

A Long-Term Study of Mine Site Rehabilitation in Australia

L. Fergusson

Prana World Consulting, PO Box 1620, Oxenford, Queensland 4210, Australia

✉ lee@pranaworldgroup.com

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Abstract: This paper presents a three-part treatment and revegetation programme conducted at a derelict metaliferous mine site in Australia from 2000-2013. Programme #1 examined the treatment of acidity and leachable metals and the impact of revegetation in a control and three treated areas of contaminated waste rock at one part of the site; Programme #2 examined the treatment of acidity, plant uptake of metals and trace elements, and tree growth in a control and four treated areas of contaminated waste rock at a second section of the site; and Programme #3 examined the treatment of acidity and leachable metals and revegetation of submerged mine tailings and an exposed tailings beach.

Programme #1, which assessed lime and biosolids amended waste rock, a standard clay-topsoil capping of waste rock, and the addition of Terra B reagent to waste rock compared to a control, showed soil pH in the control remained unchanged from years 1-14 at around 4.0, while soil pH in the limed and capped areas was initially increased but fell back to starting levels of around 4.0 within 12 months and remained low for 14 years. In contrast, soil pH in the Terra B treated area was 6.8 immediately after treatment and increased to 7.2 in year 6 and 8.7 by year 14 without further treatment; these findings were consistent for data on total actual acidity and total potential acidity.

Similarly, leachable metals were generally lower in the Terra B area than either the control or other treatment areas. Tree height and density were also measured in 2005 and again in 2013 indicating acidity in the control, limed and capped areas suppressed tree height and density, however by 2013 in the Terra B area revegetation was comparable to undisturbed, remnant forest. In Programme #2, outcomes over the 14-year period were consistent with those observed in Programme #1 for the control, biosolids and Terra B treatment areas.

Programme #3 showed the submerged and exposed tailings which had low pH of 3.5 and high concentrations of total and leachable metals before treatment, but heavy metals became non-leachable immediately after treatment with Terra B reagent and pH rose to 8.0, remaining steady for 14 years. Revegetation was achieved on the tailings beach, with almost uniform colonization of bulrush and common reed, along with bottlebrush and golden wattle; a variety of reptiles have also been observed on the revegetated tailings beach, along with migrating ducks, spoonbills and grebes. In each case, what was highly contaminated waste rock or tailings before treatment, effectively became a neutralised, stable growth medium after treatment with Terra B reagent.

Key words: Mine, revegetation, acidity, biosolids.

Introduction

One of the most significant adverse chemical and environmental processes at metaliferous mine sites throughout the world is the formation of acid mine drainage (Harries, 1997; Gurang, 2005; Hogsden and Harding, 2012). Acid mine drainage (AMD) is a

problem wherever sulfidic waste rock and tailings are exposed to water and oxygen (Lin et al., 2003; Gleisner, 2005). While the generation of naturally occurring AMD results from the weathering of sulfidic rock commonly associated with gossans, metal-enriched soils and acidic water, AMD generated by mining activities can cause significantly greater adverse environmental impacts

because the rate of AMD formation is generally quite rapid and receiving environments often do not have time to adapt to incoming acidic, metal-rich water. AMD can occur at many different types of metaliferous mine sites (such as gold, copper, lead and zinc mines) but can also occur at coal mining and beneficiation operations; in each case, AMD is most commonly associated with on-site storage of acidic overburden, waste rock, and/or mine tailings.

AMD forms as a result of sulfide decomposition and iron oxidation when sulfidic minerals in the solids are exposed to atmospheric oxygen and moisture, producing sulfuric acid, soluble metals and sulfate ions. The main mineral that reacts to produce AMD is pyrite, as shown in Table 1, but the oxidation of

the most common metal ions associated with AMD, can produce acid when they react with water to form metal hydroxide precipitates, as shown in Table 2 (McConchie et al., 2000). Because each atom of metal that precipitates can generate three hydrogen ions, it is important to know the concentrations of dissolved iron and aluminium in addition to pH when determining how much alkalinity is required to neutralize actual acidity (i.e., existing acidity as measured by millimoles of acid per kg in the solid and pH) as well as potential acidity (i.e., acidity which can be generated in future due to the presence of stored acid-generating minerals, such as sulfides, as measured by acid base accounting and other methods, e.g., Brady and Cravotta, 1992) and thereby effectively counter the polluting properties of AMD.

Table 1: Examples of pyrite oxidation reactions

Standard oxidation reaction:	$2\text{FeS}_2 + 7\text{O}_2 + 2\text{H}_2\text{O} = 2\text{Fe}^{2+} + 4\text{SO}_4^{2-} + 4\text{H}^+$
If pH is greater than about 4.0, ferrollysis can generate more acid as indicated by:	$\text{Fe}^{3+} + 3\text{H}_2\text{O} = \text{Fe}(\text{OH})_3 + 3\text{H}^+$ and $4\text{Fe}^{2+} + 6\text{H}_2\text{O} + \text{O}_2 = 4\text{FeOOH} + 8\text{H}^+$
Under pH conditions below about 3.0, reaction with ferric ions can produce more acid:	$4\text{Fe}^{2+} + \text{O}_2 + 4\text{H}^+ = 4\text{Fe}^{3+} + 2\text{H}_2\text{O}$ and $\text{FeS}_2 + 14\text{Fe}^{3+} + 8\text{H}_2\text{O} = 15\text{Fe}^{2+} + 2\text{SO}_4^{2-} + 16\text{H}^+$

sulfides can follow different pathways depending on factors such as pH and availability of ferric iron and other potential oxidants (McConchie, Clark, Hanahan and Baun, 2000). Additional pathways of decomposition may also arise where reactions are mediated by the presence of bacteria, such as *Thiobacillus thiooxidans* or *Thiobacillus ferrooxidans*, which cause an increase in the rate of oxidation and thus acid generation and metal solubilization.

In addition to the release of acid as sulfide minerals oxidize, acid can be generated when metal ions react with water. For example, iron and aluminium, two of

Table 2: Acid-generating reactions when iron and aluminium ions form hydroxides

Iron hydroxides	$(\text{Fe}(\text{OH})_3)$ $\text{Fe}^{3+} + 3\text{H}_2\text{O} = \text{Fe}(\text{OH})_{3(s)} + 3\text{H}^+ (3)$
Aluminium hydroxides	$(\text{Al}(\text{OH})_3)$ $\text{Al}^{3+} + 3\text{H}_2\text{O} = \text{Al}(\text{OH})_{3(s)} + 3\text{H}^+ (3)$

Note: The number in brackets is the moles of acid generated or consumed by the reaction.

