CSIRO PUBLISHING
Soil Research, 2019, **57**, 419–446
https://doi.org/10.1071/SR18348

Review

Soil quality and vegetation performance indicators for sustainable rehabilitation of bauxite residue disposal areas: a review

E. Di Carlo[A], C. R. Chen[B], R. J. Haynes[C], I. R. Phillips[D], and R. Courtney[A,E]

[A]Department of Biological Sciences and The Bernal Institute, University of Limerick, Ireland.
[B]Australian Rivers Institute and School of Environment and Science, Griffith University, Nathan, Qld 4111, Australia.
[C]School of Agriculture and Food Sciences/CRC CARE, The University of Queensland, St Lucia, Qld 4072, Australia.
[D]Department of Agriculture and Fisheries, Toowoomba, Qld 4350, Australia.
[E]Corresponding author. Email: ronan.courtney@ul.ie

Abstract. The generation of bauxite residue, the by-product of alumina manufacture from bauxite ore, has increased to a global stockpile of some 3 billion tonnes. In the absence of significant reuse options, the bulk of this residue is contained within bauxite residue disposal areas (BRDAs), which can occupy a significant footprint and pose potential environmental risk. Rehabilitation (amendment and vegetation establishment) is viewed as a significant strategy for eventual closure of the BRDAs. Major limitations to plant growth in residue include high pH, salinity, and sodicity, as well as deficiencies of macro- and micronutrients and potentially elevated levels of trace elements. The physical properties are also problematic as residue mud consolidates to form a solid mass that waterlogs easily or dries to form a massive structure, whereas sand has a very low water- and nutrient-holding capacity. A variety of techniques have been trialled at the pot level and at the field scale to bring about reductions in residue alkalinity and sodicity to promote plant establishment, with gypsum amendment viewed as the most promising. Other amendment strategies include use of organic additions or fertiliser applications, and a combined approach can lead to improved residue properties and successful plant establishment. Few reports have focused on longer term plant growth, self-propagation, and residue interactions under field conditions. There is some evidence that rehabilitated residue can support vegetation growth and soil development in the short to medium term (~15 years), but key issues such as nutrient availability and plant uptake require further study. Although rehabilitated residue can support diverse microbial communities and demonstrate trajectory analogous to soil, the ability of rehabilitated residue to support soil biota and key ecosystem processes warrants further study. The bioavailability of trace elements within rehabilitated sites and potential food chain transfer are relatively unexplored. These areas need careful study before definitive statements can be made regarding the sustainability of residue rehabilitation strategies.

Additional keywords: alumina refining, gypsum, organic wastes, red mud, revegetation, salinity, sodicity, tailings.

Received 29 November 2018, accepted 12 March 2019, published online 11 June 2019

Introduction

Significance of bauxite residue disposal areas (BRDAs)

Bauxite residue is the by-product generated by the extraction of alumina from bauxite ore via the Bayer process (i.e. using concentrated sodium hydroxide at high temperature and pressure) and is typically highly alkaline, saline, and sodic and may contain trace elements at elevated levels (Power *et al*. 2011). The alumina industry produces ~150 million tonnes of bauxite residue each year with an estimated global stockpile of 3 billion tonnes (Evans 2016). Despite multiple attempts to utilise bauxite residues in the construction, environmental, mining, and agronomic industries, only 2–3% of this material is currently reused or further processed (Klauber *et al*. 2011; Evans 2016; Ujaczki *et al*. 2018). Consequently, almost all bauxite residue is stored in land-based BRDAs (Burke *et al*. 2013), with many facilities reaching several hundred hectares in footprint (Residue Solutions 2007; Cristol and Greenhalgh 2018). Unless properly managed, BRDAs pose a high risk to the environment and surrounding communities (Mayes *et al*. 2011; Ruyters *et al*. 2011; Burke *et al*. 2012; Olszewska *et al*. 2016; Ren *et al*. 2018).

Recent developments have seen the industry move away from wet disposal of residue in ponds and lagoons towards dry stacking methods. Best practice in the alumina industry has evolved to employ management steps that produce residue with higher solids content, lower moisture, and recovered alkalinity through technologies such as filter presses or mud farming (amphirolling) to create a compacted and consolidated residue with reduced alkalinity (Clohessy 2015; Higgins *et al*. 2016; Arslan *et al*. 2018). Modern storage facilities are often engineered to minimise leakage of leachate from the residue mass by including a low permeability (saturated hydraulic conductivity <10^{-9} m/s) geomembrane base liner constructed from high-density polyethylene and/or compacted bentonite clay. Drainage systems installed above the liner can recycle drainage back to the refinery as a means of closing the water-balance and capturing high-strength alumina solutions before release to the environment. Despite these mitigation procedures, uncontrolled discharges from stored residues still occur, highlighting the ever-present risks to aquatic and terrestrial systems (Olszewska *et al*. 2016; Ren *et al*. 2018) and the need for effective management strategies, both during the active lifespan of a BRDA and post-closure.

Rehabilitation approaches used – capping or direct revegetation

It is generally accepted that establishing a sustainable vegetation cover (revegetation) is one of the most important steps in progressive and final closure of ore-processing residues and tailings surfaces (e.g. Tordoff *et al*. 2000; Ye *et al*. 2002). A vegetation cover provides not only a method of maintaining structural integrity of the engineered facility, but also an improved visual environment and the initiation of a new ecosystem for local flora and fauna. Like many ore-processing residues, successful rehabilitation of bauxite residues is underpinned by not only establishing the most suitable vegetation species, but, also importantly, by developing the most appropriate growing medium. In many cases, where good quality soil (topsoil or subsoil) is not available, treatment of the residues using various amendments and/or fertilisers may be required. In this respect, successful rehabilitation is a combination of both remediation of the growing medium and selection of the plant species most suited to the environmental conditions, combined with management of the chemical, physical, and microbial properties of the growing medium (Phillips 2014*a*, 2015*a*).

The approach to rehabilitation often follows one of two main strategies, these being: (1) using a surface cover of soil or soil-like material to provide a plant growth medium, or (2) improving the physical and/or chemical properties of the residue using amendments (e.g. gypsum, biosolids, or compost) (Jones and Haynes 2011) followed by direct vegetation of the tailings (residues) surface (Tordoff *et al*. 2000). Of these two generalised strategies, capping using introduced cover materials that effectively separate the growth medium (hence effective plant rooting depth) from the underlying residue is more often recommended for tailing storage facilities housing materials with extreme properties, such as hypersaline, highly acidic, or highly alkaline metalliferous residues (Tordoff *et al*. 2000; Wehr *et al*. 2005, 2006; Santini and Fey 2016).

For materials exhibiting less extreme characteristics (more consistent with soil-like properties), planting directly into the ameliorated, amended residue surface can be a more suitable alternative (Mendez and Majer 2008). However, irrespective of the method implemented, revegetation of mine tailings represents one of the most cost-effective and efficient methods of reducing the global environmental risk. This is particularly so where the rehabilitated area develops into a functional and sustainable ecosystem. Of the many properties of bauxite residue, the initially high concentrations of soluble alkalinity, coupled with the prolonged slow release of residual alkalinity, can represent a major challenge in sustainable rehabilitation of BRDAs (Meecham and Bell 1977*a*, 1977*b*; Phillips 2015*a*; Xue *et al*. 2016). Combined remediation and vegetation (or revegetation) of the residue surface represent one of the most promising strategies in bauxite residue rehabilitation (Jones and Haynes 2011).

Depending on the bauxite ore used and operational parameters within the Bayer process, residues may be dominated by a coarse-textured fraction (residue sand) or a fine-textured residue (red mud) (Courtney and Timpson 2005*b*; Wehr *et al.* 2006; Banning *et al.* 2010; Buchanan *et al.* 2010; Phillips 2010*a*; Gräfe *et al.* 2011). Different fraction residues are often separated and disposed separately and, in some cases, may be codisposed. The residue sand can be used in the construction of BRDAs (e.g. Banning *et al*. 2010) and becomes the primary growth medium in progressive rehabilitation of the outer embankment surface (e.g. Phillips 2010*a*). The major limitations to sustainable plant growth in residue sand typically include low water holding capacity, high alkalinity (soluble and residual), salinity and sodicity, negligible organic C, and very low nutrient supplying capacity. However, the use of inorganic (e.g. gypsum) and organic (e.g. compost, manure, or biosolids) amendments, coupled with inorganic fertiliser can markedly improve the physical, chemical, and microbial properties of residue sand in a relatively short timeframe (i.e. <10 years).

Finer residue materials (e.g. residue mud) also exhibit similar chemical restrictions to residue sand; however, the magnitude of these limitations is much greater due to the inherent greater micro-porosity and residual entrapped alkalinity (Gräfe *et al.* 2011). As such, bauxite mud is typically highly alkaline (pH 9.7–12.8), hypersaline (electrical conductivity 1–24 dS m^{-1}), sodic (exchangeable sodium percentage 32–91%), and can exhibit poor hydraulic properties (Gräfe *et al.* 2011). The high alkalinity is recognised as one, if not the major, limitation for establishing plant growth in residue mud, and although many techniques have been sought to address this problem (e.g. seawater treatment) (Menzies *et al*. 2009) or carbonation (Cooling *et al*. 2002), none has successfully negated the detrimental impacts of slow release of residual alkalinity over prolonged periods of time (Meecham and Bell 1977*a*, 1977*b*; Phillips 2015*a*; Xue *et al*. 2016). This release ensures residue pH can be well buffered around pH >10 for many years unless amendments are added. Nonetheless, long-term changes in properties do occur, especially under high rainfall and, with time, the material may begin to revegetate via natural seed dispersal (e.g. Khaitan *et al.* 2010; Santini and Fey 2013). For a more detailed discussion on the properties of residue, and their limitations to plant growth, see reviews by Gräfe and Klauber (2011), Gräfe *et al.* (2011) and Xue *et al.* (2016).

Rehabilitation performance criteria

Despite the numerous studies that have been undertaken to understand the properties of residue and identify strategies that allow a vegetation cover to be established in this 'plant-hostile' material, procedures to monitor and evaluate rehabilitation performance are scarce in the published literature. Rehabilitation performance can be evaluated based on a set of well defined, readily-measurable parameters related to the soil–water–plant system. Based on standard soil quality criteria guidelines, Gräfe and Klauber (2011) highlighted the 'rehabilitation goals' for several key properties in bauxite residue that should be attained for plant growth promotion. These soil physico-chemical parameters are summarised as:

- a steady-state residue pH between pH 5.5 and pH 9.0,
- an electrical conductivity (EC) of <4 dS m^{-1},
- a sodium adsorption ratio of <7 and an exchangeable sodium percentage (ESP) of <9.5,
- a residual sodium carbonate value of <1.25 and
- a bulk density of ≤ 1.6 g cm^{-3}.

Haynes (2015) suggested less stringent minimum targets before planting or seeding into bauxite residue sand and mud of pH < 8.0, EC < 5.0 dS m^{-1} and ESP <40%. There are other important factors that need to be considered. The capacity to supply the required concentrations of macronutrients (N, P, K, Ca, and Mg) and micronutrients (Cu, Zn, Mn, Fe, and B) in plant-available forms is critical (Gräfe and Klauber 2011) so guideline soil test values are required. Furthermore, bioavailable concentrations of Al and heavy metals and metalloids should be below the toxic threshold levels for plants, soil microbes, and grazing animals, and such guideline values will also be required. With the important role of soil biota in soil health, and as a mechanism for driving *in-situ* remediation, the establishment and long-term survival of diverse, functional microbial and soil faunal populations will be a critical component of residue quality and ecosystem sustainability (Shu *et al.* 2005; Biederman *et al.* 2008).

Mendez and Maier (2008) proposed that evaluation of successful revegetation of mine tailings can be divided into several criteria: (1) plant (biomass percentage cover comparable with that in uncontaminated soil, self-propagation of introduced species, establishment of native colonisers, acceptable shoot metal concentrations, and plant performance maintained for >10–29 years); (2) microbial (increases in heterotrophic bacterial and fungal communities); and (3) soil (e.g. soil aggregation improved, erosion and runoff reduced, and metal bioavailability decreased). Larson and Pierce (1991) proposed five soil quality indicators and suggested that the combined physical, chemical, and biological properties of a soil enable it to perform three functions. These are to (1) provide a medium for plant growth, (2) regulate and partition water flow through the environment, and (3) serve as an environmental filter. They defined soil quality as how effectively soils (1) accept, hold, and release nutrients and other chemical constituents; (2) accept, hold, and release water to plants, streams, and groundwater; (3) promote and sustain root growth; (4) maintain suitable soil biotic habitat; and (5) respond to management and resist degradation. The above criteria and definitions are applicable for assessing the quality of bauxite residue as a growth medium for rehabilitation.

Sustainability indicators of rehabilitation of BRDAs

Soil quality indicator – physical properties

Texture and structure

Adequate physical structure and hydraulic functions across the whole root zone are essential in the reconstructing of the profile in ore processing residues (tailings) (Huang *et al*. 2012). Plant establishment on bauxite residues can be subject to severe physical restrictions. For example, root architecture and depth of penetration can be severely restricted under the increasing compaction (bulk density) with profile depth (Fig. 1; Dobrowolski *et al.* 2009).

The textural characteristics of ore processing residues (tailings) have profound impacts on the geochemical dynamics in the tailings profile, as they determine pore sizes and the diffusion and infiltration of water and oxygen. Indeed, the physical properties of residues differ dramatically depending on whether they are residue sand or residue mud. This is particularly so as the deposit dries out. The sand fraction remains as sand after drying. By contrast, the mud fraction solidifies into solid massive structure after drying. For residue sand, the combination of high permeability with low water retention results in severe water stress, which is manifested during periods of low rainfall, particularly when this coincides with summer (e.g. south-west Western Australia's Mediterranean-type climate) (Banning *et al.* 2014).

To improve the physical and chemical properties of residue sand for supporting vegetation cover, there have been attempts to add fines back to the sand (Buchanan *et al*. 2010; Anderson *et al*. 2011; Jones *et al*. 2011; Banning *et al*. 2014). Anderson *et al*. (2011) and Jones and Haynes (2010, 2012*b*) found the combination of residue sand with residue fines improved the water and retention properties relative to residue sand alone. However, the addition of residue fines can have some adverse impacts on the physical properties (e.g. increased penetration resistance) (Buchanan *et al.* 2010).

The first phase of rehabilitation is to aid drainage in the residue to a depth of many metres to remove excess moisture

Fig. 1. Roots of *Eucalyptus gomphocephala* growing sideways with a sharp bend at the compaction layer evident at the depth of incorporation of gypsum in residue sand.

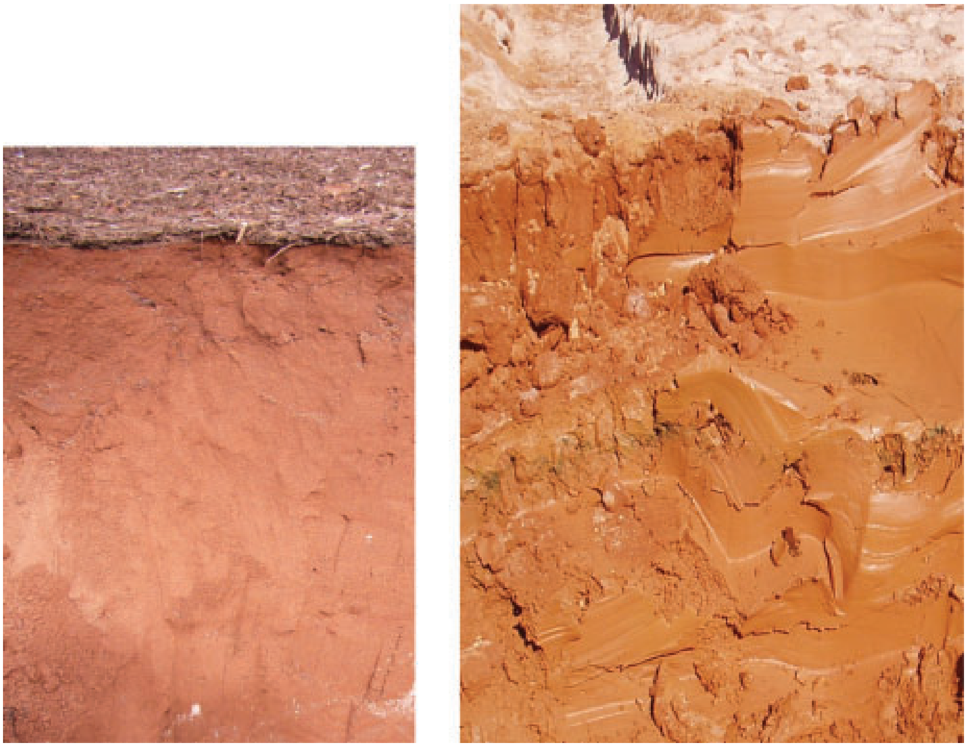

Fig. 2. Contrasting physical properties of residue sand (left) and residue mud (right).

and soluble phase alkalinity. Indeed, freshly deposited residue mud exhibits bulk densities ranging from 1.6–3.5 g cm^{-3} and very low saturated and unsaturated hydraulic conductivity. However, over a relatively short timeframe (>10 years), seasonal rains (summer in tropical climates and winter in temperate ones) coupled with extended drying conditions encourage improved physical properties of residue such as matrix consolidation (Nikraz *et al.* 2007), formation of water-stable aggregates (Utomo and Dexter 1982), and formation of cracking patterns (Figs 2, 3). The mud fraction which is deposited as a structureless pastey matrix forms a massive dense structure after drying (Wehr *et al.* 2005). That is, upon drying, the mud solidifies and loses substantial volume causing cracks and macropores to form to depth in the residue deposit (Fig. 3). Solidification occurs because of pozzolanic materials that are present in the residue, which bind the material together the first time it dries (Li *et al.* 2016). The cracks and surface-connected macropores allow for water movement and aeration to depth as well as forming channels for downward root growth. Water movement is therefore greatly increased (particularly under saturated flow conditions: K_{sat} (saturated hydraulic conductivity) $\approx 10^{-5}$ m s^{-1}) compared with 'fresh' (non-cracked) residue ($K_{sat} \approx 10^{-9}$ m s^{-1}) (Phillips 2010*a*). The contribution of macropores on water flow rates will only extend to the depth of cracking and how well the cracking is connected to produce continuous flow paths (Horn *et al*. 1994).

Over time the solidified material breaks into smaller peds (aggregates) and the surface layer can be ripped and tilled to form a stable tilth suitable for plant establishment and growth. A fine fraction of solid material is formed during tillage and natural cleavage of peds. Some of the aggregates in revegetating residue originate from breakdown of the original solidified structure and some from aggregation of the fine fraction (Khaitan *et al*. 2010; Li *et al.* 2018*a*, 2018*b*).

