

CHAPTER 7

Integrating Remediation and Resource Recovery of Industrial Alkaline Wastes: Case Studies of Steel and Alumina Industry Residues

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Abstract

With an estimated annual production of two billion tonnes globally, alkaline industrial wastes can be considered both major global waste streams, and materials that offer significant options for potential resource recovery. Alkaline wastes are usually derived from high temperature production (e.g. steel and alumina) or disposal (e.g. incineration) processes and are increasingly abundant given rising global demand for steel and alumina and the drive for waste incineration in some jurisdictions. Although relatively long-standing afteruses have been adopted for these materials (e.g. steel slag use as an aggregate) providing opportunities for value recovery, they are not sufficient to consume all residue arisings or limit potential environmental impacts. These impacts can include the generation of fugitive dusts, challenges associated with revegetation, and impacts on the water environment. These wastes can produce highly alkaline leachates that are enriched with trace metals (e.g. As, Cr, Mo, V) and persist over decades after site closure. Vanadium, one of the most severe ecotoxins in the leachate, is also a valuable commodity for renewable energy technologies, unifying the often divergent needs of resource recovery and remediation. Case studies are described illustrating routes to resource recovery from wastes from two major industrial sectors; steel production and alumina production.

7.1 Introduction

Since the Industrial Revolution, approximately 90 billion tonnes of alkaline wastes have been produced in the fabrication of steel, alumina and coal-fired power generation, with a current annual global production of two billion tonnes.¹ Presently, steel slag production can be as high as 170-250 million tonnes per year,² and bauxite processing residue (commonly known as red mud) production is estimated to be 118-177 million tonnes.^{3,4} In 2010, in Europe, 35% of the steel slag was landfilled,⁵ while in China and Ukraine the amounts were 78% and 40%,⁶ respectively. The amount of red mud stored in active and legacy sites is approximately three billion tonnes.⁷ Severe legacy and environmental impacts, in particular, affecting water bodies, can be associated with these wastes if adequate management practices are not in place¹ or in case of accidental spill.⁸ There are documented impacts on surface waters of steel slag leachate and red mud that can persist for more than 30 years.⁹⁻¹¹

In addition, the majority of the toxic trace elements that are problematic during the disposal alkaline wastes (e.g. vanadium) are also valuable commodities and critical raw materials needed for new emergent green technologies,¹² such as redox flow batteries to store renewable energies.¹³ Some of these elements are highly soluble at high pH within disposal environments,¹⁴ and so potentially recoverable from leachates. Consequently, there is growing interest in the recovery of e-tech elements from alkaline wastes and leachates as secondary sources with potentially globally significant reserves.¹⁵ Recovery of these resources would be essential for addressing the security of supply of critical elements highlighted in several strategic reviews^{16, 17} and by the European Commission,¹⁸ while simultaneously reducing the environmental cost of landfilling.

The potential for carbon sequestration using alkaline wastes has also received considerable attention.¹⁹⁻³² The presence of hydroxide, oxide and silicate minerals in alkaline wastes which naturally weather and sequester

atmospheric carbon dioxide could be a pathway for offsetting emissions associated with carbon-intensive industries. It was estimated that the worldwide red mud repositories could capture 572 Mt CO₂,³³ offsetting 3–4% of the annual CO₂ aluminium industry emissions.³³ For steel slag, the global theoretical CO₂ emissions reduction potential of steelmaking slag carbonation is 170 Mt CO₂ a⁻¹.³⁴ This would be another possibility to add value to these waste streams and minimise the environmental footprint associated with steel and alumina production.

There is also growing interest and research on the rehabilitation, and long-term management/closure plans of the alkaline waste repositories, especially for red mud.³⁵⁻⁴¹ The spontaneous revegetation of alkaline residues often occurs with rare orchid-rich calcareous grassland, a habitat type which has declined dramatically in Europe in recent decades.⁴²⁻⁴⁵ Calcium-rich, nutrient-poor substrates often have high floral diversity and, in many cases, alkaline residue disposal sites can obtain formal conservation designations.⁴⁶ This also raises the possibility of directing surface restoration at disposal sites towards these communities, offering a significant opportunity for high economic value environmental enhancement, beyond simple remediation.

This chapter reports on resource recovery and remediation of two streams of alkaline wastes – steel slag and red mud. These residues provide some synergies in both environmental impacts and resource recovery as well as some contrasting perspectives on progress to date in bringing valorisation to the forefront of industrial waste management. After a review of the environmental impacts associated with the current management practices, the opportunities for resource recovery are discussed. The challenges of both resource recovery and remediation are analysed and future directions identified.

7.2 Management Practices and Environmental Impacts

Re-use and recycling of steel slag are common practices, predominantly as fill material or road ballast in construction applications.^{5, 47} In Europe, between 70-80% of steel slag is currently reused (Figure 7.1), with the remainder being stockpiled or landfilled.⁴⁸ Slag is reused as aggregate for sand blasting, bituminous and hydraulic bound, mixtures, top layers for high skid resistance, unbound mixtures dams (road construction and noise protection), wastewater treatment, embankments and fill, railway ballast, sealing in surface layers to protect deposits roofing, armour stone, ground stabilization.^{49, 50} Steel slag can also be used for the manufacture of fertilizers, cement and other hydraulic binders,⁵¹ stonewool, ceramic materials,⁵² and glass (blended with other components).⁴⁹ Thermal energy storage is another possible use.⁴⁹ This range of potential afteruses and the significant proportion of slags produced that avoid landfill (Figure 7.1) highlight the relative maturity of valorisation efforts for steel slag.

