	1.1	16	0.04
	3A	5 - 10	8.05	15.44	356	744	121	184	1.5	69	<0.01
	4A	10 - 15	8.11	9.31	337	400	87	112	1.3	62	0.03
	5A	15 - 20	7.80	12.27	340	746	122	178	1.5	28	<0.01
	6A	20 - 30	7.29	8.52	345	568	100	127	1.3	n.a.	0.05
	7A	30 - 40	5.70	3.58	301	3980	963	1007	1.0	36	0.74
	1B	0 - 2.5	7.57	15.02	362	546	88	102	1.2	30	0.01
	2B	2.5 - 5	7.76	33.91	351	815	130	132	1.0	40	0.03
	3B	5 - 10	7.79	7.99	350	498	98	95	1.0	72	0.01
	4B	10 - 15	7.29	6.74	358	898	233	181	0.8	12	0.08
	5B	15 - 20	5.16	3.48	310	3490	1120	714	0.6	24	3.42
	6B	20 - 30	6.13	3.48	358	2102	563	449	0.8	19	0.03
	7B	30 - 40	4.59	3.19	392	4030	1250	882	0.7	22	4.94
Scald (no bioremediation)	1A	0 - 2.5	7.54	12.73	372	522	88	80	0.9	14	0.03
	2A	2.5 - 5	6.69	2.74	363	411	71	84	1.2	26	0.03
	3A	5 - 10	5.68	2.91	362	395	79	97	1.2	38	0.30
	4A	10 - 15	5.21	1.89	373	357	82	99	1.2	23	0.86
	5A	15 - 20	4.68	4.31	396	828	241	139	0.6	18	4.07
	6A	20 - 30	4.05	2.21	432	1956	866	236	0.3	57	26.72
	7A	30 - 40	3.50	3.19	459	3700	1767	430	0.2	84	41.42
	1B	0 - 2.5	6.35	7.28	280	482	111	81	0.7	9	0.08
	2B	2.5 - 5	6.08	4.64	266	658	211	88	0.4	30	1.79
	3B	5 - 10	6.66	6.22	249	543	91	95	1.0	34	0.02
	4B	10 - 15	5.42	4.44	285	323	69	84	1.2	54	0.20
	5B	15 - 20	4.91	4.44	341	405	73	82	1.1	35	0.42
	6B	20 - 30	5.11	4.21	385	386	75	82	1.1	34	0.55
	7B	30 - 40	4.63	3.69	403	401	88	98	1.1	31	0.88

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-31. Pore-water soluble cation and nutrient analyses for Tolderol in March 2012.

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	1A	0 - 2.5	23	15	73	42	0.04	<0.01	0.54	<0.01
	2A	2.5 - 5	38	21	101	13	0.13	0.01	0.70	<0.01
	3A	5 - 10	23	27	121	69	0.22	0.07	0.99	0.06
	4A	10 - 15	11	17	79	17	0.10	0.04	0.34	0.02
	5A	15 - 20	14	28	119	22	0.06	0.01	0.88	0.04
	6A	20 - 30	7	15	98	17	0.07	0.03	0.68	0.09
	7A	30 - 40	98	189	820	54	0.07	0.02	2.22	<0.01
	1B	0 - 2.5	27	19	78	12	0.15	<0.01	1.32	0.04
	2B	2.5 - 5	45	31	105	16	0.09	0.01	1.31	0.02
	3B	5 - 10	17	15	79	12	0.11	0.01	0.74	0.01
	4B	10 - 15	18	32	154	22	0.28	0.03	2.34	0.01
	5B	15 - 20	103	204	646	55	0.23	0.01	4.58	0.02
	6B	20 - 30	44	91	394	38	0.11	0.01	2.93	<0.01
	7B	30 - 40	115	216	768	59	0.37	0.03	3.74	0.01
Scald (no bioremediation)	1A	0 - 2.5	20	18	75	11	0.11	0.03	1.22	0.01
	2A	2.5 - 5	7	7	67	14	0.06	0.02	0.35	0.02
	3A	5 - 10	4	4	66	16	0.14	<0.01	0.70	<0.01
	4A	10 - 15	3	5	78	20	0.11	0.01	1.28	0.03
	5A	15 - 20	10	17	113	27	0.46	0.01	2.99	0.01
	6A	20 - 30	34	57	245	56	<0.01	0.07	5.69	0.01
	7A	30 - 40	97	171	471	49	0.63	0.08	8.40	0.01
	1B	0 - 2.5	14	14	72	10	0.03	<0.01	0.32	0.01
	2B	2.5 - 5	17	20	83	19	0.06	0.02	0.78	0.01
	3B	5 - 10	9	8	72	14	0.02	0.02	0.43	<0.01
	4B	10 - 15	4	4	65	15	0.10	0.01	0.70	0.01
	5B	15 - 20	3	4	72	15	0.07	0.01	1.08	<0.01
	6B	20 - 30	2	3	66	15	0.11	0.02	0.95	<0.01
	7B	30 - 40	3	4	67	13	0.04	0.10	1.47	<0.01

Table 9-32. Pore-water properties for Campbell Park in March 2012.

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)	Total Fe (mg/L)
Control (no bioremediation)	1A	0 - 2.5	7.11	34.08	164	4190	363	1295	3.6	19	0.09
	2A	2.5 - 5	6.82	17.83	157	4190	472	1198	2.5	21	0.73
	3A	5 - 10	5.83	6.21	230	5640	1334	1460	1.1	90	22.36
	4A	10 - 15	4.06	6.07	350	7720	2626	1785	0.7	84	78.43
	5A	15 - 20	3.57	4.82	311	8500	2930	1856	0.6	41	76.99
	6A	20 - 30	3.48	4.34	441	9000	3240	1983	0.6	66	70.93
	7A	30 - 40	3.64	4.96	452	9220	3049	2389	0.8	21	31.84
	1B	0 - 2.5	7.29	34.22	171	3950	371	1106	3.0	21	0.05
	2B	2.5 - 5	6.46	10.81	179	4480	612	1281	2.1	49	2.90
	3B	5 - 10	5.66	5.95	205	5580	1358	1459	1.1	119	18.52
	4B	10 - 15	4.03	5.32	288	6630	2333	1480	0.6	85	7.03
	5B	15 - 20	3.61	0.32	384	7960	2975	1568	0.5	120	8.88
	6B	20 - 30	3.45	3.76	403	9010	3670	1853	0.5	141	8.71
	7B	30 - 40	4.10	2.67	412	8480	2912	1978	0.7	19	3.48
2010 seeded with Bevy rye and *Puccinellia*	1A	0 - 2.5	7.19	28.57	261	4150	411	1162	2.8	55	0.17
	2A	2.5 - 5	6.46	3.60	178	4470	680	1270	1.9	24	3.16
	3A	5 - 10	4.46	3.92	316	6390	1791	1585	0.9	43	24.48
	4A	10 - 15	3.82	10.99	383	7890	2895	1791	0.6	104	63.91
	5A	15 - 20	3.55	31.98	452	8560	3470	1886	0.5	247	68.21
	6A	20 - 30	3.54	31.35	458	9190	3995	2112	0.5	47	46.13
	7A	30 - 40	3.80	32.54	452	9790	4394	2382	0.5	27	33.94
	1B	0 - 2.5	7.82	46.87	353	4160	395	1161	2.9	74	0.01
	2B	2.5 - 5	7.54	21.86	260	4060	434	1155	2.7	116	<0.01
	3B	5 - 10	6.99	13.50	215	5830	944	1684	1.8	108	<0.01
	4B	10 - 15	4.96	3.80	339	7420	2167	1964	0.9	20	16.20
	5B	15 - 20	5.11	3.96	448	8680	3029	2063	0.7	56	55.66
	6B	20 - 30	3.79	3.72	438	10900	4604	2676	0.6	47	80.41
	7B	30 - 40	3.69	3.23	446	10730	4353	2648	0.6	163	62.74

* Eh measurements are presented versus the standard hydrogen electrode

Table 9-33. Pore-water soluble cation and nutrient analyses for Campbell Park in March 2012.

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)	1A	0 - 2.5	107	150	882	144	0.13	<0.01	2.44	0.01
	2A	2.5 - 5	107	140	861	57	0.15	<0.01	2.99	<0.01
	3A	5 - 10	116	219	1088	110	0.31	0.04	7.62	0.02
	4A	10 - 15	178	335	1309	114	0.55	0.12	11.19	0.04
	5A	15 - 20	201	401	1427	95	0.51	0.12	11.56	0.06
	6A	20 - 30	240	455	1510	92	0.52	0.10	11.55	0.04
	7A	30 - 40	310	567	1743	100	0.01	0.08	9.32	0.03
	1B	0 - 2.5	84	138	853	48	0.08	0.02	2.68	0.01
	2B	2.5 - 5	75	154	942	74	0.01	0.02	4.19	0.03
	3B	5 - 10	112	244	1125	100	0.28	0.04	7.70	0.02
	4B	10 - 15	138	282	1171	97	0.53	0.06	8.40	0.03
	5B	15 - 20	179	357	1272	94	<0.01	0.13	9.37	0.08
	6B	20 - 30	250	486	1471	93	0.40	0.15	9.87	0.07
	7B	30 - 40	294	525	1498	103	0.06	0.09	6.56	0.03
2010 seeded with Bevy rye and *Puccinellia*	1A	0 - 2.5	84	143	891	44	0.07	0.01	2.41	0.01
	2A	2.5 - 5	87	166	974	64	0.21	0.03	4.25	0.02
	3A	5 - 10	150	299	1267	97	0.23	0.06	9.22	0.01
	4A	10 - 15	209	408	1462	90	0.41	0.12	11.95	0.04
	5A	15 - 20	243	471	1554	79	0.47	0.12	13.18	0.07
	6A	20 - 30	309	574	1736	81	0.55	0.09	14.05	0.04
	7A	30 - 40	389	727	1961	107	<0.01	0.07	12.80	0.04
	1B	0 - 2.5	84	137	884	50	0.26	0.01	3.53	<0.01
	2B	2.5 - 5	69	132	919	56	0.98	0.02	4.39	0.01
	3B	5 - 10	130	243	1272	71	0.11	0.10	6.82	0.11
	4B	10 - 15	210	398	1560	118	0.44	0.04	12.77	<0.01
	5B	15 - 20	246	454	1654	90	0.13	0.09	12.51	0.03
	6B	20 - 30	357	685	2182	97	<0.01	0.12	15.98	0.06
	7B	30 - 40	357	673	2119	91	<0.01	0.10	15.62	0.03

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APPENDIX 5. Pore-water plots

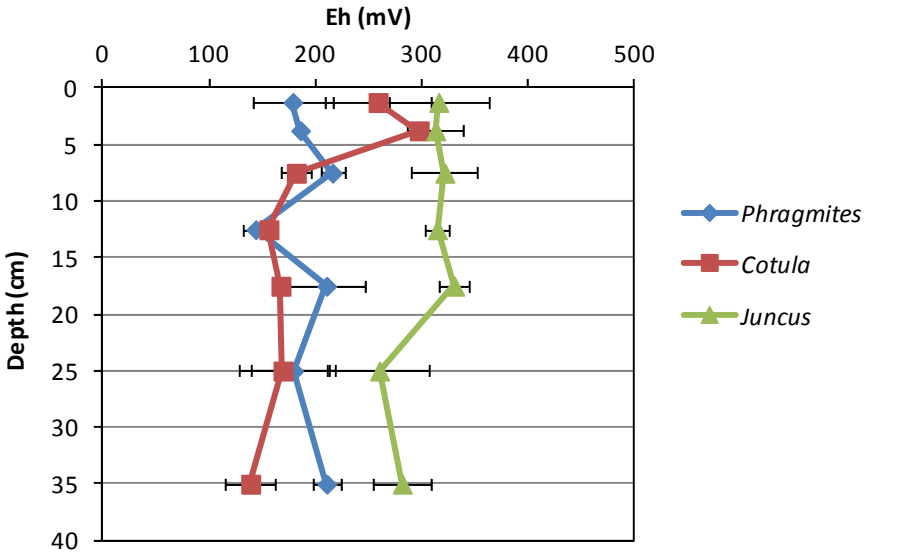

Figure 9-1. Pore-water Eh characteristics at the Waltowa study area (March 2012).

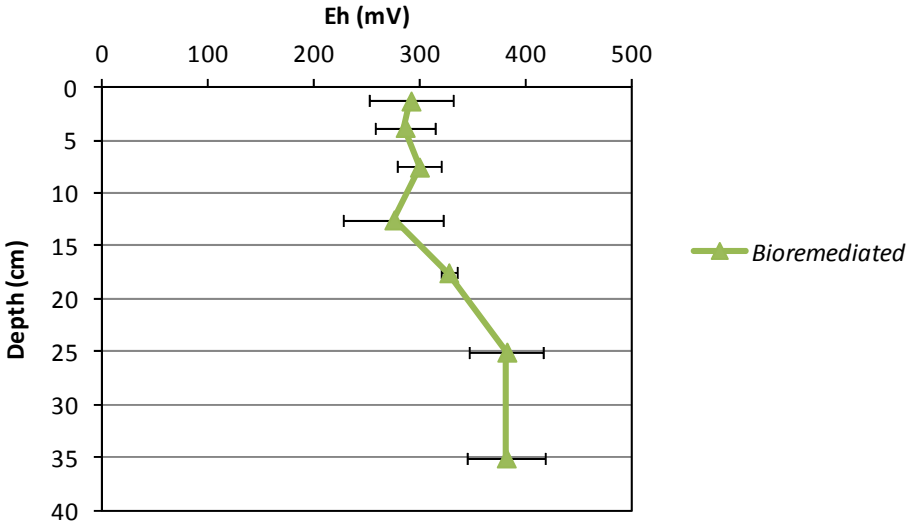

Figure 9-2. Pore-water Eh characteristics at the Poltalloch Bevy rye site (March 2012).

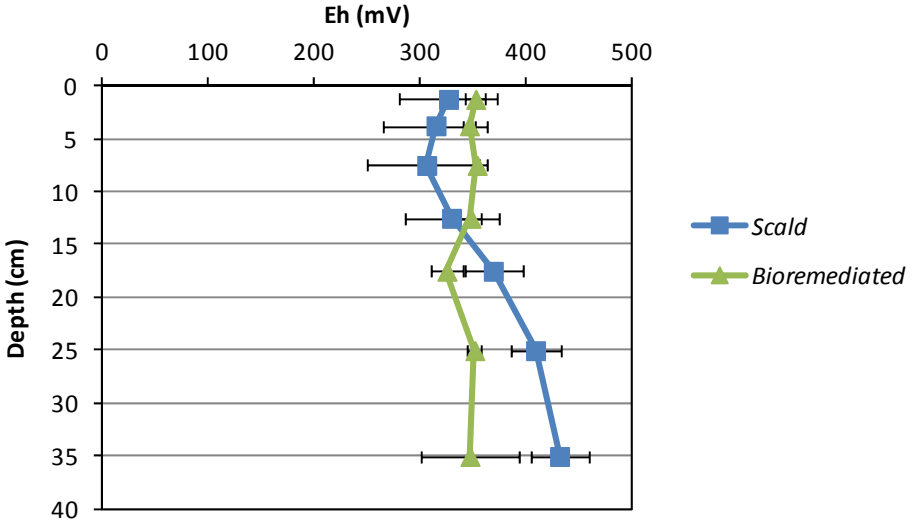

Figure 9-3. Pore-water Eh characteristics at the Tolderol study area (March 2012).

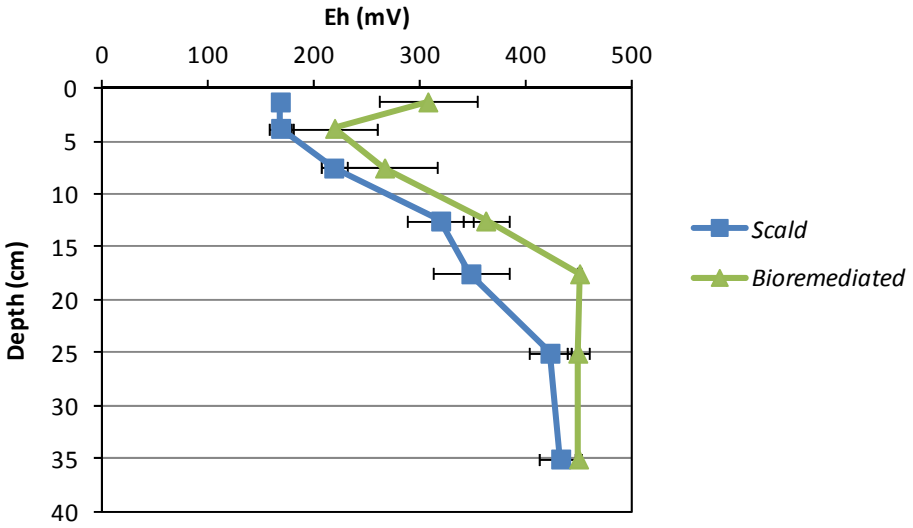

Figure 9-4. Pore-water Eh characteristics at the Campbell Park study area (March 2012).

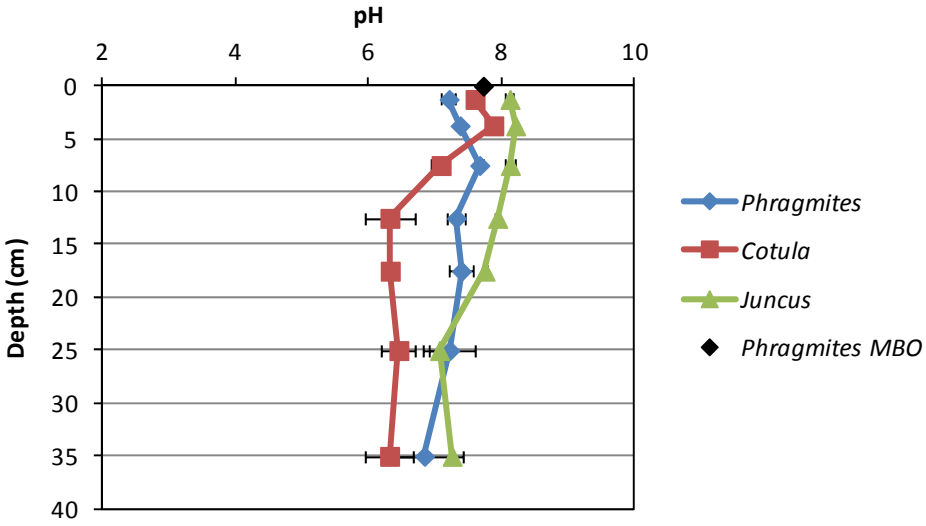

Figure 9-5. Pore-water pH characteristics at the Waltowa study area (March 2012).

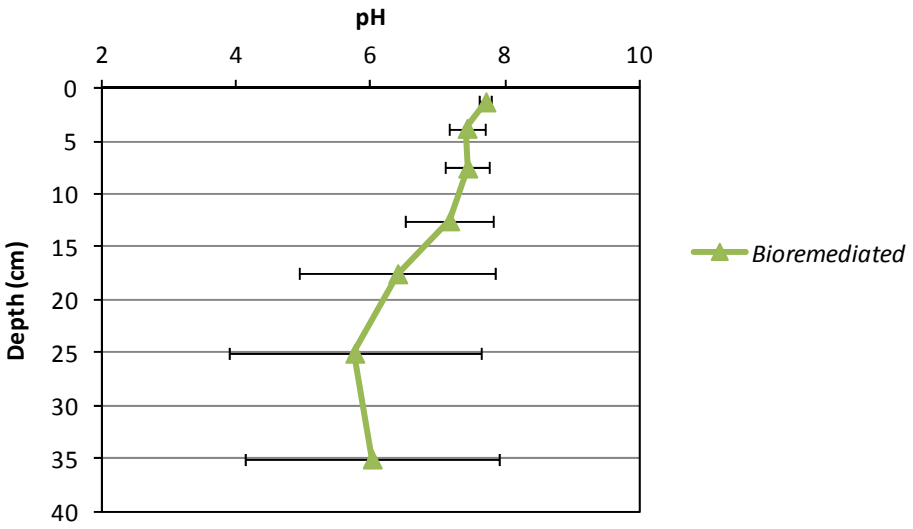

Figure 9-6. Pore-water pH characteristics at the Poltalloch Bevy rye site (March 2012).

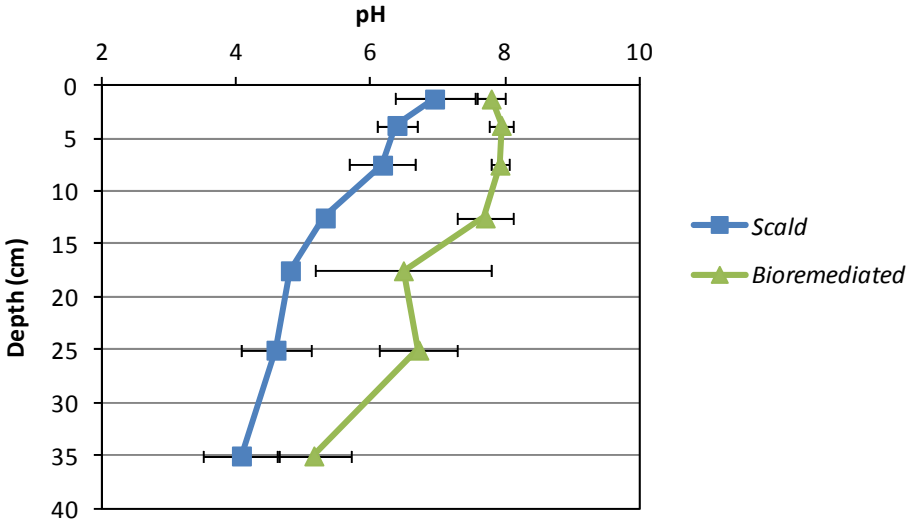

Figure 9-7. Pore-water pH characteristics at the Tolderol study area (March 2012).

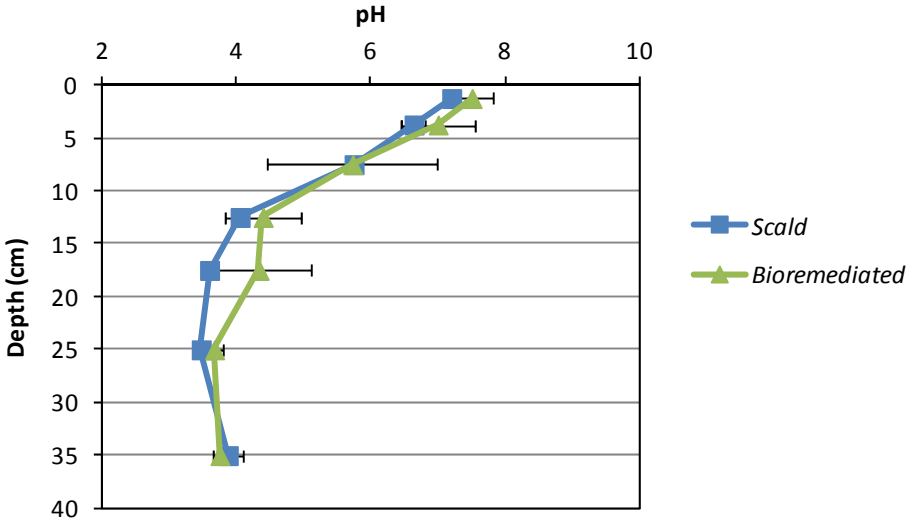

Figure 9-8. Pore-water pH characteristics at the Campbell Park study area (March 2012).

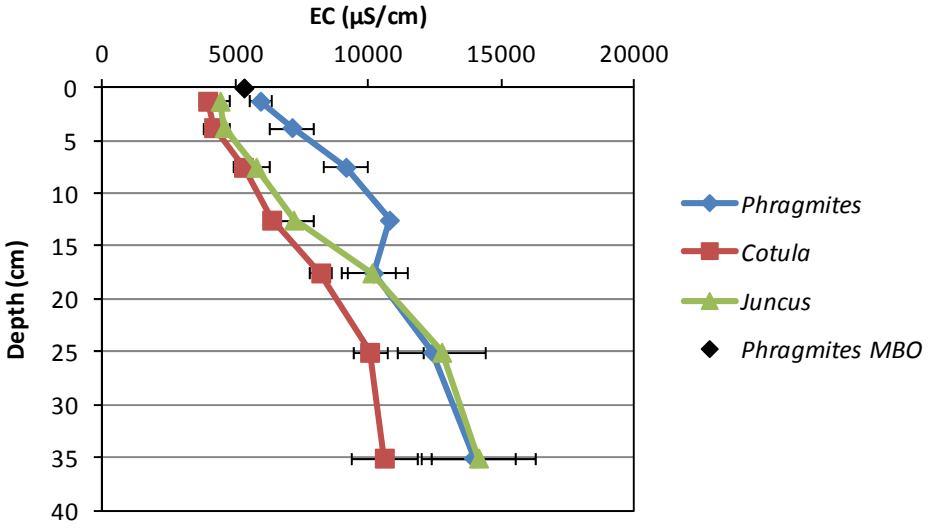

Figure 9-9. Pore-water EC characteristics at the Waltowa study area (March 2012).

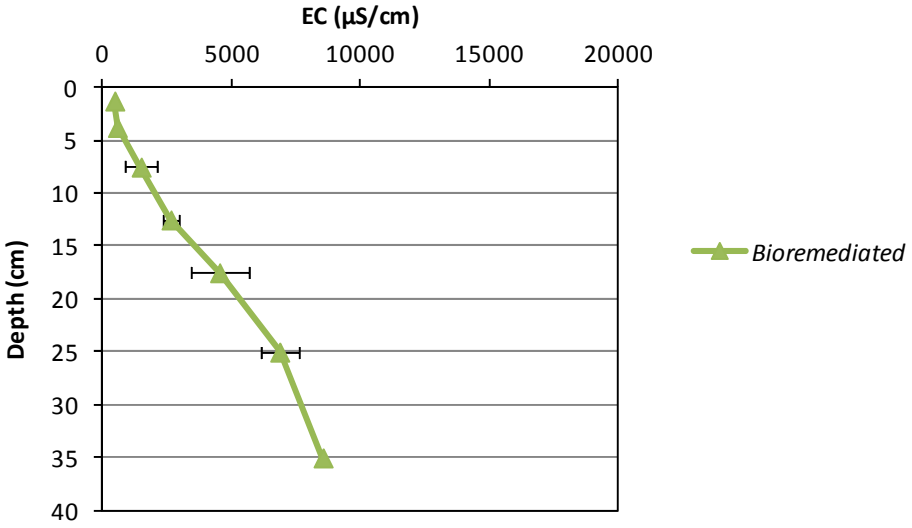

Figure 9-10. Pore-water EC characteristics at the Poltalloch Bevy rye site (March 2012).