Some workers have added residue sand back to mud (Courtney and Timpson 2005*b*; Courtney *et al*. 2009*a*). The addition of sand at ~10–25% incorporated into the surface 30 cm has been found beneficial during revegetation of residue mud (Courtney and Timpson 2005*b*). It is believed that blending can improve the hydraulic properties of mud, thereby encouraging leaching with the concomitant removal of soluble salinity and alkalinity and encouraging the dissolution of solid alkalinity formed during the Bayer process.

With both types of residues, dust production from the surface can be a problem since particles in the <0.5 mm diameter range are particularly prone to movement by wind. Although much of the visible material can be due to sodium carbonate precipitated at the residue surface (Fig. 4), soil or residue particles are still prone to wind erosion under dry, windy conditions. In many cases this can be overcome by cover systems such as water sprinklers, wood mulch, pasture, and rehabilitation (Fig. 5).

Widely used parameters for assessing soil physical quality such as bulk density, porosity, water retention, and water stable aggregation are becoming more widespread in assessing restoration of mine wastes (Shukla *et al.* 2004; Asensio *et al.* 2013), and it follows that assessment of bauxite residues should include such parameters. Indeed, several workers have examined the physical properties of residue sand or mud and the effects of age of the residue deposit and or effect of amendments on physical properties (Khaitan *et al.* 2010; Santini and Fey 2013; Courtney *et al*. 2013) (Fig. 6).

Fig. 3. Surface cracking of residue mud as it progressively dries, with evidence of cracking within the profile.

Fig. 4. Sodium carbonate forming on residue surface.

Water potential

Residues dominated by the sand fraction have very poor water-holding capacity and can be subjected to regular and prolonged surface drought. For example, in contrast to residue mud, the sand fraction can exhibit saturated hydraulic conductivities up to 40 m day^{-1} (Philips 2010*a*; Gwenzi *et al*. 2011), but its well-graded sand texture results in very poor water-retention characteristics (Phillips 2010*a*). This makes direct revegetation difficult because of severe limitations to germination and seedling establishment. For example, poor seedling emergence of Rhodes grass (*Chloris gayana*) in residue sand was attributed to low water retention (Meecham and Bell 1977*b*). Banning *et al*. (2014) found that amendment of residue sand with residue mud or compost increased water retention at matric potentials between field

Fig. 5. Examples of surface covers in bauxite reside progressive closure: wood mulch, rehabilitation and pasture.

capacity and the wilting point. A combination of residue mud and compost provided the greatest improvement in plant water availability. Investigations by Anderson *et al*. (2011) also found that water retention increased linearly with increasing percentage of residue fines at all measured water potentials, with an up to 40% increase in plant available water when residue fines were added at 20%. Jones *et al*. (2011) showed that addition of a combination of residue mud and poultry manure to residue sand resulted in substantial aggregation of sand particles and that the stability of aggregates was greatest when poultry manure was applied.

Bulk density

The bulk density of unamended residue mud is high, with average values of 2.5 g cm^{-3} being reported (Gräfe and Klauber 2011) although lower values (1.56–1.91 g cm^{-3}) have been found for non-vegetated residues (Courtney *et al*. 2009*a*; Zhu *et al*. 2016*a*). Amendment of residue mud with gypsum can cause a small decrease in bulk density, but much greater decreases have been reported with application of organic amendments (e.g. sewage sludge and spent mushroom compost) (Wong and Ho 1994*a*; Courtney *et al*. 2009*a*). At high rates of addition, decreases of 25–29% have been reported (Courtney *et al*. 2009*a*). Such decreases in bulk density are attributable to a dilution effect caused by mixing of the added organic material (with a low bulk density) with the denser mineral fraction of the residue. Weathering processes and plant colonisation of residues (with addition of organic matter in the form of plant residues) over a 20-year period also decreased bulk density to 1.39 g cm^{-3} (Zhu *et al*. 2016*b*).

Aggregate size and stability

Other physical properties also change over time. Zhu *et al*. (2016*b*) studied the physical properties of unamended residue mud with increasing age. They showed that natural plant colonisation with age (and decreased pH, EC, and exchangeable Na of residues) resulted in increased water stable aggregation and decreased erodibility. Zhu *et al*. (2016*c*) used X-ray micro-computed tomography to show that, with age, there was an increase in porosity, specific surface area, average length of paths, and average tortuosity of paths. Zhu *et al*. (2018) showed that with increasing deposit age, the dispersiveness of aggregates decreased and there was an increase in the size of the aggregate fraction (250–50 µm) with a concomitant decrease in silt and clay size fractions.

A change in aggregate size and stability has also been noted during revegetation where amendments have been applied. Zhu *et al*. (2017) showed addition of vermicompost had a positive effect on the formation and stabilisation of water stable aggregates in residue mud. Following amendment and seeding and after 1 year of plant growth, Courtney *et al*. (2009*a*) recorded dominance of less erodible, more stable aggregates in treatments with high application rates of gypsum and organic amendments. The positive effect of gypsum was demonstrated by a substantially smaller quantity of dispersed clay-sized particles in comparison with amended residues with lowered ESP values (Fig. 7). Courtney *et al*. (2013) employed a standard reclamation approach (additions of a combination of gypsum and organic residues) and found improved macroaggregate (>250 µm) stability following 1 year of plant growth. Further improvements were recorded in treatments established 9 and 11 years previously, and aggregate stability in these treatments

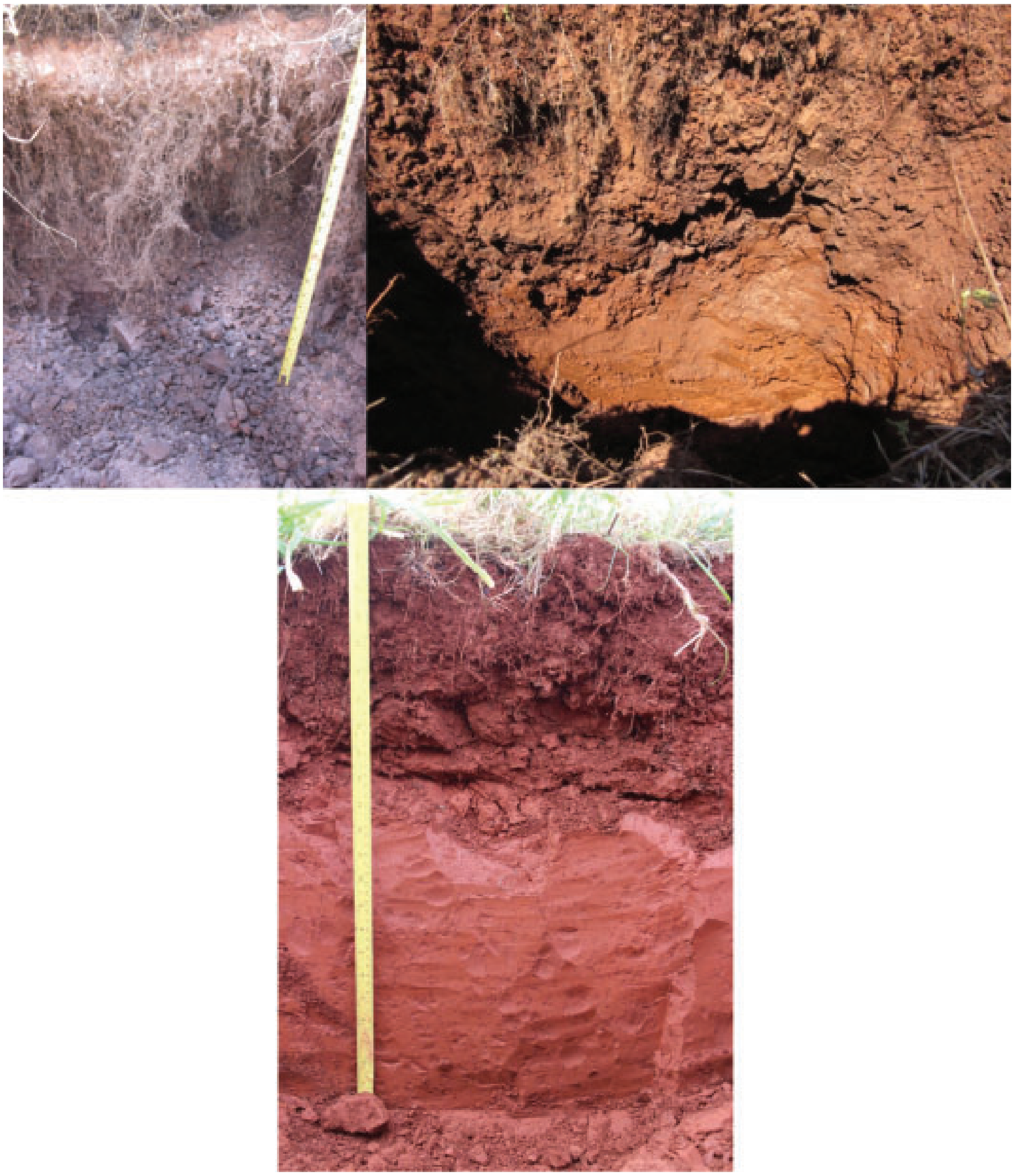

Fig. 6. Soil development of residue within the rootzone. Note structure in upper profile and typical residue material in lower profile.

was similar to that for an analogous soil. Such findings highlight the importance of plant roots (and associated rhizosphere microorganisms) as the driving factor in the formation and stabilisation of structure, as is well recognised within the wider soil sciences (Haynes 1999; Hallett *et al.* 2009).

Accumulation of soil organic matter content

Organic matter is necessary for soil functioning because it performs many important functions including (1) increasing water retentive capacity; (2) forming a stable soil structure; (3) increasing cation exchange capacity; (4) supplying nutrients (e.g. N, P, and S) through mineralisation; and (5) providing a source of nutrients and energy for soil microorganisms and many soil fauna (Stevenson 1994). The organic matter content of bauxite residue is extremely low. Small amounts of residual sodium oxalate are present which formed during caustic degradation of humic material associated with the bauxite ore (Jones *et al.* 2010). It should be noted here that the inorganic C content (HCO_3^- or CO_3^{2-}) of bauxite residue is significant and to obtain an accurate measure of its organic C content, particularly when it is first deposited is difficult and can be highly erroneous. It is necessary to react the residue with acid to remove inorganic C before total C analysis (Bray *et al.* 2018; Li *et al.* 2018a).

Because of its low organic C content, additions of organic residues to bauxite residue can be very beneficial for revegetation. Many workers have observed that additions of

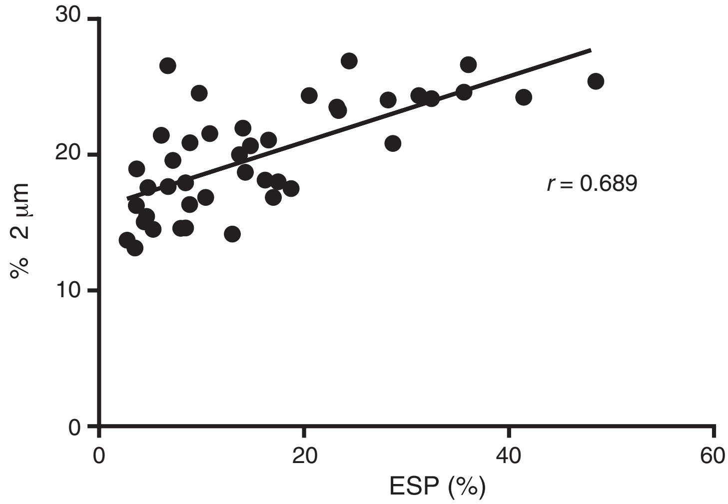

Fig. 7. Relationship of particles less than 2 microns in residue of varying ESP value (from Courtney *et al*. 2009*a*; Courtney and Harrington 2012).

various organic amendments (manures, biosolids, or composts) to bauxite residues can greatly improve plant growth both in greenhouse and field studies (Fuller *et al.* 1982; Fuller and Richardson 1986; Xenidis *et al.* 2005; Courtney *et al.* 2009*a*; Courtney and Harrington 2012; Jones *et al.* 2012*a*, 2012*b*; Li *et al.* 2018*a*, 2018*b*). Such effects are normally principally attributable to increased macronutrient supply (N, P, K, Ca, and Mg) but also to decreased pH and exchangeable Na and improved physical properties. Irrespective of whether organic residues are added or not, the organic matter content of bauxite residues will naturally increase with time during revegetation. As plants become established and grow, organic matter inputs from deposition of above- and belowground plant litter increase contributing to the formation and accumulation of soil organic matter. For example, Zhu *et al.* (2016*b*) found that over a 20-year period of natural regeneration, soil organic C content increased from 5.7 to 10.8 g kg^{-1}. As organic residues are added to the residue (as organic manures and/or plant litter) microbial decomposition will occur with the formation of metastable humic material. As a result, with time there is transfer of organic matter from particulate forms (<200 mm diameter) to the <53 µm diameter size class, which represents humic C bound to the mineral component (Courtney *et al.* 2013).

Soil quality indicator – soil chemical properties

Soil pH

Bauxite residue pH values are typically high (9.7–12.8) (Gräfe and Klauber 2011), which result from entrained caustic Bayer liquor. The main alkaline anions buffering the solution are OH^-, CO_3^{2-} or HCO_3^-, and $Al(OH)_4^-$ (Gräfe *et al.* 2011). Depending on residue management steps employed at refineries, the pH may be decreased by partial neutralisation. Practices such as carbonation with CO_2 (Nikraz *et al.* 2007), seawater treatment (Menzies *et al.* 2004, 2009), and atmospheric carbonation with amphirolling (mud farming) (Clohessy 2015) have all been applied and resulted in decreased residue pH. Natural weathering can also decrease the pH, as evidenced by Khaitan *et al.* (2010) who reported a pH of 10.5 for 14-year-old residue and pH 9.5 for 35-year-old residue. Initial leaching of the material results in the removal of soluble anionic alkalinity (HCO_3^-, CO_3^{2-}, or $Al(OH)_4^-$) as counterions for the major cation present (Na^+) and, therefore, a decrease in residue pH (Jones *et al.* 2012*a*). Further decreases occur due to slow carbonation from atmospheric CO_2. Similarly, Zhu *et al.* (2016*a*) reported a decrease in residue pH from 11.0 in 1-year-old residue to 9.5 in 40-year-old residue deposits. Consequently, initial pH values for residue undergoing revegetation vary. It is generally recognised that high pH is one of the main properties inhibiting plant growth (Courtney and Mullen 2009). This is because very high pH, with high concentrations of OH^-, CO_3^{2-}, and HCO_3^- in solution, can inhibit root growth and function (Gupta and Abrol 1990; Kopittke and Menzies 2004) and high concentrations of aluminate (present at high pH) are also phytotoxic (Fuller and Richardson 1986; Brautigan *et al.* 2012).

Although the rehabilitation objective may be to achieve a pH of ~8, the amendments used and quantities applied vary widely. The three main *in situ* neutralisation techniques used are (1) addition of an acidifying agent such as iron sulfate or elemental S; (2) addition of an amendment with a high buffering capacity that possesses a pH lower than that of the residue (e.g. composts, manures, or biosolids); or (3) application of gypsum ($CaSO_4.2H_2O$). In addition, some refineries treat their residue with seawater before its deposition in storage areas.

Seawater neutralisation

Seawater neutralisation has been promoted as a method of converting bauxite residues into a relatively benign material (Hanahan *et al.* 2004), and several bauxite refineries have installed plants to treat their residue before deposition in BRDAs (Li and Haynes 2017). During seawater treatment, the pH of residue is reduced from 11–13 down to ~9.0 and the exchangeable Na percentage is also reduced (by addition of

Ca, Mg, and K in the seawater) (Hanahan *et al.* 2004; Menzies *et al.* 2004; Palmer *et al.* 2009). Neutralisation occurs because of precipitation of soluble alkalinity (HCO_3^- or CO_3^{2-}) as sparingly soluble Ca and Mg hydroxides and hydrocarbonates (particularly hydrotalcite) (Hanahan *et al.* 2004). However, Menzies *et al.* (2009) investigated freshwater leaching of seawater neutralised residue and recorded an increase in leachate pH from 8.0 to 10.1 with increased leaching. High pH was also recorded for soil solution pH, and it was concluded that the residue media would be inhibitory to plant growth. Other workers have also noted a pH increase during leaching of seawater-neutralised residue (Li *et al.* 2016), and this is attributable to dissolution of residual alkalinity present in the residue. This may include solid phase alkalinity precipitated during seawater neutralisation as well as other compounds produced during Bayer digestion (e.g. tri-calcium aluminate (TCA), cancrinite, and sodalite) (Menzies *et al.* 2009; Li *et al.* 2016). Thus, although seawater neutralisation reduces the residue pH down to ~10, for effective revegetation, addition of amendments (e.g. gypsum and/or organic wastes) is still necessary to reduce it down to 8 (Li *et al.* 2018*a*, 2018*b*).

Sulfur-containing acidifying agents

Iron sulfate has been successfully used as an acidifying agent in several laboratory studies (Wong and Ho 1994*b*; Xenidis *et al.* 2005; Jones *et al.* 2015) (Table 1). It undergoes rapid hydrolysis with the release of SO_4^{2-} and H^+ ions and precipitation of hydroxy-Fe, and it is a highly effective acidifying agent. Elemental S has also been used successfully by some workers (Meecham and Bell 1977*a*) but unsuccessfully by others (Ippolito *et al.* 2005; Jones *et al.* 2015). Elemental S is oxidised primarily by bacteria of the genus *Acidithiobacillus*, and the activity of these microorganisms dictates the release of SO_4^{2-} and H^+ ions. Both Ippolito *et al.* (2005) and Jones *et al.* (2015) suggested that a lack of S-oxidising bacteria in the residue limited the effectiveness of

Table 1. Residue pH and effects of amendments in pot-based trials
Abbreviations: SS, sewage sludge; G, gypsum; Ww, wood waste

pH Initial	Amendment	Final pH	Comment	Reference
(SE)				
10.5	0% G	10.5	Leached with equivalent 126 mm rainfall	Wong and Ho (1993)
	2% G	9.98		
	5% G	8.68		
	8% G	8.56		
10.5	0 copperas ($FeSO_4$)	10.5	Leached with equivalent 126 mm rainfall	Wong and Ho (1994*b*)
	2% copperas	10.1		
	5% copperas	9.17		
	8% copperas	9.22		
10.2	SS 8%	10.16	Leached with equivalent 126 mm rainfall	Wong and Ho (1994*a*)
	SS 16%	9.97		
	G 8%	8.1		
	G 8% + SS 8%	8.38		
	G 8% + SS 16%	8.31		
11.9	2% G and		Leached with equivalent 396 mm rainfall	Jones *et al.* (2010)
	G	8.1		
	Biosolids	7.4–7.7		
	Mushroom c	8.1		
	Green waste	8.1–8.2		
	Biochar	8.0–8.1		
9.1	Residue sand and G at 2% (A)	8.6	Seven leaching events with typical monthly rainfall	Jones *et al.* (2012*a*)
9.0	A and red mud (B) added	8.7		
9.3	A and poultry manure (C) added	8.6		
9.1	B and C added	8.6		
		(1 : 2)		
11.9	Compost at 0%	10.5	Irrigated but not leached	Fortin and Karam (1998)
	18%	9.8		
	23%	10.1		
	29%	9.5		
	31%	9.4		
	37%	9.3		
	45%	9.4		
(1 : 1)		(1 : 2)		
10.0	Unamended	9.3	Leached with 8 pore volumes	Ippolito *et al.* (2005)
	G	8.5		
	Ww	8.6		
	G and Ww	8.1		

elemental S. Neither iron sulfate nor elemental S appears to have been used in field experiments, presumably because of the effectiveness of gypsum as a field acidifying agent (see below).