As a consequence, any estimate of AMD neutralizing requirements is typically based on the titratable acidity of AMD, because titratable acidity takes into account



Figure 1. Examples of AMD from the Savage River mine site in Tasmania (left) and from the Strauss open pit at the Mt Carrington mine site in New South Wales (right).

all potential acid-generating precipitation reactions which may occur as the AMD is neutralized, not simply on actual acidity as it exists at any given point in time. The same underlying principle applies to estimating neutralization requirements for waste rock and tailings, because metal hydroxides, particularly iron and aluminium oxyhydroxides, which are abundant in precipitates that form AMD at mine sites, have a high charge-to-mass ratio, making them extremely “surface reactive”. Hence, the fine particles in waste rock and tailings have an excellent ability to adsorb or co-precipitate trace metals, including potentially acid generating ions (such as Fe^{3+} or Al^{3+}), and can adsorb hydrogen ions, making them especially prone to AMD generation and downstream polluting consequences.

The ability of minerals in oxyhydroxide precipitates to adsorb hydrogen ions or metals depends on the sign and magnitude of the surface charge on each mineral particle, and this in turn depends primarily on the composition of the mineral particle, the pH of the solution in contact with it, and the type and concentration of any other ions in the solution. In particular, the sign and magnitude of the charge will depend on the pH of the solution relative to the point of zero charge (PZC) of a mineral precipitate; the PZC is the condition when the electrical charge density on the surface of a particle is zero. The PZC value (as measured by a negative decimal logarithm) for some components of typical acid-generating precipitates at mine sites is shown in Table 3. When pH of the precipitate is below the PZC, the charge on the mineral will be positive, but when it is above the PZC the charge will be negative, and any change in pH of the solution relative to the PZC will result in a change to which ions can be adsorbed or desorbed by the mineral contained in acidic waste rock and tailings.

In addition to the formation of low solubility metal hydroxides (the orange-brown precipitate or so-called “orange boy” commonly seen in areas affected by AMD), as the pH of AMD slowly rises (irrespective of whether it is due to natural weathering reactions or human intervention) evaporative concentrations of the acidic solution and interactions with waste rock, tailings, neutralizing agents, or other in-coming acidic water can result in the formation of metastable hydroxysulfate minerals (e.g., yellow-coloured jarosite) that may later oxidize to release more acid, thereby worsening the overall AMD contamination problem. These secondary sulfate mineral precipitates, as documented in Table 4, can include a range of heavy metals, and both metals and acid can be released to the environment if minerals dissolve as a result of heavy rainfall or a change in geochemical conditions.

Table 3: PZC values for some common mine site minerals in hydrous precipitates from AMD

Compound	PZC
$\text{Al}(\text{OH})_3$ [<i>gibbsite</i>]	5.0
$\text{Fe}(\text{OH})_3$ [<i>ferrhydrite</i>]	7.1-8.5
$\text{FeO}(\text{OH})$ [<i>goethite</i>]	3.2-6.7
Fe_2O_3 [<i>hematite</i>]	7.8
Fe_2O_3 [<i>maghemite</i>]	6.7
$\text{Mn}(\text{OH})_2$ [<i>pyrochroite</i>]	7.0
SiO_2 [<i>vitreous silica</i>]	1.8

These processes are contributors to a phenomenon known as “pH bounce back” that is sometimes observed when lime is used to treat AMD; clays and fine grained oxyhydroxide minerals in natural sediments and other mined precipitates in contact with AMD can release acid and trace metals as the composition of water changes in response to progressive acid neutralization.

Table 4: Acid generating potential of common secondary hydroxysulfate minerals

Halotrichite	$(\text{FeAl}_2(\text{SO}_4)_4 \cdot 22\text{H}_2\text{O})$ $\text{FeAl}_2(\text{SO}_4)_4 \cdot 22\text{H}_2\text{O}_{(\text{s})} + 0.25\text{O}_2 = \text{Fe}(\text{OH})_{3(\text{s})} + 2\text{Al}(\text{OH})_{3(\text{s})} + 13.5\text{H}_2\text{O} + 4\text{SO}_4^{2-} + 8\text{H}^+ \text{ (8)}$
Romerite	$(\text{Fe}_3(\text{SO}_4)_4 \cdot 14\text{H}_2\text{O})$ $\text{Fe}_3(\text{SO}_4)_4 \cdot 14\text{H}_2\text{O}_{(\text{s})} = 2\text{Fe}(\text{OH})_{3(\text{s})} + \text{Fe}^{2+} + 8\text{H}_2\text{O} + 4\text{SO}_4^{2-} + 6\text{H}^+ \text{ (6)}$
Melanterite	$(\text{FeSO}_4 \cdot 7\text{H}_2\text{O})$ $\text{FeSO}_4 \cdot 7\text{H}_2\text{O}_{(\text{s})} + 0.25\text{O}_2 = \text{Fe}(\text{OH})_{3(\text{s})} + 4.5\text{H}_2\text{O} + \text{SO}_4^{2-} + 2\text{H}^+ \text{ (2)}$
Jurbanite	$(\text{Al}(\text{SO}_4)(\text{OH})) \cdot 5\text{H}_2\text{O}$ $\text{Al}(\text{SO}_4)(\text{OH}) \cdot 5\text{H}_2\text{O}_{(\text{s})} = \text{Al}(\text{OH})_{3(\text{s})} + 3\text{H}_2\text{O} + \text{SO}_4^{2-} + 2\text{H}^+ \text{ (2)}$
Jarosite	$(\text{KFe}_3(\text{SO}_4)_2(\text{OH})_6)$ $\text{KFe}_3(\text{SO}_4)_2(\text{OH})_{6(\text{s})} + 3\text{H}_2\text{O} = \text{K}^+ + 3\text{Fe}(\text{OH})_{3(\text{s})} + 2\text{SO}_4^{2-} + 3\text{H}^+ \text{ (3)}$
Alunite	$(\text{KAl}_3(\text{SO}_4)_2(\text{OH})_6)$ $\text{KAl}_3(\text{SO}_4)_2(\text{OH})_{6(\text{s})} + 3\text{H}_2\text{O} = \text{K}^+ + 3\text{Al}(\text{OH})_3 + 2\text{SO}_4^{2-} + 3\text{H}^+ \text{ (3)}$

Note: The number in brackets is the moles of acid generated or consumed by the reaction indicated.

Reactions such as these that release additional acid and metals from waste rock and tailings can add layers of geochemical complexity to the effective treatment and long-term stabilization of AMD. However, when acidic water is treated while still in contact with sulfuric tailings, for example, the neutralization process may require more treatment reagents than calculated, because acidic tailings can re-release acid and metals back into solution; the rate of release of acid and metals from tailings usually increases as the treatment proceeds and the degree of acidic disequilibrium between treated water and tailings increases. This interdependent influence of acidic tailings on water quality and effective water treatment depends on local mineralogy and the chemical composition of tailings, and is influenced by the volume of potentially acidic and metal-generating solid precipitates in the tailings relative to the volume of water being treated. The interdependence of waste rock to water inflows are equally complex, and while the general principles of AMD are well established, they become increasingly difficult to diagnose in different geological and climatic environments when pH of the waste rock changes, atmospheric pressure changes leading to more or less oxygen infiltration, and rainwater ingress increase.