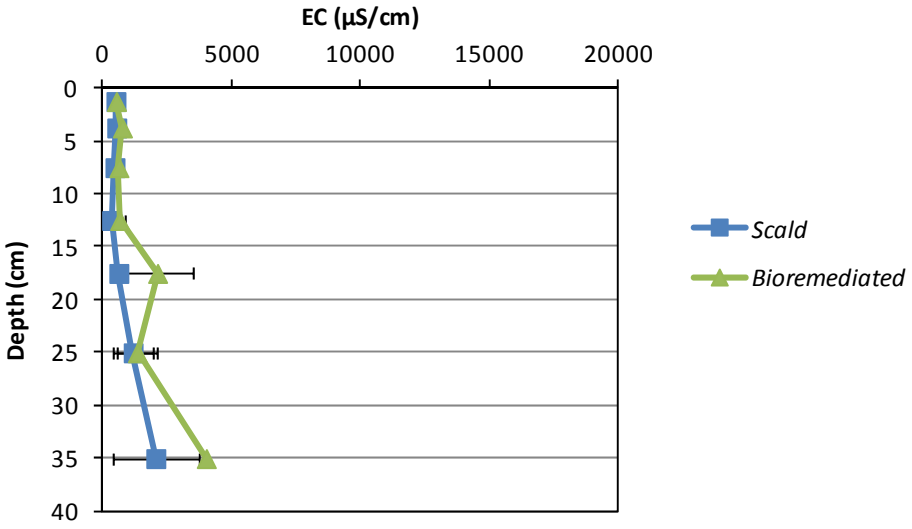

Figure 9-11. Pore-water EC characteristics at the Tolderol study area (March 2012).

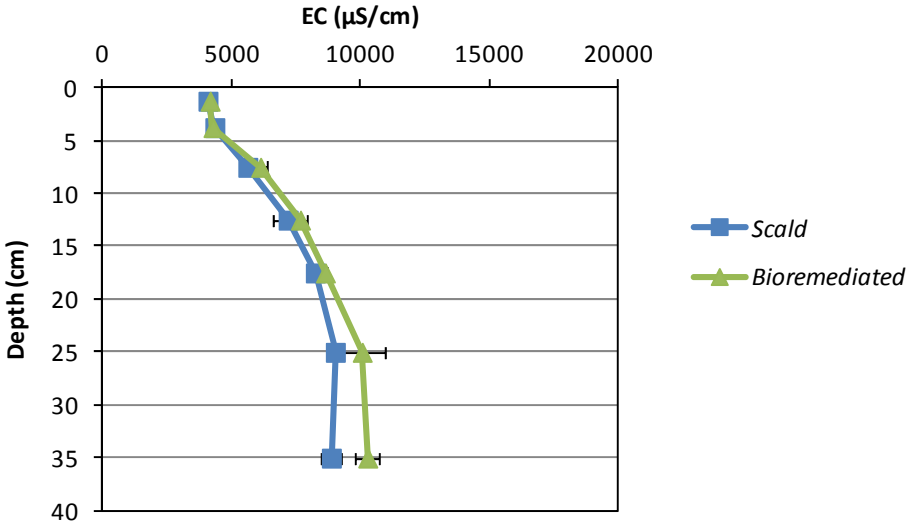

Figure 9-12. Pore-water EC characteristics at the Campbell Park study area (March 2012).

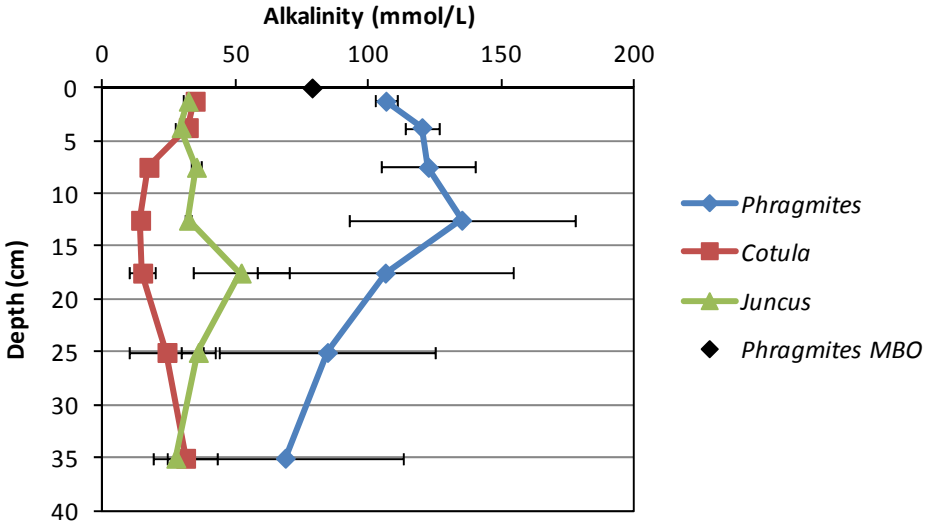

Figure 9-13. Pore-water alkalinity characteristics at the Waltowa study area (March 2012).

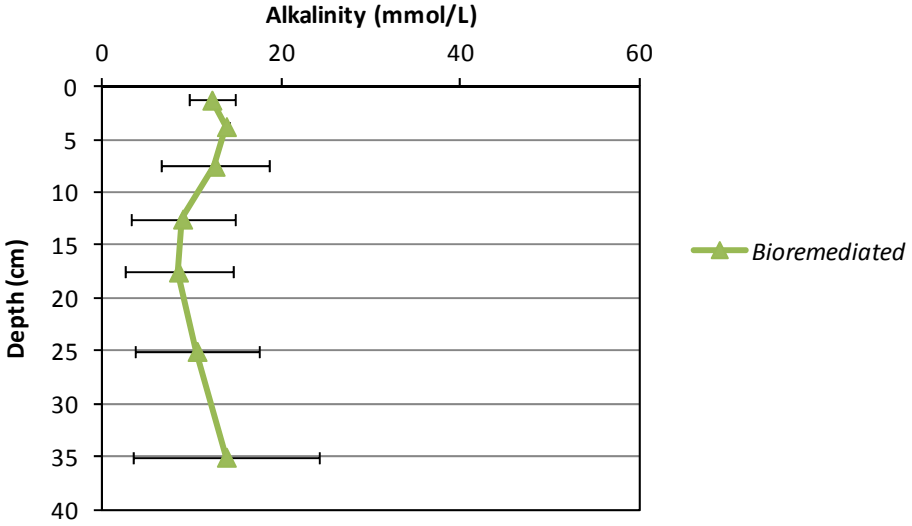

Figure 9-14. Pore-water alkalinity characteristics at the Poltalloch Bevy rye site (March 2012).

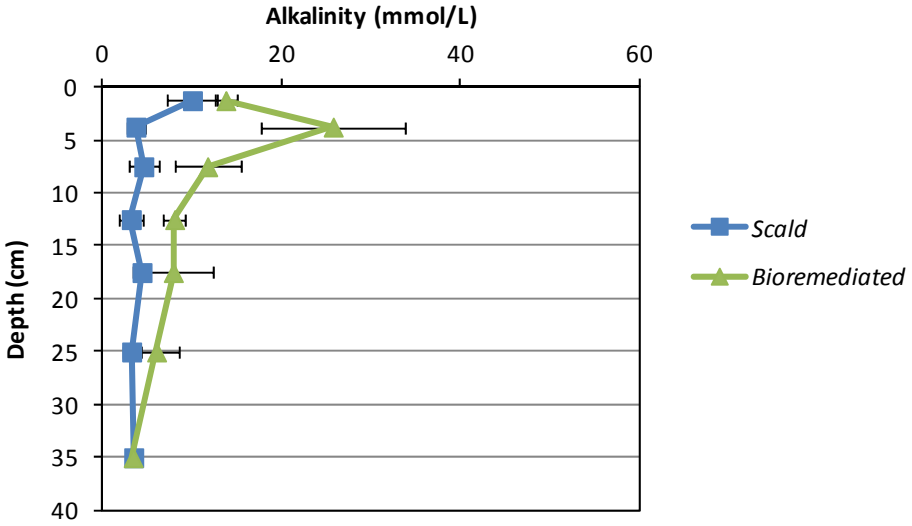

Figure 9-15. Pore-water alkalinity characteristics at the Tolderol study area (March 2012).

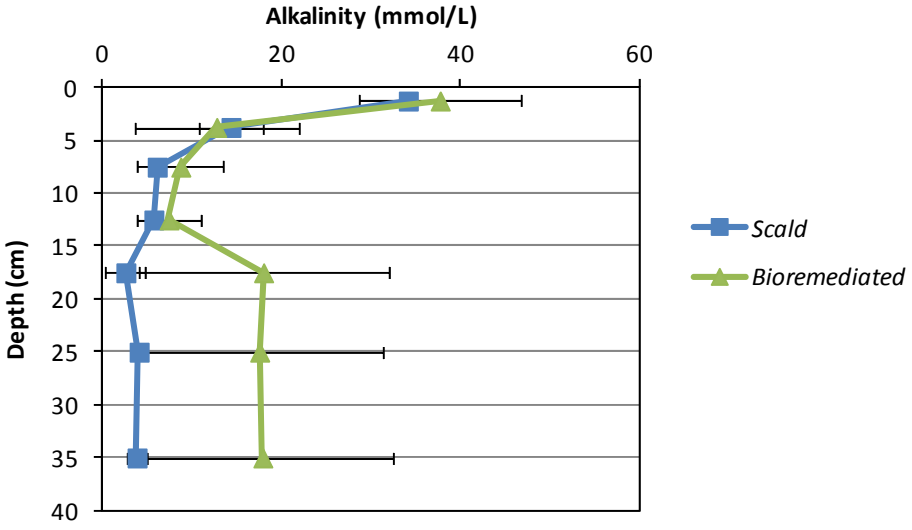

Figure 9-16. Pore-water alkalinity characteristics at the Campbell Park study area (March 2012).

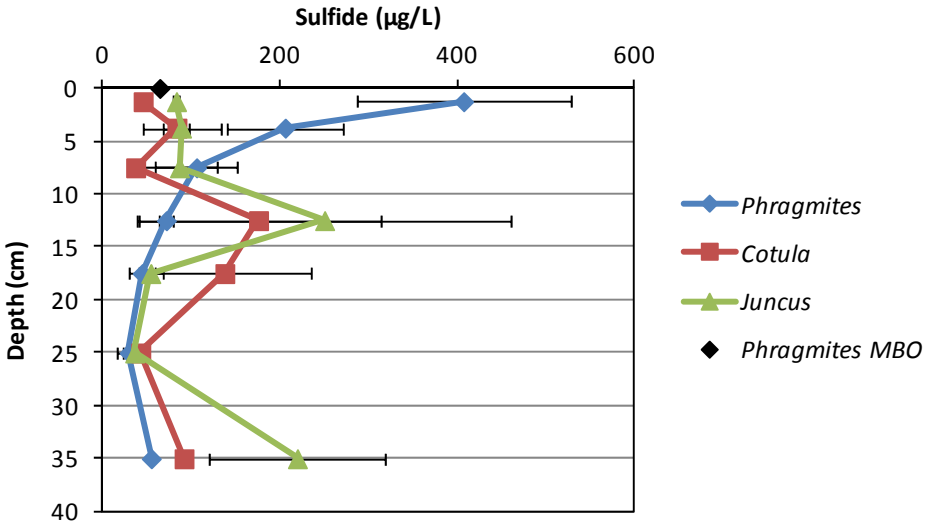

Figure 9-17. Pore-water dissolved sulfide characteristics at the Waltowa study area (March 2012).

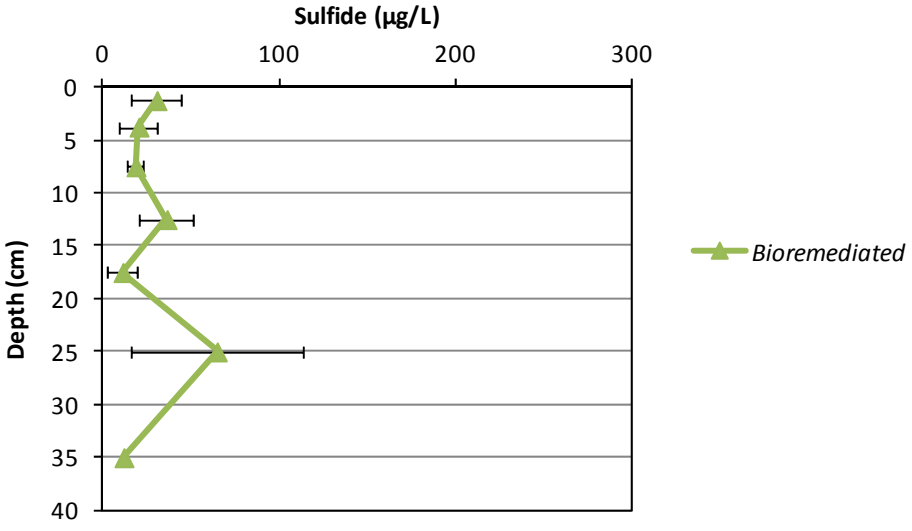

Figure 9-18. Pore-water dissolved sulfide characteristics at the Poltalloch Bevy rye site (March 2012).

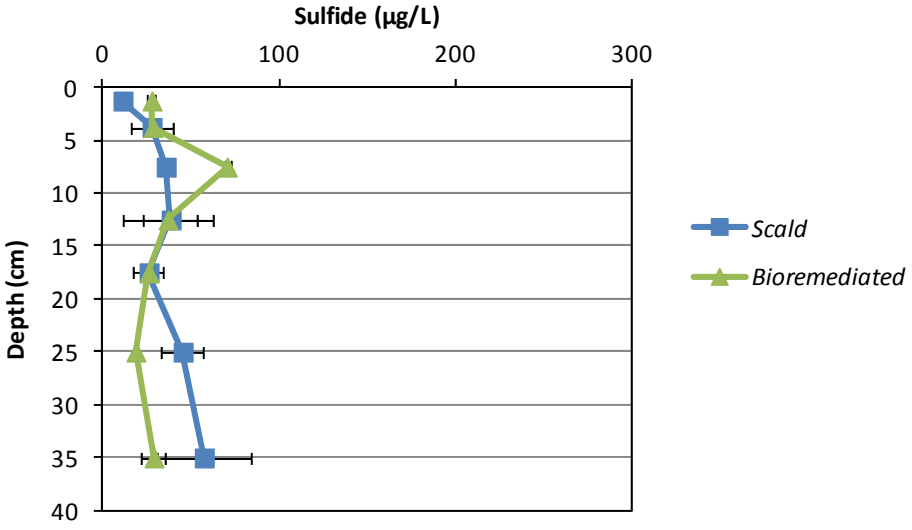

Figure 9-19. Pore-water dissolved sulfide characteristics at the Tolderol study area (March 2012).

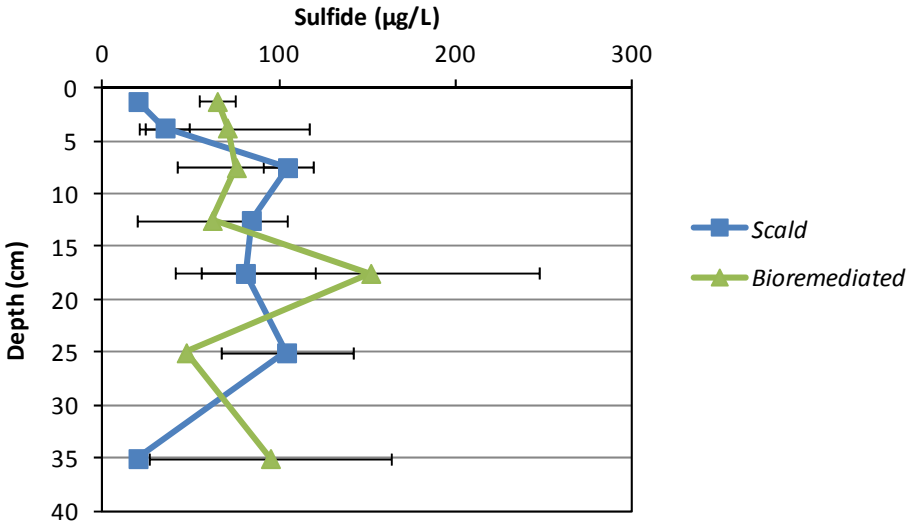

Figure 9-20. Pore-water dissolved sulfide characteristics at the Campbell Park study area (March 2012).

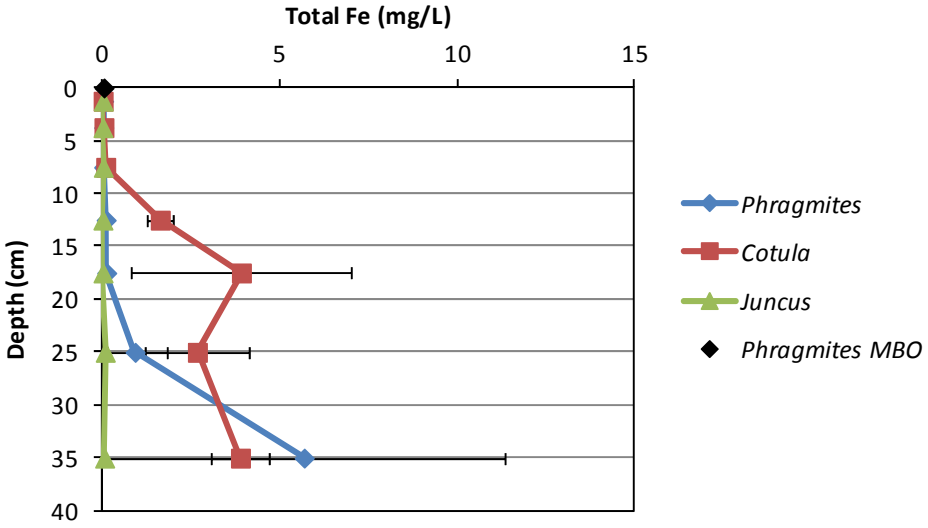

Figure 9-21. Pore-water total dissolved iron (Fe^{3+} + Fe^{2+}) characteristics at the Waltowa study area (March 2012).

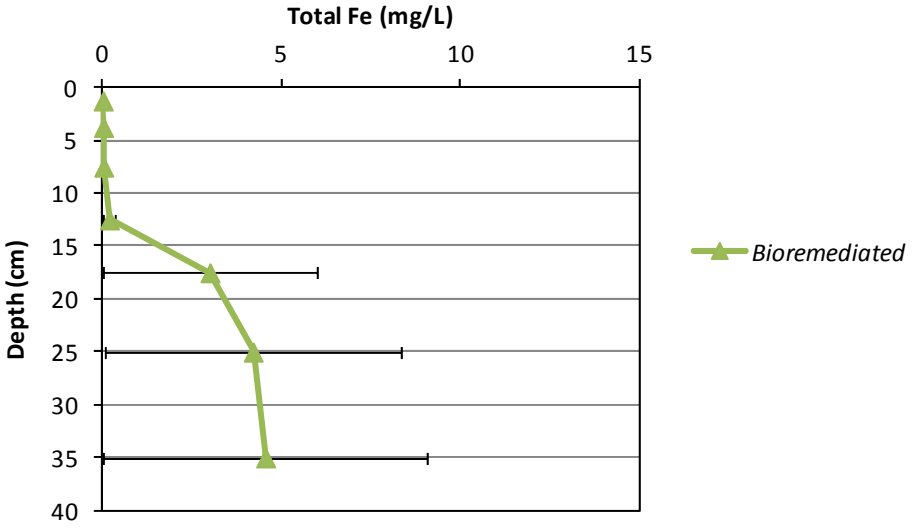

Figure 9-22. Pore-water total dissolved iron (Fe^{3+} + Fe^{2+}) characteristics at the Poltalloch Bevy rye site (March 2012).

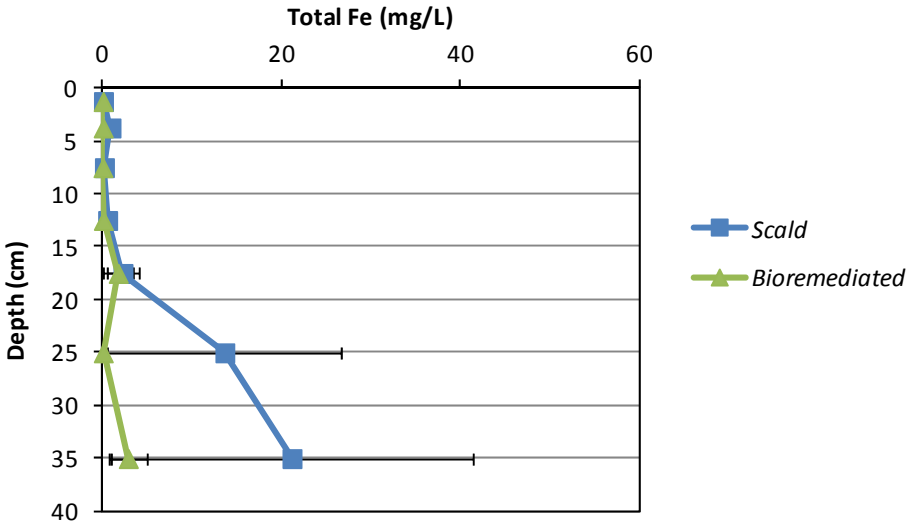

Figure 9-23. Pore-water total dissolved iron (Fe^{3+} + Fe^{2+}) characteristics at the Tolderol study area (March 2012).

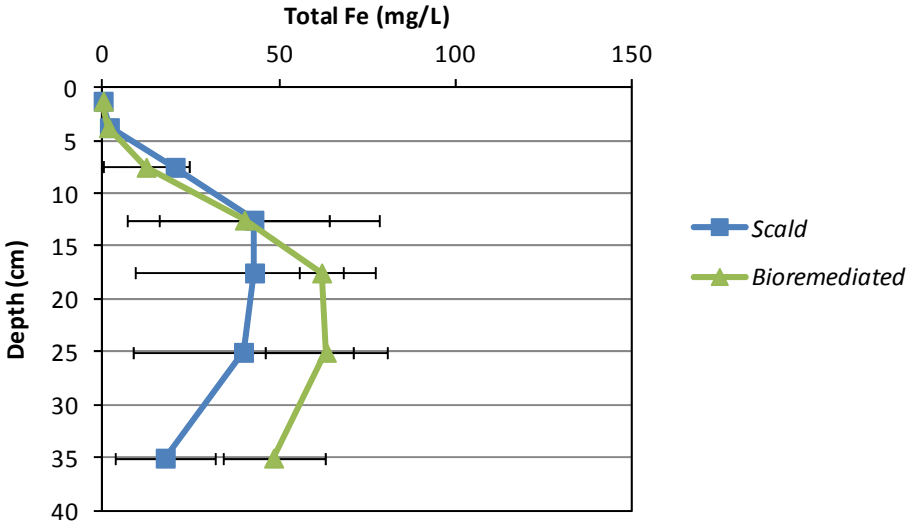

Figure 9-24. Pore-water total dissolved iron (Fe^{3+} + Fe^{2+}) characteristics at the Campbell Park study area (March 2012).

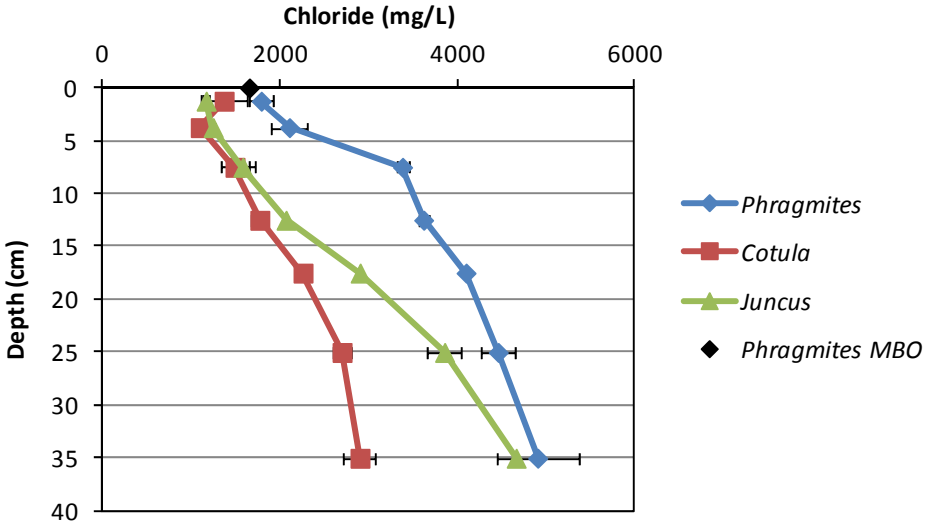

Figure 9-25. Pore-water soluble chloride characteristics at the Waltowa study area (March 2012).

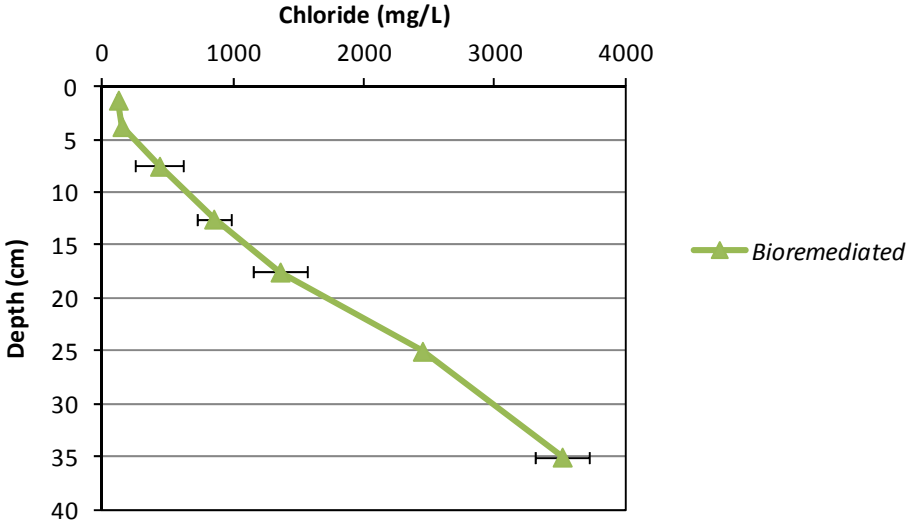

Figure 9-26. Pore-water soluble chloride characteristics at the Poltalloch Bevy rye site (March 2012).