Organic amendments

In laboratory-based studies, a range of workers have demonstrated a modest reduction in residue pH induced by addition of organic amendments such as compost or biosolids (Wong and Ho 1994*a*; Fortin and Karam 1998; Xenidis *et al.* 2005; Jones *et al*. 2010, 2011, 2015; Li *et al.* 2018*a*, 2018*b*) (Table 1). The intrinsically lower pH of these organic wastes compared with bauxite residue, their high buffering capacity associated with humic materials in the wastes, and sometimes the high N content (with ensuing mineralisation and nitrification) all contribute to these pH decreases. Where they have high Ca and Mg content, precipitation of soluble alkalinity may also occur. As noted below, addition of such materials also adds nutrients to the residue and may help in improving physical properties. The effectiveness of organic amendments in decreasing pH has also been observed in field experiments (Courtney and Timpson, 2005*a*, 2005*b*; Courtney *et al.* 2009*a*). For example, Courtney *et al*. (2009*a*) noted that 1 year after application of spent mushroom compost at 120 t ha^{-1} the pH had decreased from 9.6 to 8.4.

Gypsum

The most common amendment used in revegetation of residues is gypsum. A decrease in pH occurs because the added Ca^{2+} in the gypsum reacts with soluble alkalinity (HCO_3^-, CO_3^{2-}, $Al(OH)_4^-$, and OH^-) to form precipitates of calcite, tricalcium aluminate, and hydrocalumite (Gräfe and Klauber 2011). Because residue sand has much less buffering capacity than mud, the application rate of gypsum is typically lower (e.g. 1–2% compared with 3–8% for residue mud). Many laboratory or greenhouse studies have demonstrated substantial decreases in the pH of both residue sand and mud following gypsum application (Wong and Ho 1994*a*, 1994*b*; Xenidis *et al.* 2005; Jones *et al.* 2010, 2011, 2012*a*, 2012*b*, 2015; Li *et al.* 2018*a*, 2018*b*) (Table 1). Comparing applications of gypsum and copperas at same rates, decreases in pH were more effective for gypsum treatments (Table1; Wong and Ho 1994*b*). Applications of gypsum to seawater neutralised residue was also effective. For example, Li *et al*. (2018*a*) recorded a pH decrease in seawater neutralised residue mud from 9.1 to 8.8 with gypsum applied at 1% and a decrease to pH 8.1 with 5% gypsum.

Field experiments have also recorded both short- and long-term decreases in pH induced by gypsum application (Courtney and Timpson 2005*a*; Courtney *et al.* 2003, 2009*a*, 2009*b*, 2013). In a field study, gypsum applied at rates of 0, 40 and 90 t ha^{-1} resulted in pH values of 9.6, 8.3, and 8.0 respectively, after 1 year (Courtney *et al*. 2009*a*). One year after amendment with residue sand and gypsum (3% w/v) pH of gypsum amended plots was 8.0 compared with 8.9 where no gypsum was applied (Courtney and Timpson 2005*a*). Subsequent assessment of the field plots some 6 years later recorded a pH of 8.0 for gypsum-amended residue and a pH of 8.1 for non-gypsum treatments (Courtney *et al*. 2009*a*). In another field study Courtney *et al.* (2013) recorded unamended residue to have a pH of 10.1, which was reduced to 8.2 following gypsum application at 90 t ha^{-1}, and further decreased to pH 7.4 after 9 years. This further decrease may have been due to further leaching, microbial action, and/or production of organic acids in the rhizosphere. The field results suggest that a reduction in pH induced by gypsum applications can be sustained in the long term. Bray *et al*. (2018) found that pH in the surface 10 cm of unamended residue 16 years after initiation of a revegetation trial was 10.8 but it remained at ~8.0 in gypsum-amended plots. In the gypsum treatments, pH had decreased to a depth of 50 cm, well below the 0–10 cm depth of incorporation. Such results suggest that a gypsum-induced decrease in pH is sustained for at least 16 years.

Table 2. pH, electrical conductivity (EC) and exchangeable sodium percentage (ESP) for field-amended residue (gypsum and organic matter and seeded) after Courtney *et al.* (2013)

	Unamended	1 year old	9 years old	11 years old
pH	10.2	8.2	7.4	7.5
EC	3.1	0.96	0.36	0.32
ESP	90.2	22.6	5.4	8.1

Gypsum plus organic matter

The most effective treatment of residues to lower pH is generally considered to be a combination of gypsum plus organic wastes. This has been shown in both laboratory and greenhouse studies (Jones *et al.* 2010, 2011, 2012*a*, 2012*b*; Li *et al.* 2018*a*, 2018*b*) (Table 1) and in the field (Courtney *et al.* 2003, 2009*a*, 2013) (Table 2). Since both materials are acidifying this is not surprising. Both materials have added benefits in relation to reducing sodicity and improving physical properties (see below).

The incorporation of gypsum plus organic matter into the surface horizon may have a greater effect on acidifying the subsurface layers compared with gypsum alone. For example, in an 8-month study, Li (2017) showed that addition of gypsum plus biosolids or cattle manure into the surface 10 cm of residue had a significant interaction in decreasing pH, exchangeable Na, and extractable Al, and increasing exchangeable Ca and grass root growth in the subsoil (10–30 cm) layer compared with surface incorporation of gypsum alone. Such an effect is attributable to downward movement of acidity (e.g. as organic acids) (Santini and Fey 2013) and added Mg in organic manure displacing exchangeable Ca downward.

Microbial activity

The potential for microbial activity to initiate decreases in residue pH was explored by Hamdy and Williams (2001), who treated bauxite residue with hay and yard waste. The resulting mixtures supported growth of microflora such as *Lactobacillus*, *Micrococcus*, *Staphylococcus*, *and Pseudomonas*, and there was a reduction in pH from 13 to 7. At bench scale, acid fermentation through the application of microbial inoculants to alkaline residue (~pH 10–12) resulted in reduction to pH 7–9 over a 16-day test period (Santini *et al.* 2016) and testing at field scale is warranted. Reduction in alkalinity in bauxite residue treatments with fungal (*Aspergillus tubingensis*) growth was observed by Krishna *et al*. (2005), who also noted improved plant performance. In the field, Chauhan and Ganguly (2011) used acid-producing bacteria with vermicompost and gypsum as amendments for promoting vegetation growth on residues. Specific effects of bacteria inoculation were not examined in the study.

Soil salinity (EC)

High salinity results in a more negative water potential in soil solution and as a result water uptake by plants is reduced and root-pressure-driven xylem transport of water and solutes is reduced as is shoot and root growth (Lauchli and Grattan 2007; Yadev *et al.* 2011). However, the sensitivity of plants to salinity is species and cultivar dependent, and often salt tolerant plants such as *Atriplex*, *Agropyron*, and *Distichlis* are used in revegetation of bauxite residues (Fuller and Richardson 1986; Wong and Ho 1994*a*). For soils, an EC measured in a saturation paste extract (EC_{SE}) of <0.95 dS m^{-1} is usually considered low, >4.5 dS m^{-1} high and above 12 dS m^{-1} is too saline for most plants to survive (Shaw 1999). Some studies measured EC in 1 : 5 residue : water ($EC_{1:5}$) extracts, and equivalent values to those quoted above in bauxite residues are ~0.31, 1.5, and 3.9 dS m^{-1} (Li *et al.* 2018*a*). Deposited residue normally has a high EC. For example, Wong and Ho (1994*a*) reported an EC_{SE} of 7.7 dS m^{-1}, Fortin and Karam (1998) reported a value of 11.4 dS m^{-1}, and Meecham and Bell (1977*a*) reported values of 30–36 dS m^{-1}.

As a result, the salinity of freshly deposited residue is a growth limiting factor. For unamended residues, strong correlations between EC and sodium content occur (r = 0.99) (Kong *et al.* 2017) since the major cation in soil solution is Na^+. Even so, in the absence of amendment strategies, the EC of residue progressively decreases due to leaching of salts down the profile induced by natural rainfall events. Zhu *et al*. (2016*a*), for example, showed a progressive decrease in $EC_{1:5}$ from 3.73 dS m^{-1} 1 year after deposition to 0.95 dS m^{-1} after 10 years and to 0.36 dS m^{-1} after 20 years. Similarly, in a pot experiment using residue with an initial EC_{SE} 7.7 dS m^{-1}, a reduction to 3.92 dS m^{-1} was recorded following leaching with rainfall equivalent of 126 mm (Wong and Ho 1993). A potential problem with seawater neutralisation of residue is that the salinity is increased due to the addition of seawater (Menzies *et al.* 2004; Li *et al.* 2016), so that even more intensive leaching is required to reduce EC to acceptable levels (Li and Haynes 2017).

The EC of the mud fraction can be significantly higher than that of residue sand (Jones *et al.* 2011, 2012*a*; Li 2017). The addition of residue sand to the mud fraction can, therefore, decrease salinity and greatly improve the permeability and hence leachability of the salinity, thus further decreasing salinity (Courtney *et al.* 2009*a*). By contrast, addition of mud to the sand fraction to improve water and nutrient retention can have the opposite effect. Jones *et al*. (2011) added residue mud to residue sand at rates up to 20%. All residue mud-amended treatments were highly saline and would, therefore, require intensive leaching. It was, however, concluded that under field conditions, the salts would leach out of the surface layers of residue sand and down the profile.

Because of their high initial soluble salt content, addition of organic amendments such as animal manures, mushroom compost, and particularly biosolids, may initially increase the EC of amended residues (Jones *et al.* 2010, 2011); however, following leaching and nutrient uptake by revegetating plants, the EC decreases to low levels (Jones *et al.* 2012*b*; Courtney *et al*. 2009*a*).

Amendment of residues with gypsum characteristically results in elevated EC values (e.g. Wong and Ho 1993; Table 3). In addition, Li *et al.* (2018*a*) reported that the EC_{SE} in bauxite residue increased from 2.4 dS m^{-1} to 9.4 dS m^{-1} with the reaction of 5% gypsum for 4 weeks followed by leaching with 6 pore volumes of water. Although EC decreases over time in field treatments (Table 3), elevated values following gypsum amendment can also be evident in the longer term, with 10-year-old gypsum rehabilitated residues displaying an $EC_{1:5}$ of 0.52 dS m^{-1} compared with 0.36 dS m^{-1} where no gypsum was applied (Courtney *et al*. 2009*a*). Such an effect occurs because gypsum is sparingly soluble and dissolves over a period of years. It is commonly applied to residues at high rates (e.g. 1–5%) and its dissolution releases Ca^{2+} and SO_4^{2-} into solution (with some of the Ca^{2+} displacing Na^+ on the exchange sites). As a result, ionic strength in soil solution is maintained at a higher level than would otherwise be the case.

The dominance of Ca^{2+} as soluble cation in gypsum-amended residue may well mean that the elevated EC values are not as inhibitory to plant growth as similar values in non-amended residue (where Na^+ is the dominant soluble cation in solution). That is, high concentrations of Na can be toxic to plants (see below), but the presence of Ca can alleviate the phytotoxic effects of Na (Kinraide 1999). Thus, the ratio of Ca : Na in soil solution is an important consideration. For example, Courtney and Mullen (2009) reported an EC in a 1 : 2 residue : water extract of >4.5 dS m^{-1}, 6 months after gypsum application, yet seedling emergence and growth were promoted. Finngean *et al*. (2018) reported that the earthworm *Eisenia fetida* displayed a preference for bauxite residues with an $EC_{1:5}$ value of 2.38 dS m^{-1} and suggested that this was because Ca was the dominant cation in the gypsum-treated residue (i.e. 3% gypsum addition). Thus, not only EC but also the dominant cation in soil solution needs to be considered when contemplating revegetation.

Exchangeable sodium percentage

Remnant sodium hydroxide (NaOH) and sodium carbonate (Na_2CO_3) from the Bayer process plus the formation of desilication products (DSP), sodalite and cancrinite, result in Na being the dominant ion in unamended residue. Amounts of Na will vary between residues depending on variations in refining parameters (e.g. residue washing and disposal practice) (Gräfe and Klauber 2011). Nonetheless, Na is the dominant cation present both on the cation exchange sites on the residue and in solution. The sodicity of soils and bauxite residue is commonly expressed as the ESP, which is calculated as the concentration of exchangeable Na^+ as a percentage of the total exchangeable cations (Ca^{2+} + Mg^{2+} + K^+ + Na^+). For

Table 3. Residue electrical conductivity (EC) and effects of amendments in pot-based trial
Abbreviations: SS, sewage sludge; G, gypsum; Ww, wood waste

Initial dS m^{-1}	Amendment	Final	Comment	Reference
(SE)				
7.7	0% G	3.92 (1 : 5)	Leached with equivalent 126 mm rainfall	Wong and Ho (1993)
	2% G	2.69		
	5% G	4.45		
	8% G	4.04		
	0 copperas ($FeSO_4$)	3.92 (1 : 5)	Leached with equivalent 126 mm rainfall	Wong and Ho (1994*b*)
	2% copperas	1.86		
	5% copperas	1.53		
	8% copperas	1.66		
	SS 8%	1.8		Wong and Ho (1994*a*)
	SS 16%	1.53		
	G 8%	4.39		
	G 8% and SS 8%	4.14		
	G 8% and SS 16%	4.06		
2.5	2% G and	0.59	Leached with equivalent 396 mm rainfall	Jones *et al*. (2010)
	G	1.09–1.25		
	Biosolids	0.79–0.9		
	Mushroom c	0.59–0.65		
	Biochar	0.46–0.51		
2.7	Residue sand and G at 2% (A)	0.01	Seven leaching events with typical monthly rainfall	Jones *et al*. (2012*a*)
3.3	A and red mud (B) added	0.10		
2.9	A and poultry manure (C) added	0.11		
3.0	B and C added	0.12		
(SE)		(1 : 2)		
10.3	Compost at 0%	8.7	Irrigated but not leached	Fortin and Karam (1998)
	18%	4.4		
	23%	4.2		
	29%	4.0		
	31%	3.5		
	37%	3.5		
	45%	3.1		
3.77	Unamended	0.82	Leached with 8 pore volumes	Ippolito *et al*. (2005)
	G	0.80		
	Ww	0.77		
	G and Ww	0.84		

freshly deposited residues, ESP is typically 60–90% (Meecham and Bell 1977*a*, Fuller *et al.* 1982). Soil sodicity is usually recognised when ESP values are >15% (Sumner 1995). Plant growth is inhibited by excessive uptake and accumulation of Na. Dehydration of leaf cells can occur when Na accumulates in the leaf apoplast and enzyme reactions can also be inhibited (Keren 2000). High concentrations of Na in soil solution can reduce Ca uptake, and Ca deficiency often results (Kopittke and Menzies 2005). A deficiency of Ca can influence membrane permeability and, therefore, restrict uptake of other ions resulting in deficiencies of N, K, Mn, Zn, and Cu (Yadev *et al.* 2011).

To overcome this high sodicity, gypsum amendment is frequently applied to the residue to promote plant growth (Wong and Ho 1994*a*; Courtney and Timpson 2005ab; Courtney and Mullen 2009; Jones *et al*. 2011; Anderson *et al*. 2011; Table 4). The divalent cation Ca^{2+} in gypsum is more strongly held to many of the cation exchange sites than Na^+, so added Ca^{2+} displaces Na^+ from exchange sites and it can leach down the profile with the added SO_4^{2-} as a counterion. Thus, with gypsum addition there is an immediate decrease in ESP due to Ca addition followed by a slower increase, as Na is leached out of the profile. Several laboratory-based experiments have demonstrated that addition of gypsum effectively decreases the sodicity of residues and that the effect increases with increasing rates of application. Wong and Ho (1994*a*; Table 3) observed decreases in ESP from 70% down to 43% when residue mud was amended with organic matter alone, whereas gypsum addition at a very high rate (8%) decreased values to 13%. Similarly, Jones *et al*. (2010, 2011) found that gypsum applied at 2% to residue sand followed by leaching was effective in decreasing ESP down to 19–24%. Using residue mud, Li *et al*. (2018*a*) recorded a decrease in ESP from 64% to 38% after an application of 5% gypsum followed by leaching. In a field trial on residue mud, Courtney and Harrington (2012) recorded a decrease in ESP from 31% to 5.7% with an addition of 90 t ha^{-1} of gypsum. Similarly, Zhu *et al*. (2016*a*) reported an ESP of 72% (exchangeable Na = 20 cmol kg^{-1}) in 1-year-old deposited residue, which decreased to an ESP of 29% (exchangeable Na = 9.8 cmol kg^{-1}) in 20-year-old residue on which grass had become established. A reduction in ESP to 9.5%, as suggested by Gräfe and Klauber (2011), may not be practicable in many situations, whereas the suggestion of Haynes (2015) of <40% may be too high. A reduction in ESP to <25–30% is suggested as a reasonable goal.

It should be noted here that gypsum dissolves slowly over a period of years, but the Ca in the residual gypsum may be extracted when exchangeable cations are extracted (e.g. with ammonium acetate) and this will lead to magnified values for exchangeable Ca and consequently low values for calculated ESP. There are several other factors that can also affect ESP values. For example, where significant soluble salts are present, the ammonium acetate-extractable fraction will include both soluble and exchangeable cations (Jones *et al*. 2012*b*). This results in elevated ESP values (unless the concentration of cations in soil solution is subtracted from the extractable fraction) because Na is least strongly held to exchange sites and, therefore, has the greatest percentage in the soluble fraction. There can also be some difficulty in extracting all the exchangeable Na^+, and Wong and Ho (1995) showed divalent cations were ineffective at extracting exchangeable Na held in the structure of the sodalite and/or cancrinite present in bauxite residue. Nonetheless, extraction with ammonium acetate and calculation of ESP based on the ammonium acetate-extractable Na as a percentage of effective cation exchange capacity (i.e. ammonium acetate extractable Na^+ + K^+ + Mg^{2+} + Ca^{2+}) is the common way of calculating ESP in rehabilitating residues.

A sustained reduction in residue ESP is paramount for sustainable revegetation. A desilication product (known as DSP) (e.g. sodalite and cancrinite) is considered an important component of bauxite residues because Na^+ and OH^- ions are slowly released from its mineral structure (Wong and Ho 1995; Whittington *et al.* 1998). Residual gypsum present in the residue may, therefore, be important, because it will continue to dissolve releasing Ca^{2+} which will help neutralise newly solubilised alkalinity as well as displace released Na^+ and promote its leaching (Li and Haynes 2017). Indeed, long-term field trials on revegetation of bauxite residues have not revealed any discernible increase in exchangeable Na^+ or pH over time (Table 2; Bray *et al*. 2018). In all likelihood, under field conditions, the natural soil processes of acidification and leaching counteract the effect of DSP dissolution.