Although acid generating reactions are the major reactions normally associated with the formation of AMD, there are also natural acid neutralizing reactions that eventually neutralize stored or “potential” acidity rather than the actual acidity present in a precipitate at any given point in time. Many weathering reactions are “acid consumers”, and therefore improvements over time can be expected, but most of these naturally occurring reactions are extremely slow, sometimes hundreds of years in the case of one tailings beach (e.g., Taylor et al., 1996). Whether AMD is produced at all in any given environment depends on the relative proportions of “acid generating” and “acid neutralizing” minerals in each setting, their rates of reaction, and other geochemical and biological conditions that may affect rates of reaction. However, the continual discharge of AMD at mine sites can result in both cumulative and chronic long-term environmental impacts, and AMD is therefore widely credited with being one of the most significant environmental problems facing the worldwide mining industry, with serious impacts caused by both acidity and liberated metal ions (e.g., Davies et al., 1996).

For this reason, both preventing the formation and escape of AMD from existing mineral recovery operations as well as managing and remediating the

legacy stockpiles of acidic waste rock and tailings at abandoned or derelict mines represent a significant challenge to operators and government agencies responsible for long-term mine management; as noted above, without prevention and remediation of AMD mine sites become a “perpetual pollution machine” (Price and Errington, 1997).

Most AMD problems are associated with acidity, but the mobilization of toxic metals as pH decreases also causes significant damage to the environment because most metals are soluble as pH drops below 6.0. Some metals like cadmium and lead are largely insoluble in neutral pH, but other metals like arsenic, iron and copper remain relatively soluble in circumneutral pH, and the leaching of metals, even in neutral pH, can have a negative impact on receiving environments, including waterways and vegetation. For example, it has been observed that elevated concentrations of arsenic and copper in acidic soils at a mine site in Peru significantly retarded plant growth (Bech et al., 1997). This finding is particularly true in soils with reduced organic content, such as acidic soils, tailings and rock dumps, which are common at most mine sites.

Adverse environmental impacts of AMD can be avoided either by preventing its formation or by treating it once it has occurred. However, once AMD has formed there are few viable (let alone sustainable) treatment options available, and most typically involve some form of chemical intervention, although sulfur reducing bacteria have been used with some limited success (Bai et al., 2013; Doshi, 2006). The most widely used approach to managing sulfidic waste rock and tailings is to isolate them from atmospheric oxygen and water by placing rock in lined and capped impoundments (Eger et al., 2000) or by capping it with clay and topsoil, and isolating tailings under an anoxic water layer or so-called “free water cap” (Jiang et al., 2013) in order to disallow AMD from forming. Where the escape of acidic leachate from waste rock dumps is anticipated, permeable reactive barriers or other interception methods have been used to intercept leachate (e.g., Monroe et al., 2004); leachate can also be diverted to a dam or chemical processing facility for treatment, but is most commonly pumped back to the top of the waste rock dump untreated, thereby further concentrating AMD reactions in the dump and compounding any future AMD management efforts. Depositing sulfidic tailings below the water table to restrict atmospheric infiltration can be effective, provided the required level of standing water is continuously maintained over the tailings to avoid atmospheric oxygen infiltration; old

mine pits have been used for this purpose, but the strategy can be expensive because of the large volumes of acidic material typically involved in this type of endeavour. Each of these strategies requires ongoing and long-term management and expenditure, and is generally not considered a “walk away” solution now favoured by sustainable mine closure.

Where chemical interventions have been applied at mine sites, sulfidic waste rock and tailings are typically blended with acid neutralizing chemicals, such as lime, or encapsulated in unreactive materials, such as silica or phosphates (McCloskey, 2005). Blending sulfidic waste with lime is relatively cheap, but because lime is slightly soluble in rainwater (and some groundwater), if a series of wet seasons precede a dry period, much of the blended lime may leach from the waste rock or tailings before stored acid has been produced, subsequently leading to poor neutralization outcomes. Furthermore, the presence of lime increases the availability of bicarbonate ions that can catalyze the decomposition of sulfides and thereby accelerate acid and trace metal releases to the environment. The development of coatings on particles of lime mixed with sulfidic tailings and the precipitation of gypsum may also reduce the effectiveness of treatment due to the reduction of reactive surfaces.

The present study examined the use of two chemical reagents, Terra B and Acid B Extra which are said to avoid this problem. Each of these reagents is a benign, non-hazardous and non-toxic additive used in the treatment of mine site contamination and pollution. Terra B and Acid B Extra reagents are derived from modified alumina refinery residue mixed with other benign chemical agents, such as magnesium oxide (MgO); the powdered chemical formulation of these reagents has been used to neutralize acid and sequester heavy metals at mine sites.

Unlike the simple ionic binding of cations through changes in pH and adsorption normally associated with immobilizing chemicals such as calcium hydroxide (CaOH) and sodium hydroxide (NaOH), the metals sequestered in Terra B and Acid B Extra become more tightly bound as time passes. This phenomenon is due to the fact that ViroBind is composed of a complex cocktail of metals and minerals, including hematite (Fe_2O_3), beohmite ($\delta\text{-AlOOH}$), gibbsite ($\text{Al}[\text{OH}]_3$) and sodalite ($\text{Na}_4\text{Al}_3\text{Si}_3\text{O}_{12}\text{Cl}$), anatase (TiO_2), aragonite (CaCO_3), brucite ($\text{Mg}[\text{OH}]_2$), diaspore ($\beta\text{-Al}_2\text{O}_3 \cdot \text{H}_2\text{O}$), ferrihydrite ($\text{Fe}_5\text{O}_7[\text{OH}] \cdot 4\text{H}_2\text{O}$), gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$), hydrocalumite ($\text{Ca}_2\text{Al}[\text{OH}]_7 \cdot 3\text{H}_2\text{O}$), hydrotalcite ($\text{Mg}_6\text{Al}_2\text{CO}_3[\text{OH}]_{16} \cdot 4\text{H}_2\text{O}$), and p-aluminohydrocalcite

($\text{CaAl}_2[\text{CO}_3]_2[\text{OH}]_4 \cdot 3\text{H}_2\text{O}$). Of significance in these formulae is the presence of hydroxides and oxyhydroxide compounds which contribute to the acid neutralizing capacity of Terra B and Acid B Extra, both of which have extremely low solubility and hence react slowly with acid. At least 48 hours are required for complete “on contact” reactions, and tests involving the addition of sulfuric acid to these reagents show about 40% of the acid neutralizing capacity of reagents is exhausted in five minutes, about 70% in four hours, and about 95% in 24 hours (McConchie et al., 2000).