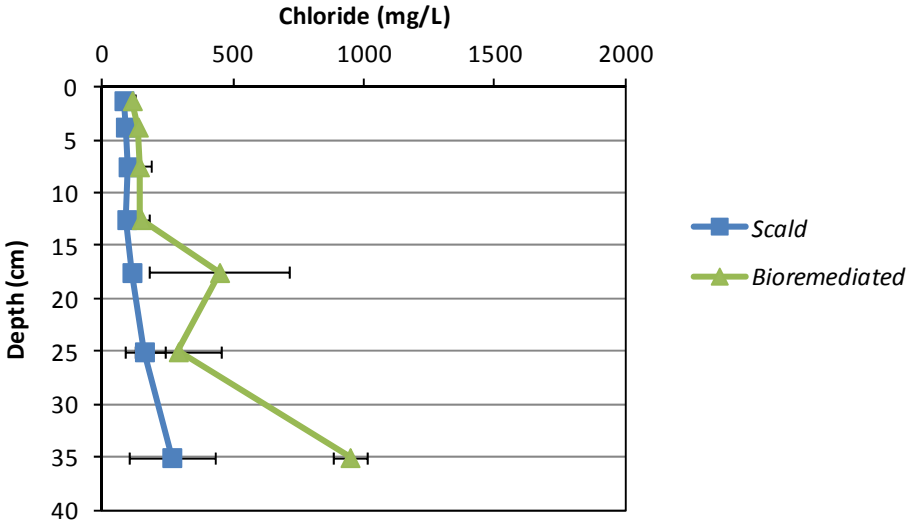

Figure 9-27. Pore-water soluble chloride characteristics at the Tolderol study area (March 2012).

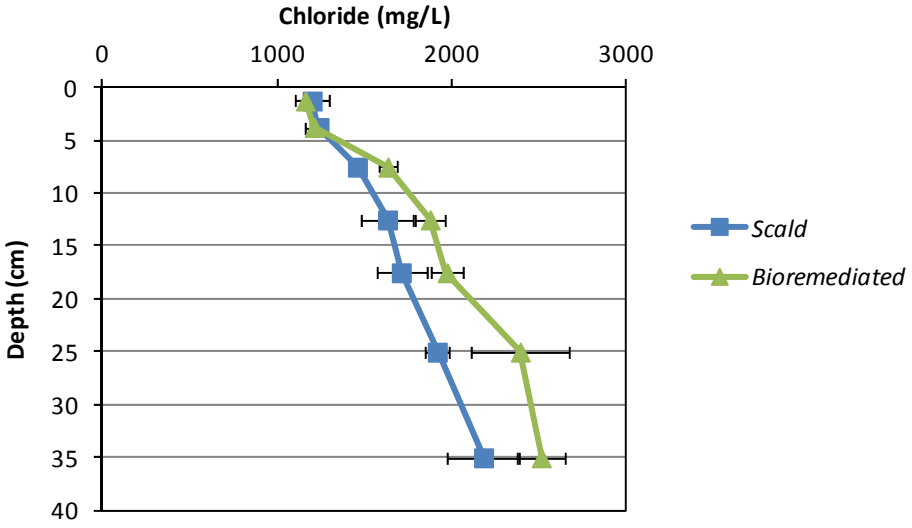

Figure 9-28. Pore-water soluble chloride characteristics at the Campbell Park study area (March 2012).

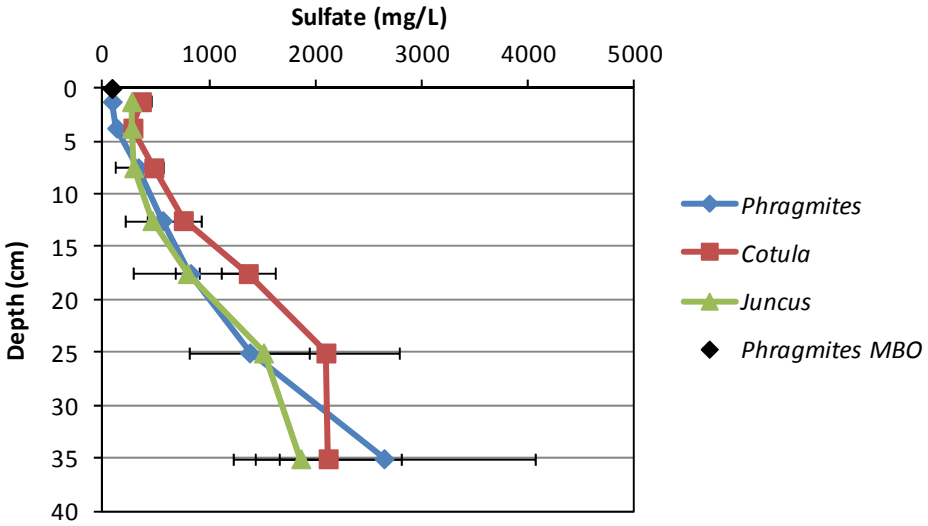

Figure 9-29. Pore-water soluble sulfate characteristics at the Waltowa study area (March 2012).

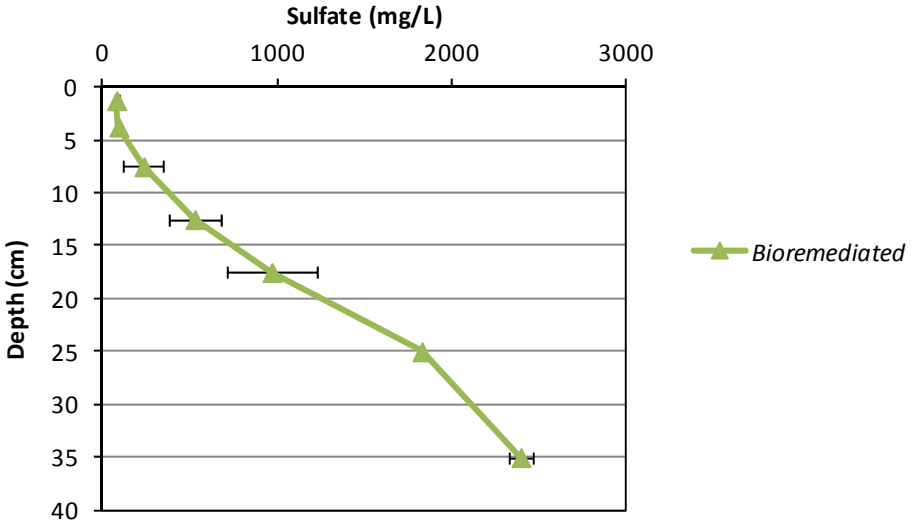

Figure 9-30. Pore-water soluble sulfate characteristics at the Poltalloch Bevy rye site (March 2012).

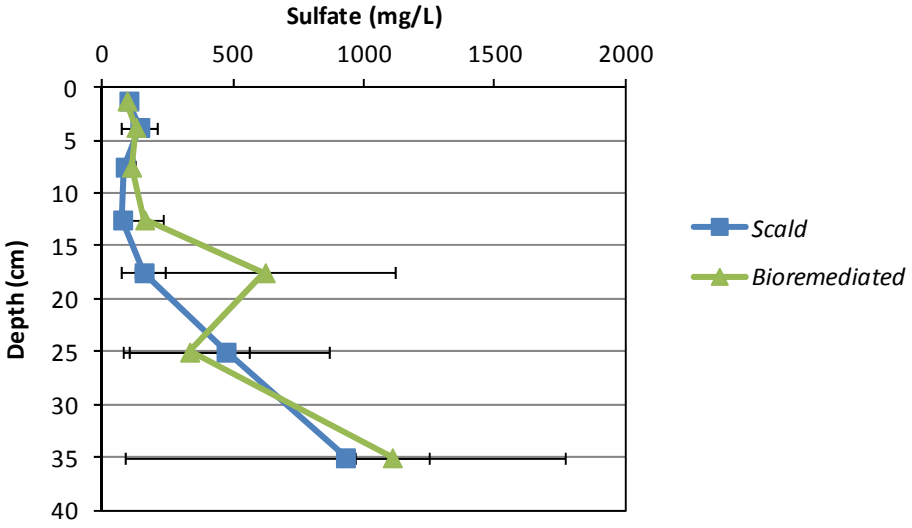

Figure 9-31. Pore-water soluble sulfate characteristics at the Tolderol study area (March 2012).

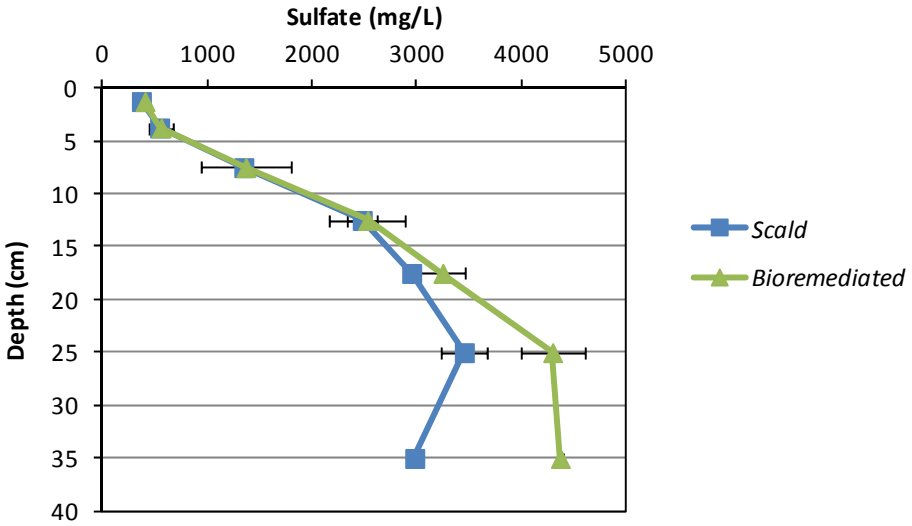

Figure 9-32. Pore-water soluble sulfate characteristics at the Campbell Park study area (March 2012).

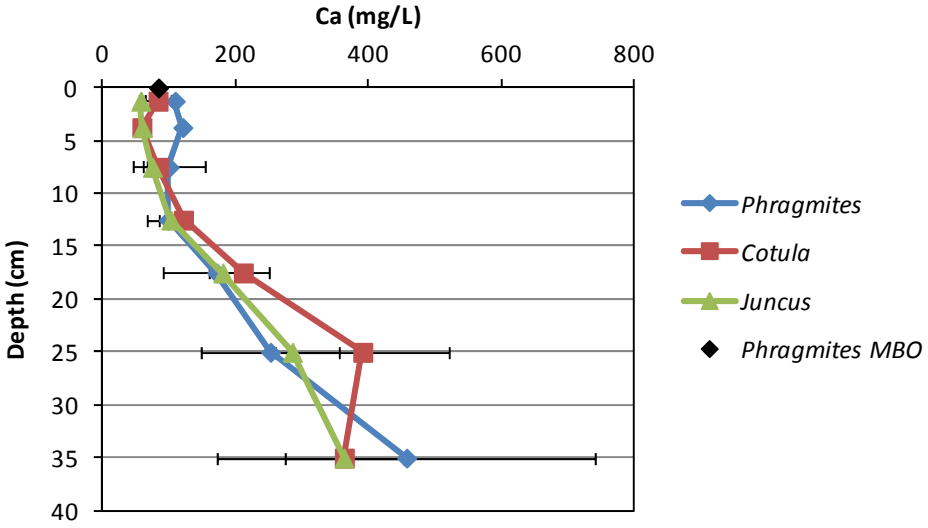

Figure 9-33. Pore-water soluble calcium characteristics at the Waltowa study area (March 2012).

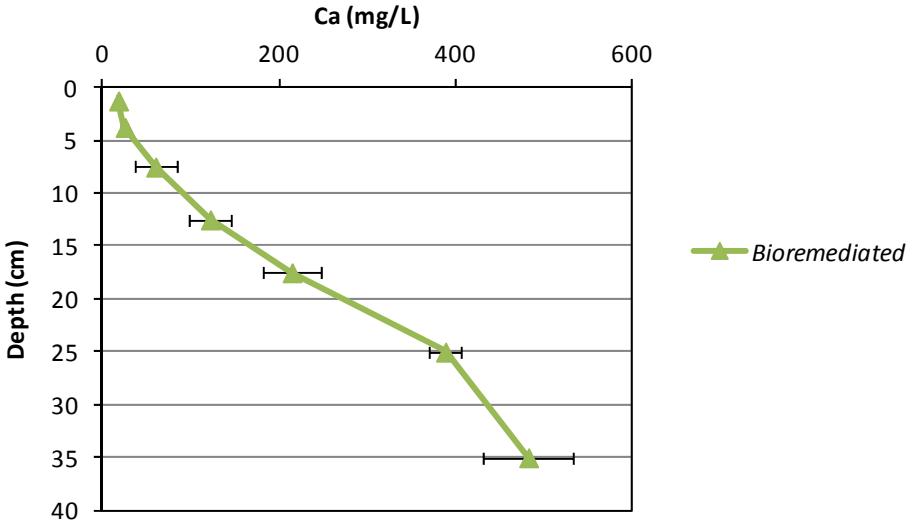

Figure 9-34. Pore-water soluble calcium characteristics at the Poltalloch Bevy rye site (March 2012).

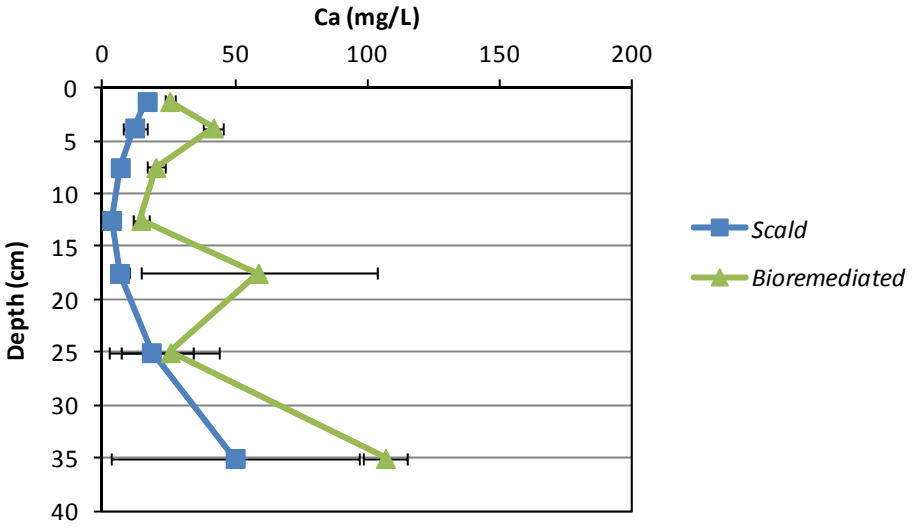

Figure 9-35. Pore-water soluble calcium characteristics at the Tolderol study area (March 2012).

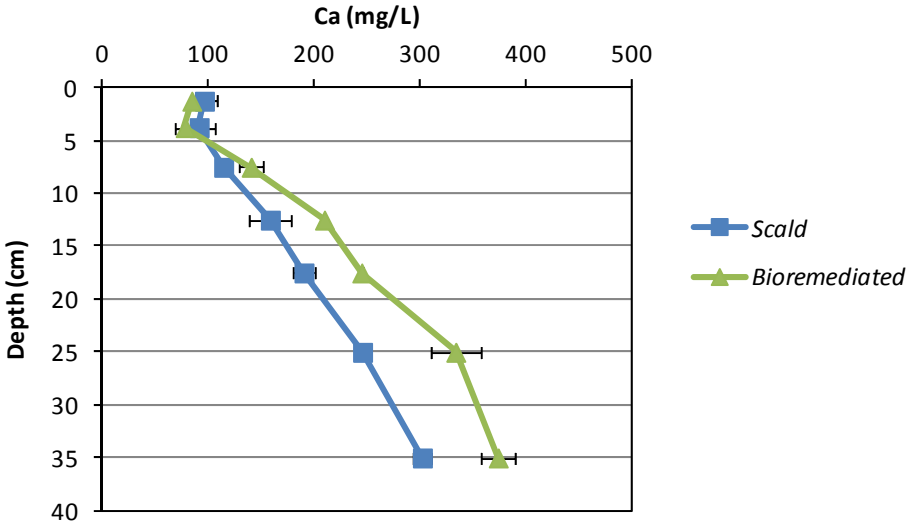

Figure 9-36. Pore-water soluble calcium characteristics at the Campbell Park study area (March 2012).

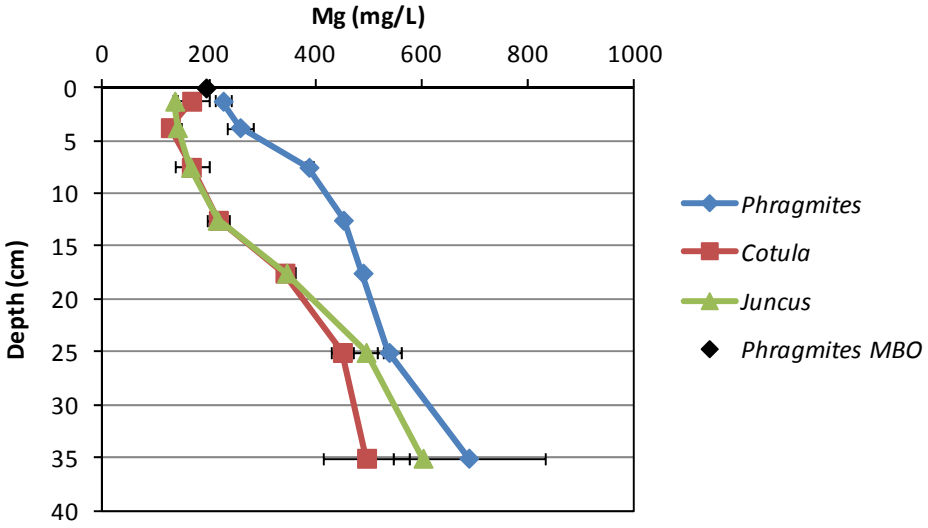

Figure 9-37. Pore-water soluble magnesium characteristics at the Waltowa study area (March 2012).

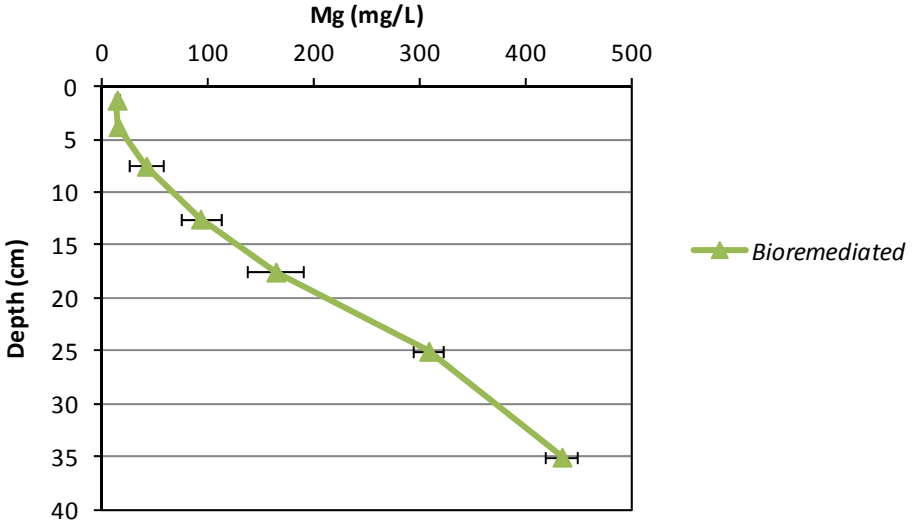

Figure 9-38. Pore-water soluble magnesium characteristics at the Poltalloch Bevy rye site (March 2012).

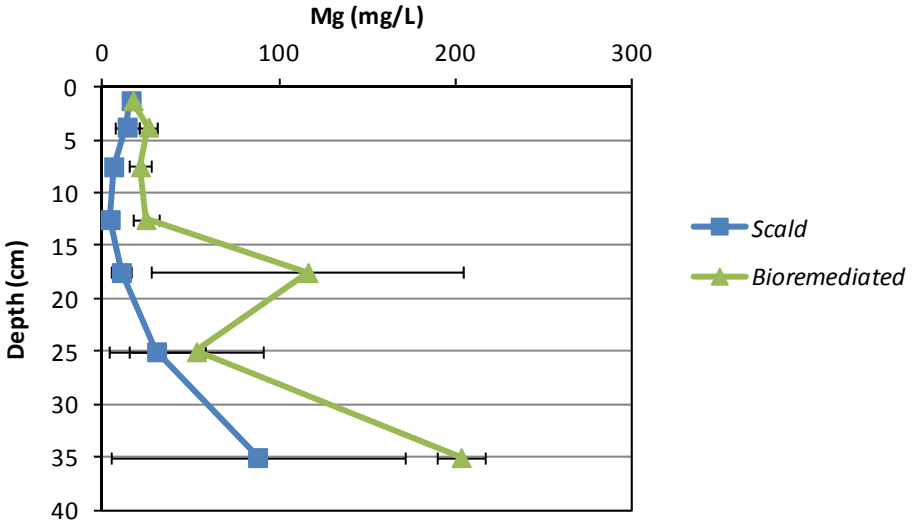

Figure 9-39. Pore-water soluble magnesium characteristics at the Tolderol study area (March 2012).

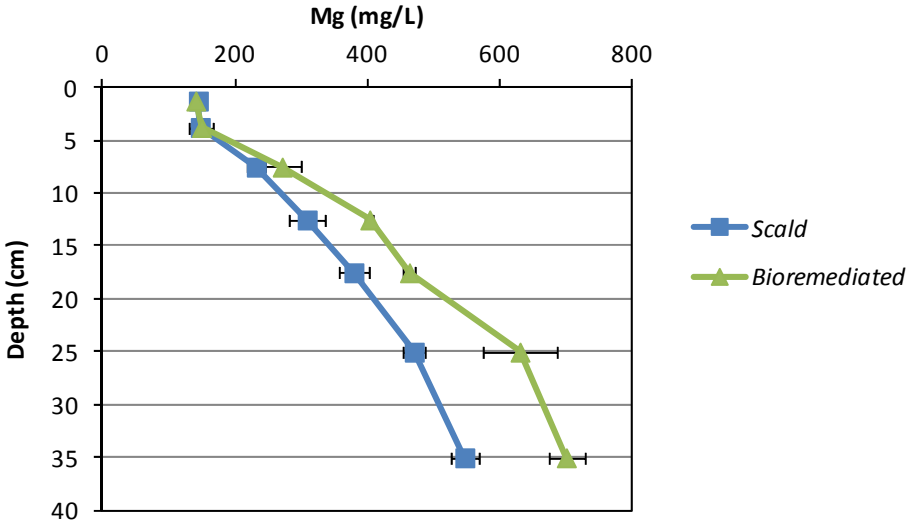

Figure 9-40. Pore-water soluble magnesium characteristics at the Campbell Park study area (March 2012).

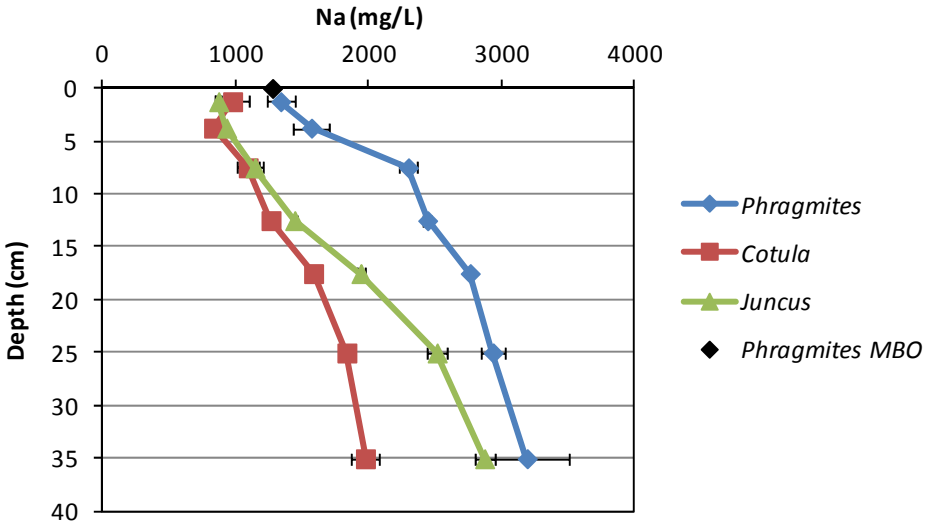

Figure 9-41. Pore-water soluble sodium characteristics at the Waltowa study area (March 2012).

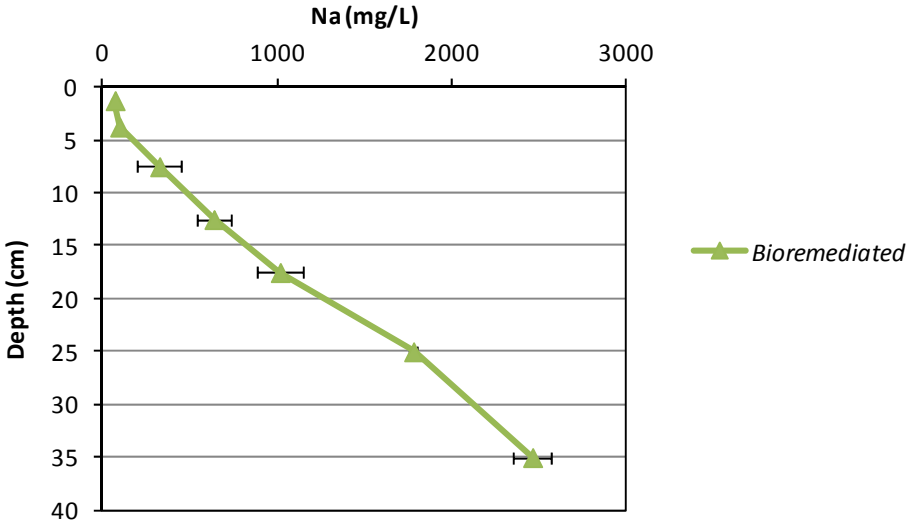

Figure 9-42. Pore-water soluble sodium characteristics at the Poltalloch Bevy rye site (March 2012).

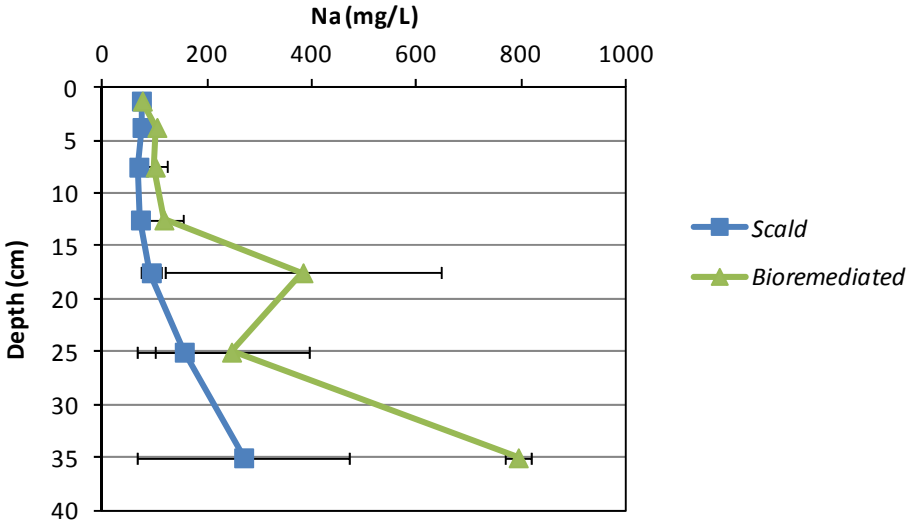

Figure 9-43. Pore-water soluble sodium characteristics at the Tolderol study area (March 2012).