As already noted, one of the main nutritional disorders associated with sodic soil conditions is Ca deficiency. Amendment with gypsum results in Ca^{2+} becoming the dominant exchangeable and soluble cation present (although there is also a significant amount of Na^+ remaining), and, as a result, the Ca content of plants growing in the residue is typically sufficient (Courtney and Harrington 2012). Nonetheless, the predominance of Ca and Na in the residue means that deficiencies of other cations (Mg and K) can occur (Courtney and Timpson 2005*b*; Eastham and Morald 2006; Anderson *et al.* 2011). These occur because of (1) the low intrinsic content of exchangeable Mg and K in deposited residue, (2) leaching of these cations induced by gypsum application, and (3) competitive effects of Ca and Na on plant uptake of K and Mg (Qadir and Schubert 2000; Kopittke and Menzies 2005). A balanced nutrition is required so that additions of K and Mg are often necessary. These can be supplied in mineral fertilisers (Eastham and Morald 2006; Kaur *et al.* 2016), as organic manures or residues (Courtney and Harrington 2012), or as a combination (Eastham *et al*. 2006).

Balanced supply of essential nutrients

The ability of residue (fines and sand) to retain and supply nutrients to growing plants varies considerably depending on factors such as the form of fertiliser (organic versus inorganic), timing of application (leaching potential), rate of application, and number of applications (single versus split). Numerous studies have been undertaken under laboratory, greenhouse, and field conditions to understand nutrient dynamics in bauxite residue (Thiyagarajan *et al*. 2009, 2011, 2012; Chen *et al.* 2010*a*, 2010*b*; Banning *et al*. 2010, 2014; Goloran *et al.* 2014*a*, 2015*a*, 2015*b*, 2017; Jones *et al.* 2010, 2011, 2012*a*, 2012*b*, 2015; Kaur *et al*. 2016). Outcomes from these studies clearly demonstrate the importance of nutrient addition to achieve a vegetation cover. Protocols for evaluating rhizosphere behaviour, particularly in terms of nutrient dynamics, need to be developed (Rezaei Rashti *et al*. 2019).

There is a need to develop availability indices for macro- and micronutrients in bauxite residues. Because residues have properties that differ somewhat from most soils, it may not be appropriate to simply use critical values developed in agricultural soils. Soil tests may well need to be recalibrated for plants growing in residues. For example, residue has a particularly high nutrient retention capacity (P, Cu, Zn, Mn, Fe, and B) through adsorption to Fe and Al hydrous oxides (Summers *et al.* 2002; Carter *et al.* 2009; Phillips and Chen 2010; Thiyagarajan *et al.* 2009, 2011). It is also important to note that the target values will be greatly species- and ecosystem- dependent so studies in analogue ecosystems may be required. Where studies on soil tests have been carried out, they are noted below.

Nitrogen

Bauxite residue has a very low organic matter content and, therefore, a low total N content ranging from 0.002% to 0.03%, depending on its source (e.g. Courtney *et al.* 2009*a*; Chen *et al.* 2010*a*; Jones and Haynes 2011). Nitrogen supply to plants, via microbial mineralisation of native organic N, is therefore minimal. As a result, application of sufficient N reserves in bauxite residue rehabilitation is an essential management step. Nitrogen inputs can come from fertiliser N (urea, NH_4^+, or NO_3^--containing materials) or organic wastes (e.g. animal manures, biosolids, or composts). Where organic materials are added, this will increase the organic matter (and total N) content of the residue. In addition, as revegetation proceeds, organic matter will accumulate in the residue due to deposition of above- and belowground litter by the growing plants. Atmospheric N can also enter the soil organic pool via decomposition of leguminous plant litter where legumes ae used in revegetation (Courtney *et al.* 2009*b*). Thus, the total N content of the residue will increase over time and an increasing proportion of plant available mineral N (NH_4^+-N and NO_3^--N) will originate from microbial mineralisation of the accumulated organic N. Particle size fractionation of residue by Courtney *et al*. (2013) showed that in residue that had been revegetated for 1 year, the bulk of the C and N was in the >200 mm fraction, but, after 9–11 years, there was an increasing proportion in the 200–53 μm and <53 μm fractions (and in a similar proportion to that in an adjacent soil). This reflects input of organic matter as large particulates (added manure and decomposing plant debris) and the decomposition and humification of this organic matter over time. The organic matter in the <53 μm fraction represents humic material bound to the soil mineral fraction.

Nitrogen availability indices have been incorporated in classifying ecological conditions in soils due to the influence of N on plant growth performance and species composition (Binkley and Hart 1989). The use of a suitable soil N index that fully reflects its ecological significance is important (Wilson *et al.* 2005). Such indexes usually take account of easily mineralisable 'labile' soil organic N as well as mineral N (extractable NH_4^+-N and NO_3^--N) already present in the profile. However, Goloran *et al.* (2013) showed that for bauxite residue sand, under both pot experiment and field conditions, KCl-extractable NO_3^--N was best correlated with plant biomass N. Presumably, the soil organic N pool had not yet built up to a level where potentially mineralisable N was significantly contributing to N uptake by plants.

Many researchers have demonstrated the importance of inputs of N when revegetating bauxite residues. For example, Wong and Ho (1993) reported a deficiency for nitrogen in plants grown in red mud, whereas Courtney and Harrington (2012) recorded a low N content in *Holcus lanatus* in residues amended with spent mushroom compost. However, they observed that high application rates of mushroom compost 120 t ha^{-1} alleviated such N deficiency. In a two-year field assessment of grass growth in residue, Courtney and Timpson (2005*a*) recorded low-to-deficient N in tissues of *H. lanatus* and *Lolium perenne* after one year. Subsequent foliar analysis following applications of inorganic N, P, and K yielded adequate N content in both *H. lanatus* and *Trifolium pratense* (Courtney *et al*. 2009*b*). The much higher values recorded in *T. pratense* highlighted the benefits of including leguminous species in residue rehabilitation. Indeed, the study of Courtney *et al*. (2009*b*) also found six Fabaceae species in rehabilitated residue indicating that they can play an important role in N supply.

The initially high concentrations of salinity, sodicity, and alkalinity in residues inhibit both plant growth and N uptake. Kaur *et al*. (2016), for instance, showed that N uptake by *Cenchrus clandestinus* (kikuyu grass) grown in residue sand was enhanced after leaching of excess salinity and alkalinity from the residue profile. This suggests that these factors initially limited grass growth and N uptake. Similarly, under field conditions, Goloran *et al*. (2015*b*) observed a general negative relationship between exchangeable soil Na and leaf N and the leaf N : P ratio in shrubs growing in a revegetated area of residue sand.

Nitrogen losses from soil can occur via leaching down the profile (e.g. NO_3^- leaching) or gaseous losses to the atmosphere N (e.g. NH_3 volatilisation). Losses from soils are affected by numerous factors including soil moisture content, temperature, source, and application rate of N and soil pH (Kumar and Goh 1999). The high pH of bauxite residues favours NH_3 volatilisation losses when NH_4^+-containing fertilisers are surface applied (Phillips and Chen 2010; Jones and Haynes 2011), because NH_3 volatilisation is a physicochemical process favoured by high pH (e.g. >7) and high concentrations of NH_3 at the soil and atmosphere interface. In laboratory studies, Chen *et al*. (2010*a*) measured massive losses of NH_3 (up to 95% of added N) when di-ammonium phosphate (DAP) was applied to un-neutralised residue sand (pH 11.6), whereas Chen *et al.* (2010*b*) recorded up to an 85% loss of applied N within 7 days via volatilisation in sand at pH 9. Such losses can be of great practical significance. For example, rehabilitation undertaken by Alcoa of Australia includes application of inorganic di-ammonium phosphate fertiliser applied in conjunction with trace elements and potassium sulfate at rates of 2.7 t ha^{-1} (Eastham and Morald 2006). This rate of DAP applies ~265 kg N ha^{-1}, but very little N remains in the residue profile after 12 months. In fact, most of the applied N is probably lost via volatilisation during the first 24 h after application.

Application of biochar to residue can reduce NH_3 volatilisation substantially, because of its adsorption capacity for NH_4^+ and NH_3 (Chen *et al.* 2010*a*). Esfandbod *et al*. (2017) found that application of acidic biochar to residue reduced the cumulative loss of NH_3 and that KCl-extractable NH_4^+-N was also much higher in this treatment compared with control or where alkaline biochars were used. From a practical viewpoint, incorporation of fertiliser NH_4^+ into the topsoil layer by tillage will greatly reduce losses by reducing NH_3 concentrations in soil solution near the soil surface (Jones and Haynes 2011).

Another possible disadvantage of applying NH_4^+ fertilisers is that NH_4 can accumulate due to lack of autotrophic nitrifier bacterial activity and high concentrations of NH_4^+ can be phytotoxic (Meecham and Bell 1977*b*). Jones *et al*. (2011) also observed accumulation of NH_4^+-N in bauxite residue after applications of biosolids and poultry manure. An initial lack of microbial activity in bauxite residue is to be expected, because it is effectively both a chemically (NaOH) and heat-treated material (Jones and Haynes 2011). Nonetheless, once deposited in storage areas, colonisation by a microbial community (including nitrifiers) is likely to be rapid via aeolian movement (e.g. dust with adhering bacterial cells and fungal spores) (Haynes 2014). Thus, autotrophic nitrifying bacteria colonise residue deposits rapidly (Goloran *et al.* 2015*a*), and NO_3^- becomes the dominant form of N present in the residue even where N was initially supplied in the NH_4^+ form (Jones *et al.* 2011; Goloran *et al.* 2013, 2015*a*). Goloran *et al.* (2013) found that the concentration of soil NO_3 increased with increasing rehabilitation age from a 7-year-old site to one of 15 years. An advantage of nitrification of applied fertiliser NH_4^+ is that two moles of H^+ are produced per mole of NO_3^--N produced (Rodriguez *et al*. 2008), which will help maintain and/or lower the pH of the residue.

Fertiliser NO_3^- is an alternative that can be used; however, the NO_3^- anion is highly mobile and could easily leach during periods of heavy rain. In a growth chamber experiment Goloran *et al*. (2015*a*) recorded significant NO_3 leaching from residue sand and Chen *et al*. (2010*a*) noted that significant NO_3 leaching would be expected under field conditions. Even so, Goloran *et al*. (2015*a*) showed plant uptake of N was greater from KNO_3 than NH_4SO_4 despite leaching losses of NO_3^- occurring. They suggested that when NH_4^+ fertilisers are used, NH_3^- volatilisation loss dominated over plant uptake of N. Leaching of NO_3^- may well become a significant loss mechanism in older rehabilitation sites when the microbial population has developed capability for mineralisation and nitrification of organic N (Banning *et al.* 2014).

The use of controlled-release fertilisers that release their nutrients slowly over a period of months or years is probably the most practicable solution during revegetation of residues. In addition, a slow release of N via microbial mineralisation of organic N in organic wastes and manures will supply N. Ammonification will release NH_4^+-N which will then be nitrified to NO_3^--N as long as nitrifying bacteria are present. The entire process will release one H^+ ion per mole of NO_3^- produced and will, therefore, tend to lower pH.

Phosphorous availability

The P status of residue is partially dependent on the P content of the parent bauxite and also on the processes used in the alumina refinery. Nevertheless, a low P status commonly limits plant establishment and growth in bauxite residue (Goloran *et al.* 2014*a*). Many bauxite deposits are intrinsically low in P and furthermore residue has a characteristically high P fixation capacity. Indeed, the high content of Fe oxides (and presence of Al oxides) in residues means their surfaces possess variable charge characteristics and have a high capacity to adsorb phosphate (Phillips and Chen 2010). Phosphate adsorption increases with decreasing pH (Phillips and Chen 2010). This high P fixation capacity of residues has led to pilot trials on its use as an absorbent to remove P from P-rich wastewaters (López *et al.* 1998; Snars *et al.* 2004; Cusack *et al*. 2018) and for land application to sandy soils to prevent leaching losses of P (Summers *et al.* 1996). The low P content and high P fixation capacity of most residues means that P status is low and tends to decline over time (Bendfeldt *et al.* 2001). Particle size within the residue matrix also effects P adsorption as shown by Courtney and Harrington (2010), who found that P adsorption in amended residue substrate had a stronger correlation with particles in the <2 μm fraction compared with the larger 20 μm size. This is because smaller particles have a greater surface area and thus more surfaces available for P adsorption. In addition, the adsorption capacity of residue mud is much greater than that of residue sand because of the much greater Fe oxide content of mud. That is, residue sand is principally quartz sand particles. Biochar has a capacity to adsorb P, and its addition to bauxite residue was shown by Goloran *et al.* (2014*b*) to increase extractable P levels and increase plant P uptake.

Because of the low P status, rehabilitation and revegetation practices employed on bauxite residues involve the application of P rich organic waste and/ or inorganic P fertilisers (Courtney and Harrington 2010; Eastham *et al*. 2006; Goloran *et al*. 2014*a*, 2014*b*). Biosolids applications can give particularly high available P levels due to their high P content (Courtney and Mullen 2008). Goloran *et al.* (2014*b*) showed that in residue sand there was an interaction between N and P applications. Applied P along with NO_3^--N resulted in much greater utilisation of P by *Lolium rigidum* compared with NH_4^+-N, but the mechanism involved was unclear. Even with high P application rates, deficiencies in plant P content have been recorded in pot trials (Anderson *et al*. 2011) and field studies (Courtney and Timpson 2005*b*; Eastham *et al.* 2006). Indeed, as well as P fixation by Fe and Al oxides, the presence of $CaCO_3$ in residues plus the addition of high rates of gypsum can further decrease P availability. That is, at high pH and high Ca concentrations calcium phosphates can precipitate and P can also be adsorbed to the surfaces of calcite (Eastham and Morald 2006; Phillips and Chen 2010). Goloran *et al*. (2015*b*) found that Ca content in residues was negatively correlated with extractable P in both aged and freshly amended residues.

Use of standard agronomic soil P extraction techniques on rehabilitated or revegetated residues have provided varying results. Courtney and Harrington (2010) assessed P availability in 1-year-old revegetated residue using Morgan's reagent (acetic acid, pH 4.8) and reported all treatments had levels greater than >10 mg L^{-1} (and in soils crops should not respond to P application), yet herbage P content was low to deficient in all samples. This 'overestimation' of plant available P in the residue substrate may be explained by the acidity of the extractant removing both available and non-mobile phosphorus (due to dissolution of Al and Fe-bound P). Similarly, Meecham and Bell (1977*a*) reported that acid-extractable P was eight times higher than $NaHCO_3$-extractable P. Colwell ($NaHCO_3$)-extractable P also substantially overestimated amounts of plant-available P in bauxite residue (Courtney and Harrington 2010; Goloran *et al.* 2014*b*). However, both these studies reported significant correlations between P-Colwell and plant dry matter yield. Such results suggest that critical soil test P values derived from plant growth in soils cannot necessarily be used for plants growing in bauxite residues and that the test values need to be recalibrated using plant growth and P uptake data from growth in residues. Goloran *et al.* 2014*b*) reported that plant P uptake was best correlated with P extracted using anion exchange strips, but no further studies were carried out.

Micronutrients – Mn, Cu and Zn

Micronutrient deficiencies in rehabilitated residues have been highlighted in several different studies (Table 5). Early work by Meecham and Bell (1977*a*) highlighted the low-to-deficient levels of Mn, Zn and Cu in both residue mud and sand. The dominance of hydrous Fe and/or Al oxides in residues along with their high pH leads to deficiency levels for extractable trace elements (Gherardi and Rengel 2001; Thiyagarajan *et al.* 2009). That is, these ions are strongly adsorbed to metal oxide surfaces and the high pH in residues favours their strong adsorption as well as well as precipitation reactions. As a result, micronutrient deficiencies have been recorded by many workers in plants grown in residues (Fuller *et al.* 1982; Gherardi and Rengel 2001; Thiyagarajan *et al*. 2009).

Reduced availability of added Mn can become problematic in bauxite residues under field conditions. Fuller and Richardson (1986) reported that *Distichlis spicata* (desert saltgrass) became deficient in Mn after 1–2 years of growth and exhibited necrotic lesions. Although Mn supplied through organic additions resulted in adequate plant content after 1 year of *L. perenne* and *H. lanatus* growth, plant concentrations

Table 4. Residue exchangeable sodium percentage (ESP) and effects of amendments in pot-based trials
Abbreviations: SS, sewage sludge; G, gypsum; Ww, wood waste

Initial	Amendment	Final	Comment	Reference
70.4	0% G	70.4	Leached with equivalent 126 mm rainfall	Wong and Ho (1993)
	2% G	55.9		
	5% G	11.6		
	8% G	10.8		
70.4	0 copperas ($FeSO_4$)		Leached with equivalent 126 mm rainfall	Wong and Ho (1994*b*)
	2% copperas	46.4		
	5% copperas	42.5		
	8% copperas	39.8		
70.4	SS 8%	47.2		Wong and Ho (1994*a*)
	SS 16%	42.8		
	G 8%	12.7		
	G 8% + SS 8%	11.0		
	G 8% + SS 16%	9.5		
75	G	19.2	Leached with equivalent 396 mm rainfall	Jones *et al*. (2010)
	Biosolids	15.6–15.9		
	Mushroom c	15.4–17.7		
	Green waste	18.5–19.5		
	Biochar	18.5 – 19.4		
50	Residue sand and gypsum @ 2% (A)	2.1	7 leaching events with typical monthly rainfall	Jones *et al*. (2012*a*)
53	A and red mud (B) added	9.1		
47	A and poultry manure (C) added	3.6		
52	B and C added	7.7		
60	Unamended	7	Leached with 8 pore volumes	Ippolito *et al*. (2005)
	G	4		
	Ww	6		
	G and Ww	5		

recorded following the second year of growth demonstrated significant decreases to deficiency levels (Courtney and Timpson 2005*a*). Courtney *et al*. (2009*b*) also reported low Mn status in residue 7 years after rehabilitation, and they identified Mn deficiencies in *T. pratense* and *H. lanatus*. Gherardi and Rengel (2001) added Mn as $MnSO_4.H_2O$ to residue and found that plant-available Mn, as estimated by diethylenetriaminepentaacetic acid (DTPA) extraction, decreased markedly with time. The high pH and abundance of electron acceptors (e.g. HCO_3^- and CO_3^{2-}) favours oxidation of soluble Mn^{2+} to insoluble Mn^{4+} (Gherardi and Rengel 2001). They concluded that banding fertiliser Mn is more effective than broadcast fertilisation, because the applied Mn persisted in available form in the banded volume for longer periods of time.