Similarly, the positively charged iron-, aluminium-, magnesium- and titanium-based molecules in Terra B and Acid B Extra not only initially adsorb metals but also lead to the long-term “sequestration” phenomena observed with inorganic species (Fergusson, 2009). Thus, the ability of minerals in Terra B and Acid B Extra to sequester trace metals is time dependent; while most initial acid neutralization and metal sequestration reactions are completed within the first 24 hours, research has shown the longer these reagents are left in-situ the more tightly sequestered metals become, indicating that long-term co-precipitation and isomorphic substitution reactions are occurring at a molecular level. These and other relevant phenomena identified with Terra B and Acid B Extra reagents at metaliferous mine sites around the world have been discussed elsewhere (Fergusson, 2010, 2012a, 2012b), and applications utilizing these and related reagents in the treatment of coal waste and radioactive elements, such as radium, and in industrial site remediation to sequester mercury, for example, have also been examined (Akhurst et al., 2011; Clark et al., 2004; Fergusson, 2013; Genç-Fuhrman et al., 2004; Hutson and Attwood, 2008; McConchie et al., 2002); other core technical issues associated with this technology have been the subject of specialist scientific papers (e.g., Taylor et al., 2011).

This paper examines a three-part acidic waste rock and mine tailings rehabilitation programme conducted at the Mt Carrington gold and silver mine at Drake in northern New South Wales from 2000-2013. Initial findings conducted at the site between 2000 and 2005 were reported earlier (Clark et al., 2001; Davies-McConchie et al., 2002; Fergusson, 2012c); this paper updates those findings with empirical, photographic and descriptive evidence of rehabilitation outcomes after 14 years. The study therefore asked: (a) how does the application of Terra B reagent in the treatment and revegetation of acidic waste rock compare with standard waste rock treatment methods and a control over a 14-year period; and (b) how does the addition

of Terra B and Acid B Extra reagents affect long-term outcomes associated with acidic mine tailings treatment and revegetation over the same period?

Method

The epithermal mineral deposit at Mt Carrington was first worked when alluvial gold was discovered in Plumbago Creek in 1868, with the original processing plant and many of the shafts, adits and open cut pits dug between the 1920s and 1980s still visible at the site today. These pits were relatively large mines in themselves and were given names such as White Rock, Carrington, Strauss and Lady Hampden, all of which have generated AMD since mining operations ceased in the 1990s. In 2000, a series of trials were designed to examine rehabilitation of waste rock and tailings, and this study documents those trials over a 14-year period. The areas chosen for these trials include two waste rock dumps and the main exposed tailings beach, areas which provided good access for researchers and were considered representative of the site.

Programme #1: Waste Rock Treatment and Revegetation

This rehabilitation programme conducted on a waste rock dump located 500 m south of Strauss adit monitored the effects of three different intervention strategies for treating and revegetating contaminated acidic waste rock from 2000-2013. The programme included four areas: an untreated waste rock control (Area One); a standard mixture of waste rock, lime (as CaCO_3) and biosolids (a source of nutrients, as reported by Brown et al., 2000) (Area Two); a standard capping treatment of waste rock in which the rock was capped with a semi-permeable clay-topsoil cover (Area Three); and treatment with a mixture of waste rock and Terra B reagent (Area Four).

Each area of waste rock was approximately 50 m \times 35 m in size. Particle size distribution of rock in all four areas ranged from fine clay to boulders (up to one metre in diameter), and contained little or no vegetation, as can be seen in Figure 2. As shown in Table 5, soil pH in all four areas was 4.3 prior to treatment. In Area Two, waste rock was mixed with lime and biosolids and Area Four with Terra B reagent to a depth of 300 mm, respectively (each area included about a 5% vol/vol waste rock to additive application rate). Area Three was capped with a semi-permeable covering of approximately 300 mm of clay and topsoil. All four areas were planted with seedlings immediately

after treatment in 2000 using the same number and size of Australian native tree species: narrow-leaved grey ironbark (*Eucalyptus paniculata*) and golden (or fringed) wattle (*Acacia frimbriata*). These two species were chosen because they are common throughout the area, would potentially form both an understorey and canopy, and were planted approximately every 10 m².

Table 5 shows the chemical characteristics of waste rock in all four areas prior to and after intervention through year 14, and Figures 2 and 4 (left) show what the waste rock looked like in all four areas before treatment and revegetation in 2000. The primary criteria for “revegetation” used in this programme were tree height and density. It is acknowledged that these criteria are relatively simplistic, and a more rigorous study should use more sophisticated criteria, including health of species, rate of growth, soil health, recruitment rates, species diversity, and other parameters associated with a more systematic investigation into on-site rehabilitation. However, for the purposes of this initial trial, tree height and density were considered a reasonable starting point from which to draw conclusions.

Programme #2: Waste Rock Treatment and Revegetation

Programme #2 was also carried out over a 14-year period from 2000-2013 on an acidic waste rock dump located about 2.0 km to the southeast of Programme #1 immediately contiguous with the Lady Hampden open cut, and included an untreated waste rock control (Area One) and four treatment areas (Areas Two to Five). In these areas, waste rock was vol/vol blended with different amounts of Terra B reagent and/or biosolids (added to supply macronutrients to the soil) according to the following: Area Two, 8% Terra B reagent; Area Three, 8% Terra B reagent + 2% biosolids; Area Four, 12% Terra B reagent + 3% biosolids; and Area Five, 2% biosolids only. Each area was approximately 25 m \times 25 m in size. Additives in Areas Two through Five were mixed into waste rock to a depth of approximately 300 mm, in which the same number and size of *E. paniculata* were planted as seedlings every 10 m². All five areas were measured for soil pH, total metal concentrations (Al, Mn, Fe, Zn and Cu in mg/kg), metal concentrations (Al, Mn, Fe, Zn and Cu as a %) and element concentrations (Ca, K, Mg and Na as a %) in leaves compared to metal and element concentrations in leaves of *E. paniculate* found grown in undisturbed Australian woodlands, and tree growth. Figure 9 (left) shows the waste rock dump area used in Programme #2 before treatment and revegetation.

Table 5: Soil pH, TAA, TPA and leachable metals in four waste rock areas, 2000-2013