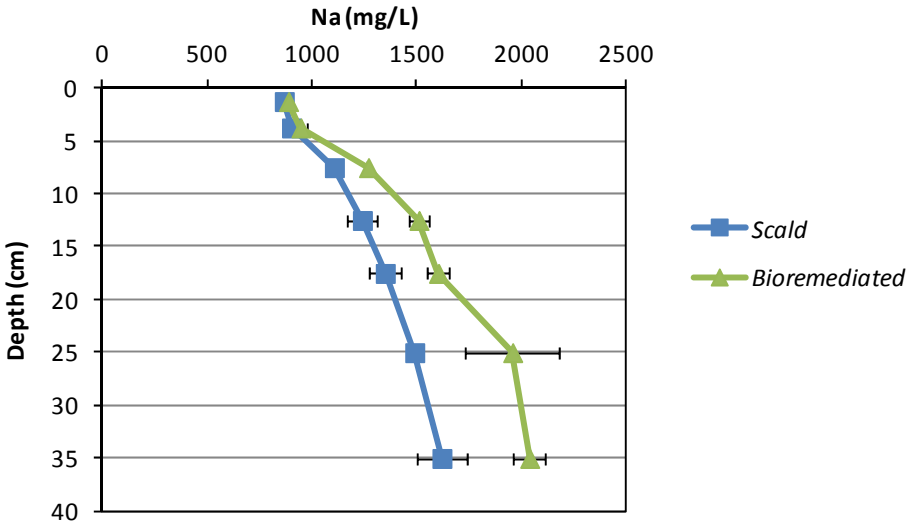

Figure 9-44. Pore-water soluble sodium characteristics at the Campbell Park study area (March 2012).

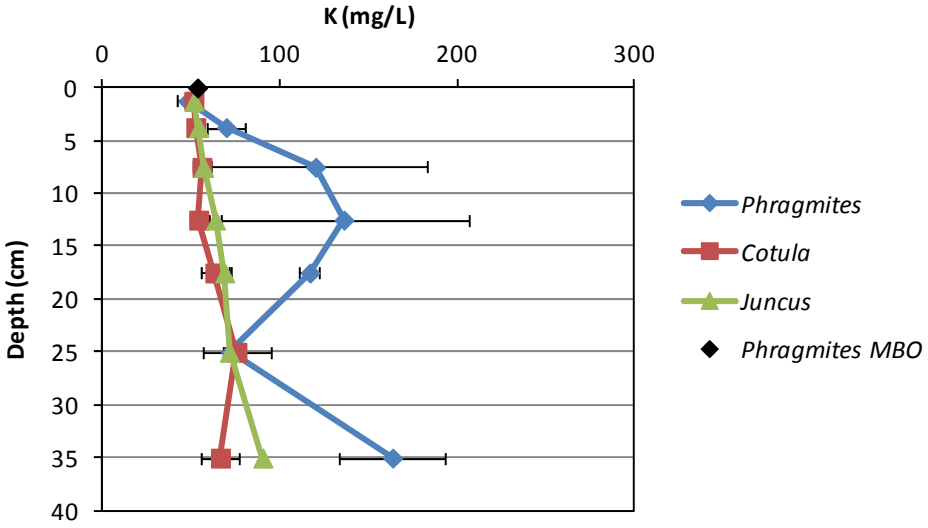

Figure 9-45. Pore-water soluble potassium characteristics at the Waltowa study area (March 2012).

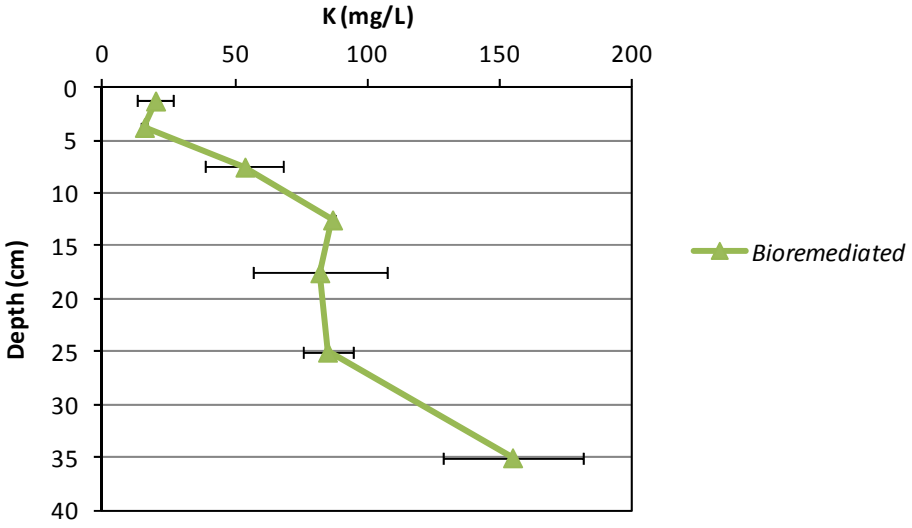

Figure 9-46. Pore-water soluble potassium characteristics at the Poltalloch Bevy rye site (March 2012).

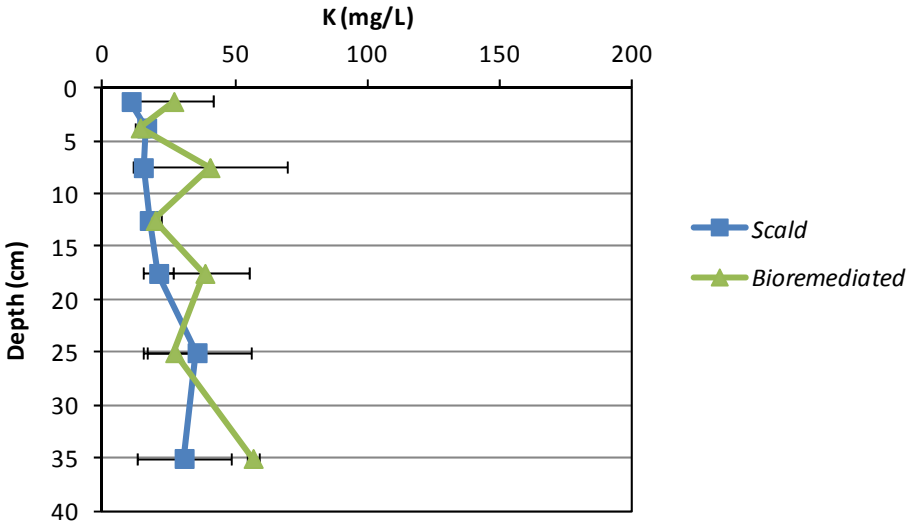

Figure 9-47. Pore-water soluble potassium characteristics at the Tolderol study area (March 2012).

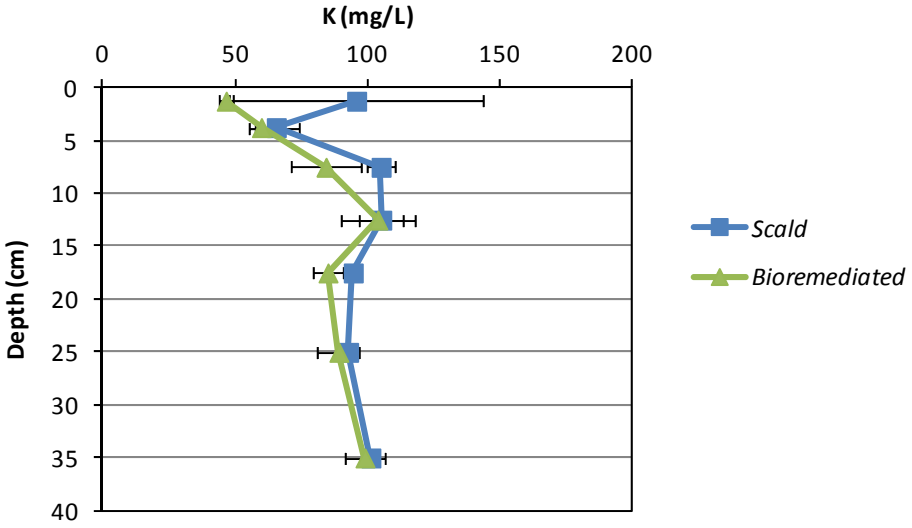

Figure 9-48. Pore-water soluble potassium characteristics at the Campbell Park study area (March 2012).

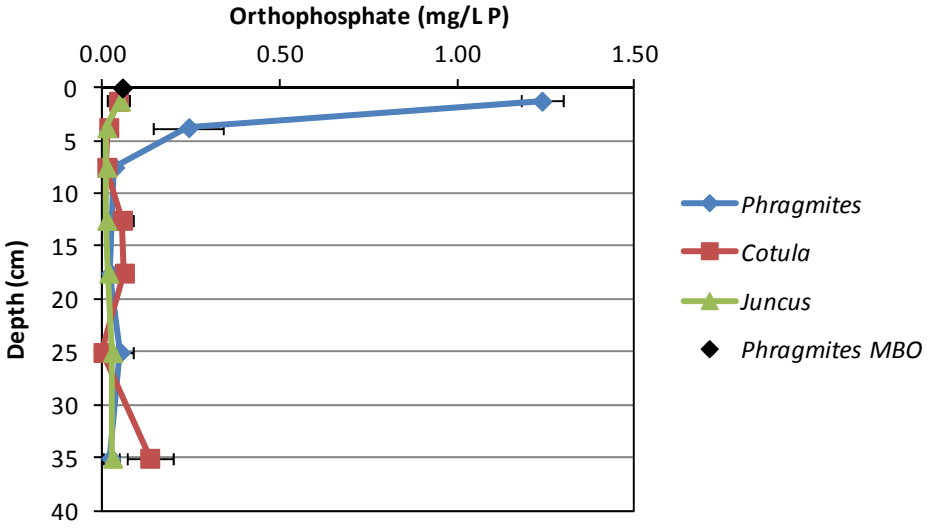

Figure 9-49. Pore-water orthophosphate characteristics at the Waltowa study area (March 2012).

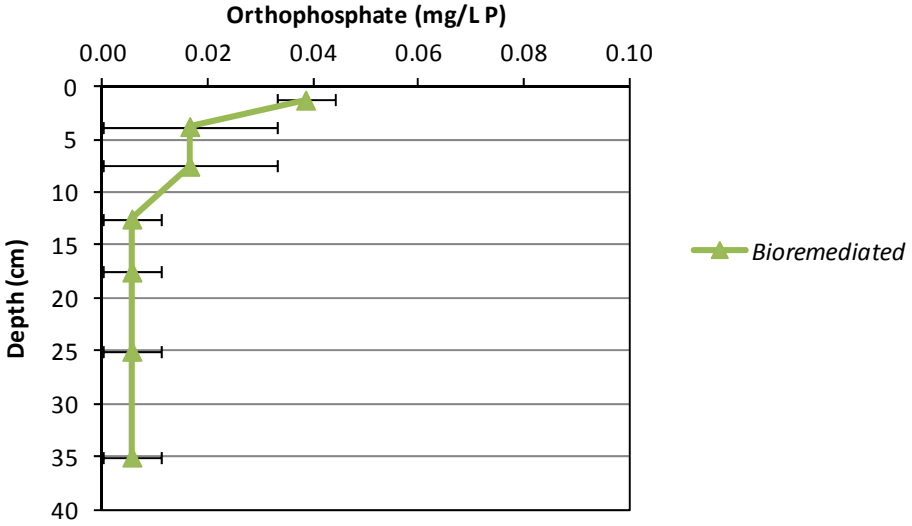

Figure 9-50. Pore-water orthophosphate characteristics at the Poltalloch Bevy rye site (March 2012).

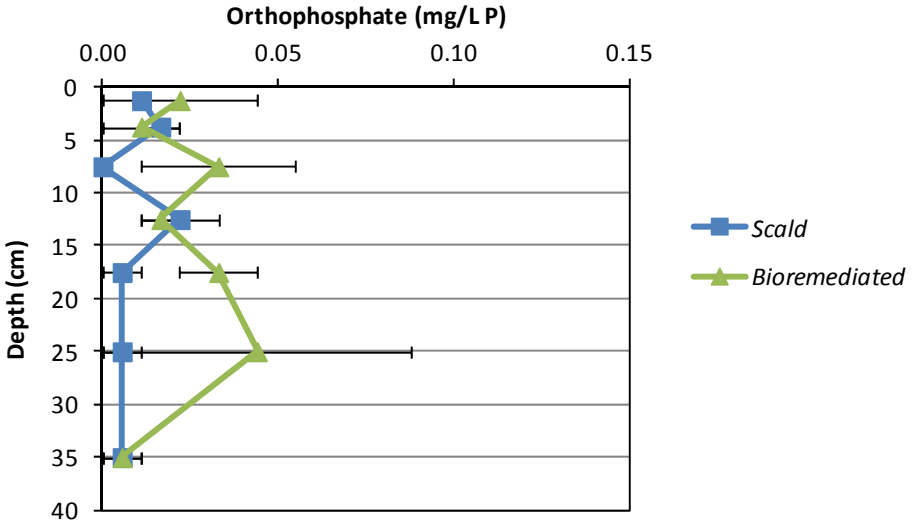

Figure 9-51. Pore-water orthophosphate characteristics at the Tolderol study area (March 2012).

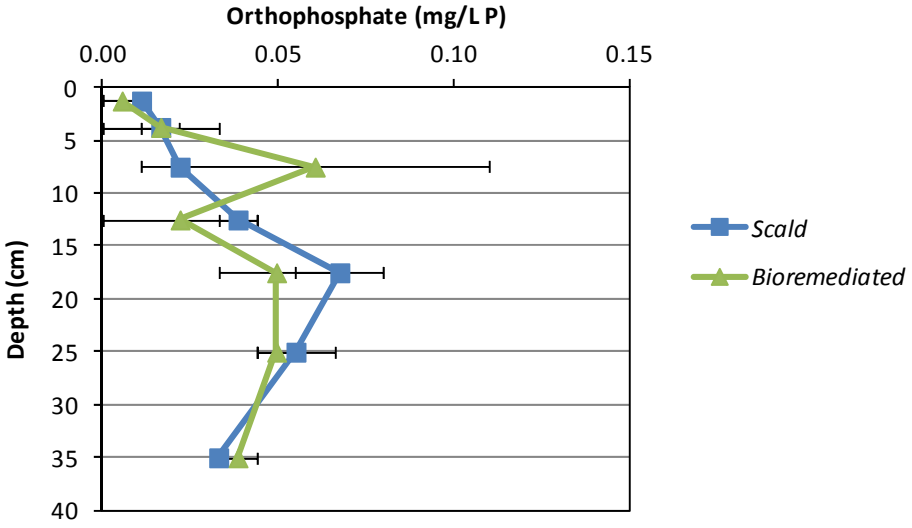

Figure 9-52. Pore-water orthophosphate characteristics at the Campbell Park study area (March 2012).

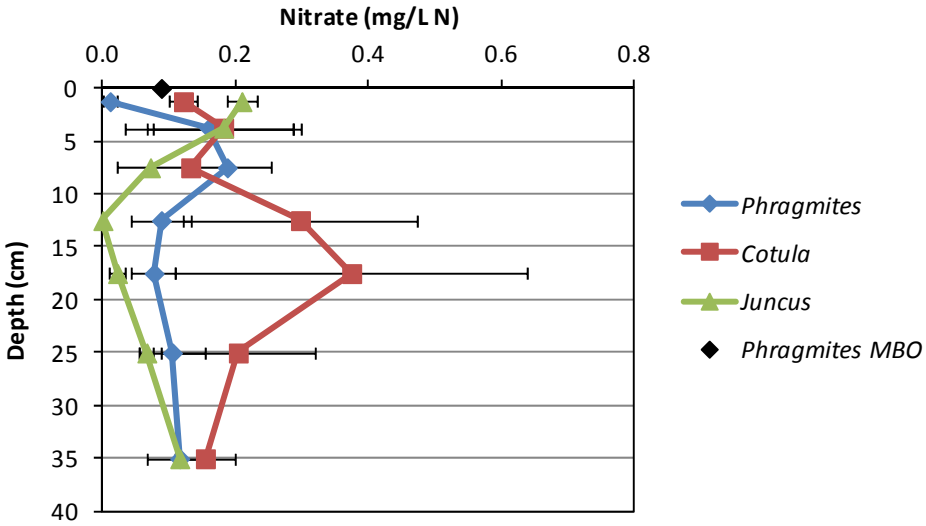

Figure 9-53. Pore-water nitrate characteristics at the Waltowa study area (March 2012).

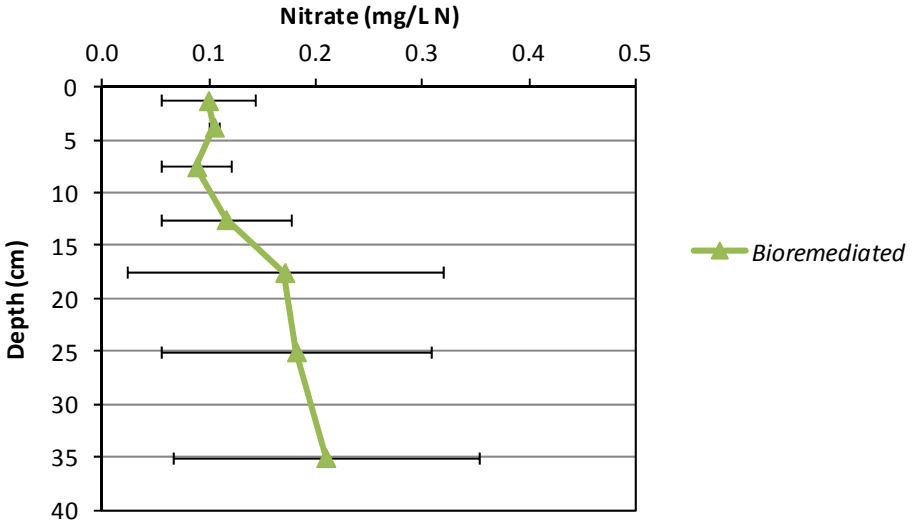

Figure 9-54. Pore-water nitrate characteristics at the Poltalloch Bevy rye site (March 2012).

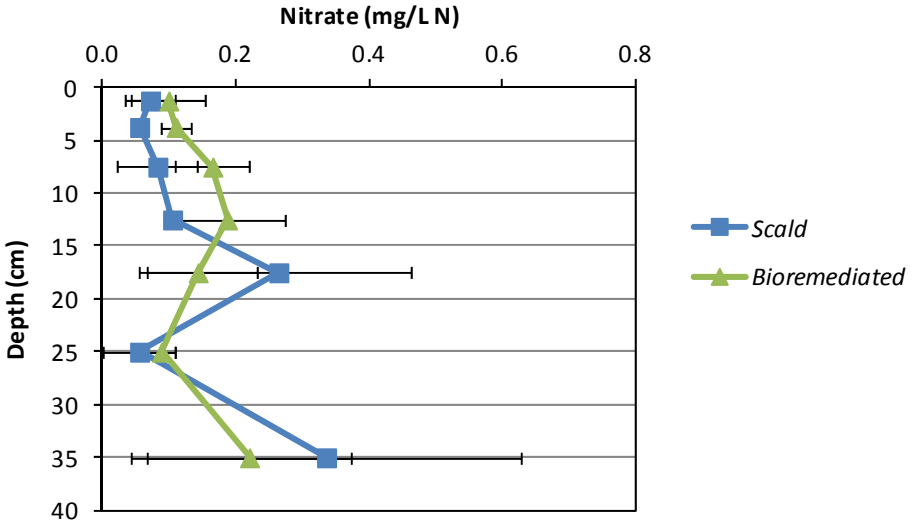

Figure 9-55. Pore-water nitrate characteristics at the Tolderol study area (March 2012).

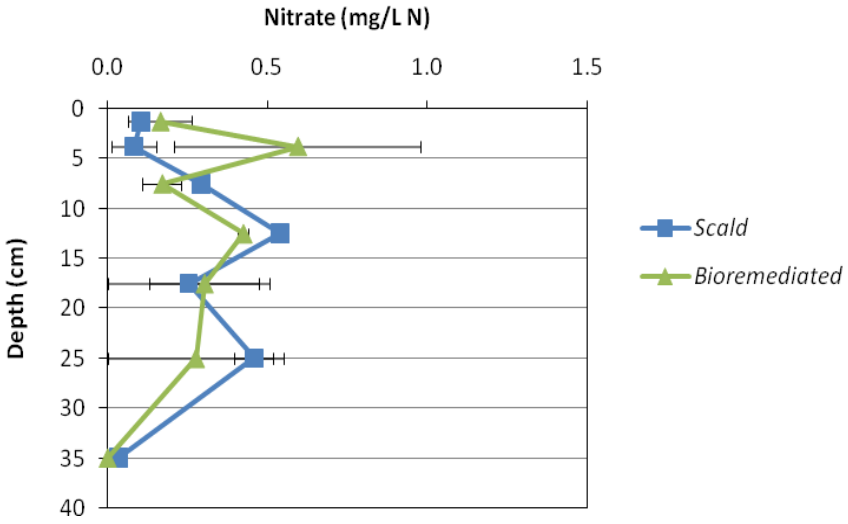

Figure 9-56. Pore-water nitrate characteristics at the Campbell Park study area (March 2012).

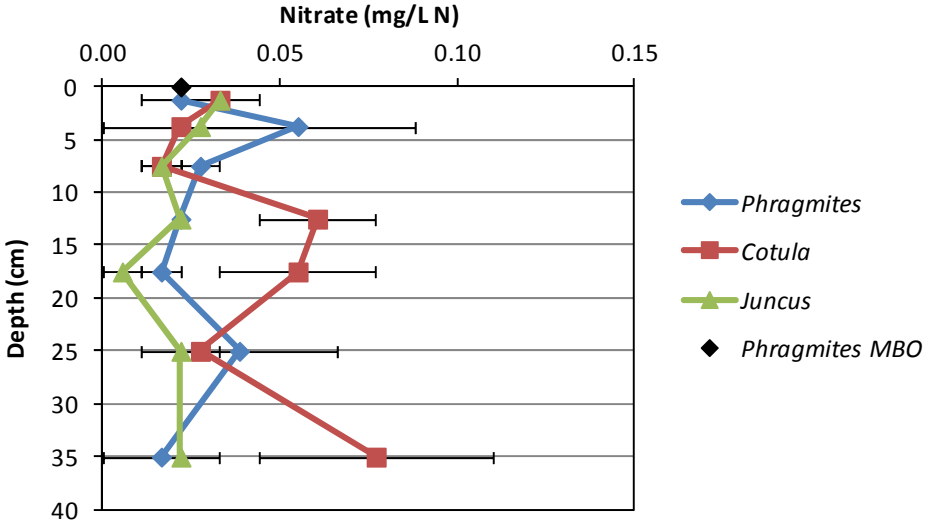

Figure 9-57. Pore-water nitrite characteristics at the Waltowa study area (March 2012).

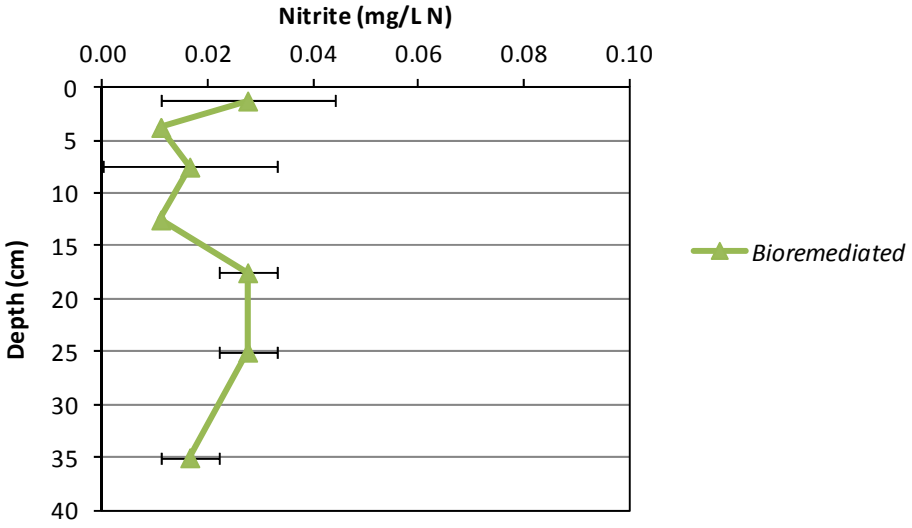

Figure 9-58. Pore-water nitrite characteristics at the Poltalloch Bevy rye site (March 2012).

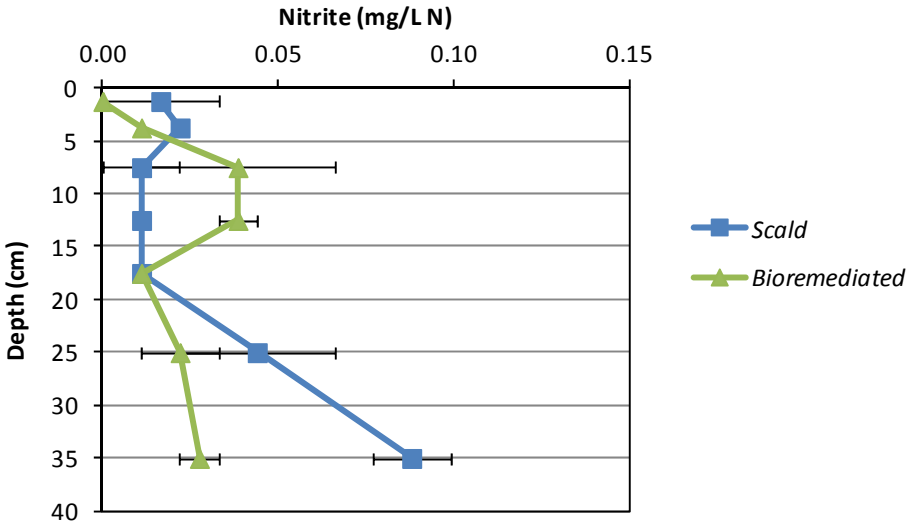

Figure 9-59. Pore-water nitrite characteristics at the Tolderol study area (March 2012).