Due to the above considerations, regular application of fertiliser micronutrients may be important for sustained plant production in residues. Because of the very high pH of residues and, therefore, rapid adsorption and immobilisation of added micronutrients in some cases, soil application of micronutrient fertilisers might not be effective in correcting micronutrient deficiencies (Eastham *et al*. 2006). In such cases foliar applications of micronutrients in chelated form may be required. It is also important to note that addition of organic manures may add significant amounts of micronutrients. Fuller and Richardson (1986) found that additions of sewage sludge resulted in adequate Zn uptake. Similarly, Courtney and Harrington (2012) added spent mushroom compost at rates of 60–120 t ha^{-1} and reported all treatments to have DTPA-extractable Mn, Cu, and Zn above critical levels. Furthermore, foliar concentrations in *H. lanatus* after 1 year of growth were also all well above reported deficiency levels.

Potentially toxic elements: Al availability and plant content

Bauxite residue has significant residual Al content (e.g. 2–23% Al_2O_3; Gräfe and Klauber 2011). The solubility of Al-bearing minerals (e.g. gibbsite) is highly pH-dependent. That is, between pH 6 and 8, Al is present as insoluble $Al(OH)_3$. With increasing pH, Al sequentially dissociates H^+ ions to form increasingly soluble negatively charged aluminates. Between pH 8 and 9, $Al(OH)_4^-$ predominates, between 9–10 it is mainly as $Al(OH)_5^{2-}$, and above 10 as $Al(OH)_6^{3-}$. Thus, soluble Al is present in high concentrations when residue is first deposited, and this aluminate has been shown to be phytotoxic (Kopittke *et al.* 2005). Indeed, early revegetation trials identified Al toxicity as a potentially limiting factor (Fuller and Richardson 1986). Lowering pH to 8 or below causes precipitation of Al as $Al(OH)_3$ and alleviation of such toxicity.

The risk of Al toxicity at high pH has been highlighted by Kopittke *et al*. (2005), who suggested that at high pH, $Al(OH)_4^-$ constitutes >99% of the monomeric hydroxy-Al. However, it was also suggested that $(Al(OH)_4^-$ becomes non-toxic when Ca is added to soil solution and Al^{13} forms. In pot trials using *D. spicata*, Fuller and Richardson (1986) reported that plant yield and nutrient content was negatively correlated with aluminium content. In unamended residue, Al content was

Table 5. Diethylenetriaminepentaacetic acid (DTPA)-extractable nutrient elements in bauxite residue and amended treatments
Note: all results in mg kg^{-1}

	Critical values[A]	Bauxite residue	Source
		Unamended	
Mn	1–5	0.02–0.42	Wong and Ho (1994*a*);
Fe	2.5–4.5	2.16–6.81	Thiyagarajan *et al*. (2009);
Zn	0.2–2.0	0.07–1.27	Jones *et al*. (2010);
Cu	0.1–2.5	0.25–0.17	Courtney and Harrington (2012)
		Sewage sludge amended	
Mn		0.22–1.9	Wong and Ho (1994*a*)
Fe		7.1–21.5	
		Micronutrients added	
Zn		0.08–0.153	Thiyagarajan *et al*. (2009)
Mn		0.46 – 0.06	
Cu		0.16–0.24	
Fe		2.69–6.15	
		4-year-old field samples	
Zn		0.867–1.08	
Mn		0.25–0.26	
Cu		0.27 – 0.417	
Fe		1.83–2.41	
		Organic additions	
Zn		0.02–7.7	Jones *et al*. (2010)
Mn		0.05–1.38	
Cu		0.02–6.43	
Fe		2.06–17.9	
		Organic amended 1-year-old field samples	
Zn		5.3–7.3	Courtney and Timpson (2005*a*)
Mn		0.68–1.06	
		1-year-old field samples	
Mn		0.29–2.81	Courtney and Harrington (2012)
Zn		0.24–2.96	
Cu		0.23–0.62	
		9-year-old field samples	
Mn		0.36–0.37	Courtney *et al*. (2009*b*)

[A]After Sims and Johnson (1991).

up to eight times that of a reference plant (i.e. 1650 compared to 230 mg kg^{-1}).

Addition of organic matter alone may not be sufficient to mitigate against the high levels of soluble Al present in residues, because the pH is not lowered sufficiently. Courtney *et al*. (2018) reported decreases from 21 mg kg^{-1} in unamended residue to ~10 mg kg^{-1} with organic amendment alone. Wong and Ho (1993) found that addition of sewage sludge alone to bauxite residue did not lower the pH and high soluble Al contributed to the low biomass of *Agropyron elongatum*. Addition of gypsum to lower pH is, however, a successful strategy to lower soluble Al. For example, pot trial investigations using gypsum amendment have reported significant decreases in soluble Al down to very low concentrations (Wong and Ho 1993; Xenidis *et al*. 2005; Courtney *et al*. 2018).

Acidifying procedures other than gypsum treatment are probably not as effective as gypsum in decreasing levels of (bio)available Al in residue. For example, comparing the acidulants gypsum, $MgSO_4$, and H_2SO_4 Courtney and Kirwan (2012) found that acid neutralised treatments had a higher soluble Al content than Ca and Mg sulfate treatments at the same pH. They suggested that an excess of divalent cations is required to suppress dissolution of $Ca_3Al_2(OH)_{12}$ (TCA) and sustain lower pH values. The most effective way to achieve this is to supply an excess of a slightly soluble salt, such as gypsum.

$$3Ca^{2+} + 4OH^- + 2Al(OH)_4^- \rightarrow Ca_3Al_2(OH)_{12}$$

Addition of organic wastes containing elevated Ca and Mg concentrations could have a similar effect.

As noted previously, field experiments have shown that the pH of gypsum-amended residue remains relatively constant over time. As a result, sustained decreases in exchangeable and soluble Al also occur. Bray *et al*. (2018) assessed field plots 16 years after amendment and found that in untreated plots soluble Al was ~10 mg kg^{-1} at the surface, increasing to ~65 mg kg^{-1} at 50 cm depth. In plots amended with gypsum plus organic matter, soluble Al concentrations were near the lower detection limit (0.09 mg kg^{-1}) to a depth of 30 cm. Similarly, field trials have shown that gypsum applications during revegetation result in low (non-toxic) concentrations of Al in tissues of plants growing in the residue (Courtney and Kirwan 2012).

Potentially toxic elements: other elements

Bauxite residue can also contain elevated concentration of potentially toxic elements As, Cr, Ni, Pb, Mo, and V (Gräfe *et al.* 2011, Klebercz *et al.* 2012; Lockwood *et al*. 2014; Mišík *et al*. 2014). Due to residual NaOH, water in contact with residue can be alkaline and saline (Mayes *et al*. 2011; Higgins *et al*. 2016) leading to potential for enhanced mobility of such elements (Lockwood *et al*. 2014). Addition of amendments such as gypsum or seawater to near-neutral pH in residue enhances trace element sorption of Al, As, and V (Burke *et al*. 2013; Lehoux *et al*. 2013; Lockwood *et al*. 2014).

Bray *et al*. (2018) reported that rehabilitation procedures (gypsum and organic amendment) resulted in appreciable decreases in soluble As and V, both at the depth of amendment and further down the profile (up to 50 cm). Within the zone of amendment (0–20 cm), values were mostly below the limit of detection and this trend was also found for V in pot-based trials (Courtney *et al*. 2018). However, the bioavailability and plant uptake potential of these metals in rehabilitated residue remains unexplored.

Soil quality indicator – biological properties

Microbial colonisation and activity

The importance of microbial community structure and function is well recognised in the assessment of reclaimed mine residues or tailings. Sustained plant growth is largely dependent on microbes recycling and mobilising soil macronutrients, and, as noted previously, microbial activity can also be effective in decreasing residue pH. When it is first deposited, bauxite residue is effectively a heat- and chemically- treated sterile inorganic component (with a very low organic matter content). It therefore has a very low microbial activity. Microbial community assembly occurs mainly by via aeolian transport, and diverse assemblages of bacteria and fungi can occur on newly exposed parent material within a matter of months (Banning *et al.* 2011; Haynes 2014). However microbial activity is limited by low C and N availability so that the early stages of community assembly comprises microflora which typically exist in resting stages. As a result, the size of the community (microbial biomass) and its activity (respiratory CO_2 evolution) are extremely low.

Early colonisation of residues by alkali-tolerant bacterial populations (Chitinophagaceae, Beijerinckiaceae, Xanthomonadaceae, and Acetobacteraceae) occurs within a relatively short time frame (Santini *et al.* 2015*a*, 2015*b*; Schmalenberger *et al.* 2013). These taxonomic groups are normally associated with alkaline salt lakes and sediments. Acetobacteraceae are organic acid producing bacteria, and it is possible they may contribute to the decreasing pH that occurs on bare, unamended residues (Schmalenberger *et al*. 2013). Phospholipid fatty acid (PLFA) analyses by Courtney *et al.* (2014*a*) revealed decreases in Gram-negative bacteria with time after revegetation. This was interpreted as ecosystem development towards a more mature system and was attributed to a shift from chemolithotrophic to heterotrophic communities. The diversity of bacterial communities increases with rehabilitation age (Banning *et al.* 2011; Schmalenberger *et al*. 2013). Bacterial communities in a residue site that had been revegetated for 10–12 years contained populations typical of soils including Acidobacteriaceae, Nitrosomonadaceae, and Caulobacteraceae (Schmalenberger *et al*. 2013). In addition, sequences closely related to endobacteria from arbuscular mycorrhizal (AM) fungi were identified. PLFA analysis of the same site demonstrated that the AM fungi bioindicator ($16:1\omega5$) was substantially increased after only 1 year of rehabilitation and levels were sustained after 12 years (Courtney *et al*. 2014*a*). Increased AM fungi in residue sites will be beneficial for sustained plant growth. The speed with which these biota are dispersed to the site may be site specific, and, where mycorrhizal fungi are not established, inoculation is likely to be beneficial for plant establishment. Babu and Reddy (2011), for example, found that inoculation with AM fungi increased the growth of Bermuda grass (*Cynodon dactylon*) in bauxite residues.

For a substantial microbial biomass and high respiratory activity to develop in residues, a supply of C is necessary. Banning *et al*. (2014) reported that compost amendment increased microbial biomass C by 7–10 times that of the control at a 2% addition rate and 16–19 times that of the control at a 5% addition. Respiration activity was also increased by 3–8 times that of the control at the 2% amendment rate and 19–28 times that of the control at the 5% amendment rate. Other workers have recorded similar results (Jones *et al.* 2010, 2011, 2012*a*; Li *et al.* 2018*a*, 2018*b*). Microbial activity can also be assayed based on soil enzyme activities. Jones *et al*. (2011) detected no activity for β-glucosidase, L-asparaginase, and alkaline phosphatase enzymes in unamended residue and recorded greatly increased activities in residues amended with poultry manure or biosolids.

As discussed previously, with increasing time after revegetation soil organic matter will accumulate in the residue and, therefore, the size and activity of the microbial community will also increase (Haynes 2014). As a result, Banning *et al.* (2011) showed an increase in microbial biomass with increasing age after revegetation. Increased activity for dehydrogenase and β-glucosidase activities with time since revegetation has also been observed (Courtney *et al*. 2014*b*). Similarly, Banning *et al.* (2011) recorded almost zero amino acid breakdown in a freshly deposited residue, but decomposition rates in amended residues increased with rehabilitation age and, after only 2 years, they were similar to that of an analogue soil system. Banning *et al.* (2011) also showed that the metabolic quotient (an indicator of microbial stress) decreased with increasing time since revegetation. PLFA analysis showed an increased fungal : bacterial ratio with rehabilitation time, which is indicative of improved nutrient cycling ability within the soil system (Courtney *et al*. 2014*a*). Changes in community composition with increasing residue age are generally correlated with decreases in pH, exchangeable Na, and ESP and increases in organic C and total N (Banning *et al.* 2011; Courtney *et al.* 2014*a*; Schmalenberger *et al.* 2013).

Soil faunal colonisation

It is increasingly being recognised that a functioning belowground ecosystem including diverse microbial and soil faunal communities is critical for reclamation strategies to become sustainable (Frouz *et al*. 2006, 2007; Biederman *et al*. 2008; Courtney *et al*. 2014*b*, 2018). To date, the ability of soil faunal groups to become established and to function in bauxite residues has received scant attention. In general, the development of a soil faunal community is less rapid than that of the microbial community, because dispersal is slower and some faunal species require a certain depth of topsoil or a litter layer before high populations develop (Haynes 2014).

Certainly, the high pH, salinity, and sodicity of bauxite residue will be inhibitory to soil fauna. Southwell and Majer (1982) attributed the high mortality rate of earthworms (*Eisenia fetida*) in residue to high pH resulting in high mortality rates. Finngean *et al*. (2018) found that earthworms (*E. fetida*) avoided residue with elevated Na levels (i.e. unamended residues) but choose residue high in Ca (gypsum amended). They also found that the springtail *Folsomia candida* showed avoidance and high mortality rates in unamended bauxite residue and preferential movement towards amended and revegetated treatments, where it exhibited low mortality rates. Similarly, Courtney *et al*. (2018) reported survival of earthworm *Allobophora chlorotica* in saline residue resulting from gypsum amendment. Improved residue conditions (amendment and time since revegetation) were shown by Courtney *et al*. (2011) to alter nematode trophic composition and increase taxa richness. Nematode taxa, indicative of more advanced successional stages and reduced environmental stress, were recorded in greater numbers or only existed in residue treatments rehabilitated the longest and exhibiting lower pH, EC, and sodicity. Courtney *et al*. (2011) also observed an increased maturity index with time since revegetation, indicating that the nematode assemblage was moving along a successional trajectory towards a more natural system.

Courtney *et al*. (2014*b*) surveyed the presence and absence of soil dwelling invertebrates in revegetated bauxite residues. They recorded natural establishment of large invertebrates such as earthworms (Lumbricidae) and ants (Formicidae) in sites revegetated for 10 and 12 years, whereas a site that had been revegetated for 1 year exhibited a large quantity of springtails and Dipteran larvae, which typify early succession. Canonical correspondence analysis demonstrated a relationship between opportunistic invertebrates and the youngest rehabilitated site, whereas the older sites were associated with a wider range of invertebrate species. Furthermore, decomposition rates (as determined by the litterbag method) were higher at the older rehabilitated sites and this was attributed to higher faunal densities. Investigations of 12-year-old revegetated residue by Courtney *et al*. (2018) reported the presence of five earthworm species representing both epigeic (surface dwelling) and endogeic groups, whereas further pot-based experiments found that the anecic (deep burrowing) species *Aporrectodea longa* can survive in some revegetated residue.

Whilst it is evident that rehabilitated residues can support a variety of key soil faunal groups, the inhibitory characteristics of the residue are also obvious. Tolerance values for key faunal groups remain unknown for bauxite residues. Whether inoculation, as practiced in other reclamation strategies, could provide a useful way to accelerate ecosystem development in rehabilitated residues is also unknown. Further work in the area of soil faunal (e.g. earthworms) colonisation and function in rehabilitated residues is warranted.

Vegetation indicators

The success of a reclamation and revegetation program is dependent upon the selection of several appropriate perennial grasses or shrubs and trees (Mendez and Maier 2008). Good plant performance at germination and initial seedling development is extremely important because poor plant production often results from plants failing to achieve rapid establishment (Williamson *et al*. 1982).

A large number of criteria can be considered in selecting plant species for seed mixtures used in mine waste revegetation. These include: (1) use of natural colonisers (such species often possess adaptations to particular facets of environmental stress); (2) inherent tolerance to adverse chemical conditions (e.g. high pH and high salinity); (3) ability of the species to withstand adverse environmental conditions, such as drought and other conditions prevalent at the site; (4) possession of a well-developed root system that can stabilise the substrate; (5) eventual land use of the area; and (6) commercial availability of seed. Grasses provide a quick ground cover and temporarily limit aeolian dispersion of tailings (Williams and Currey 2002), whereas shrubs and trees become established over longer periods of time (Williams and Currey 2002). For shrub and tree communities, supplementary seeding or planting to improve species diversity of target plants is often required (Wehr *et al.* 2006; Huang *et al*. 2012). Shrubs and trees can provide an extensive canopy cover and establish a deeper root network to prevent erosion over the longer term (Mendez and Maier 2008). As shown in Table 6, a wide range of grass species have been used globally in residue revegetation in both pot and field studies, whereas only a few workers have studied shrub and tree communities (e.g. Xenidis *et al.* 2005; Thiyagarajan *et al.* 2009; Chauhan and Ganguly 2011).

It is important to note here that the choice of vegetation cover should be made in conjunction with amendment and management of the residue deposit. Rehabilitation involves management of the entire plant–soil system. For example, rehabilitation studies from other mine tailings facilities have demonstrated how failure to effectively stimulate the development of soil structure and functions in the root zone can lead to poor plant survival and failure of newly established plant communities within a few years (e.g. Bell and Jones 1987; McNearny and Wheeler 1995; Courtney 2013; Huang *et al*. 2012). Successful rehabilitation will involve management of the physical, chemical, and microbial characteristics of residue and their above- and belowground interactions with the vegetation cover.