<i>Year</i>	<i>Area One</i>	<i>Area Two</i>	<i>Area Three</i>	<i>Area Four</i>
Soil pH				
2000 (before treatment)	4.3	4.3	4.3	4.3
2001	4.6	5.5	5.5	6.8
2005	4.0	4.7	4.7	7.2
2013	—	5.5	4.0	8.7
TAA (mmoles/kg)				
2000 (before treatment)	87	87	87	87
2013	—	45	24	0
TPA (%)				
2000 (before treatment)	0.7	0.7	0.7	0.7
2013	—	0.18	0.68	0.08
Leachable Al (mg/L)				
2000 (before treatment)	5.0	5.0	5.0	5.0
2000 (immediately after treatment)	5.0	1.2	—	<0.01 [†]
2001	46	4.0	—	<0.01 [†]
2002	33	4.0	—	<0.01 [†]
2005	29	3.0	—	<0.01 [†]
2013	—	5.6	1.2	3.0
Leachable Cd (mg/L)				
2000 (before treatment)	1.2	1.2	1.2	1.2
2000 (immediately after treatment)	1.2	1.0	—	0.1
2001	5.5	1.8	—	0.2
2002	5.1	1.8	—	0.1
2005	5.4	2.0	—	<0.01 [†]
2013	—	0.45	<0.01 [†]	<0.01 [†]
Leachable Cu (mg/L)				
2000 (before treatment)	1.2	1.2	1.2	1.2
2000 (immediately after treatment)	1.2	1.5	—	0.5
2001	13	3.8	—	0.1
2002	12.5	2.0	—	0.1
2005	1.6	5.9	—	<0.01 [†]
2013	—	0.4	0.6	<0.01 [†]
Leachable Pb (mg/L)				
2000 (before treatment)	0.25	0.25	0.25	0.25
2000 (immediately after treatment)	0.25	0.3	—	0.01
2001	0.11	0.43	—	<0.01 [†]
2002	0.35	0.14	—	0.01
2005	0.15	0.13	—	0.01
2013	—	<0.01 [†]	<0.01 [†]	<0.01 [†]

[†] Below level of detection**Programme #3: Tailings Treatment and Revegetation**

In 2000-2001, the tailings dam wastewater, submerged tailings, and exposed “tailings beach” at the site were treated using Acid B Extra and Terra B reagent. Programme #3 focuses on the physical and chemical status of the submerged tailings and the exposed tailings beach after revegetation over a 14-year period.

Treatment protocols for submerged tailings and the tailings beach consisted of two different methods. The submerged tailings were not treated “directly” in 2000 but “indirectly” as a consequence of dam wastewater treatment. In other words, as a result of treating the acidic wastewater in the dam by direct addition of Acid B Extra reagent, a “blanket” of chemical reagent was formed incidentally over the surface of the submerged



Figure 2: Close up photographs of Area One before treatment and revegetation in 2000, Programme #1.

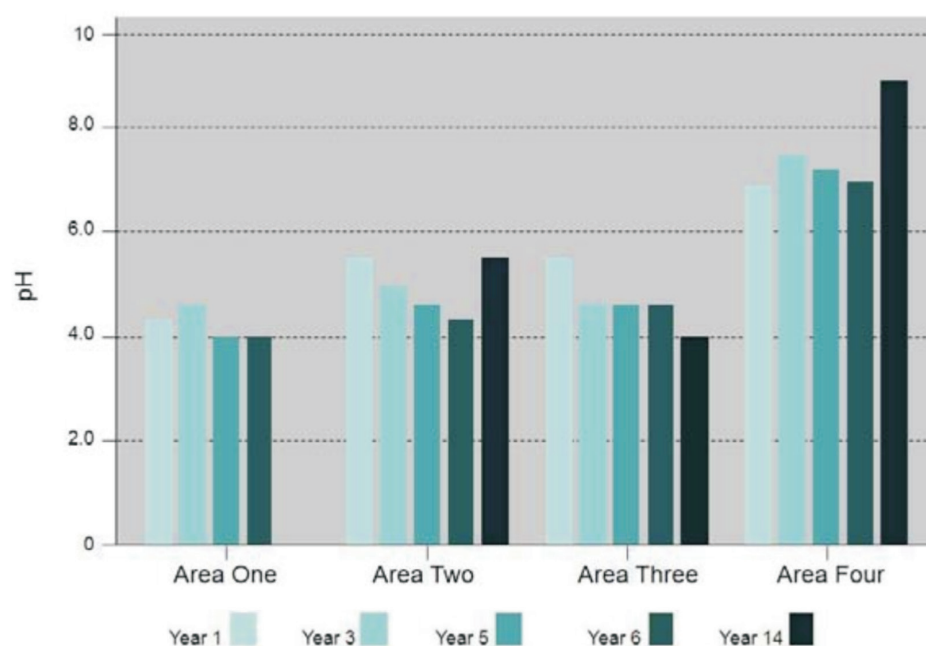


Figure 3: Soil pH of untreated waste rock (Area One), lime-treated waste rock (Area Two), clay-topsoil capped waste rock (Area Three), and Terra B-treated waste rock (Area Four) over a 14-year period.



Figure 4: Area One waste rock before treatment in 2000 (left); Area One after 14 years of revegetation in 2013, Programme #1.

tailings as the solid fine particles of the reagent settled through the water column at a rate of about 10 m in 48 hours (see Figure 14, left). This blanket of reagent, created as a result of direct addition of approximately 10 g/L to the dam water column, was ultimately about 3.0-5.0 mm thick after settling, thereby effectively “capping” the submerged tailings.

It has been presumed that some degree of physical mixing may have occurred as a result of wind and wave action in the dam because untreated submerged tailings and the “blanketed” submerged tailings had significantly different chemical properties; however, the details of this submarine process have never been fully investigated. Irrespective of the actual physical mechanisms involved, blanketed submerged tailings were successfully treated as a consequence of wastewater treatment, and any by-product effect on the tailings were incidental to the larger dam treatment.

In contrast, the tailings beach, which is shown in Figure 15 immediately after treatment and dewatering but before revegetation, was treated by direct addition of Terra B reagent in 2001. This included the addition of about 3% vol/vol of reagent to tailings; the reagent was blended into the surface of the exposed tailings to a depth of approximately 200-300 mm with a rotary hoe or in some cases was deposited directly on the tailings where access to tailings was limited.

The total exposed tailings beach was approximately 100 m × 100 m, with parts of the beach planted randomly with seedlings of bottlebrush (*Callistemon* spp., including *Callistemon citrinus*, *Callistemon salignus*, and *Callistemon viminalis*) and golden wattle (*Acacia pycnantha*). As dust had earlier been an issue with the exposed tailings, the revegetation effort was designed to both treat the contaminated tailings by raising pH and sequestering metals into non-bioavailable forms and to revegetate the tailings beach and thereby suppress dust.

Results

Programme #1: Waste Rock Treatment and Revegetation

Table 5 shows that soil pH in all four areas was 4.3 before treatment; soil pH in Area One was 4.6 in year one, and 4.0 in years two through six. It has been assumed that soil pH remained unchanged by year 14 given its stability over the first six years. Soil pH in Area Two was 5.5 in year one, dropped back to about 5.0 by year two, remained steady at about 4.7 through to year six, and was 5.5 in year 14. Soil pH in Area Three was initially raised to 5.5 immediately after treatment,

but fell to 4.7 within 12 months and stayed there for six years, and was 4.0 in year 14.

Soil pH in Area Four was 6.8 immediately after treatment in year one, 7.2 in the sixth year without further treatment, and actually rose to 8.7 by year 14, as seen in Figure 3. Soil pH data were derived from leachate samples collected in lysimeters but soil pH data in 2013 were obtained by testing the waste rock pH itself due to an absence of leachate in lysimeters as a result of a dry season. For this reason, 2013 pH results should be taken as indicative only.