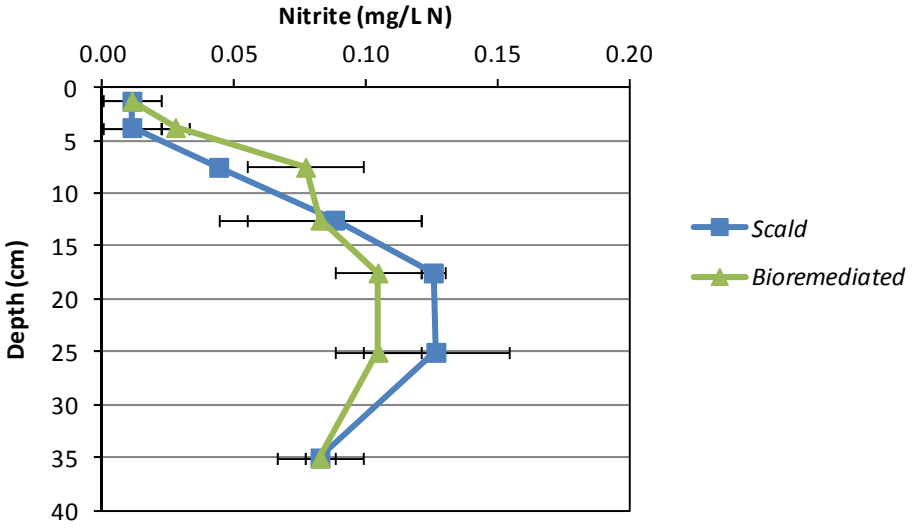

Figure 9-60. Pore-water nitrite characteristics at the Campbell Park study area (March 2012).

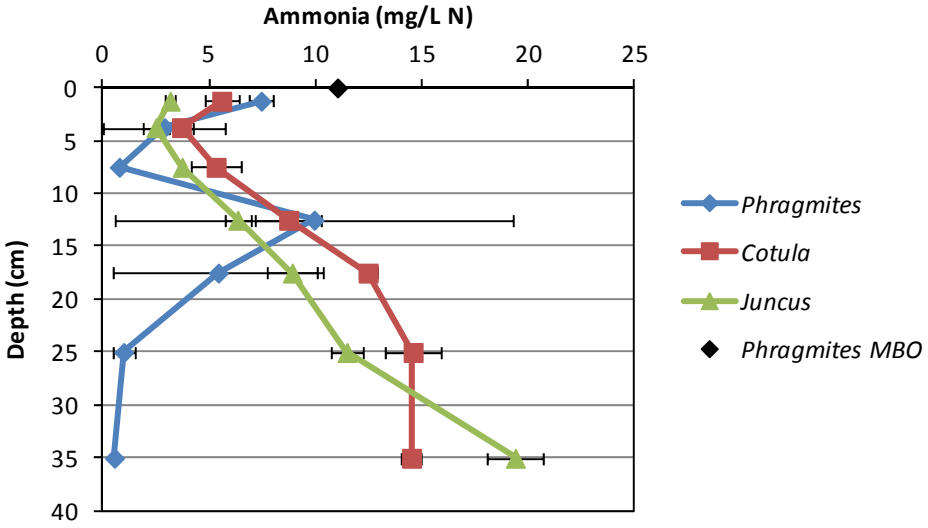

Figure 9-61. Pore-water ammonia characteristics at the Waltowa study area (March 2012).

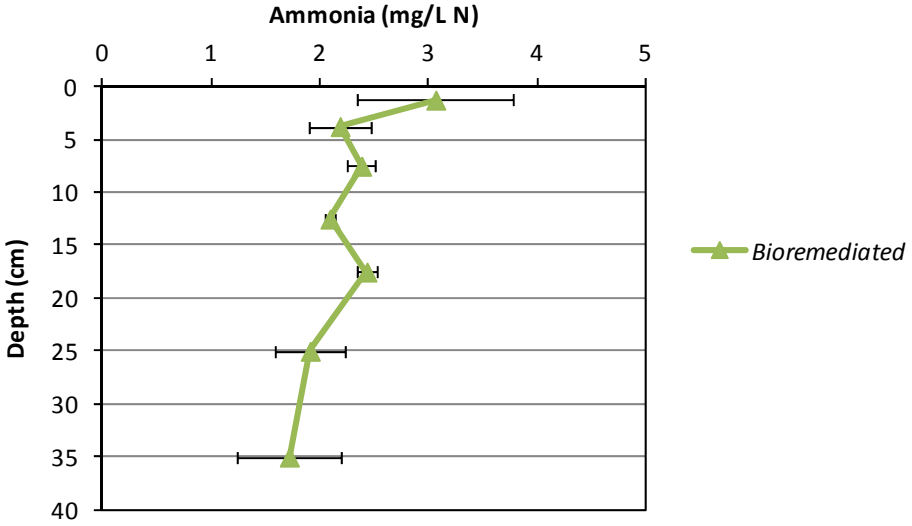

Figure 9-62. Pore-water ammonia characteristics at the Poltalloch Bevy rye site (March 2012).

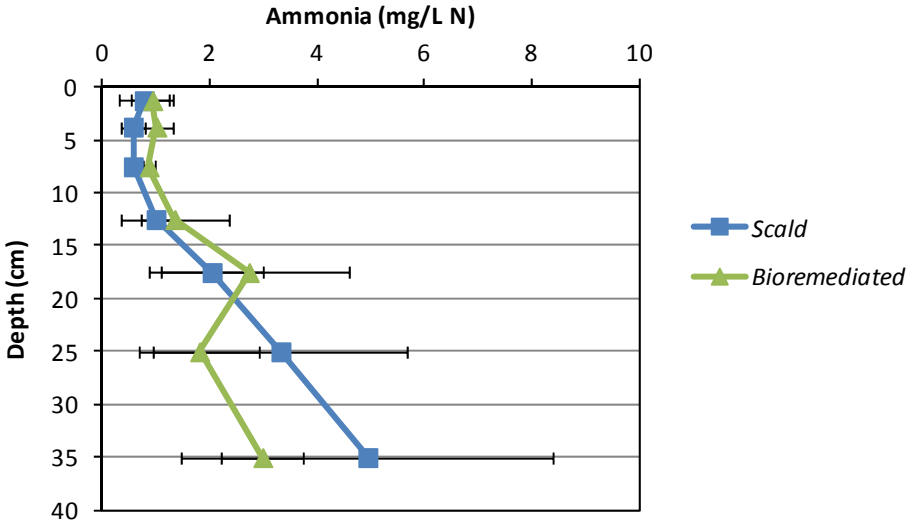

Figure 9-63. Pore-water ammonia characteristics at the Tolderol study area (March 2012).

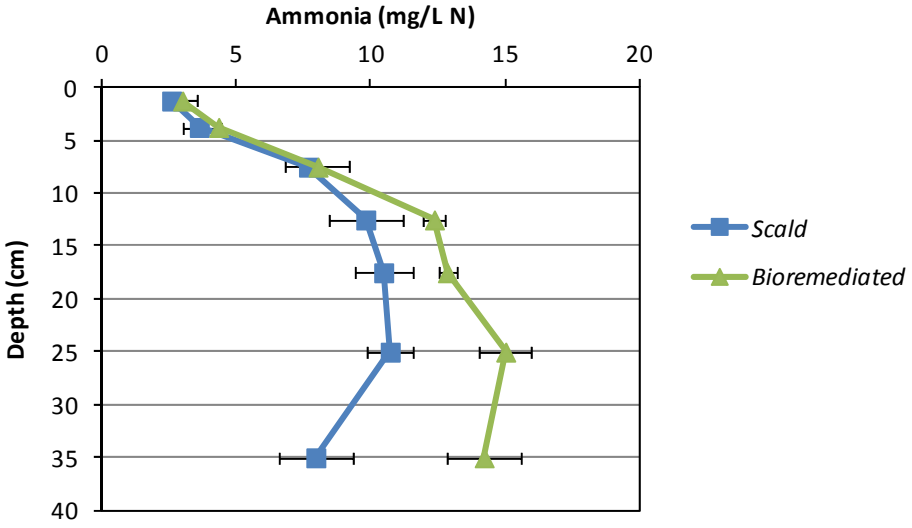

Figure 9-64. Pore-water ammonia characteristics at the Campbell Park study area (March 2012).

APPENDIX 6. HCl extractable metal plots

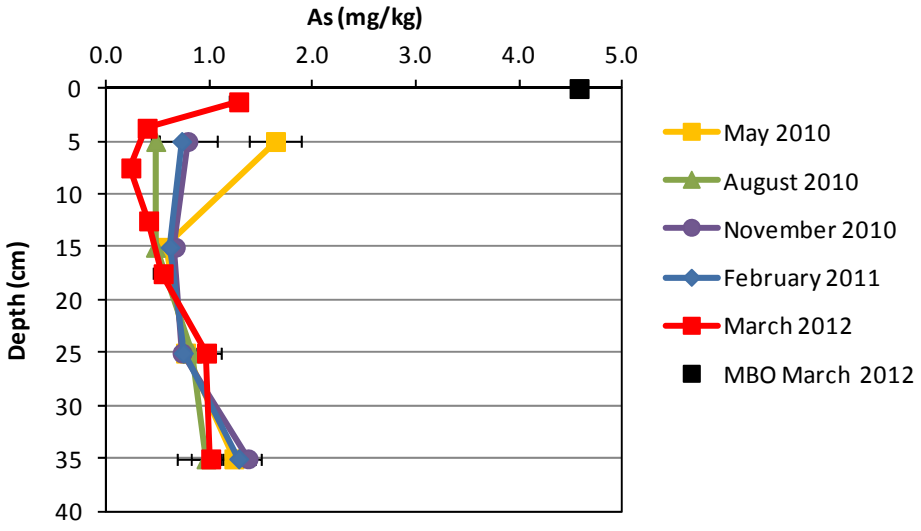

Figure 9-65. Waltowa HCl extractable arsenic dynamics at the established *Phragmites* site (May 2010 – March 2012).

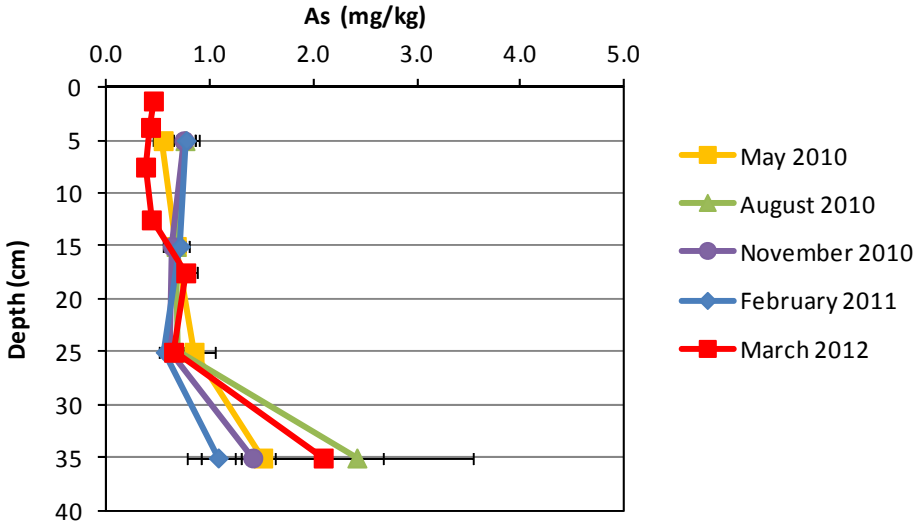

Figure 9-66. Waltowa HCl extractable arsenic dynamics at the established *Cotula* site (May 2010 – March 2012).

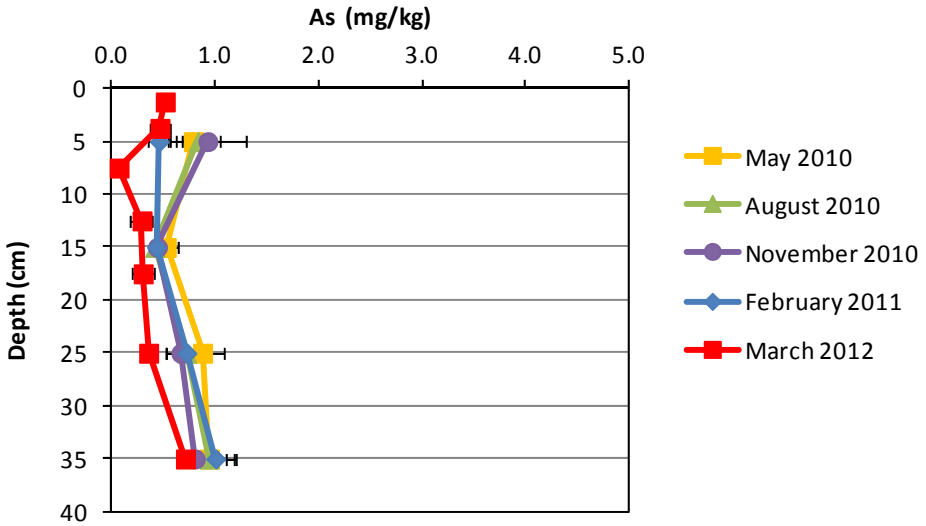

Figure 9-67. Waltowa HCl extractable arsenic dynamics at the established *Juncus* site (May 2010 – March 2012).

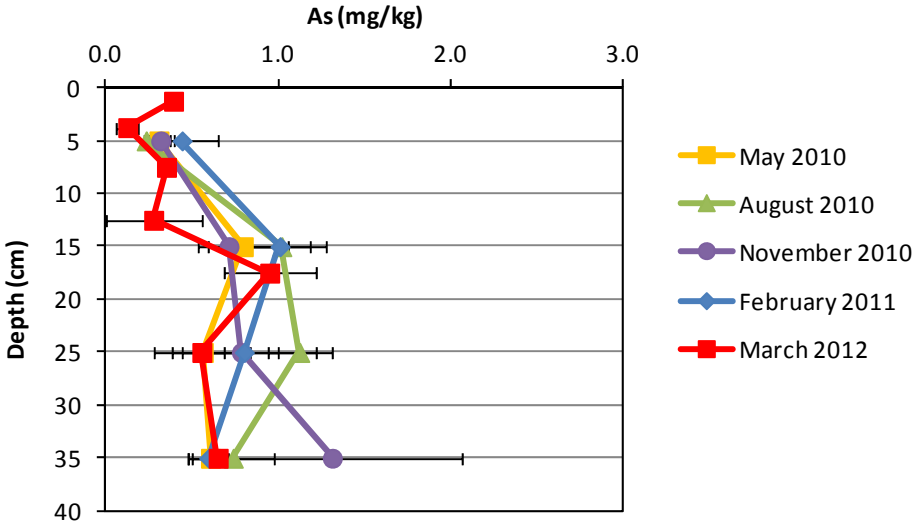

Figure 9-68. Poltalloch HCl extractable arsenic dynamics at the *Juncus* plantings in Bevy rye site (May 2010 – March 2012).

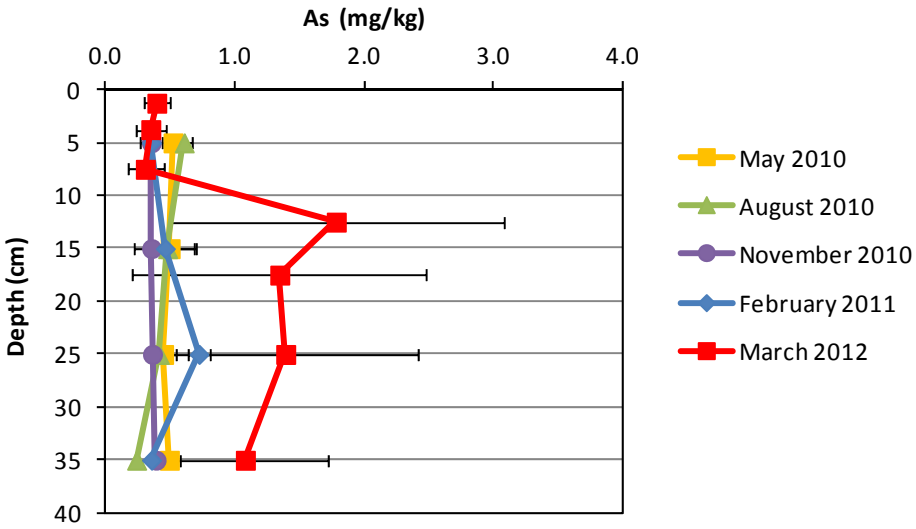

Figure 9-69. Tolderol HCl extractable arsenic dynamics at the control site (May 2010 – March 2012).

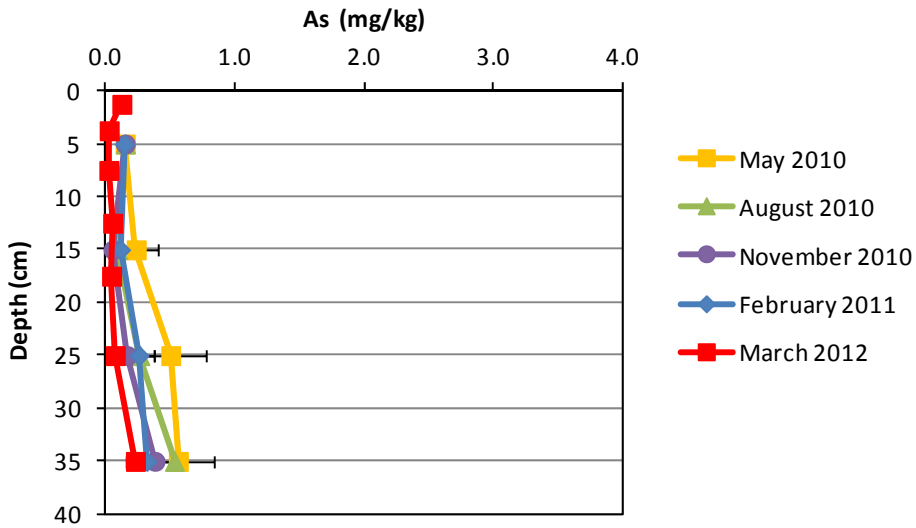

Figure 9-70. Tolderol HCl extractable arsenic dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

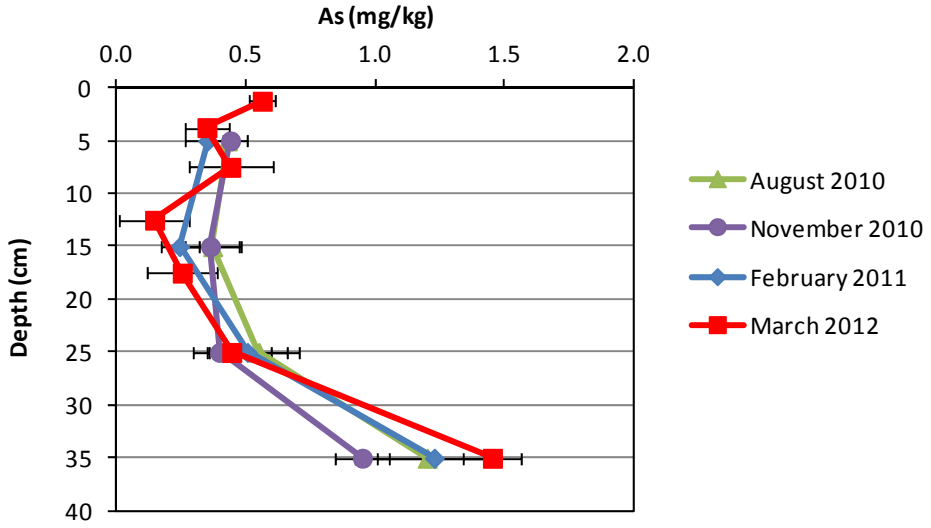

Figure 9-71. Campbell Park HCl extractable arsenic dynamics at the control site (August 2010 – March 2012).

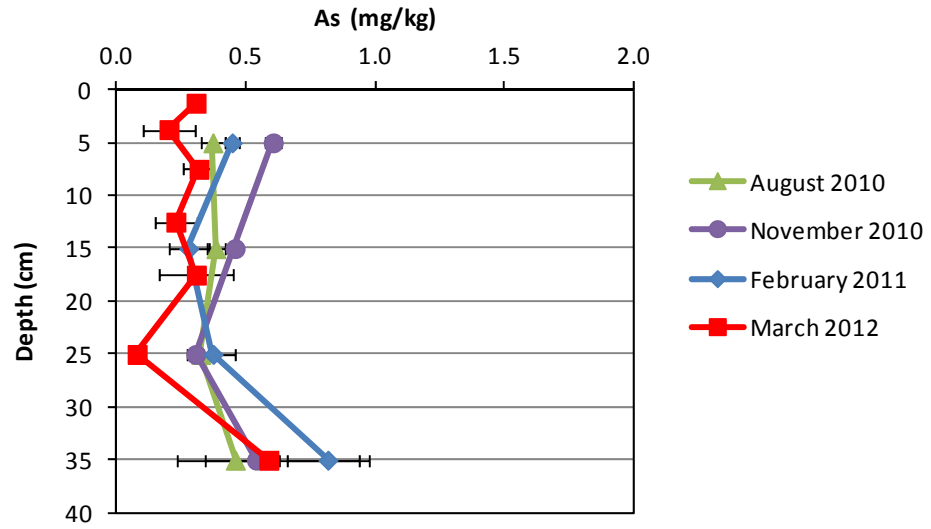

Figure 9-72. Campbell Park HCl extractable arsenic dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

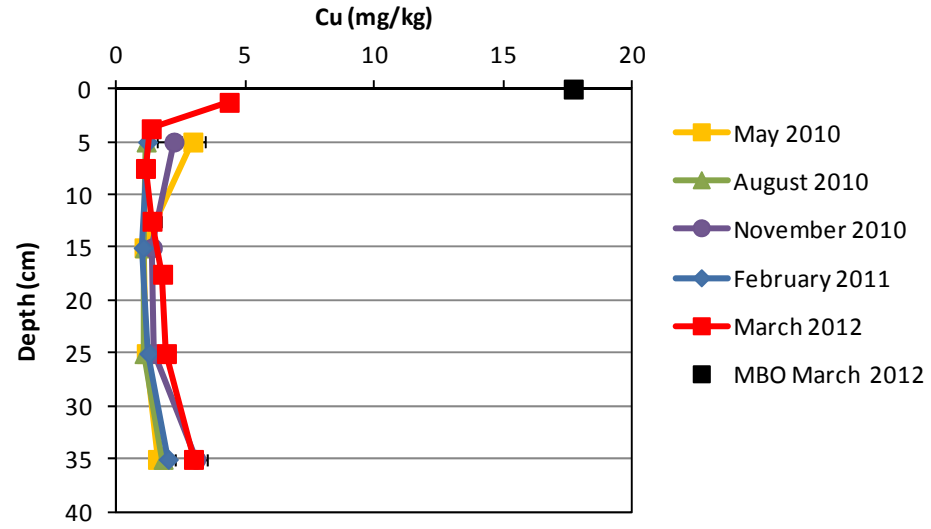

Figure 9-73. Waltowa HCl extractable copper dynamics at the established *Phragmites* site (May 2010 – March 2012).

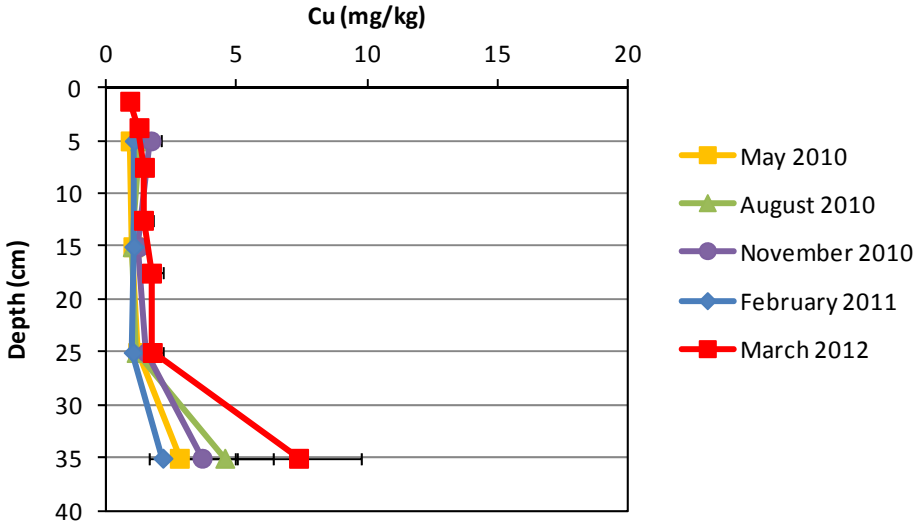

Figure 9-74. Waltowa HCl extractable copper dynamics at the established *Cotula* site (May 2010 – March 2012).

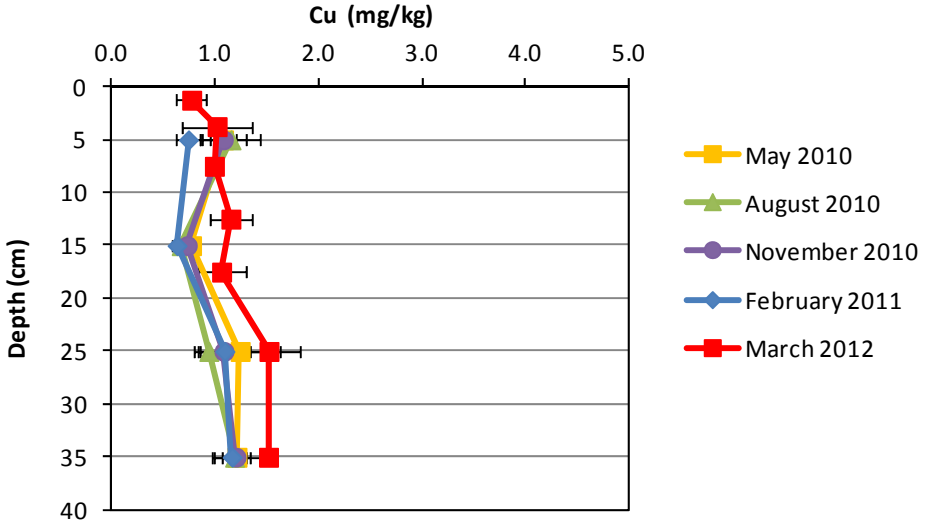

Figure 9-75. Waltowa HCl extractable copper dynamics at the established *Juncus* site (May 2010 – March 2012).