Species selection

Good plant performance at germination and initial seedling development under saline-sodic conditions is essential. Plant performance can be inhibited by excess salts in soil solution that prevent water uptake due to the low osmotic potential of the medium and by toxic effects of ions (e.g. Na) (Waisel 1972). Selection of suitable plants for bauxite residue reclamation has included saline-sodic tolerant plants such as *Chloris gayana*

Table 6. Vegetation species used in residue rehabilitation trials and parameters assessed

Species used	Duration of experiment	Parameters	Location of study	Source
		Pot-based studies		
Atriplex canescens	8 days	Reactive oxygen species, malondialdehyde, and protein carbonyl	China	Shi *et al*. (2017)
Lolium rigidum	7 weeks	Biomass	Western Australia	Banning *et al*. (2014)
L. rigidum, *Acacia saligna*	18 weeks	Shoot and root yield, nutrient content, Na content,	Western Australia	Jones *et al*. (2012*a*)
Distichlis spicata	4.5 months	Shoot yield, rhizome length	Alabama, US	Fuller *et al*. (1982)
D. spicata	18 weeks	Biomass, plant elemental content	Alabama, US	Fuller and Richardson (1986)
Chloris gayana	56 days	Seedling emergence, biomass	Queensland, Australia	Meecham and Bell (1977*b*)
Agropyron elongatum, *Cynodon dactylon*	10 weeks	Seedling emergence, biomass elemental content	Western Australia	Wong and Ho (1993)
A. elongatum	20 days	Seedling emergence	Western Australia	Wong and Ho (1994*a*)
A. elongatum	Not stated	Nutrient content Al content	Western Australia	Wong and Ho (1994*a*)
Pucinellia distans	60 days	Biomass	Québec, Canada	Fortin and Karam (1998)
Tamarix sp., *Pittosporum chinense*, *Pistacia lentiscus*, *Atriplex halimus*, *Centaurea Spinosa Tamarix* sp., *Pistacia lentiscus*, *Quercus coccifera*	6 months, 15 months	Survival height	Greece, Europe	Xenidis *et al*. (2005)
Cenchrus clandestinus (*Chiov.*) *Morrone*	3 weeks	Plant nutrient content	Western Australia	Kaur *et al*. (2016)
Lolium rigidum	3 months	Biomass N content	Western Australia	Goloran *et al*. (2013)
Acacia saligna	13 weeks	Root and shoot biomass, nutrient content	Western Australia	Anderson *et al*. (2011)
			Texas, US	Woodard *et al*. (2008)
L. perenne, Lattuca sativa, Trifolium pratense and Lepidium sativum	7 days	Germination index	Ireland, Europe	Courtney and Mullen (2009)
		Field-based studies		
Terminalia arjuna (*Arjun*), *Acacia nilotica* (*Babool*), *Pongamia pinnata* (*Pongamia*), *Bauhinia variegata* (*Kachnar*), *Albizia lebbeck* (*Kala Siris*) *Pennisetum pedicellatum* (*Dinanath grass*) *Stylosanthes hamata*	19 months	Physiological	Jharkhand State, India	Chauhan and Ganguly (2011)
D. spicata	2.5 months, 1 year	Stem length, total cover	Alabama, US	Fuller *et al*. (1982)
A. elongatum	June–December	Plant cover biomass	Western Australia	Wong and Ho (1991), (1994*a*)
Cynodon dactylon		Root biomass	Texas, US	Khaitan *et al*. (2010)
Secale cereal, *L. rigidum*	5 weeks, 10 weeks	Yield nutrient content, yield nutrient content	Western Australia	Eastham and Morald (2006), Eastham *et al*. (2006)
T. pratense	11 months	Biomass, nutrient content	Ireland, Europe	Courtney *et al*. (2003)
L. perenne, *H. lanatus*	2 years	Biomass, nutrient content	Ireland, Europe	Courtney and Timpson (2005*a*)
T. pratense	11 months	Biomass, nutrient content, Al content	Ireland, Europe	Courtney and Timpson (2005*b*)
H. lanatus	10 months	Biomass, nutrient content, Al content	Ireland, Europe	Courtney and Harrington (2012)
H. lanatus, *T. pratense*	5 years	Nutrient content, Al content	Ireland, Europe	Courtney and Kirwan (2012)
Hardenbergia comptoniana, Acacia cyclops Grevillea crithmifolia, Eucalyptus gomphocephala	4 years	Nutrient content	Western Australia	Thiyagarajan *et al*. (2009)

(Meecham and Bell 1977*b*; Wong and Ho 1993), *D. spicata* (Fuller and Richardson 1986), *C. dactylon* (Woodard *et al.* 2008), *Atriplex* spp. (Woodard *et al.* 2008), and *Agropyron* spp. such as *A. smithii* (Fuller *et al*. 1982), *A. elongatum* (Fuller *et al*. 1982; Wong and Ho 1991), and *L. rigidum* and *Acacia saligna* (Jones *at al.* 2012*a*, 2012*b*).

Limiting factors

A wide range of factors can limit plant growth in bauxite residues, and tolerance of a plant species to one (e.g. salinity) does not guarantee a plant will grow successfully in residues. Khaitan *et al*. (2010) reported establishment of salt tolerant Bermuda grass (*C. dactylon*) on unamended residue and suggested its suitability for revegetation programs. Nevertheless, poor germination and seedling emergence for several other salt tolerant species have been attributed to high pH, ESP, and soluble Al content in the residues (Fuller *et al*. 1982; Wong and Ho 1993; Woodard *et al*. 2008; Finngean *et al.* 2018). Such limitations will be decreased following amendment of residues with gypsum and/or organic residues. Shi *et al*. (2017), for example, recorded oxidative stress indicators in *Atriplex canescens* growing in a range of bauxite residue treatments and noted these were decreased following gypsum application. Unsurprisingly, germination indices (% germination and relative root growth) for less salt-tolerant species (e.g. *L. sativum*, *Lolium perenne*) have been found to be negatively correlated with alkalinity, sodicity, and salinity (Courtney and Mullen 2009; Jones *et al*. 2012*a*, 2012*b*; Finngean *et al.* 2018) In addition, increasing extractable Ca content promoted seedling development for *L. perenne* and *T. pratense* in the study of Courtney and Mullen (2009) but there was no similar relationship for *Agropyron* spp. seedling emergence in the study of Wong and Ho (1993). This highlights the variability of plants in their sensitivity to particular factors and the difficulty in ascribing absolute threshold values for key parameters such as EC and ESP.

Poor physical properties in residues can also limit early growth of seedlings. For example, mechanical impedance to root penetration and low unsaturated hydraulic conductivity have been shown to result in poor seedling emergence of *C. gayana* in residue sand (Meecham and Bell 1977*b*).

Plant yield and nutrient content

The majority of plant growth trials in bauxite residues have been for relatively short periods (Table 2), and there is a definite need for longer-term growth trials. Poor plant yields have also been attributed to nutrient deficiencies, particularly N, P, and Mn (Meecham and Bell 1977*b*), and residue pH and EC were negatively correlated with leaf P content and biomass (Goloran *et al*. 2013). Although additions of organic matter and/or fertilisers are used to provide nutrients and improve plant performance (Table 6) (Fortin and Karam 1998; Eastham and Morald 2006; Eastham *et al*. 2006; Courtney and Harrington 2012; Banning *et al*. 2014), nutrient deficiencies often remain (Courtney and Timpson 2005*a*; Thiyagarajan *et al.* 2009).

The high exchangeable Na content in residues can result in high Na uptake into vegetation. Anderson *et al*. (2011) attributed the yellowing and scorching of tips and margins and lower biomass in *A. saligna* to Na toxicity. Conversely, Na content in *H. lanatus* was not excessive in gypsum-amended field plots (Courtney and Harrington 2012), and further decreases in plant Na content over time have been recorded (Courtney and Kirwan 2012). The ratio of residue mud to sand can also affect plant uptake of Na. Courtney and Timpson (2005*b*) reported lower Na content in *T. pratense* in field treatments with higher rates of sand application. Similarly, Jones *et al*. (2012*a*) reported increased Na content and decreased root mass in *A. saligna* when increased rates of residue fines were added to sand. Cation imbalances due to high Na levels in residue and from Ca additions in gypsum may also result in low K and Mg uptake (Eastham *et al*. 2006). Although low foliar K was observed for grassland growing on bauxite residues amended with sewage sludge (Courtney and Timpson 2005*a*), sufficient levels were recorded for grassland established following applications of spent mushroom compost (Courtney and Harrington 2012). Micronutrient deficiencies can also occur. For example, field monitoring of *Acacia cyclops* growing in residue sand revealed symptoms of nutrient deficiencies such as reduced leaf size, curling, crinkling, and malformation of leaves (Thiyagarajan *et al*. 2009) and leaf analysis revealed deficiencies of Mn and B.

Courtney and Timpson (2005*a*) recorded a significant decrease in vegetation biomass following a second year's growth on field-established vegetation treatments, and this was concomitant with decreased plant nutrient content. However, grassland vegetation established on bauxite residues were sustained for at least 9 years, and, although nutrient contents remained low, species indicative of seminatural grasslands replaced the more agriculturally dominant species such as *L. perenne* (Courtney *et al*. 2009*b*). Thus, nutrient availability can greatly influence the species composition of vegetation growing on residue.

Field approaches

Two main field approaches to bauxite residue revegetation have been reported in the literature. One method is to replicate the grassland approaches developed for other metalliferous mine wastes, whereas the other is to use native bush and tree species that blend into the surrounding environment. Williamson *et al*. (1982) highlighted the suitability of agricultural grassland species and amenity grasses for mine waste revegetation once sufficient amendments and nutrients have been applied. As long-term growth of vegetation depends on adequate N supply, inclusion of legume species that fix atmospheric N_2 is also an important consideration. The grassland species *L. perenne*, *H. lanatus*, and *T. pratense* have all been established in a range of field trials (Courtney *et al*. 2003, 2013; Courtney and Timpson 2005*a*).

In a field study in Ireland, four grass species and two legumes were seeded onto bauxite residue amended with gypsum and spent mushroom compost. A vegetation survey 10 years later showed that the bauxite residues supported a variety of plant types and species. Though the initial grassland seed mixes comprised only 6 species, the sites supported a much more diverse plant community comprising 47 species belonging to

38 genera and 15 families (Courtney *et al*. 2009*b*). Asteraceae and Poaceae were the dominant families with 14 and 9 species respectively. In general, species that invade minelands are those with effective seed dispersal mechanisms, high seed production rates in the local area, and show tolerance to the chemical and physical soil conditions present.

Gautam and Agrawal (2017) examined plant species colonising a residue disposal area that had been abandoned 18 years previously and identified 37 herbaceous, 8 woody, and 9 shrub species that had colonised the residue area. The site was dominated by *C. dactylon* and, compared with surrounding land, it was concluded that the site was species poor with many native species absent. The pollution-tolerant, halophytic, and alkaliphilic species, *Stylosanthes scabra*, was only found on the residue site. Less diversity of herbaceous and woody species on the residue site may have been due to higher soil alkalinity, salinity, exchangeable cations, poor physical properties, and poor phytoavailable levels of trace elements as well as lower nutrient levels. Similarly, drought-resistant species, *Albizia lebbeck* and *Acacia nilotica*, dominated rehabilitated residues in India (Chauhan and Ganguly 2011).

At Alcoa in Western Australia, native plants from a coastal sand dune system were used for revegetation of residue sand (Phillips 2014*a*, 2015*a*). A seed mix of 55 plant species is employed and, in addition, seedlings of species that cannot become established from seed are planted by hand. However, in terms of residue rehabilitation performance, a general finding from these studies was that of the >60 species used in rehabilitation since the early-2000s, <20 species dominate the rehabilitation species make-up within 10 years. The remaining >40 species were present in only small densities or had disappeared entirely. Furthermore, vegetation cover decreased from ~50% within 2 years of rehabilitation establishment to <40% within 10 years. The decline in species composition may have been related to stress factors due to limited water availability, emphasising the importance of the properties of residue to provide long-term support of the vegetation cover. The change in species mix with time requires redefining of the 'target' ecosystem. For example, the initial species mix at Alcoa's sites target a 'coastal dune' ecosystem; however, within 10 years the rehabilitation best reflected a 'scrub–shrubland–heath' ecosystem. Clearly, the initial species mix is a dynamic property of residue rehabilitation. Long-term monitoring is essential in evaluating rehabilitation performance and sustainability and in providing excellent guidance to selecting and refining the initial species mix, which will determine the long-term trajectory of the steady-state ecosystem.

Conclusions and future research

This review has highlighted recommendations in addition to those proposed by Gräfe and Klauber (2011) and Haynes (2015) for developing success criteria for assessing bauxite residue rehabilitation. Critical to supporting plant cover on bauxite residue is the wealth of information on approaches to decreasing the high pH of residues. In the main, these have been relatively short-term studies but indicate that gypsum additions result in a sustained pH decrease to <9. Decreases in salinity also occur to levels that do not pose salinity stresses, but high exchangeable Na (ESP) may persist. Refinement of methods for assessing residue ESP and calibration of various nutrient soil test extraction methods to plant yields and plant nutrient uptake *in situ* in bauxite residue are warranted.

The suggested soil criteria of Mendez and Maier (2008) of improved aggregation and reduced erosion and runoff have been well studied for bauxite residues, but the requirement for reduced metal bioavailability and mobility is considerably less well defined. Some recent work has investigated the solubility and availability of potentially toxic elements in residues but most of these have been at the pot-based level and there are few data from revegetated sites under field conditions. Threshold guideline values for heavy metals and metalloids in bauxite residues are not currently available, and these need to be determined.

Soil microbial diversity and function in residues has been well reported in both laboratory and field studies, and such studies highlight the need for amendments (e.g. gypsum and organic composts) to confer optimum soil conditions for soil microbiota. Where such rehabilitation is practiced, a microbial community trajectory towards that of analogous soil conditions is possible. Soil faunal colonisation and activity in mine wastes are increasingly being studied, but there is very little information reported for bauxite residues and this is an area where further research is required.

Mendez and Maier (2008) recommend that plant biomass and percent cover in rehabilitated tailings should be comparable to that of undisturbed sites, but this is an unrealistic target where fully productive agricultural land surrounds the residue area. Nevertheless, a stable vegetative ecosystem that achieves soil stabilisation is required. A major objective in revegetation of amended tailings can be that plants stabilise metals in the root zone and there is little to no shoot accumulation (Gil-Loaiza *et al*. 2016). Most bauxite residue studies have only investigated plant nutrient contents (and sometimes Al content). The recommendations that shoot metal concentrations do not exceed domestic animal toxicity limits (Mendez and Maier 2008) have not been investigated in bauxite residue revegetation, and this is an area requiring future attention. This is particularly the case in light of the more recent publications highlighting the As, Hg, and V risks from residues when poorly managed (Mišík *et al*. 2014; Olszewska *et al*. 2016).

The requirement for self-propagation of introduced plant species (Mendez and Maier 2008) has not been investigated in bauxite residues. Although a limited number of studies have shown the ability of other plant species to colonise revegetated areas, the long-term assessment (>10 years) is lacking and highlights the need to further examine the sustainability of revegetated residues. Furthermore, the ability of the introduced plant community to withstand and recover from stress such as fire, drought, waterlogging, and predation in rehabilitated residues is unknown.

Areas requiring attention

Targeted studies are required to translate laboratory-based research findings to the field level. Long-term field trials that are properly monitored for many decades will be required to answer most of the unanswered questions with regard to bauxite residue revegetation. Indeed, whether revegetation efforts instituted today are sustainable or not will be determined by future generations.

Long-term field trials should be designed to target the following.

- The availability and plant content of key nutrients. The potential for severe deficiencies in Mn and P has been highlighted in several studies but field performance is largely unknown.
- The development of target soil test values for macro- and micronutrients calibrated in revegetating bauxite residue for the plant being used for revegetation.
- The availability and plant content of potentially toxic heavy metals and metalloids.
- Determination of threshold levels of these elements for plants and microbes living in bauxite residue.
- The risk of transfer of such metals and metalloids to grazing animals and wildlife.
- Soil faunal colonisation activity and their effects on ecosystem function.
- The effectiveness of inoculation of soil fauna on ecosystem function.
- Ability of introduced plant communities to withstand environmental stresses.
- Capacity of introduced plant communities for self-propagation.

Geochemical transformation of residue properties will occur via mechanisms such as weathering, mineral dissolution and precipitation, ion exchange, redox, organic matter complexation, and rhizosphere-induced changes. Assessment of key criteria over time will contribute to form the basis for eco-hydrological models combining the water balance, water and chemical transport, and geochemical speciation modelling. This will provide a basic theoretical framework for predicting key quality criteria for rehabilitated BRDAs and demonstrating ultimate success.

Conflict of interest

The authors declare no conflicts of interest.

Acknowledgements

This study was partly supported by grants by Science Foundation Ireland (SFI 17/CDA/4778) and authors E. Di Carlo and R. Courtney would also like to acknowledge the support of the International Aluminium Institute, Rio Tinto and Alteo.

References

Anderson JD, Bell RW, Phillips IR (2011) Bauxite residue fines as an amendment to residue sands to enhance plant growth potential—a glasshouse study. *Journal of Soils and Sediments* **11**, 889–902. doi:10.1007/s11368-011-0379-7

Arslan S, Demir G, Çelikel B, Baygül M (2018) Implementation and optimization of filter press in red mud washing process at Eti Aluminium, Travaux 47. In 'Proceedings of the 36th International ICSOBA Conference', 29 October–1 November 2018, Belem, Brazil.

Asensio V, Vega FA, Andrade ML, Covelo EF (2013) Tree vegetation and waste amendments to improve the physical condition of copper mine soils. *Chemosphere* **90**, 603–610. doi:10.1016/j.chemosphere.2012.08.050

Babu AG, Reddy MS (2011) Influence of arbuscular mycorrhizal fungi on the growth and nutrient status of bermudagrass grown in alkaline bauxite processing residue. *Environmental Pollution* **159**, 25–29. doi:10.1016/j.envpol.2010.09.032

Banning NC, Phillips IR, Jones DL, Murphy DV (2011) Development of microbial diversity and functional potential in bauxite residue sand under rehabilitation. *Restoration Ecology* **19**, 78–87. doi:10.1111/j.1526-100X.2009.00637.x

Banning NC, Sawada Y, Phillips IR, Murphy DV (2014) Amendment of bauxite residue sand can alleviate constraints to plant establishment and nutrient cycling capacity in a water-limited environment. *Ecological Engineering* **62**, 179–187. doi:10.1016/j.ecoleng.2013.10.034

Bell LC, Jones CJ (1987) Rehabilitation of base metal tailings – Australian case study. *Mining Rehabilitation* **87**, 88–102.

Bendfeldt ES, Burger JA, Daniels WL (2001) Quality of amended mine soils after sixteen years. *Soil Science Society of America Journal* **65**, 1736–1744. doi:10.2136/sssaj2001.1736

Biederman LA, Boutton TW, Whisenant SG (2008) Nematode community development early in ecological restoration: the role of organic amendments. *Soil Biology & Biochemistry* **40**, 2366–2374. doi:10.1016/j.soilbio.2008.05.017

Binkley D, Hart SC (1989) The components of nitrogen availability assessments in forest soils. In 'Advances in soil science'. (Ed. BA Stewart) pp. 57–112. (Springer: New York)

Brautigan DJ, Rengasamy P, Chittleborough DJ (2012) Aluminium speciation and phytotoxicity in alkaline soils. *Plant and Soil* **360**, 187–196. doi:10.1007/s11104-012-1232-5

Bray AB, Stewart DI, Courtney R, Rout SP, Humphreys PN, Mayes WM, Burke IT (2018) Sustained bauxite residue rehabilitation with gypsum and organic matter 16 years after initial treatment. *Environmental Science & Technology* **52**, 152–161. doi:10.1021/acs.est.7b03568

Buchanan SJ, So HB, Kopottke PM, Menzies NW (2010) Influence of texture in bauxite residues on void ratio, water holding characteristics, and penetration resistance *Geoderma* **158**, 421–426. doi:10.1016/j.geoderma.2010.06.016

Burke IT, Mayes WM, Peacock CL, Brown AP, Jarvis AP, Gruiz K (2012) Speciation of arsenic, chromium, and vanadium in red mud samples from the Ajka spill site, Hungary. *Environmental Science & Technology* **46**, 3085–3092. doi:10.1021/es3003475

Burke IT, Peacock CL, Lockwood CL, Stewart DI, Mortimer RJ, Ward MB, Renforth P, Gruiz K, Mayes WM (2013) Behavior of aluminum, arsenic, and vanadium during the neutralization of red mud leachate by HCl, gypsum, or seawater. *Environmental Science & Technology* **47**, 6527–6535. doi:10.1021/es4010834

Carter CM, Van der Sloot HA, Cooling D (2009) pH-dependent extraction of soil and soil amendments to understand the factors controlling element mobility. *European Journal of Soil Science* **60**, 622–637. doi:10.1111/j.1365-2389.2009.01139.x

Chauhan S, Ganguly A (2011) Standardizing rehabilitation protocol using vegetation cover for bauxite waste (red mud) in eastern India. *Ecological Engineering* **37**, 504–510. doi:10.1016/j.ecoleng.2010.12.017

Chen CR, Phillips IR, Wei LL, Xu ZH (2010*a*) Behaviour and dynamics of diammonium phosphate in bauxite processing sand in Western Australia – I. NH_3 volatilization and residual nitrogen availability. *Environmental Science and Pollution Research International* **17**, 1098–1109. doi:10.1007/s11356-009-0267-5

Chen CR, Phillips IR, Wei LL, Xu ZH (2010*b*) Behaviour and dynamics of diammonium phosphate in bauxite processing sand in Western Australia – II. Phosphorus fractions and availability. *Environmental Science and*

Pollution Research International **17**, 1110–1118. doi:10.1007/s11356-009-0268-4

Clohessy J (2015) Closure and rehabilitation of Rusal Aughinish BRDA, presented at the Residue Closure and Rehabilitation Workshop, 10th Alumina Quality Workshop, Perth, Western Australia.