Total actual acidity (TAA) in Area One was 87 mmoles/kg in year one, but was not measured in year 14. However, by year 14, TAA was 45 mmoles/kg in Area Two (a 49% decrease), 24 mmoles/kg in Area Three (a 73% decrease), and less than the detection limit in Area Four (a 100% decrease). Similarly, total potential acidity (TPA) in Area One was 0.7% in year one (but was not measured in year 14), 0.18% in Area Two by year 14 (a 75% decrease), 0.68% in Area Three by year 14 (a 3% decrease), and 0.08% in Area Four by year 14 (an 89% decrease). These two findings suggest that Terra B totally neutralised both the actual acidity as well as mostly eliminated the potential for waste rock to generate acid in the future, while treatment methods used in Areas Two and Three did not.

Table 5 also presents results for leachable Al, Cd, Cu and Pb over 14 years for Areas One, Two, Three and Four. Leachable metals data from 2000-2005 were derived from leachate samples collected in lysimeters, but pH and leachable metals data in 2013 were obtained by conducting a TCLP leach test on waste rock samples due to an absence of leachate in lysimeters; for this reason, 2013 results should be taken as indicative only. Leachable aluminium was 5.0 mg/L in Area One immediately after treatment, 46 mg/L after two years, settling back to a 29 mg/L after six years; leachable aluminium was not tested in Area One in year 14; leachable aluminium remained relatively stable in Area Two at around 3.0 mg/L over the six years and was 5.6 mg/L in year 14; leachable aluminium was not tested in Area Three over the six years but was 1.2 mg/L in year 14; and leachable aluminium was beyond detection limit in for Area Four over six years, but was 3.0 mg/L in year 14, possibly due to sampling differences from previous years.

Leachable cadmium was 1.3 mg/L one year after initial testing in Area One, rising to 5.4 mg/L over six years; leachable cadmium was not tested in Area One in year 14; leachable cadmium rose from 1.0 mg/L to 2.0 mg/L in Area Two by year six but was <0.45 mg/L

by year 14; leachable cadmium was not tested in Area Three over the six years but was below the detection limit by year 14; and leachable cadmium decrease was below the detection limit in Area Four by year 14.

Leachable copper was 1.2 mg/L one year after initial testing in Area One, rising to 1.6 mg/L over six years;

leachable copper was not tested in Area One in year 14; leachable copper rose from 1.5 mg/L to 5.9mg/L in Area Two by year six but was 0.4 mg/L by year 14; leachable copper was not tested in Area Three over the six years but was 0.6 mg/L by year 14; and leachable copper was below the detection limit in Area Four by year 14.



Figure 5: Results of revegetating Area Two (lime and biosolids) in 2013.



Figure 6: Results of revegetating Area Three (clay-topsoil capping) in 2013.



Figure 7: Results of revegetating Area Four (Terra B addition) in 2013, Programme #1.

Leachable lead was 0.25 mg/L one year after initial testing in Area One, dropping to 0.15 mg/L by year six; leachable lead was not tested in Area One in year 14; leachable lead dropped from 0.3 mg/L to 0.13 mg/L in Area Two by year six, but was below detection limit by year 14; leachable lead was not tested in Area Three over the six years but was below the detection limit by year 14; and leachable lead was 0.01 mg/L in year one, but dropped below the detection limit in Area Four by year 14.

Figure 8 summarises the results of tree growth in all four areas, and Figures 4, 5, 6 and 7 provide photographic evidence of the same findings. Figure 4 shows that virtually no trees grew in Area One either before the programme or over the 14-year period, with a generally declining trend in tree height of the few trees in the area over 14 years as recorded in Figure 8. Approximately 20% of planted trees were alive in the lime and biosolids-treated Area Two after three years, although their relative growth was minimal with an average height of 1.0 m after six years. There was an indication of tree growth but not tree density in Area Two by year 14, with tree heights averaging about 4.0 m by 2013; Figure 5 provides photographic evidence of tree growth in 2013, with a note that the vast majority of revegetation in Area Two was grass not trees. A similar growth pattern was observed in Area Three, although with decidedly more bald, un-grassed areas, with photographic evidence provided in Figure 6.

More than 80% of planted trees in the Terra B remediated Area Four were alive after two years, and their relative tree heights were the most significant of all areas with an average height of 2.1 m by year six, rising to about 9.0 m by 2013, as shown in Figure 7. As evidenced in Figures 5, 6 and 7, when comparing Area Four to Areas Two and Three, it was also visually clear that Area Four had developed a more obvious understorey of golden wattle and self-seeded native species, whereas Areas Two and Three did not; these two areas were mostly populated with self-seeded grass species and trees with reduced height and density. Tree height and density in 2013 indicated acidity in Area One mostly disallowed tree growth, and despite increasing acidity in Areas Two and Three, tree growth occurred but was minimal; in Area Four, tree height and density were comparable to undisturbed, remnant forest, which was populated mostly with ironbark and other eucalypt species.

Results from this 14-year study indicate that a lime-biosolids treatment and clay-topsoil capping of waste rock were not beneficial in neutralising acid over a

14-year period, with soil pH levels in both Areas Two and Three returning to near untreated waste rock levels over that time. Similarly, while TAA in both areas was lower than the control, TPA remained unchanged in Area Three but was reduced in Area Two. However, when compared to Area Four, pH had increased significantly over the 14-year period and both TAA and TPA had been markedly reduced.

Leachable metal trends generally indicate that Terra B reagent was more effective in sequestering metals over the 14-year experimental period than other treatments, although some data indicate that Areas Two and Three also either bound metals or had levels of leachable metals comparable to the control. As noted above, higher concentrations of leachable heavy metals, such as arsenic and copper, have been linked to minimizing effects in tree growth. However, the leachate data must be weighed against the difference in sampling methods between the 2000-2005 sampling period and the 2013 sampling period.

Programme #2: Waste Rock Treatment and Revegetation

Prior to treatment, waste rock used in Programme #2 had a soil pH of 3.5. Figure 10 shows that soil pH over an eight-month monitoring period was generally stable in all areas after treatment. Soil pH was low and did not change over the eight-month period in Area One (control) and increased marginally in Area Five (biosolids only), but rose to near-neutral levels in Areas Two, Three and Four that were treated with Terra B reagent.

Figure 11 shows concentrations of heavy metals present in the leaves of *E. paniculata* two years after planting in 2002 in the five areas when compared to metal concentrations in leaves of the same species growing in undisturbed Australian woodlands (i.e., “average natural”). From these data it is clear that plant uptake of aluminium, copper, iron, manganese and zinc was generally lower and at average natural levels for all three Terra B treatments, but that plant uptake of metals was significantly higher in both Areas One and Five when compared to both Areas Two, Three and Four and average natural levels. This finding indicates that Terra B was more effective in sequestering metal contaminants into non-bioavailable when compared to both doing nothing or biosolids only.