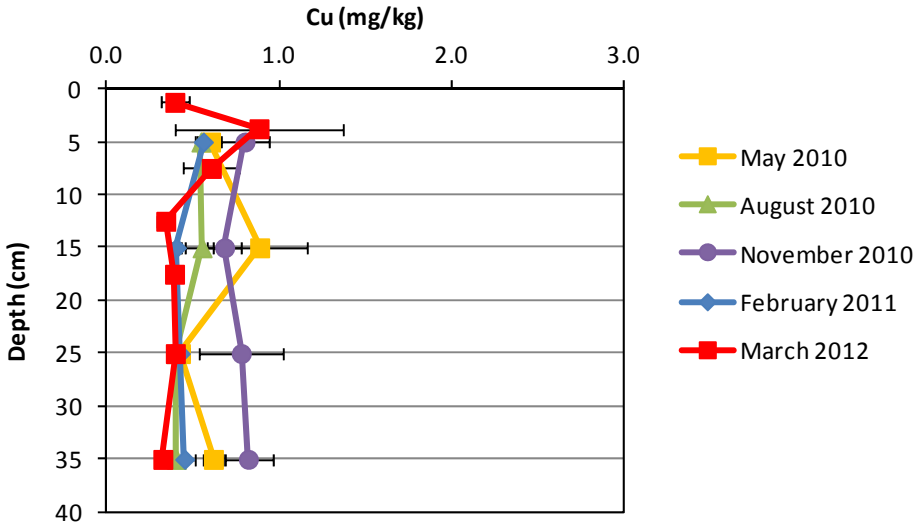

Figure 9-76. Poltalloch HCl extractable copper dynamics at the *Juncus* plantings in Bevy rye site (May 2010 – March 2012).

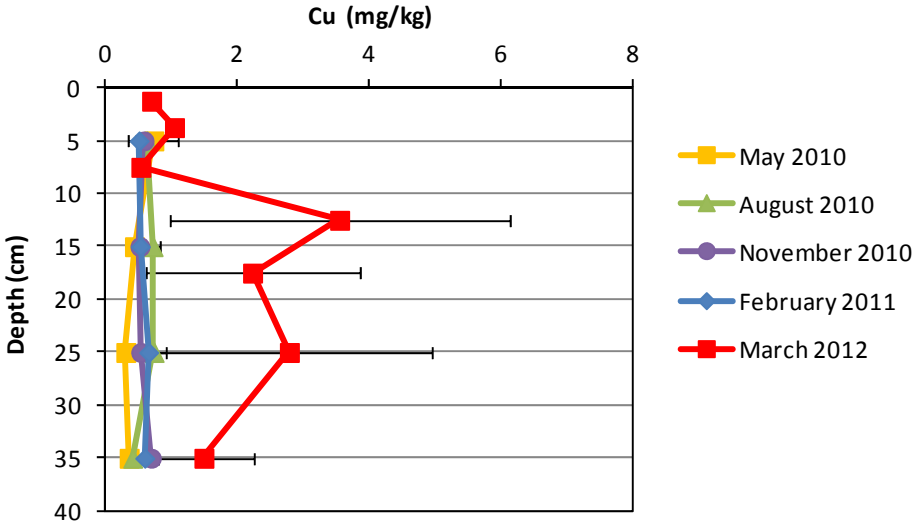

Figure 9-77. Tolderol HCl extractable copper dynamics at the control site (May 2010 – March 2012).

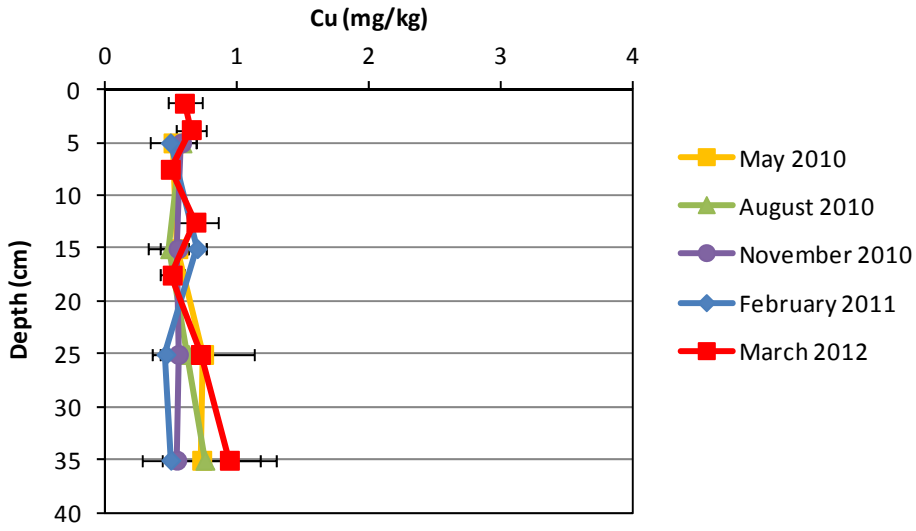

Figure 9-78. Tolderol HCl extractable copper dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

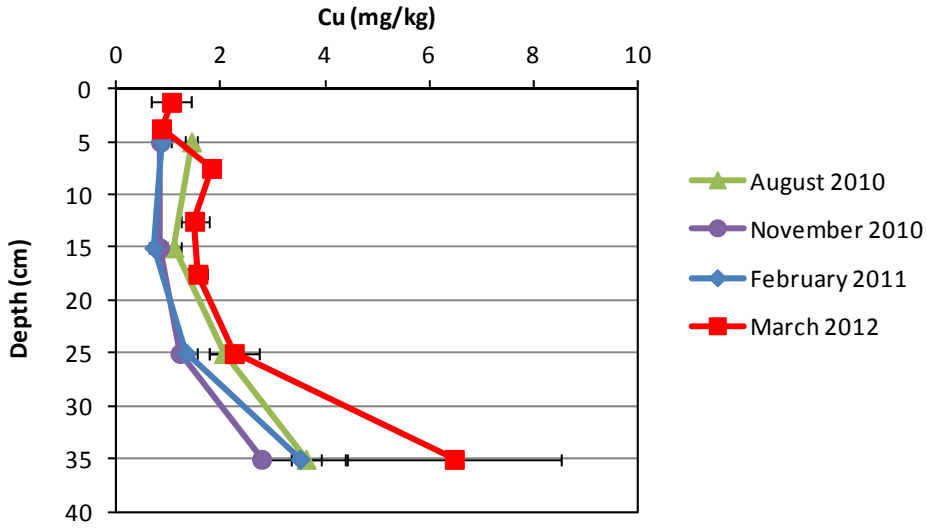

Figure 9-79. Campbell Park HCl extractable copper dynamics at the control site (August 2010 – March 2012).

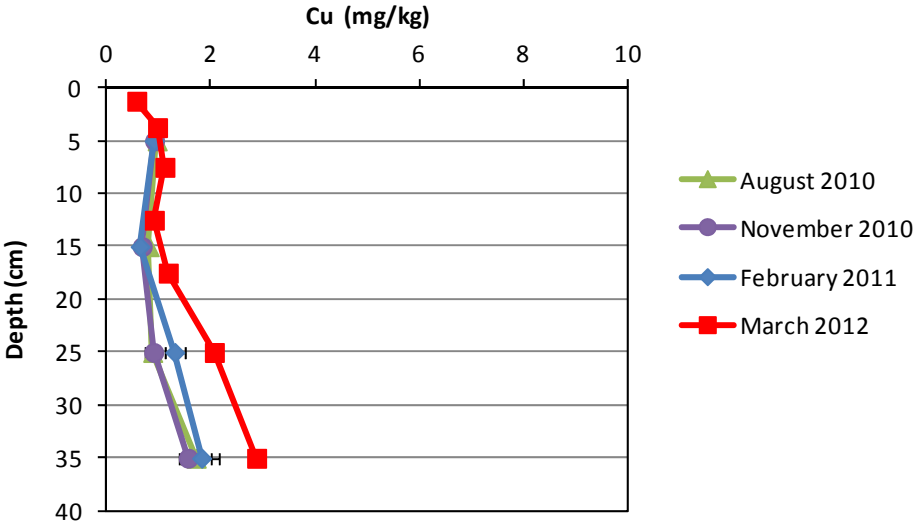

Figure 9-80. Campbell Park HCl extractable copper dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

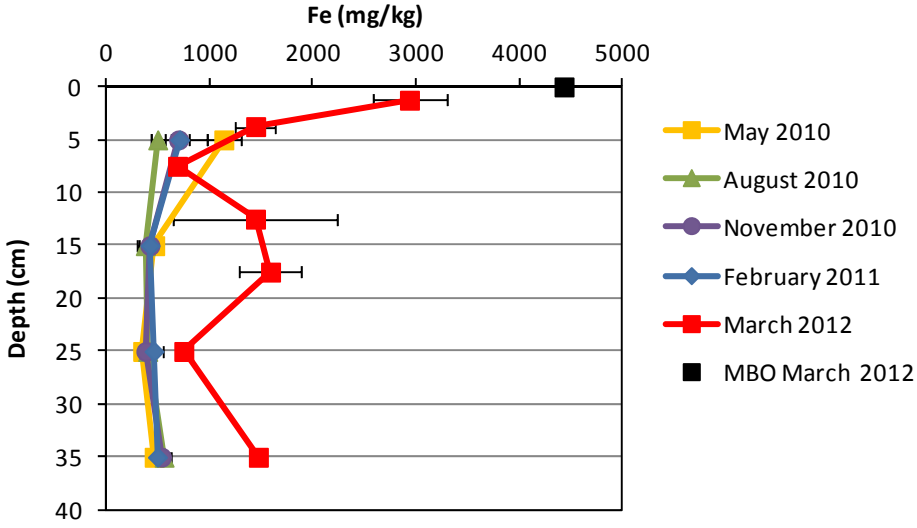

Figure 9-81. Waltowa HCl extractable iron dynamics at the established *Phragmites* site (May 2010 – March 2012).

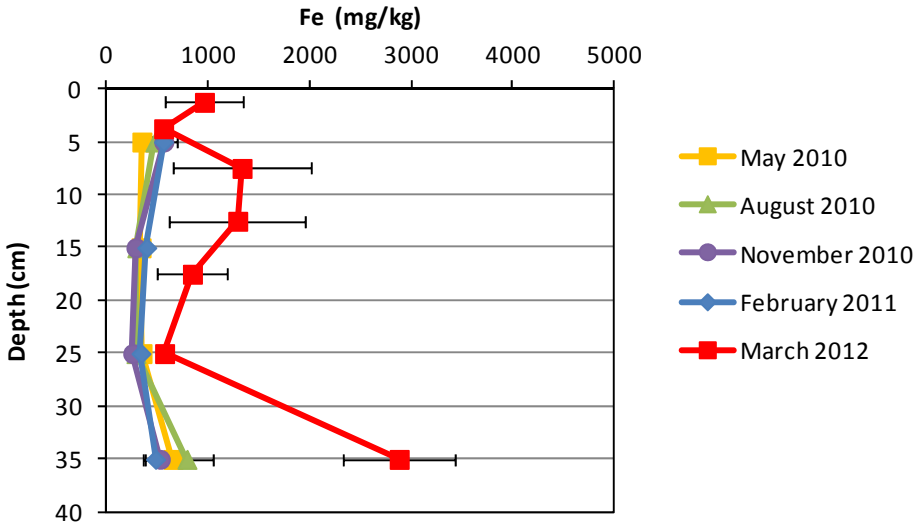

Figure 9-82. Waltowa HCl extractable iron dynamics at the established *Cotula* site (May 2010 – March 2012).

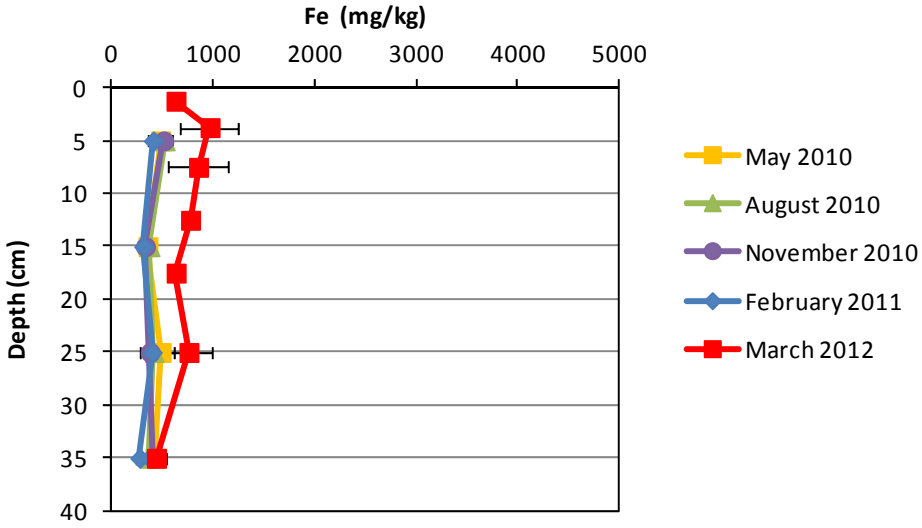

Figure 9-83. Waltowa HCl extractable iron dynamics at the established *Juncus* site (May 2010 – March 2012).

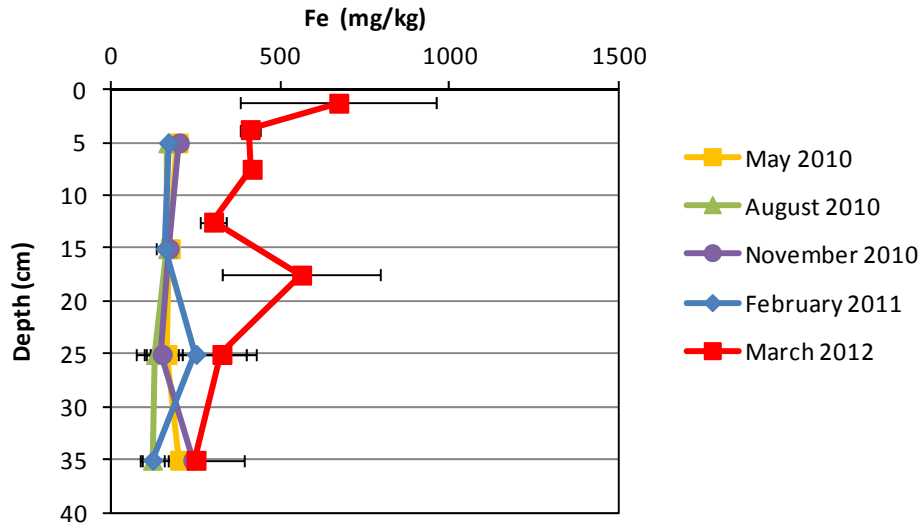

Figure 9-84. Poltalloch HCl extractable iron dynamics at the *Juncus* plantings in Bevy rye site (May 2010 – March 2012).

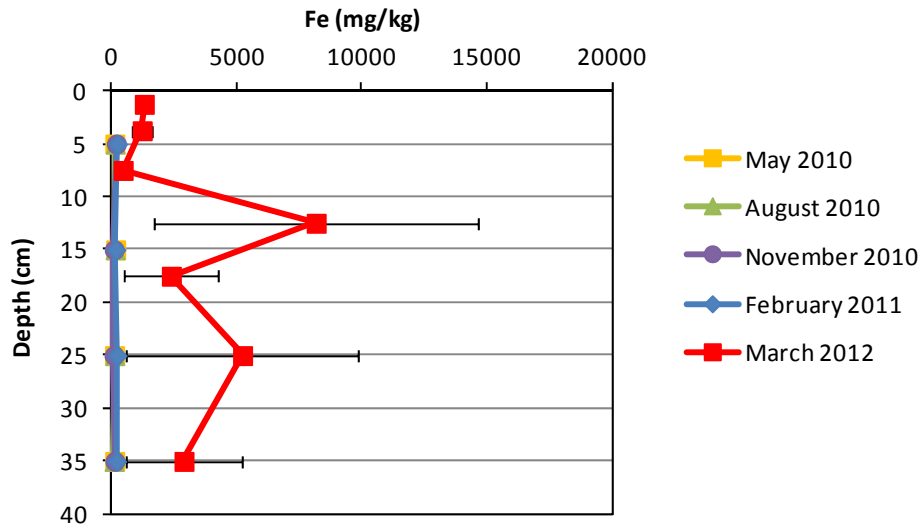

Figure 9-85. Tolderol HCl extractable iron dynamics at the control site (May 2010 – March 2012).

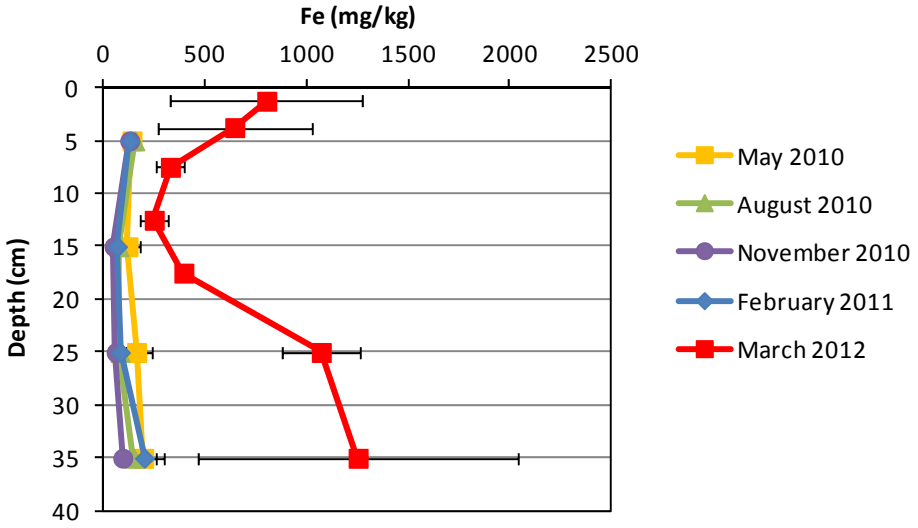

Figure 9-86. Tolderol HCl extractable iron dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

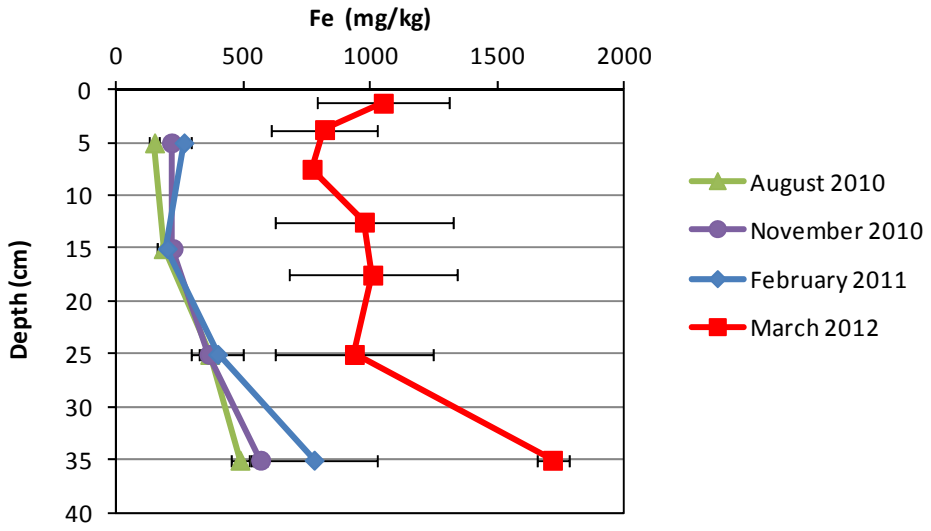

Figure 9-87. Campbell Park HCl extractable iron dynamics at the control site (August 2010 – March 2012).

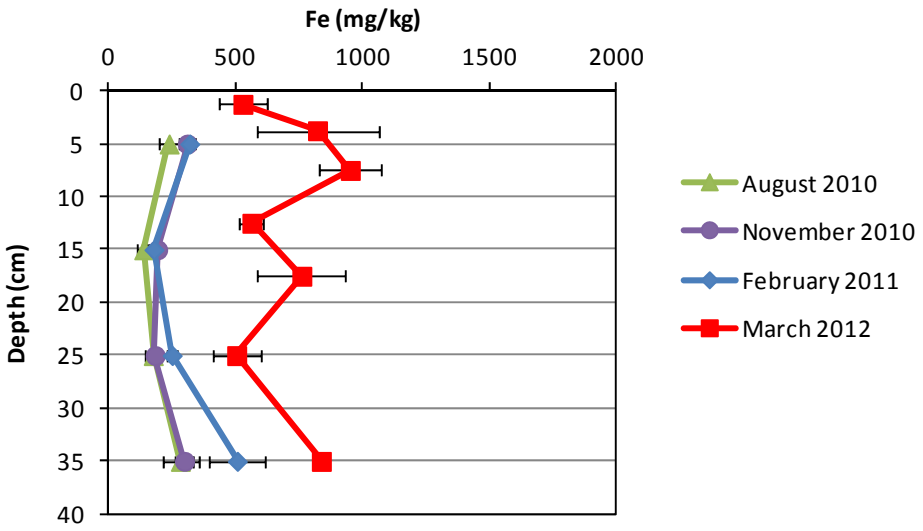

Figure 9-88. Campbell Park HCl extractable iron dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

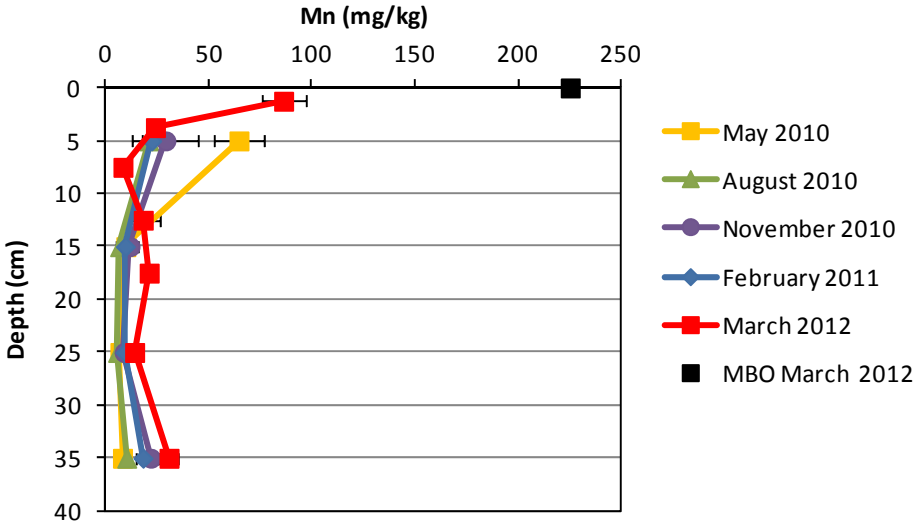

Figure 9-89. Waltowa HCl extractable manganese dynamics at the established *Phragmites* site (May 2010 – March 2012).

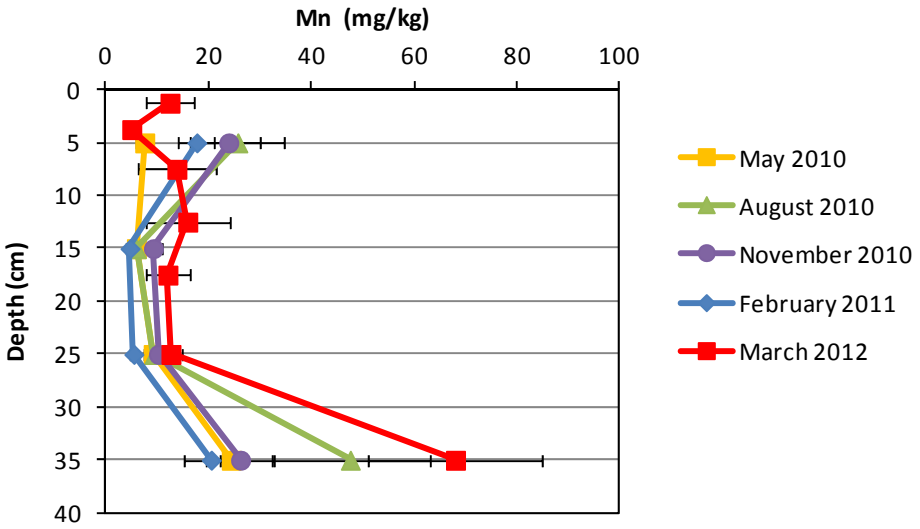

Figure 9-90. Waltowa HCl extractable manganese dynamics at the established *Cotula* site (May 2010 – March 2012).

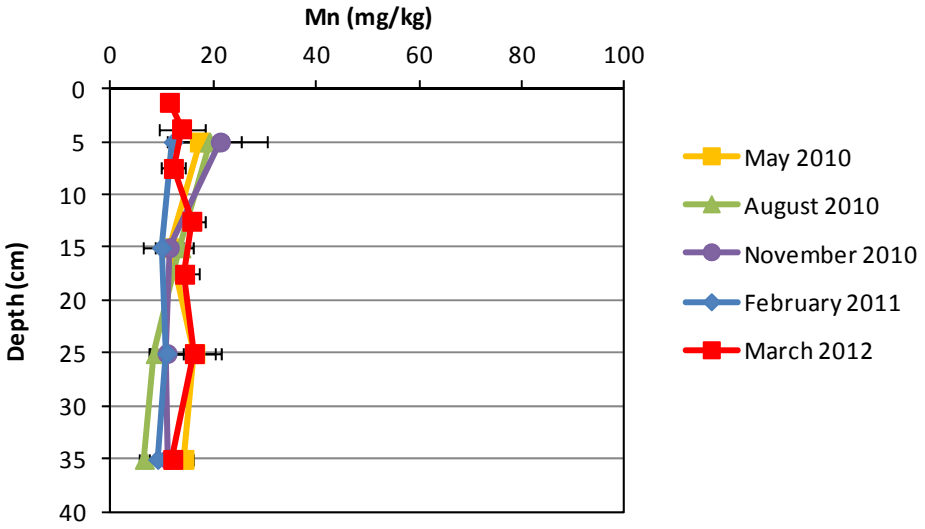

Figure 9-91. Waltowa HCl extractable manganese dynamics at the established *Juncus* site (May 2010 – March 2012).

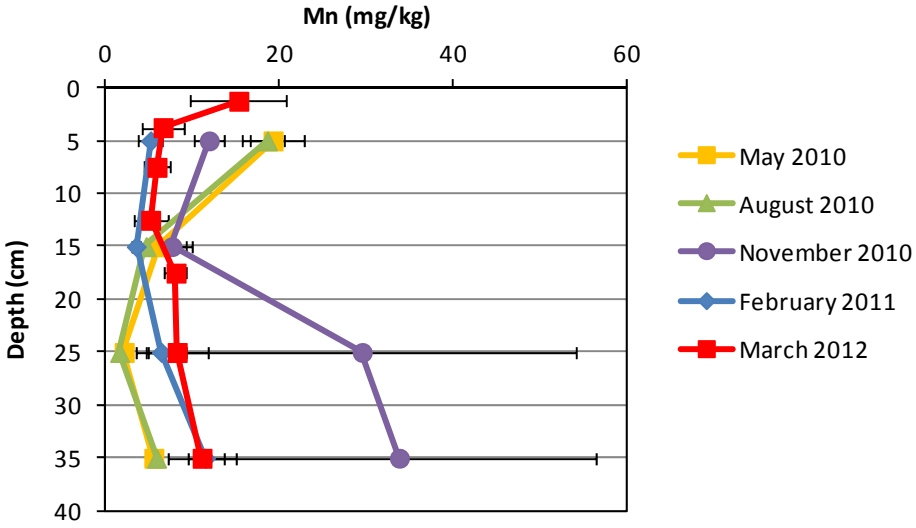

Figure 9-92. Poltalloch HCl extractable manganese dynamics at the *Juncus* plantings in Bevy rye site (May 2010 – March 2012).