Cooling DJ, Hay PS, Guilfoyle L (2002) Carbonation of bauxite residue. In 'Proceedings of the 6th international alumina quality workshop', pp. 185–190. (AQW Inc.: Brisbane)

Courtney R (2013) Mine tailings composition in a historic site: implications for ecological restoration. *Environmental Geochemistry and Health* **35**, 79–88. doi:10.1007/s10653-012-9465-z

Courtney R, Harrington T (2010) Assessment of plant-available phosphorus in a fine textured sodic substrate. *Ecological Engineering* **36**, 542–547. doi:10.1016/j.ecoleng.2009.12.001

Courtney R, Harrington T (2012) Growth and nutrition of *Holcus lanatus* in bauxite residue amended with combinations of spent mushroom compost and gypsum. *Land Degradation & Development* **23**, 144–149. doi:10.1002/ldr.1062

Courtney R, Kirwan L (2012) Gypsum amendment of alkaline bauxite residue–plant available aluminium and implications for grassland restoration. *Ecological Engineering* **42**, 279–282. doi:10.1016/j.ecoleng.2012.02.025

Courtney RG, Mullen GJ (2008) Soil quality and barley growth as influenced by the land application of two compost types. *Bioresource Technology* **99**, 2913–2918. doi:10.1016/j.biortech.2007.06.034

Courtney R, Mullen G (2009) Use of germination and seedling performance bioassays for assessing revegetation strategies on bauxite residue. *Water, Air, and Soil Pollution* **197**, 15–22. doi:10.1007/s11270-008-9787-8

Courtney RG, Timpson JP (2005*a*) Nutrient status of vegetation grown in alkaline bauxite processing residue amended with gypsum and thermally dried sewage sludge – A two year field study. *Plant and Soil* **266**, 187–194. doi:10.1007/s11104-005-0872-0

Courtney RG, Timpson JP (2005*b*) Reclamation of fine fraction bauxite processing residue (red mud) amended with coarse fraction residue and gypsum. *Water, Air, and Soil Pollution* **164**, 91–102. doi:10.1007/s11270-005-2251-0

Courtney R, Timpson JP, Grennan E (2003) Growth of *Trifolium pratense* in red mud amended with process sand, gypsum and thermally dried sewage sludge. *International Journal of Surface Mining Reclamation and Environment* **17**, 227–233. doi:10.1076/ijsm.17.4.227.17481

Courtney RG, Jordan SN, Harrington T (2009*a*) Physico-chemical changes in bauxite residue following application of spent mushroom compost and gypsum. *Land Degradation & Development* **20**, 572–581. doi:10.1002/ldr.926

Courtney R, Mullen G, Harrington T (2009*b*) An evaluation of revegetation success on bauxite residue. *Restoration Ecology* **17**, 350–358. doi:10.1111/j.1526-100X.2008.00375.x

Courtney R, Keith AM, Harrington T (2011) Nematode assemblages in bauxite residue with different restoration histories. *Restoration Ecology* **19**, 758–764. doi:10.1111/j.1526-100X.2010.00734.x

Courtney R, Harrington T, Byrne KA (2013) Indicators of soil formation in restored bauxite residues. *Ecological Engineering* **58**, 63–68. doi:10.1016/j.ecoleng.2013.06.022

Courtney R, Harris JA, Pawlett M (2014*a*) Microbial community composition in a rehabilitated bauxite residue disposal area: a case study for improving microbial community composition. *Restoration Ecology* **22**, 798–805. doi:10.1111/rec.12143

Courtney R, Feeney E, O'Grady A (2014*b*) An ecological assessment of rehabilitated bauxite residue. *Ecological Engineering* **73**, 373–379. doi:10.1016/j.ecoleng.2014.09.064

Courtney R, Di Carlo E, Schmidt O (2018) Soil properties and earthworm populations associated with bauxite residue rehabilitation strategies. *Environmental Science and Pollution Research International* doi:10.1007/s11356-018-3973-z

Cristol B, Greenhalgh R (2018) QAL bauxite residue storage using sea water neutralisation. 2nd Bauxite Residue Valorisation and Best Practices Conference, Athens, 7–10 May 2018.

Cusack PB, Healy MG, Ryan PC, Burke IT, O'Donoghue LM, Ujaczki É, Courtney R (2018) Enhancement of bauxite residue as a low-cost adsorbent for phosphorus in aqueous solution, using seawater and gypsum treatments. *Journal of Cleaner Production* **179**, 217–224. doi:10.1016/j.jclepro.2018.01.092

Dobrowolski MP, Mullins RG, Phillips IR (2009) Factors affecting plant root distribution in sand embankments of bauxite residue disposal areas. Environmental Department Research Note 29, Alcoa of Australia. January 2009.

Eastham J, Morald T (2006) Effective nutrient sources for plant growth on bauxite residue: II. Evaluating the response to inorganic fertilizers. *Water, Air, and Soil Pollution* **171**, 315–331. doi:10.1007/s11270-005-9055-0

Eastham J, Morald T, Aylmore P (2006) Effective nutrient sources for plant growth on bauxite residue. *Water, Air, and Soil Pollution* **176**, 5–19. doi:10.1007/s11270-005-8870-7

Esfandbod M, Phillips IR, Miller B, Rezaei Rashti M, Lan ZM, Srivastava P, Singh B, Chen CR (2017) Aged acidic biochar increases nitrogen retention and decreases ammonia volatilization in alkaline bauxite residue sand. *Ecological Engineering* **98**, 157–165. doi:10.1016/j.ecoleng.2016.10.077

Evans K (2016) The history, challenges, and new developments in the management and use of bauxite residue. *Journal of Sustainable Metallurgy* **2**, 316–331. doi:10.1007/s40831-016-0060-x

Finngean G, O'Grady A, Courtney R (2018) Plant assays and avoidance tests with collembola and earthworms demonstrate rehabilitation success in bauxite residue. *Environmental Science and Pollution Research International* **25**, 2157–2166. doi:10.1007/s11356-017-0632-8

Fortin J, Karam A (1998) Effect of a commercial peat moss-shrimp wastes compost on p*uccinellia* growth in red mud. I*nternational Journal of Surface Mining, Reclamation and Environment* **12**, 105–109. doi:10.1080/09208118908944032

Frouz J, Elhottová D, Kuráž V, Šourková M (2006) Effects of soil macrofauna on other soil biota and soil formation in reclaimed and unreclaimed post mining sites: results of a field microcosm experiment. *Applied Soil Ecology* **33**, 308–320. doi:10.1016/j.apsoil.2005.11.001

Frouz J, Elhottova D, Pižl V, Tajovský K, Šourková M, Picek T, Malý S (2007) The effect of litter quality and soil faunal composition on organic matter dynamics in post-mining soil: a laboratory study. *Applied Soil Ecology* **37**, 72–80. doi:10.1016/j.apsoil.2007.04.001

Fuller RD, Richardson CJ (1986) Aluminate toxicity as a factor controlling plant growth in bauxite residue. *Environmental Toxicology and Chemistry* **5**, 905–915. doi:10.1002/etc.5620051007

Fuller RD, Nelson EDP, Richardson CJ (1982) Reclamation of red mud (bauxite residues) using alkaline-tolerant grasses with organic amendments. *Journal of Environmental Quality* **11**, 533–539. doi:10.2134/jeq1982.00472425001100030040x

Gautam M, Agrawal M (2017) Phytoremediation of metals using vetiver (*Chrysopogon zizanioides* (L.) *Roberty*) grown under different levels of red mud in sludge amended soil. *Journal of Geochemical Exploration* **182**, 218–227. doi:10.1016/j.gexplo.2017.03.003

Gherardi MJ, Rengel Z (2001) Bauxite residue sand has the capacity to rapidly decrease availability of added manganese. *Plant and Soil* **234**, 143–151. doi:10.1023/A:1017992529531

Gil-Loaiza J, White SA, Root RA, Solís-Dominguez FA, Hammond CM, Chorover J, Maier RM (2016) Phytostabilization of mine tailings using compost-assisted direct planting: translating greenhouse results to the field. *The Science of the Total Environment* **565**, 451–461. doi:10.1016/j.scitotenv.2016.04.168

Goloran J, Chen CR, Phillips IR, Xu ZH, Condron LM (2013) Selecting a nitrogen availability index for understanding plant nutrient dynamics in rehabilitated bauxite-processing residue sand. *Ecological Engineering* **58**, 228–237. doi:10.1016/j.ecoleng.2013.07.004

Goloran J, Phillips IR, Chen C (2014*a*) Forms of nitrogen alter plant phosphorus uptake and pathways in rehabilitated highly alkaline bauxite processing residue sand. *Land Degradation & Development* **28**, 628–637. doi:10.1002/ldr.2630

Goloran J, Phillips IR, Xu ZH, Condron LM, Chen C (2014*b*) Effects of amendments and fertilization on plant growth, nitrogen and phosphorus availability in rehabilitated highly alkaline bauxite-processing residue sand. *Soil Use and Management* **30**, 198–208.

Goloran JB, Chen CR, Phillips IR, Liu X (2015*a*) Pathways of different forms of nitrogen and role of ammonia-oxidizing bacteria in alkaline residue sand from bauxite processing. *European Journal of Soil Science* **66**, 942–950. doi:10.1111/ejss.12274

Goloran J, Phillips IR, Chen C, Elser JJ (2015*b*) Shifts in leaf N : P stoichiometry during rehabilitation in highly alkaline bauxite processing residue sand. *Scientific Reports* **5**, 1–12.

Goloran JB, Phillips IR, Chen C (2017) Forms of nitrogen alter plant phosphorus uptake and pathways in rehabilitated highly alkaline bauxite processing residue sand. *Land Degradation & Development* **28**, 628–637. doi:10.1002/ldr.2630

Gräfe M, Klauber C (2011) Bauxite residue issues: IV. Old obstacles and new pathways for in situ residue bioremediation. *Hydrometallurgy* **108**, 46–59. doi:10.1016/j.hydromet.2011.02.005

Gräfe M, Power G, Klauber C (2011) Bauxite residue issues: III. Alkalinity and associated chemistry. *Hydrometallurgy* **108**, 60–79.

Gupta RK, Abrol IP (1990) Salt affected soils: their reclamation and management for crop production *Advances in Soil Science* **11**, 224–288.

Gwenzi W, Hinz C, Holmes K, Phillips IR, Mullins IJ (2011) Field-scale spatial variability of saturated hydraulic conductivity on a recently constructed artificial ecosystem. *Geoderma* **166**, 43–56. doi:10.1016/j.geoderma.2011.06.010

Hallett PD, Feeney DS, Bengough AG, Rillig MC, Scrimgeour CM, Young IM (2009) Disentangling the impact of AM fungi versus roots on soil structure and water transport. *Plant and Soil* **314**, 183–196. doi:10.1007/s11104-008-9717-y

Hamdy MK, Williams FS (2001) Bacterial amelioration of bauxite residue waste of industrial alumina plants. *Journal of Industrial Microbiology & Biotechnology* **27**, 228–233. doi:10.1038/sj.jim.7000181

Hanahan C, McConchie D, Pohl J, Creelman R, Clark M, Stocksiek C (2004) Chemistry of seawater neutralization of bauxite refinery residues (red mud) *Environmental Engineering Science* **21**, 125–138. doi:10.1089/109287504773087309

Haynes RJ (1999) Labile organic matter fractions and aggregate stability under short-term, grass-based leys. *Soil Biology & Biochemistry* **31**, 1821–1830. doi:10.1016/S0038-0717(99)00102-9

Haynes RJ (2014) Nature of the belowground ecosystem and its development during pedogenesis. *Advances in Agronomy* **127**, 43–109. doi:10.1016/B978-0-12-800131-8.00002-9

Haynes RJ (2015) Sustainable vegetation. In 'Proceedings of the 10th International Alumina Quality Workshop, Post Conference Workshop – Maintaining a flexible approach to closure and rehabilitation of residue storage areas.' 19–24 April 2015, Perth, Australia.

Higgins D, Curtin T, Pawlett M, Courtney R (2016) The potential for constructed wetlands to treat alkaline bauxite-residue leachate: *Phragmites australis* growth. *Environmental Science and Pollution Research International* **23**, 24305–24315. doi:10.1007/s11356-016-7702-1

Horn R, Taubner H, Wuttke M, Baumgartl T (1994) Soil physical properties related to soil structure. *Soil & Tillage Research* **30**, 187–216. doi:10.1016/0167-1987(94)90005-1

Huang L, Baumgartl T, Mulligan D (2012) Is rhizosphere remediation sufficient for sustainable revegetation of mine tailings? *Annals of Botany* **110**, 223–238. doi:10.1093/aob/mcs115

Ippolito JA, Redente EF, Barbaick KA (2005) Amendment effects on pH and salt content of bauxite residue *Soil Science* **170**, 832–841. doi:10.1097/01.ss.0000190510.56545.8d

Jones BE, Haynes RJ (2011) Bauxite processing residue: a critical review of its formation, properties, storage and revegetation *Critical Reviews in Environmental Science and Technology* **41**, 271–315. doi:10.1080/10643380902800000

Jones BE, Haynes RJ, Phillips IR (2010) Effect of amendment of bauxite processing sand with organic materials on its chemical, physical and microbial properties *Journal of Environmental Management* **91**, 2281–2288. doi:10.1016/j.jenvman.2010.06.013

Jones BE, Haynes RJ, Phillips IR (2011) Influence of organic waste and residue mud additions on chemical, physical and microbial properties of bauxite residue sand *Environmental Science and Pollution Research International* **18**, 199–211. doi:10.1007/s11356-010-0364-5

Jones BE, Haynes RJ, Phillips IR (2012*a*) Cation and anion leaching and growth of *Acacia saligna* in bauxite residue sand amended with residue mud, poultry manure and phosphogypsum *Environmental Science and Pollution Research International* **19**, 835–846. doi:10.1007/s11356-011-0630-1

Jones BE, Haynes RJ, Phillips IR (2012*b*) Addition of an organic amendment and/or residue mud to bauxite residue sand in order to improve its properties as a growth medium *Journal of Environmental Management* **95**, 29–38. doi:10.1016/j.jenvman.2011.09.014

Jones BE, Haynes RJ, Phillips IR (2015) Influence of amendments on acidification and leaching of Na from bauxite processing sand. *Ecological Engineering* **84**, 435–442. doi:10.1016/j.ecoleng.2015.09.054

Kaur N, Phillips I, Fey MV (2016) Amelioration of bauxite residue sand by intermittent additions of nitrogen fertiliser and leaching fractions: the effect on growth of kikuyu grass and fate of applied nutrients. *The Science of the Total Environment* **550**, 362–371. doi:10.1016/j.scitotenv.2016.01.012

Keren R (2000) Salinity. In 'Salinity'. (Ed. ME Summer) pp. G3-G25. (CRC Press: Boca Raton, FL, USA)

Khaitan S, Dzombak DA, Swallow P, Schmidt K, Fu J, Lowry GV (2010) Field evaluation of bauxite residue neutralization by carbon dioxide, vegetation, and organic amendments. *Journal of Environmental Engineering* **136**, 1045–1053. doi:10.1061/(ASCE)EE.1943-7870.0000230

Kinraide TB (1999) Interactions among Ca^{2+}, Na^{+} and K^{+} in salinity toxicity: quantitative resolution of multiple toxic and ameliorative effects. *Journal of Experimental Botany* **50**, 1495–1505. doi:10.1093/jxb/50.338.1495

Klauber C, Gräfe M, Power G (2011) Bauxite residue issues: II. Options for residue utilization. *Hydrometallurgy* **108**, 11–32. doi:10.1016/j.hydromet.2011.02.007

Klebercz O, Mayes WM, Anton ÁD, Feigl V, Jarvis AP, Gruiz K (2012) Ecotoxicity of fluvial sediments downstream of the Ajka red mud spill, Hungary. *Journal of Environmental Monitoring* **14**, 2063–2071. doi:10.1039/c2em30155e

Kong X, Guo Y, Xue S, Hartley W, Wu C, Ye Y, Cheng Q (2017) Natural evolution of alkaline characteristics in bauxite residue. *Journal of Cleaner Production* **143**, 224–230. doi:10.1016/j.jclepro.2016.12.125

Kopittke PM, Menzies NW (2004) Effect of Mn deficiency and legume inoculation on rhizosphere pH in highly alkaline soils. *Plant and Soil* **262**, 13–21. doi:10.1023/B:PLSO.0000037023.18127.7a

Kopittke PM, Menzies NW (2005) Effect of pH on Na induced Ca deficiency *Plant and Soil* **269**, 119–129. doi:10.1007/s11104-004-0395-0

Kopittke PM, Menzies NW, Blamey FPC (2005) Rhizotoxicity of aluminate and polycationic aluminium at high pH *Plant and Soil* **266**, 177–186. doi:10.1007/s11104-005-2229-0

Krishna P, Reddy MS, Patnaik SK (2005) *Aspergillus tubingensis* reduces the pH of the bauxite residue (red mud) amended soils. *Water, Air, and Soil Pollution* **167**, 201–209. doi:10.1007/s11270-005-0242-9

Kumar K, Goh KM (1999) Crop residues and management practices: effects on soil quality, soil nitrogen dynamics, crop yield, and nitrogen recovery. *Advances in Agronomy* **68**, 197–319. doi:10.1016/S0065-2113(08)60846-9

Larson WE, Pierce FJ (1991) Conservation and enhancement of soil quality. In 'Evaluation for sustainable land management in the developing world: proceedings of the International Workshop on Evaluation for Sustainable Land Management in the Developing World', Chiang Rai, Thailand, 15–21 September 1991.