In order to maintain representative sampling, in this study, leaves were removed from similar parts of each plant and included the main leaves along the stem and upper leaf tips and fully extended leaves below the

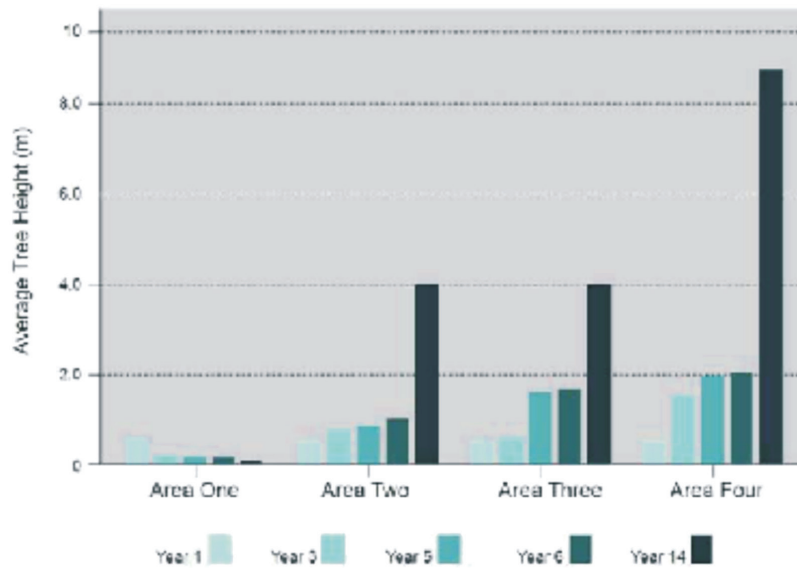


Figure 8: Average growth rate of trees planted in untreated waste rock (Area One), lime-treated waste rock (Area Two), clay-topsoil capped waste rock (Area Three), and Terra B-treated waste rock (Area Four) over a 14-year period in Programme# 1.



Figure 9: The waste rock dump used in all five areas of Programme #2 before treatment and revegetation, 2000 (left); Area One, the control, in 2013 (right).

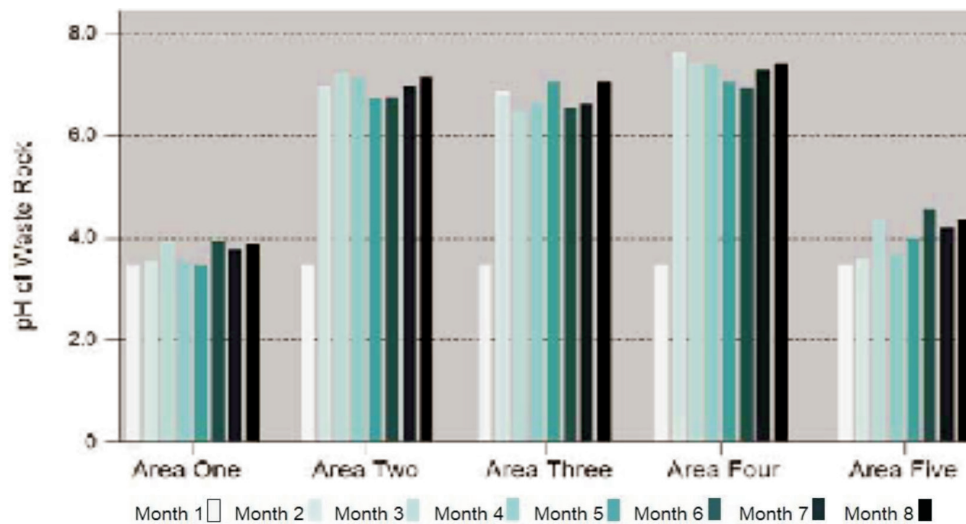


Figure 10: Average soil pH of five revegetation areas over an eight-month period in Programme #2.

main tip. Variations in growth of *E. paniculata* between treatments may have been impacted by variations in element concentrations, soil pH, and other chemical and physical soil parameters. However, the end result of these parameters is the associated trace element toxicities or deficiencies that maintain plant physiology. Figure 12 shows total concentrations of trace elements present in the leaves of *E. paniculata* planted in the five areas after two years in 2002 when compared to total trace element concentrations in leaves of the same species grown in undisturbed Australian woodlands (i.e., “eucalypt average”).

Results indicate, for example, concentrations of sodium in leaves were: Area One = 0.04 mg/kg; Area Two = 1.05 mg/kg; Area Three = 0.8 mg/kg; Area Four = 0.7 mg/kg; and Area Five = 0.45 mg/kg (eucalypt average = 0.2 mg/kg), and concentrations of potassium in leaves were: Area One = 0.9 mg/kg; Area Two = 12.2 mg/kg; Area Three = 0.8 mg/kg; Area Four = 0.9 mg/kg; and Area Five = 0.9 mg/kg (eucalypt average = 0.4 mg/kg). From the data presented in Figure 12 it can be concluded that plant uptake of calcium and magnesium was generally near to or lower than the eucalypt average for all three Terra B treatments but were lower than Areas One and Five; for sodium and potassium, Areas Two, Three and Four were higher in sodium and potassium than eucalypt average levels and for Areas One and Five.

The final outcome of revegetation in Programme #2 is shown in Figure 13. Figure 9 (left) shows the waste rock dump before any treatment or revegetation was carried out, and Figure 9 (right) shows Area One in 2013. Note, virtually no grass or trees established in the control area during the 14-year period. However, as shown in Figure 13 (left and right) Areas Two, Three, Four and Five were largely indistinguishable from each other by 2013; in fact the research team was unable to identify demarcation points between Areas Two, Three, Four or Five as revegetation of grass and trees made identification of these separate areas impossible.

Programme #3: Tailings Treatment and Revegetation

Previously published data on the tailings dam wastewater treatment programme showed the addition of about 10g/L of Acid B Extra reagent, increased dam pH from 3.5 to 8.0, while decreasing soluble Al from 60 mg/L to 0.03 mg/L, Cd from 1.6 mg/L to <0.001 mg/L, Cu from 42 mg/L to 0.003 mg/L, Fe from 7.0 mg/L to 0.03 mg/L, Pb from 0.7 mg/L to 0.03 mg/L, Ni from 0.1 mg/L to 0.01 mg/L, and Zn from 64 mg/L to 0.01

mg/L (Davies-McConchie et al., 2002; McConchie et al., 2000; HazNews, 2000). This treatment facilitated the establishment of silver perch (*Bidyanus bidyanus*), as shown in Figure 14 (right), and eastern snake-necked turtles (*Chelodina longicollis*) in the dam.

Accompanying research also investigated the properties and treatment of an adjacent adit contaminated with extensive AMD using the same method (Clark et al., 2001). Similarly, research conducted on acidity and major acidic cations contained within the tailings showed that TAA and TPA were present in water-soluble forms, thereby making treatment of the actual solid tailings imperative (Lin et al., 2003). For example, data showed the submerged tailings were a classifiable “hazardous waste” under the then NSW Environmental Protection Agency’s guidelines for assessment, classification and management of liquid and non-liquid wastes.