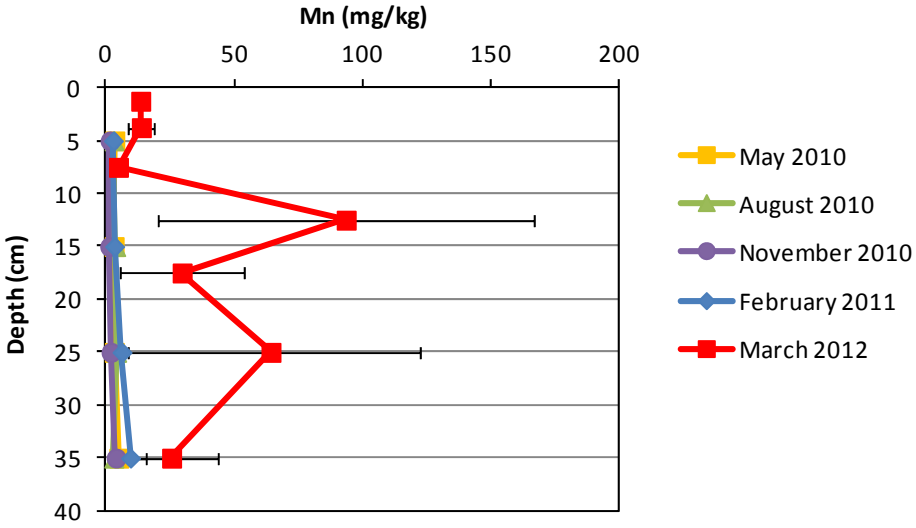

Figure 9-93. Tolderol HCl extractable manganese dynamics at the control site (May 2010 – March 2012).

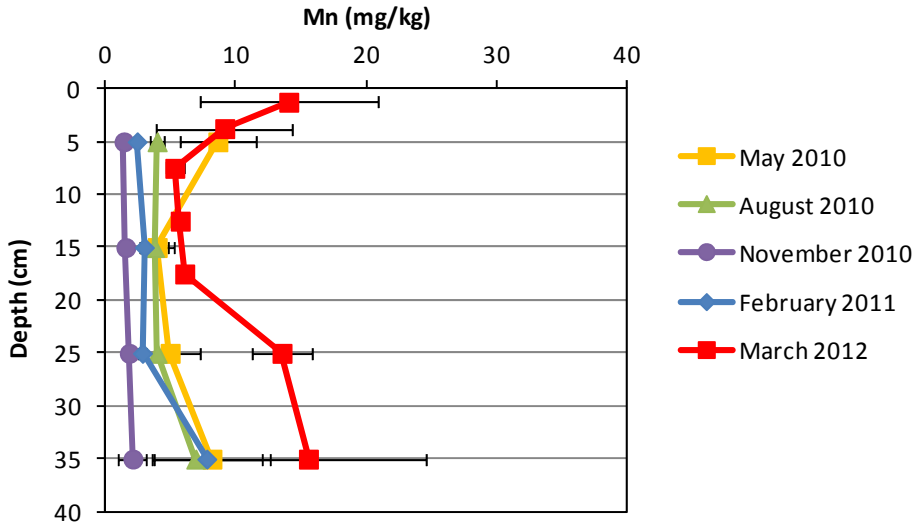

Figure 9-94. Tolderol HCl extractable manganese dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

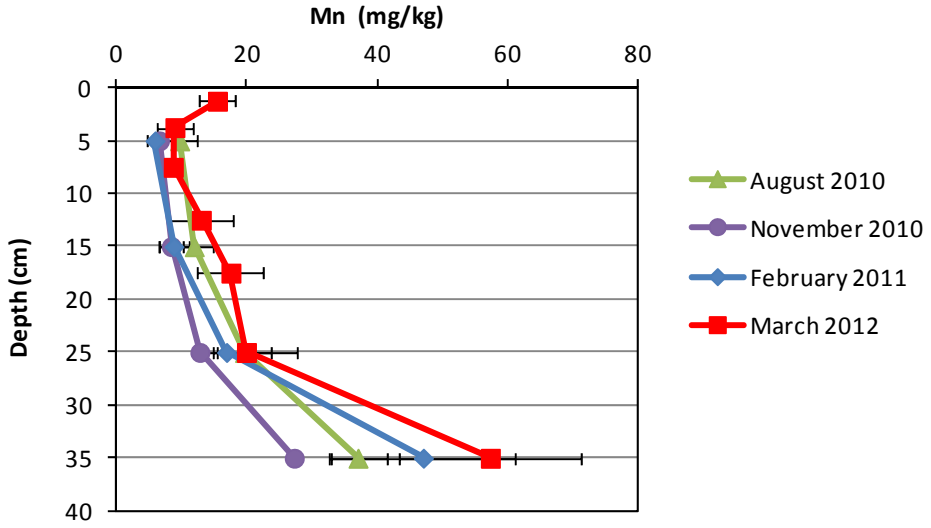

Figure 9-95. Campbell Park HCl extractable manganese dynamics at the control site (August 2010 – March 2012).

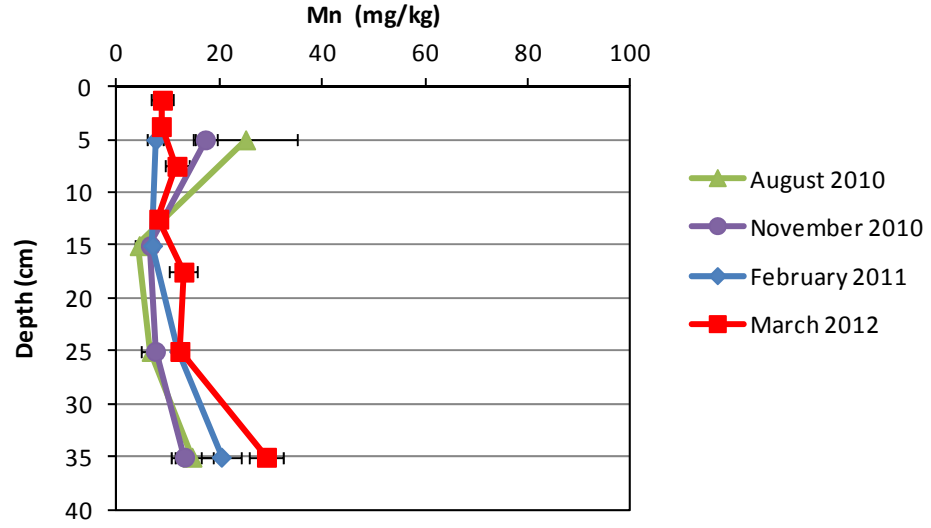

Figure 9-96. Campbell Park HCl extractable manganese dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

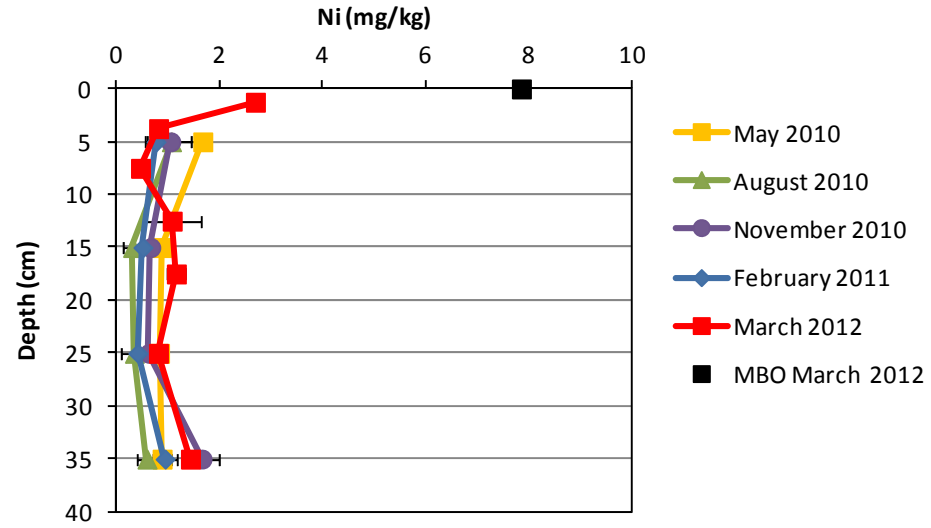

Figure 9-97. Waltowa HCl extractable nickel dynamics at the established *Phragmites* site (May 2010 – March 2012).

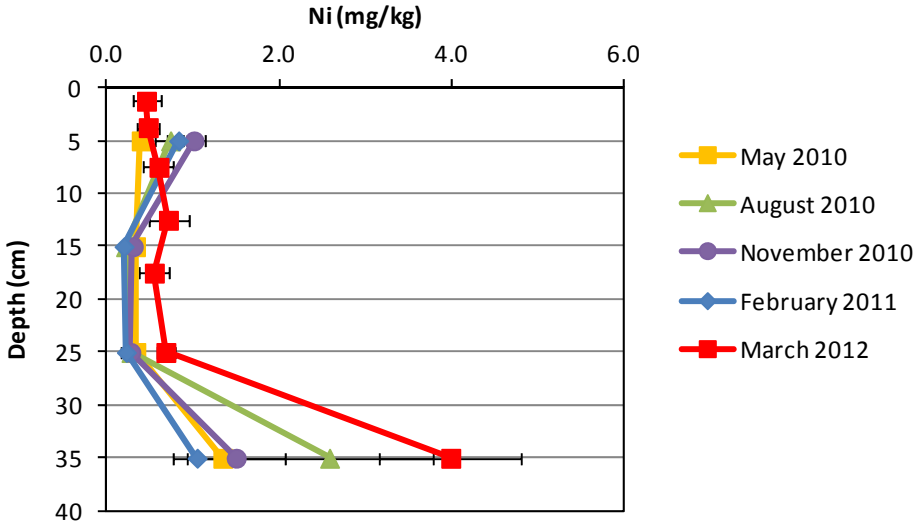

Figure 9-98. Waltowa HCl extractable nickel dynamics at the established *Cotula* site (May 2010 – March 2012).

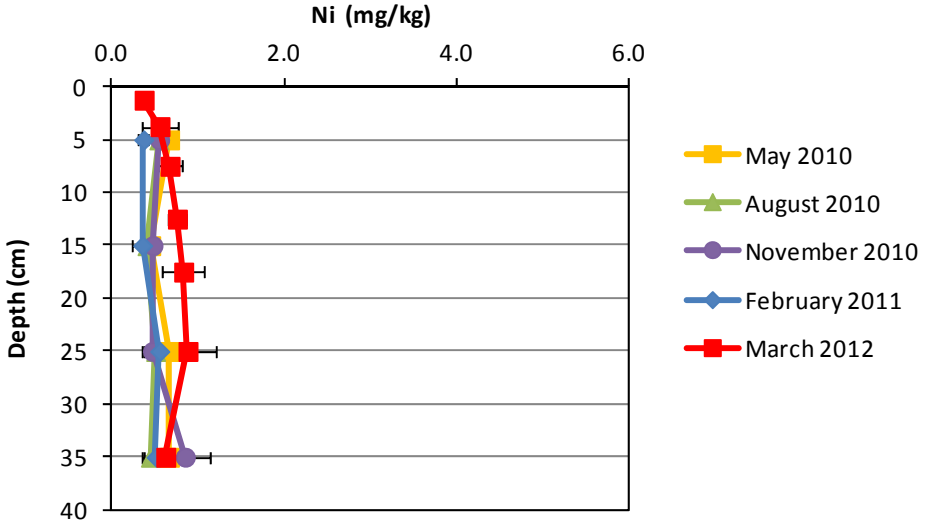

Figure 9-99. Waltowa HCl extractable nickel dynamics at the established *Juncus* site (May 2010 – March 2012).

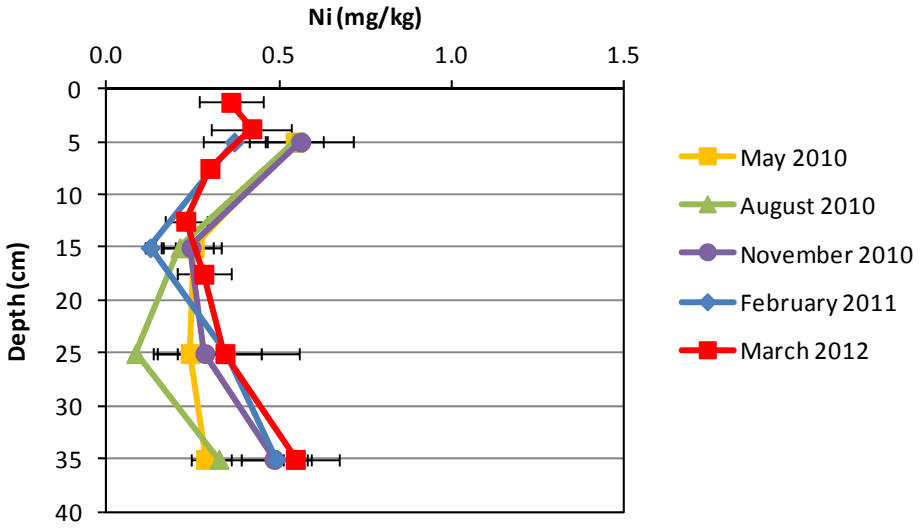

Figure 9-100. Poltalloch HCl extractable nickel dynamics at the *Juncus* plantings in Bevy rye site (May 2010 – March 2012).

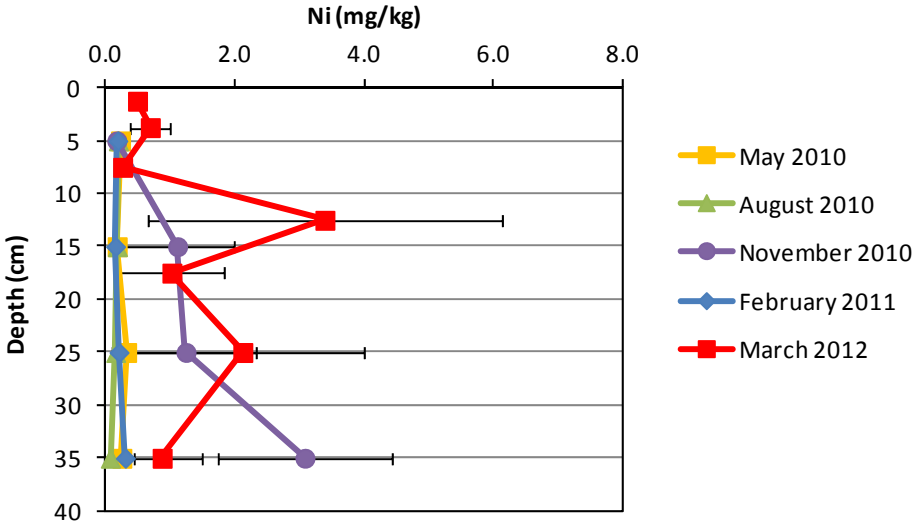

Figure 9-101. Tolderol HCl extractable nickel dynamics at the control site (May 2010 – March 2012).

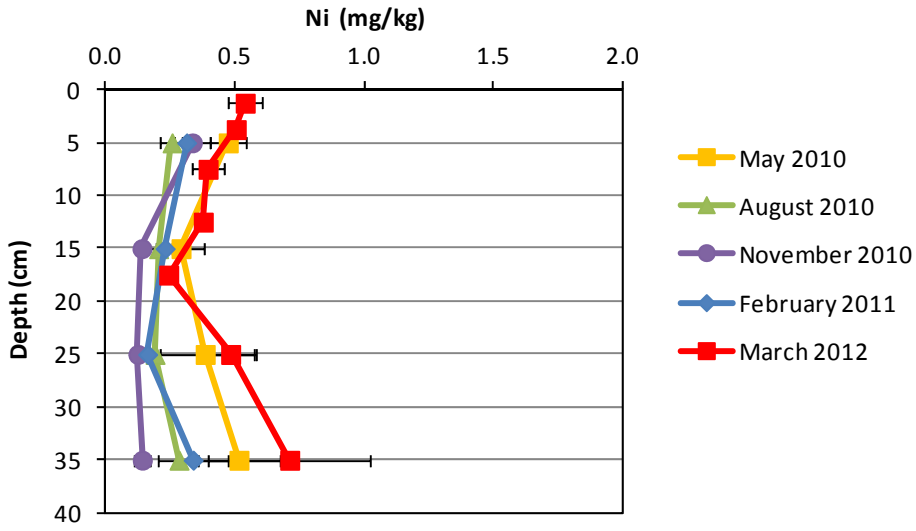

Figure 9-102. Tolderol HCl extractable nickel dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

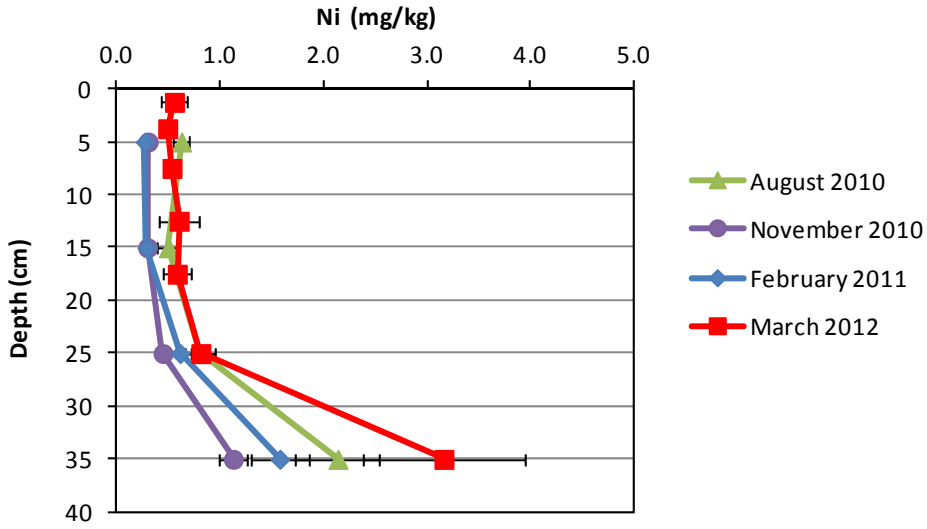

Figure 9-103. Campbell Park HCl extractable nickel dynamics at the control site (August 2010 – March 2012).

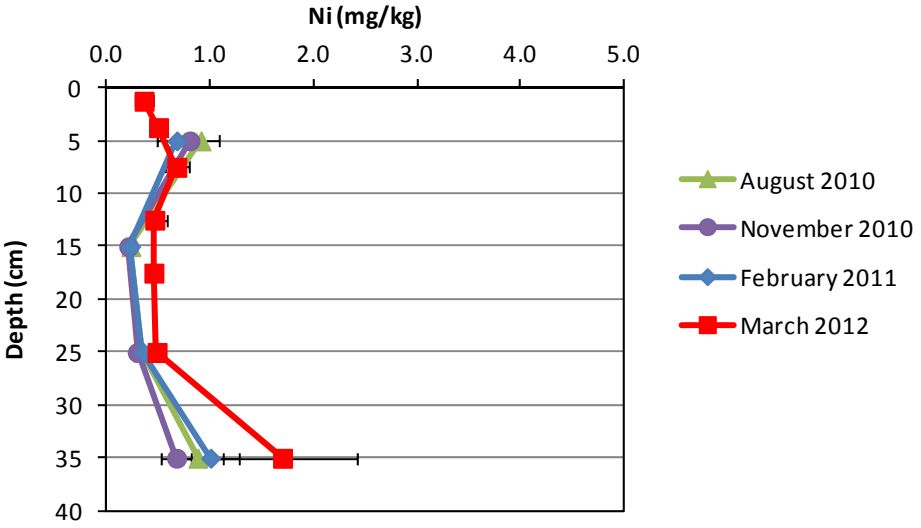

Figure 9-104. Campbell Park HCl extractable nickel dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

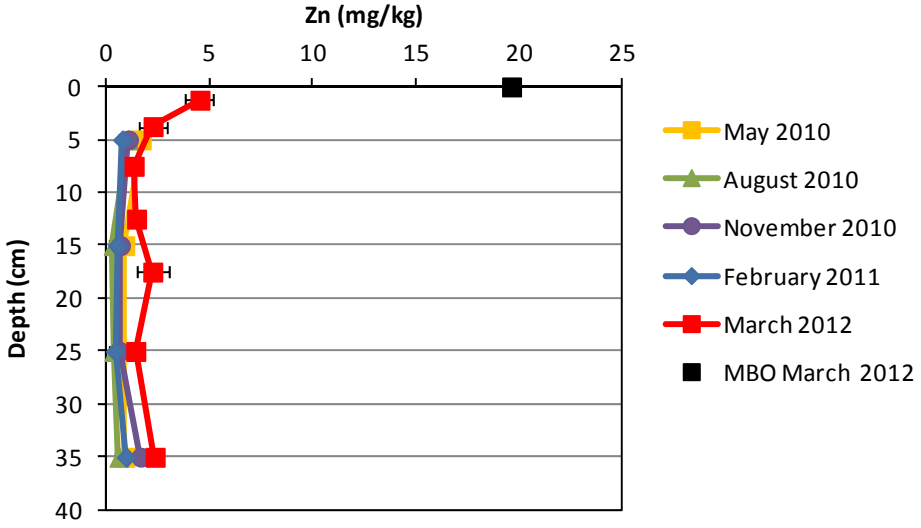

Figure 9-105. Waltowa HCl extractable zinc dynamics at the established *Phragmites* site (May 2010 – March 2012).

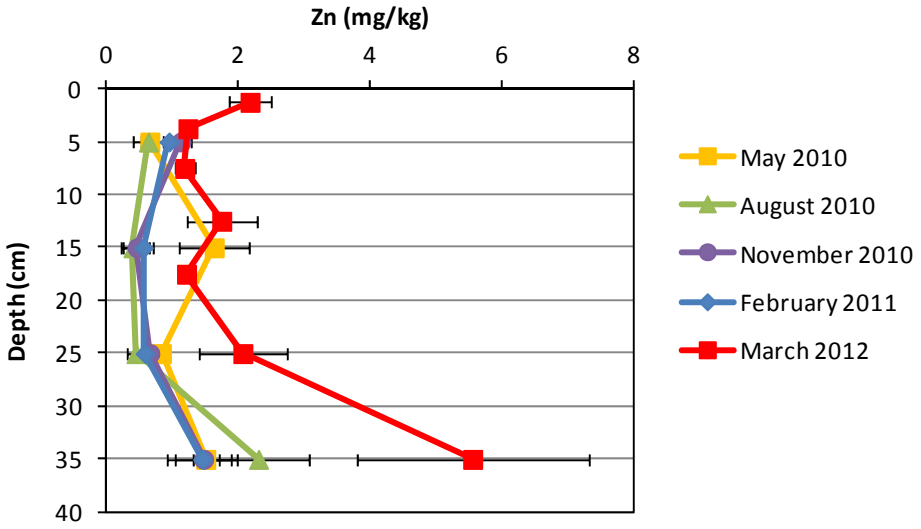

Figure 9-106. Waltowa HCl extractable zinc dynamics at the established *Cotula* site (May 2010 – March 2012).

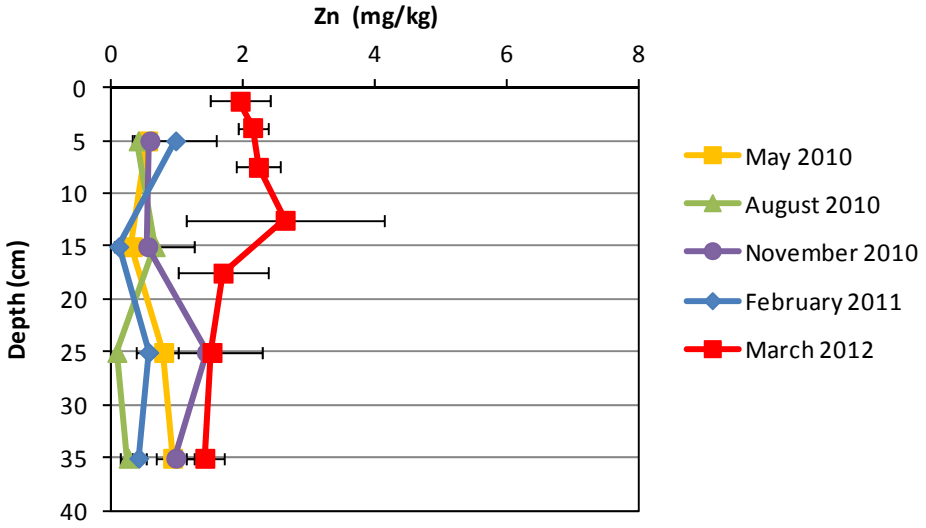

Figure 9-107. Waltowa HCl extractable zinc dynamics at the established *Juncus* site (May 2010 – March 2012).

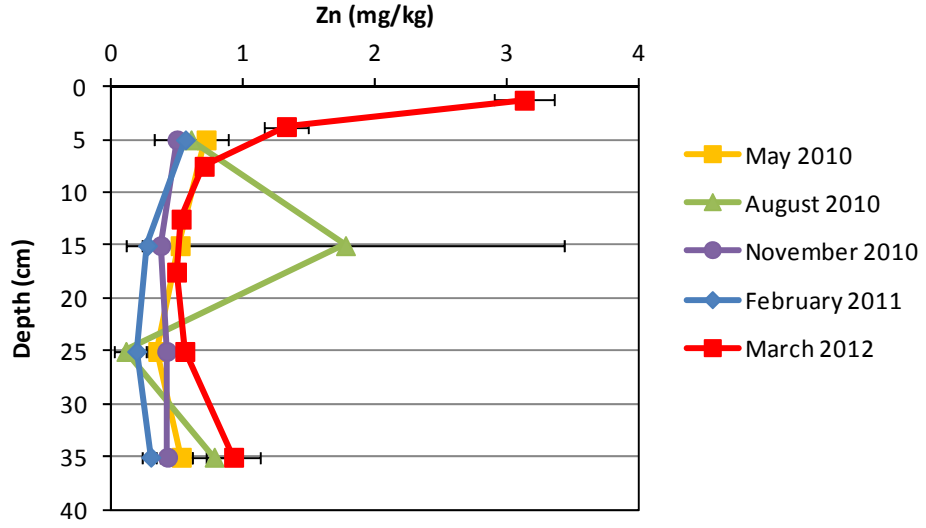

Figure 9-108. Poltalloch HCl extractable zinc dynamics at the *Juncus* plantings in Bevy rye site (May 2010 – March 2012).