Lauchli A, Grattan S (2007) Plant growth and development under salinity stress. In 'Advances in molecular breeding toward drought and salt tolerant crops'. (Eds MA Jenks, PM Hasegawa, JS Mohan) pp. 1–32. (Springer: Berlin)

Lehoux AP, Lockwood CL, Mayes WM, Stewart DI, Mortimer RJ, Gruiz K, Burke IT (2013) Gypsum addition to soils contaminated by red mud: implications for aluminium, arsenic, molybdenum and vanadium solubility. *Environmental Geochemistry and Health* **35**, 643–656. doi:10.1007/s10653-013-9547-6

Li Y (2017) Revegetation strategies for seawater neutralized bauxite residue at Rio Tinto Alcan using various amendments. PhD thesis, University of Queensland, Brisbane, Qld.

Li Y, Haynes RJ (2017) Formation, properties and revegetation prospects for bauxite processing residue and the effects of seawater neutralization. *International Journal of Environmental Engineering* **9**, 11–39. doi:10.1504/IJEE.2017.087984

Li Y, Haynes RJ, Zhou YF, Chandrawana I (2016) Chemical and physical properties of seawater-neutralized bauxite residue mud and sand and the effects of leaching. *International Journal of Environmental Sciences and Development* **7**, 273–277. doi:10.7763/IJESD.2016.V7.783

Li Y, Haynes RJ, Chandrawana I, Zhou YF (2018*a*) Properties of seawater neutralized bauxite residues and changes in chemical, physical and microbial properties induced by additions of gypsum and organic matter. *Journal of Environmental Management* **223**, 489–494. doi:10.1016/j.jenvman.2018.06.070

Li Y, Haynes RJ, Chandrawana I, Zhou YF (2018*b*) Increased leaching and addition of amendments improves the properties of seawater neutralized bauxite residue as a growth medium. *Environmental Science and Pollution Research International* **25**, 25476–25485. doi:10.1007/s11356-018-2564-3

Lockwood CL, Mortimer RJ, Stewart DI, Mayes WM, Peacock CL, Polya DA, Lythgoe PR, Lehoux AP, Gruiz K, Burke IT (2014) Mobilisation of arsenic from bauxite residue (red mud) affected soils: effect of pH and redox conditions. *Applied Geochemistry* **51**, 268–277. doi:10.1016/j.apgeochem.2014.10.009

López E, Soto B, Arias M, Nunez A, Rubinos D, Barral MT (1998) Adsorbent properties of red mud and its use for wastewater treatment. *Water Research* **32**, 1314–1322. doi:10.1016/S0043-1354(97)00326-6

Mayes WM, Jarvis AP, Burke IT, Walton M, Feigl V, Klebercz O, Gruiz K (2011) Dispersal and attenuation of trace contaminants downstream of the Ajka bauxite residue (red mud) depository failure, Hungary. *Environmental Science & Technology* **45**, 5147–5155. doi:10.1021/es200850y

McNearny RL, Wheeler KR (1995) Knight Mine reclamation: a study of revegetation difficulties in a semi-arid environment. *International Journal of Mining, Reclamation and Environment* **9**, 113–119. doi:10.1080/09208119508964730

Meecham JR, Bell LC (1977*a*) Revegetation of alumina refinery wastes. 1. Properties and amelioration of the materials. *Australian Journal of Experimental Agriculture* **17**, 679–688. doi:10.1071/EA9770679

Meecham JR, Bell LC (1977*b*) Revegetation of alumina refinery wastes. 2. Glasshouse experiments. *Australian Journal of Experimental Agriculture* **17**, 689–695. doi:10.1071/EA9770689

Mendez MO, Maier RM (2008) Phytostabilization of mine tailings in arid and semiarid environments: an emerging remediation technology. *Environmental Health Perspectives* **116**, 278–283. doi:10.1289/ehp.10608

Menzies NW, Fulton IM, Morrell WJ (2004) Seawater neutralization of alkaline bauxite residue and implications for revegetation. *Journal of Environmental Quality* **33**, 1877–1884. doi:10.2134/jeq2004.1877

Menzies NW, Fulton IM, Kopittke RA, Kopittke PM (2009) Fresh water leaching of alkaline bauxite residue after sea water neutralization. *Journal of Environmental Quality* **38**, 2050–2057. doi:10.2134/jeq2008.0511

Mišík M, Burke IT, Reismüller M, Pichler C, Rainer B, Mišíková K, Mayes WM, Knasmueller S (2014) Red mud a byproduct of aluminum production contains soluble vanadium that causes genotoxic and cytotoxic effects in higher plants. *The Science of the Total Environment* **493**, 883–890. doi:10.1016/j.scitotenv.2014.06.052

Nikraz HR, Bodley AJ, Cooling DJ, Kong PYL, Soomro M (2007) Comparison of physical properties between treated and untreated bauxite residue mud. *Journal of Materials in Civil Engineering* **19**, 2–9. doi:10.1061/(ASCE)0899-1561(2007)19:1(2)

Olszewska JP, Meharg AA, Heal KV, Carey M, Gunn ID, Searle KR, Winfield IJ, Spears BM (2016) Assessing the legacy of red mud pollution in a shallow freshwater lake: arsenic accumulation and speciation in macrophytes. *Environmental Science & Technology* **50**, 9044–9052. doi:10.1021/acs.est.6b00942

Palmer S, Frost R, Nguyen T (2009) Hydrotalcites and their role in coordination of anions in Bayer liquors: anion binding in layered double hydroxides *Coordination Chemistry Reviews* **253**, 250–267. doi:10.1016/j.ccr.2008.01.012

Phillips IR (2010*a*) Procedures for measuring hydraulic properties of residue sand and implications for store-release covers in residue storage area closure design. Environmental Department Technical Series No. 7, Alcoa of Australia. October 2010.

Phillips IR (2010*b*) Evaluating techniques for reducing pH of bauxite processing residue sand at depth using gypsum and irrigation. Environmental Department Research Bulletin 37, Alcoa of Australia. August 2010.

Phillips IR (2014*a*) Pore water ionic composition and ion exchange equilibria of residue sand under rehabilitation. Environmental Department Technical Series No. 16, Alcoa of Australia. May 2014.

Phillips IR (2014*b*) Viability, germination and seedling establishment using seeds produced on rehabilitated residue sand embankments. Environmental Department Research Note No. 37, Alcoa of Australia. March 2014.

Phillips IR (2015*a*) Botanical monitoring of residue rehabilitation established in 2012: General rehabilitation, fenced trial and residue sand/mud mixture. Environmental Department Technical Series No. 17, Alcoa of Australia. August 2015.

Phillips IR (2015*b*) Nutrient and gypsum transport in residue sand profiles under rehabilitation – a field and column leaching study. Environmental Department Technical Series No. 19, Alcoa of Australia. October 2015.

Phillips IR, Chen C (2010) Surface charge characteristics and sorption properties of bauxite-processing residue sand. *Australian Journal of Soil Research* **48**, 77–87. doi:10.1071/SR09056

Power G, Gräfe M, Klauber C (2011) Bauxite residue issues: I. Current management, disposal and storage practices. *Hydrometallurgy* **108**, 33–45. doi:10.1016/j.hydromet.2011.02.006

Qadir M, Schubert S (2002) Degradation processes and nutrient constraints in sodic soils. *Land Degradation & Development* **13**, 275–294. doi:10.1002/ldr.504

Ren J, Chen J, Han L, Wang M, Yang B, Du P, Li F (2018) Spatial distribution of heavy metals, salinity and alkalinity in soils around bauxite residue disposal area. *The Science of the Total Environment* **628–629**, 1200–1208. doi:10.1016/j.scitotenv.2018.02.149

Residue Solutions (2007) Residue management sustainability review: Aughinish Alumina Limited. Available at http://www.epa.ie/licences/lic_eDMS/090151b280195475.pdf [verified 7 October 2018].

Rezaei Rashti M, Esfandbod M, Phillips IR, Chen C (2019) Biochar amendment and water stress alter rhizosphere carbon and nitrogen budgets in bauxite-processing residue sand under rehabilitation. *Journal of Environmental Management* **230**, 446–455. doi:10.1016/j.jenvman.2018.09.093

Rodriguez MB, Godeas A, Lavado RS (2008) Soil acidity changes in bulk soil and maize rhizosphere in response to nitrogen fertilization. *Communications in Soil Science and Plant Analysis* **39**, 2597–2607. doi:10.1080/00103620802358656

Ruyters S, Mertens J, Vassilieva E, Dehandschutter B, Poffijn A, Smolders E (2011) The red mud accident in Ajka (Hungary): plant toxicity and trace metal bioavailability in red mud contaminated soil. *Environmental Science & Technology* **45**, 1616–1622. doi:10.1021/es104000m

Santini TC, Fey MV (2013) Spontaneous vegetation encroachment upon bauxite residue (red mud) as an indicator and facilitator of in situ remediation processes. *Environmental Science & Technology* **47**, 12089–12096. doi:10.1021/es402924g

Santini TC, Fey MV (2016) Assessment of Technosol formation and in situ remediation in capped alkaline tailings. *Catena* **136**, 17–29. doi:10.1016/j.catena.2015.08.006

Santini TC, Kerr JL, Warren LA (2015*a*) Microbial driven strategies for bioremediation of bauxite residue. *Journal of Hazardous Materials* **293**, 131–157. doi:10.1016/j.jhazmat.2015.03.024

Santini TC, Warren LA, Kendra KE (2015*b*) Microbial diversity in engineered haloalkaline environments shaped by shared geochemical drivers observed in natural analogues. *Applied and Environmental Microbiology* **81**, 5026–5036. doi:10.1128/AEM.01238-15

Santini TC, Malcolm LI, Tyson GW, Warren LA (2016) pH and organic carbon dose rates control microbially driven bioremediation efficacy in alkaline bauxite residue. *Environmental Science & Technology* **50**, 11164–11173. doi:10.1021/acs.est.6b01973

Schmalenberger A, O'Sullivan O, Gahan J, Cotter PD, Courtney R (2013) Bacterial communities established in bauxite residues with different restoration histories. *Environmental Science & Technology* **47**, 7110–7119. doi:10.1021/es401124w

Shaw RJ (1999) Soil salinity-electrical conductivity and chloride. In 'Soil analysis: an interpretation manual'. (Eds KI Peverill, LA Sparrow, DJ Reuter) pp. 129–140 (CSIRO Publishing: Melbourne, Vic.)

Shi B, Qu Y, Li H (2017) Gypsum alleviated hydroxyl radical-mediated oxidative damages caused by alkaline bauxite residue in leaves of *Atriplex canescens*. *Ecological Engineering* **98**, 166–171. doi:10.1016/j.ecoleng.2016.10.008

Shu WS, Ye ZH, Zhang ZQ, Lan CY, Wong MH (2005) Natural colonization of plants on five lead/zinc mine tailings in Southern China. *Restoration Ecology* **13**, 49–60. doi:10.1111/j.1526-100X.2005.00007.x

Shukla MK, Lal R, Underwood J, Ebinger M (2004) Physical and hydrological characteristics of reclaimed minesoils in southeastern Ohio. *Soil Science Society of America Journal* **68**, 1352–1359. doi:10.2136/sssaj2004.1352

Sims JT, Johnson GV. (1991) Micronutrients soil tests. In Micronutrients in Agriculture, 2nd edn. (Eds JJ Mortvedt., FR Cox, LM Shuman, RM Welch) SSSA Book Series, No. 4. pp. 427–476. (Soil Science Society of America: Madison, WI)

Snars K, Hughes JC, Gilkes RJ (2004) The effects of addition of bauxite red mud to soil on P uptake by plants. *Australian Journal of Agricultural Research* **55**, 25–31.

Southwell LT, Majer JD (1982) The survival and growth of the earthworm *Eisenia foetida* (Lumbricidae: Oligochaeta) in alkaline residues associated with the bauxite refining process. *Pedobiologia* **23**, 42–52.

Stevenson FJ (1994) 'Humus chemistry: genesis, composition, reactions.' (John Wiley & Sons: Hoboken, NJ, USA)

Summers RN, Smirk DD, Karafilis D (1996) Phosphorus retention and leachates from sandy soil amended with bauxite residue (red mud). *Soil Research* **34**, 555–567.

Summers RN, Rivers MR, Clarke MF (2002) The use of bauxite residue to control diffuse phosphorus pollution in Western Australia – a win-win-win outcome. In 'Proceedings of the 6th International Alumina Quality Workshop,' Brisbane, Australia. pp. 262–269.

Sumner ME (1995) Sodic soils: new perspectives. In 'Australian sodic soils: distribution, properties and management'. (Eds R Naidu, ME Sumner, P Rengasamy) pp. 1–34. (CSIRO: Melbourne, Vic.)

Thiyagarajan C, Phillips IR, Dell B, Bell RW (2009) Micronutrient fractionation and plant availability in bauxite-processing residue sand. *Soil Research* **47**, 518–528. doi:10.1071/SR08201

Thiyagarajan C, Bell RW, Anderson JD, Phillips IR (2011) Zinc forms in compost- and red mud-amended bauxite residue sand. *Journal of Soils and Sediments* **11**, 101–114. doi:10.1007/s11368-010-0279-2

Thiyagarajan C, Bell RW, Anderson J, Phillips IR (2012) Plant-available manganese in bauxite residue sand amended with compost and residue mud *Soil Research* **50**, 416–423. doi:10.1071/SR11342

Tordoff GM, Baker AJM, Willis AJ (2000) Current approaches to the revegetation and reclamation of metalliferous mine wastes. *Chemosphere* **41**, 219–228. doi:10.1016/S0045-6535(99)00414-2

Ujaczki É, Feigl V, Molnár M, Cusack P, Curtin T, Courtney R, O'Donoghue L, Davris P, Hugi C, Evangelou MW, Balomenos E (2018) Re-using bauxite residues: benefits beyond (critical raw) material recovery. *Journal of Chemical Technology and Biotechnology (Oxford, Oxfordshire)* **93**, 2498–2510. doi:10.1002/jctb.5687

Utomo WH, Dexter AR (1982) Changes in soil aggregate water stability induced by wetting and drying cycles in non-saturated soil. *Journal of Soil Science* **33**, 623–637. doi:10.1111/j.1365-2389.1982.tb01794.x

Waisel Y (1972) 'Biology of halophytes.' (Academic Press: New York)

Wehr JB, So HB, Menzies NW, Fulton I (2005) Hydraulic properties of layered soils influence survival of Rhodes grass (*Chloris gayana* Kunth.) during water stress. *Plant and Soil* **270**, 287–297. doi:10.1007/s11104-004-1651-z

Wehr JB, Fulton I, Menzies NW (2006) Revegetation strategies for bauxite refinery residue: a case study of Alcan Gove in Northern Territory, Australia. *Environmental Management* **37**, 297–306. doi:10.1007/s00267-004-0385-2

Whittington BI, Fletcher BL, Talbot C (1998) 1–22. doi:10.1016/S0304-386X(98)00021-8

Williams DJ, Currey NA (2002) 270–288. doi:10.1076/ijsm.16.4.270.8632

Williamson NA, Johnson MS, Bradshaw AD (1982) 'Mine waste rehabilitation: the establishment of vegetation on metal mine waste.' (Mining Journal Books: London)

Wilson SM, Pyatt DG, Ray D, Malcolm DC, Connolly T (2005) Indices of soil nitrogen availability for an ecological site classification of British forests. *Forest Ecology and Management* **220**, 51–65. doi:10.1016/j.foreco.2005.08.004

Wong JWC, Ho GE (1991) Effects of gypsum and sewage sludge amendment on physical properties of fine bauxite refining residue. *Soil Science* **152**, 326–332. doi:10.1097/00010694-199111000-00003

Wong JWC, Ho GE (1993) Use of waste gypsum in the revegetation on red mud deposits: a greenhouse study. *Waste Management & Research* **11**, 249–256. doi:10.1177/0734242X9301100306

Wong JW, Ho G (1994*a*) Sewage sludge as organic ameliorant for revegetation of fine bauxite refining residue. *Resources, Conservation and Recycling* **11**, 297–309. doi:10.1016/0921-3449(94)90097-3

Wong JW, Ho G (1994*b*) Effectiveness of acidic industrial wastes for reclaiming fine bauxite refining residue (red mud) *Soil Science* **158**, 115–123. doi:10.1097/00010694-199408000-00005

Wong JW, Ho G (1995) Cation exchange behavior of bauxite refining residues from Western Australia. *Journal of Environmental Quality* **24**, 461–466.

Woodard HJ, Hossner L, Bush J (2008) Ameliorating caustic properties of aluminum extraction residue to establish a vegetative cover. *Journal of Environmental Science and Health. Part A* **43**, 1157–1166.

Xenidis A, Harokopou AD, Mylona E, Brofas G (2005) Modifying alumina red mud to support a revegetation cover. *JOM* **57**, 42–46. doi:10.1007/s11837-005-0214-y

Xue S, Zhu F, Kong X, Wu C, Huang L, Huang N, Hartley W (2016) A review of the characterization and revegetation of bauxite residues (red mud). *Environmental Science and Pollution Research International* **23**, 1120–1132. doi:10.1007/s11356-015-4558-8

Yadev S, Irfan M, Ahmad A, Hayat S (2011) Causes of salinity and plant manifestations to salt stress: a review *Journal of Environmental Biology* **32**, 667–685.

Ye ZH, Shu WS, Zhang ZQ, Lan CY, Wong MH (2002) Evaluation of major constraints to revegetation of lead/zinc mine tailings using bioassay techniques. *Chemosphere* **47**, 1103–1111. doi:10.1016/S0045-6535(02)00054-1

Zhu F, Xue S, Hartley W, Huang L, Wu C, Li X (2016*a*) Novel predictors of soil genesis following natural weathering processes of bauxite residues. *Environmental Science and Pollution Research International* **23**, 2856–2863. doi:10.1007/s11356-015-5537-9

Zhu F, Zhou J, Xue S, Hartley W, Wu C, Guo Y (2016*b*) Aging of bauxite residue in association of regeneration: a comparison of methods to determine aggregate stability and erosion resistance. *Ecological Engineering* **92**, 47–54. doi:10.1016/j.ecoleng.2016.03.025

Zhu F, Liao J, Xue S, Hartley W, Zou Q, Wu H (2016*c*) Evaluation of microstructures following natural regeneration in bauxite residue as characterized by synchrotron-based X-ray micro-computed tomography. *The Science of the Total Environment* **573**, 155–163. doi:10.1016/j.scitotenv.2016.08.108

Zhu F, Hou J, Xue S, Wu C, Wang Q, Hartley W (2017) Vermicompost and gypsum amendments improve aggregate formation in bauxite residue. *Land Degradation & Development* **28**, 2109–2120. doi:10.1002/ldr.2737

Zhu F, Cheng Q, Xue S, Li C, Hartley W, Wu C, Tian T (2018) Influence of natural regeneration on fractal features of residue microaggregates in bauxite residue disposal areas. *Land Degradation & Development* **29**, 138–149. doi:10.1002/ldr.2848

Handling Editor: Mary Beth Kirkham