Prior to treatment with Terra B reagent, submerged tailings had a pH of 3.8, and total metal concentrations of cadmium 17 mg/kg, lead 59 mg/kg, and nickel 25 mg/kg, with leachable metal concentrations of cadmium 0.4 mg/L, copper 6.8 mg/L, iron 18.7 mg/L, lead 4.2 mg/L, nickel 2.1 mg/L, and zinc 42.2 mg/L. As a result of treatment with Terra B reagent, pH was increased in the tailings to 7.3, and leachable metals became largely non-bioavailable, with leachable cadmium <0.001 mg/L, copper 0.15 mg/L, iron 3.2 mg/L, lead <0.01 mg/L, nickel <0.01 mg/L, and zinc 2.1 mg/L. Therefore, the submerged tailings stored in the tailings dam were reclassified as an “inert solid” under NSW EPA guidelines.

Revegetation of the exposed tailings beach since 2000 has been even more dramatic. Figure 15 shows the tailings beach before it was treated. As shown in Figures 16 and 17, the beach has now been completely populated with bulrush (*Scirpus lacustris*) and common reed (*Phragmites australis*), and parts of the beach had been populated with bottlebrush and goldenwattle, and the beach has become home to reptiles, such as the spotted python (*Antaresia maculosa*) as shown in Figure 16, and visiting ducks, spoonbills and grebes.

Conclusion

From these findings it can be concluded that Terra B and Acid B Extra have largely been effective in rehabilitating waste rock, submerged tailings and an exposed tailings beach at the Mt Carrington mine site in Australia. Programme #1 showed that Terra B was more effective in treating waste rock than either lime and biosolids, standard clay-topsoil capping, or doing

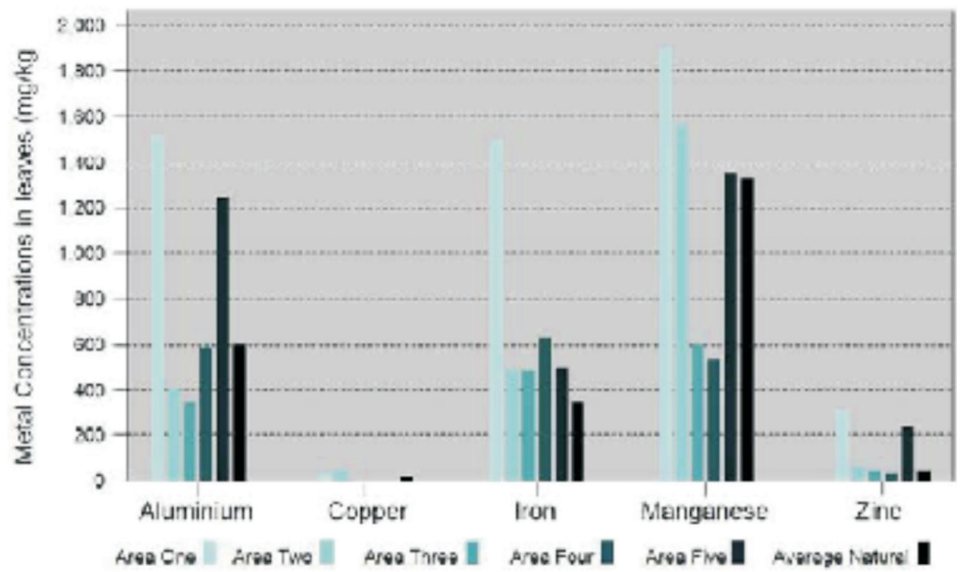


Figure 11: Metal concentrations in leaves as a result of uptake in Programme #2.

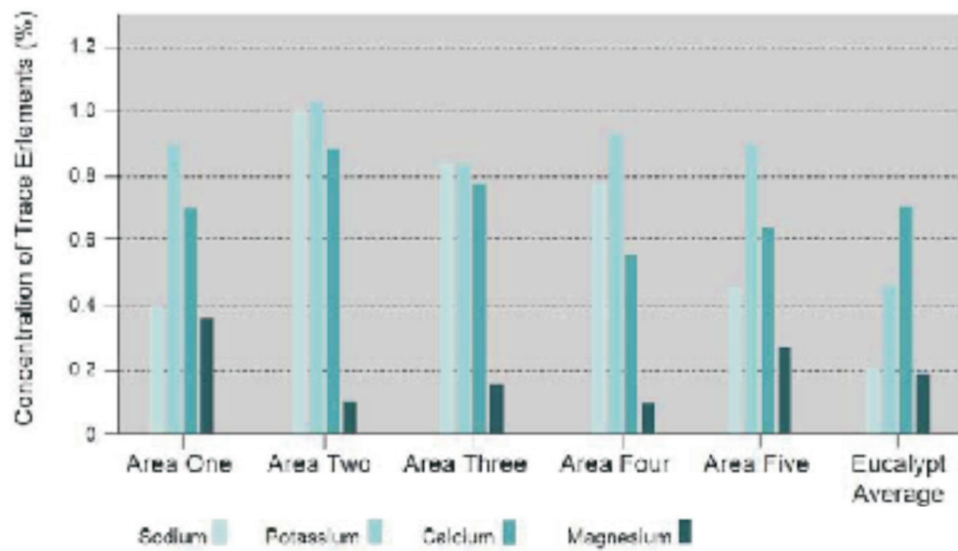


Figure 12: Trace element concentrations in leaves as a result of uptake in Programme #2.



Figure 13: Examples of revegetated Areas Two, Three, Four and Five, 2013 in Programme #2.



Figure 14: Close-up photograph of treated tailings (left) in 2000; close-up photograph of trout in treated tailings dam (right).



Figure 15: Examples of exposed tailings beach immediately after treatment but before revegetation in 2000.



Figure 16: Two examples of revegetated exposed tailings beach in 2013.



Figure 17: Close-up examples of bottlebrush growing on revegetated tailings beach in 2013 (left); spotted python on revegetated tailings beach in 2013 (right).

nothing in raising soil pH, reducing or eliminating both actual as well as potential acid forming minerals, reducing heavy metal leachates, and promoting tree growth over a 14-year period. This finding was largely replicated in Programme #2, which showed that Terra B raised soil pH of acidic waste rock, significantly reduced the uptake of heavy metals and trace elements in the leaves of trees grown in the Terra B-treated waste rock, and promoted tree growth in revegetated areas over the same 14-year period.

Similarly, as shown in Programme #3, Terra B and Acid B Extra were effective in treating both submerged mine tailings and exposed tailings at this mine site, and was successful in rehabilitating the exposed tailings beach to a point where plants and animals now treat the revegetated beach as home. These various elements of the rehabilitation programme at Mt Carrington have contributed to our overall understanding of the sustainable management and revegetation at this metaliferous mine site in Australia, and thereby serve as a workable example of long-term rehabilitation in mine site closure.

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