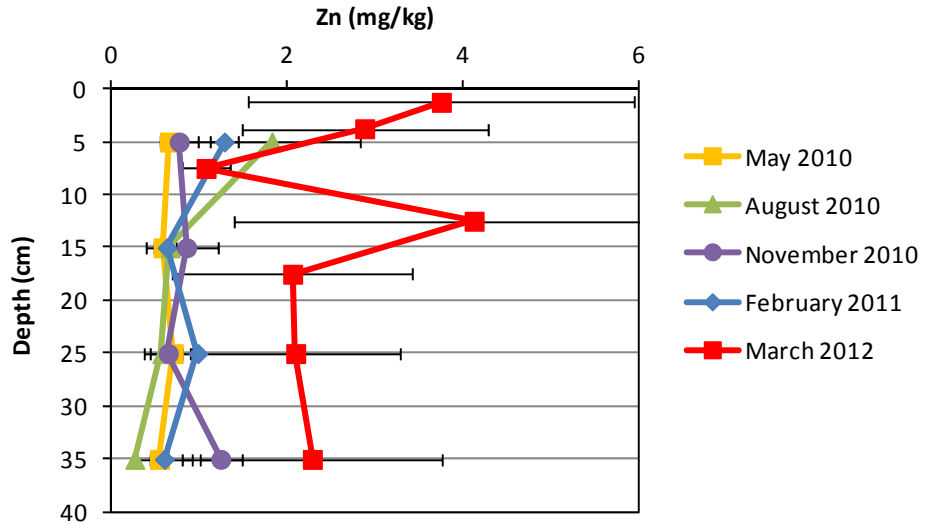

Figure 9-109. Tolderol HCl extractable zinc dynamics at the control site (May 2010 – March 2012).

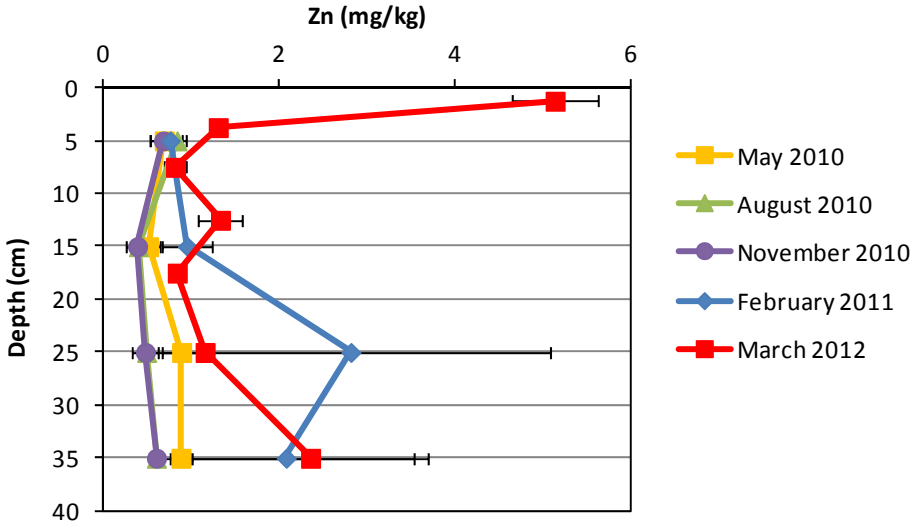

Figure 9-110. Tolderol HCl extractable zinc dynamics at the *Juncus* in Bevy rye site (May 2010 – March 2012).

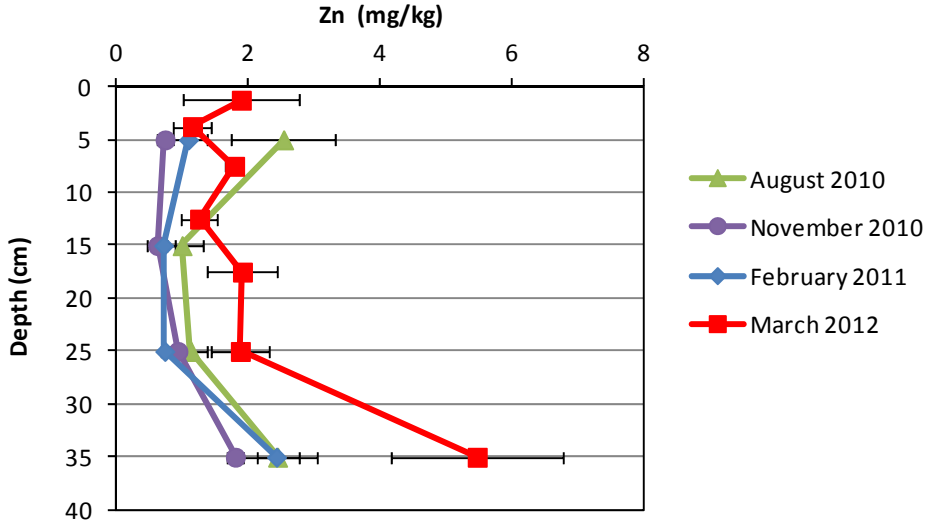

Figure 9-111. Campbell Park HCl extractable zinc dynamics at the control site (August 2010 – March 2012).

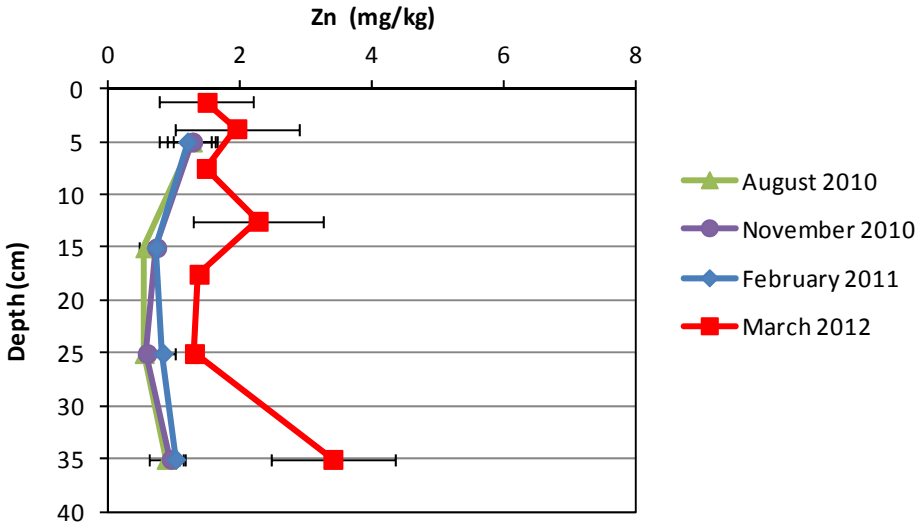

Figure 9-112. Campbell Park HCl extractable zinc dynamics at the Bevy rye/*Puccinellia* site (August 2010 – March 2012).

APPENDIX 7. Additional information

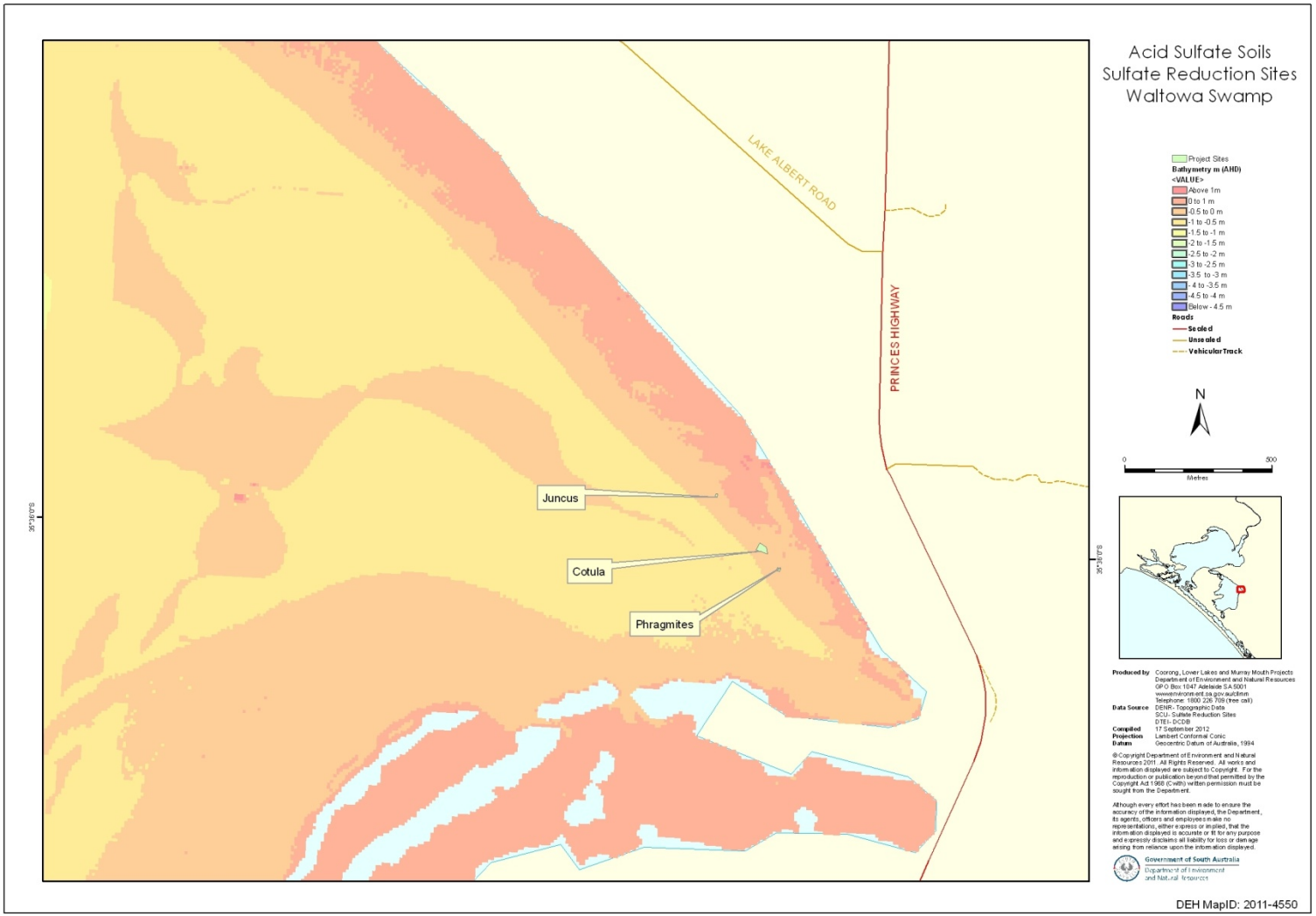

Figure 9-113. Bathymetry map for the Waltowa study area (Source: DEWNR).

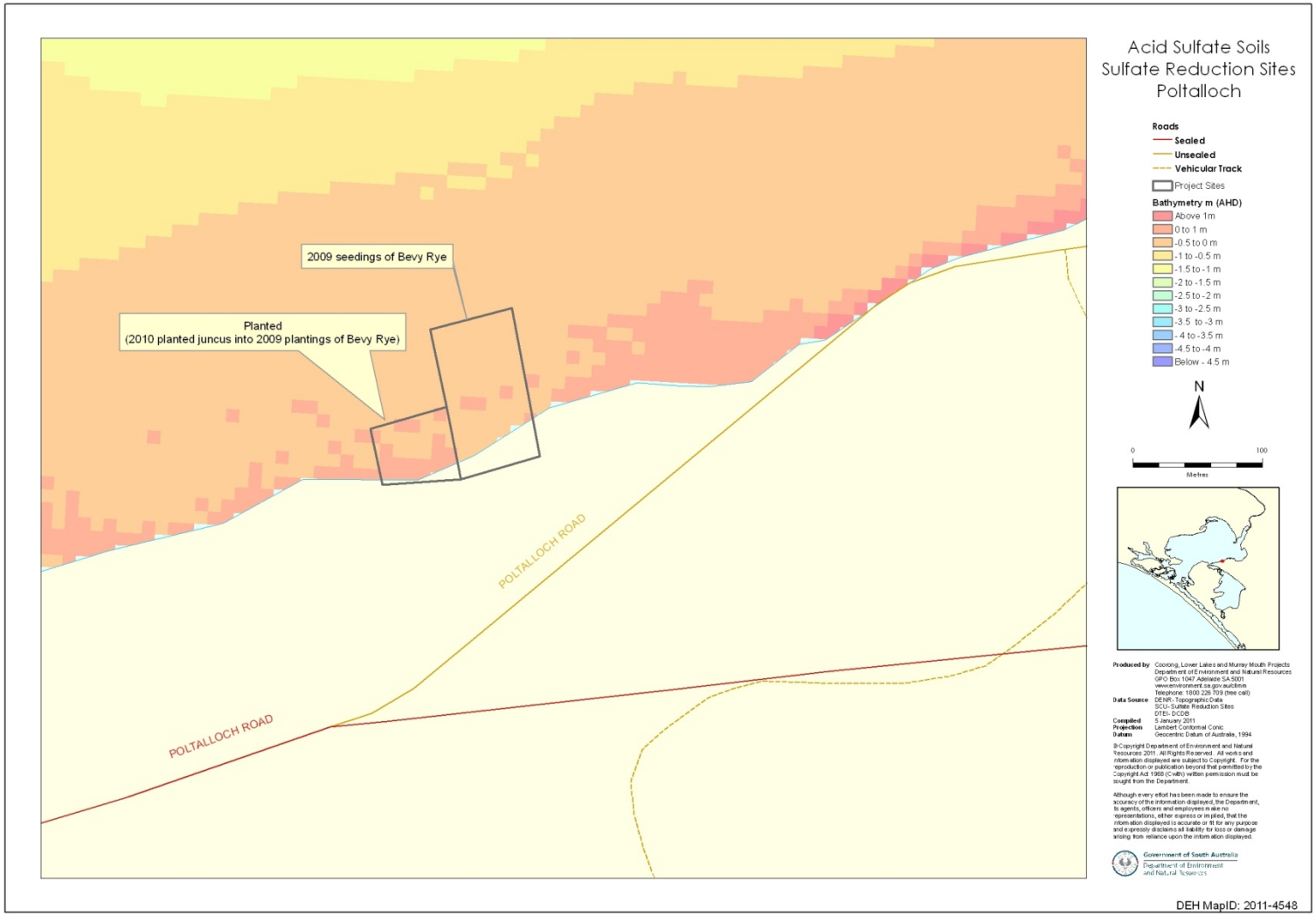

Figure 9-114. Bathymetry map for the Poltalloch study area (Source: DEWNR).

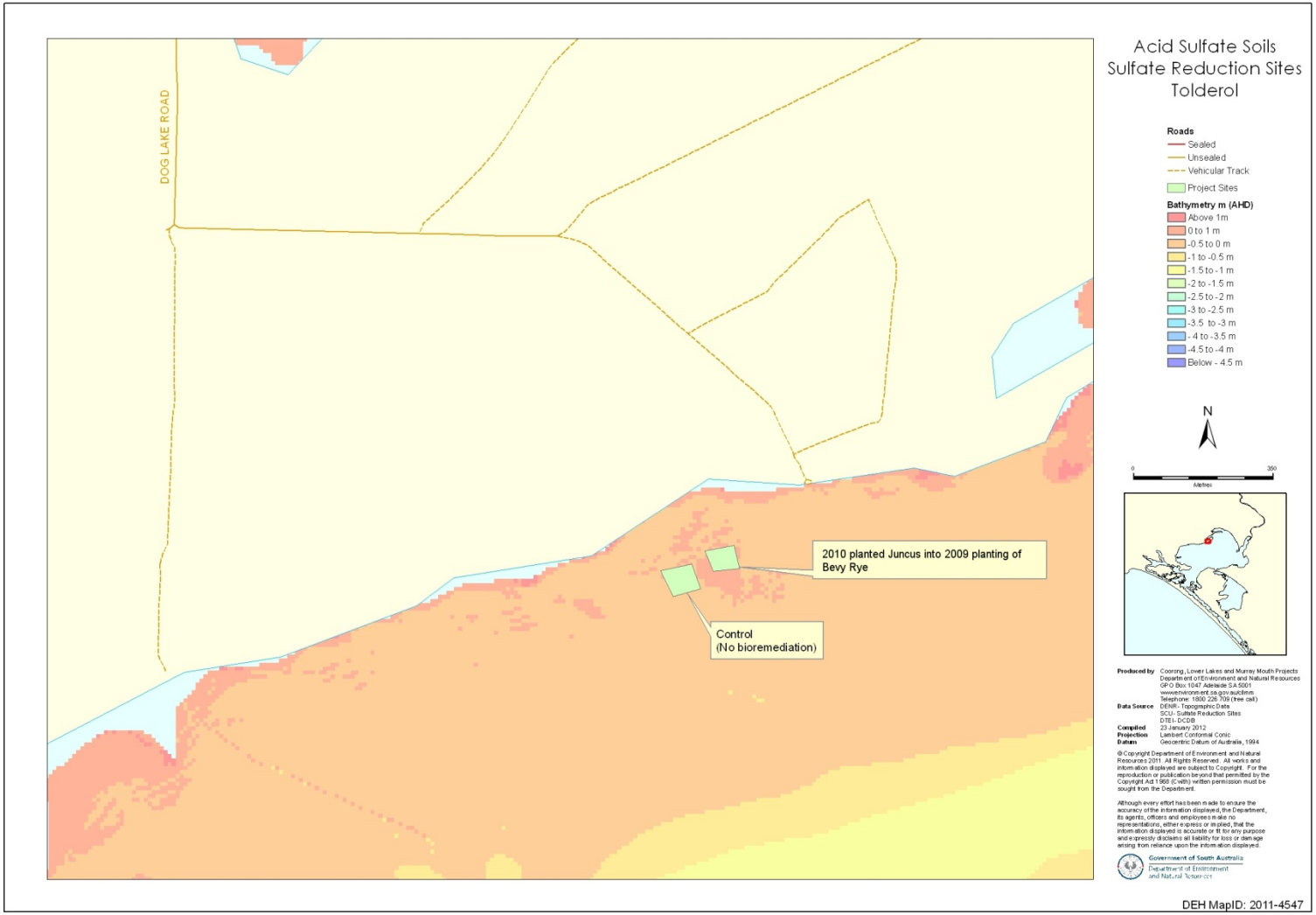

Figure 9-115. Bathymetry map for the Tolderol study area (Source: DEWNR).

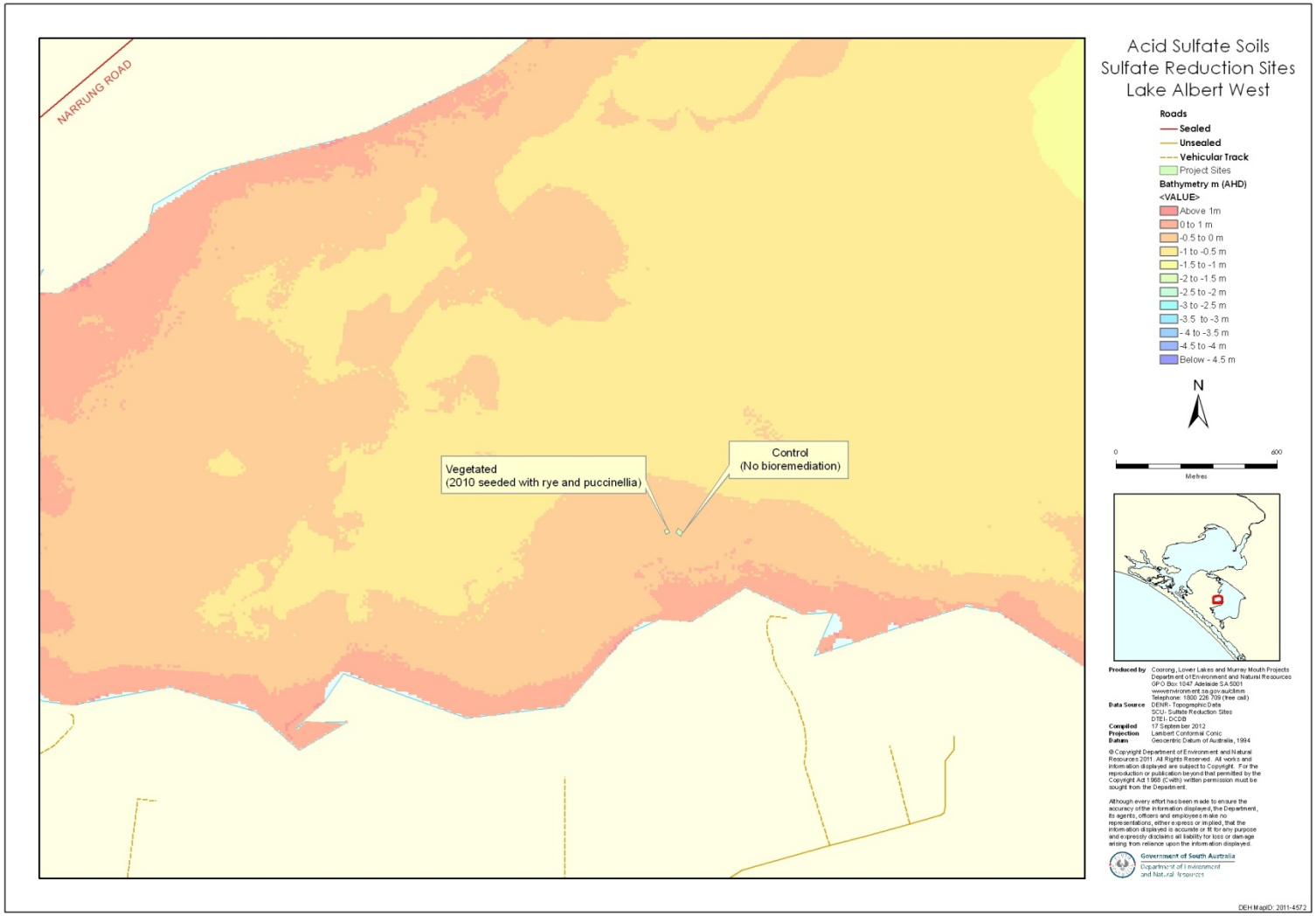

Figure 9-116. Bathymetry map for the Campbell Park study area (Source: DEWNR).

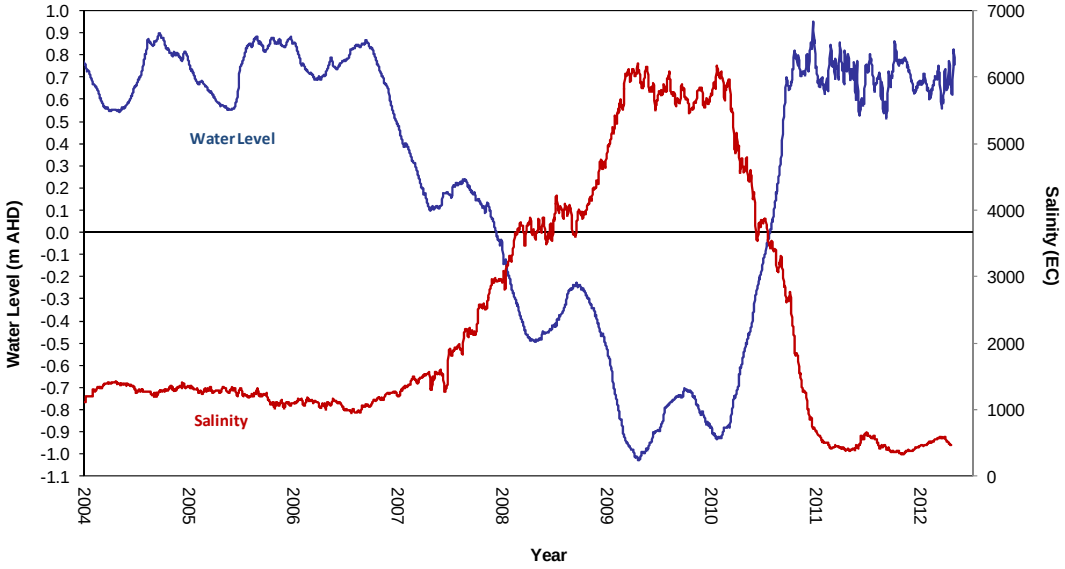

Figure 9-117. Lake Alexandrina historical water level and salinity data (Source: DEWNR).

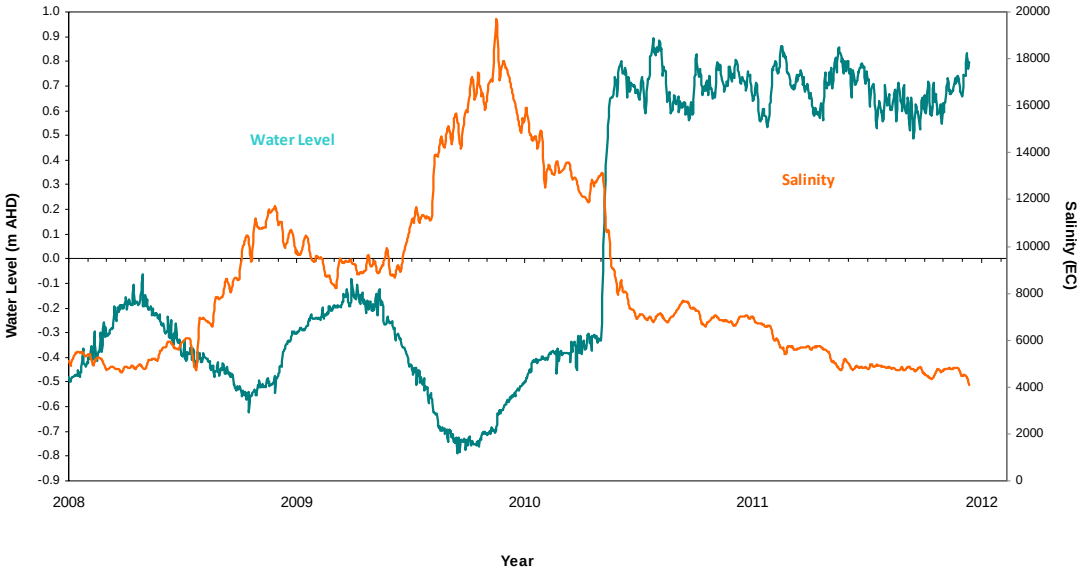

Figure 9-118. Lake Albert historical water level and salinity data (Source: DEWNR).

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