[Figure 7.1 near here]

The weathering of the slag to reduce calcium oxide (CaO) content before reuse and during disposal can generate highly alkaline waters (pH > 11.5) leaching to surface or ground waters.^{9, 53} These leachates result from the dissolution of calcium and magnesium oxides, alongside calcium aluminosilicate minerals in the slag.¹ The leaching products of these minerals elevate alkalinity in receiving waters and can originate pH levels up to 12.4; much higher than the pH range typically found in nature.⁹ At elevated pH, there are also potential ecotoxicological impacts with high concentrations of some metals and metalloids; notably, those that form oxyanions mobile under alkaline conditions (e.g. Cr, Mo, V).⁵⁴⁻⁵⁶

The extreme pH and enrichment on calcium, fluoride, and chromium were reported in groundwaters and spring

resurgence from a large-scale slag-fill aquifer in Illinois, USA.⁵³ Koryak et al. observed diminished diversity of invertebrate and fish populations in a stream affected by steel slag leachate from a riparian slag disposal area.⁵⁷ Mayes et al.^{9, 58} and Hull et al.⁵⁹ assessed the impact of slag leachates at a range of UK sites, showing the enrichment of potentially toxic trace elements, such as vanadium, lithium and molybdenum. Chromium, barium, and molybdenum have also been highlighted as elements of concern in steel slag leachate.^{54, 55} There are few studies on the longevity of these impacts. However, leachates from steel mill sites can continue for over 30 years after closure.^{9, 10}

High rates of precipitation of secondary carbonate minerals (mostly calcite), which can smother the stream beds, are also associated with steel slag leachates.^{53, 57} Total invertebrate abundance and diversity was found to be negatively correlated with pH in streams affected by leachate, with an impoverished benthic fauna around source areas (pH 10.4–11.8).⁵⁹ The precipitates can smother macroinvertebrate communities and reduce light penetration amplifying the ecological cost of leachate release.⁵⁹ As soon as pH lowers to values close to 9, invertebrate communities were seen to recover over relatively small spatial scales.⁵⁹

Storage and use of steel slag may generate the emission of fugitive dust, which if inhaled may pose a risk to public health, due to the physical characteristics of dust itself and the concentration of contaminants in the dust.⁶⁰ The UK Environment Agency performed a generic risk assessment for the risks to public health from the emission of wind-blown dusts from a working steel slag stockpile. Compared with relevant health benchmarks for chemical concentrations in air, most risks were negligible assuming a worst-case emission scenario.⁶⁰ Only As and Mn posed a risk in a scenario which assumed worst-case emissions from a continuously disturbed working stockpile. However, after considering good working practice with aggregates, which includes either the stockpile being constructed in a silo or the material being conditioned with water, the risks were considered to be low.⁶⁰

Widespread re-use and recycling of bauxite processing residue (red mud) are at a far less developed stage than steel slag. Figure 7.1 shows that of the 118-170 million tonnes produced annually, less than 3% currently avoids final disposal in waste residue depositories.⁴ This rather modest resource recovery effort is apparent despite over 1200 patents being filed for resource recovery or re-use options for bauxite processing residue.⁴ The after-uses that currently consume the majority of the 3% avoiding disposal are uses in cement and iron production, or as an aggregate in road and building construction. Efforts to encourage re-use and resource recovery from red mud have been catalysed in part by the public attention on the alumina industry in the aftermath of the Ajka (Hungary) red mud spill in 2010. Major initiatives that have ensued have seen global industry best-practice guidelines trying to embed industrial symbiosis in residue management processes,⁷ as well as significant investment in resource recovery initiatives.⁶¹

The accidental release of around 1 million m³ of highly alkaline and caustic red mud slurry at Ajka also significantly improved the research base on the environmental impacts associated with red mud. There are various synergies with steel slag leachate given high alkalinity (albeit NaOH-derived with red mud) and the potential mobility of various oxyanion-forming contaminants (e.g. As, Cr, Mo, V) at high pH. The main difference, however, lies with the high salinity associated with the NaOH digestant used in the Bayer process and the presence of fine colloidal particulate matter in red mud suspension.

In aquatic environment, studies on the effects of red mud have highlighted the acute impact of red mud on a range of receptors such as microbial communities, macroinvertebrates and aquatic plants. The combined stressors of alkalinity, salinity, metal-enrichment and fine particle size have limited determinations of causal agents in some of the post-Ajka studies. The high salinity and alkalinity impacts were relatively short-lived at Ajka, with concern focussing on potential cycling and uptake of metal(loid)s. Long term studies at an enclosed

lake subject to historical red mud disposal in Scotland have demonstrated the multi-decadal scales over which arsenic from red mud can cycle between lake sediments, water column and aquatic macrophytes.⁶² The potential for oxyanion-forming metal(loid)s to cycle with ubiquitous anions such as phosphate has been highlighted in numerous studies.^{8, 63}

In terrestrial settings, the salinity associated with red mud has been shown to be the dominant constraint on plant growth,⁶⁴ although the genotoxic effects of vanadium on higher plants have been highlighted in one study.⁶⁵ Efforts to rehabilitate and revegetate bauxite residue disposal areas are critical to longer term residue management and are typically focussed to overcome the constraints of salinity, alkalinity and dominance of exchangeable sodium (sodicity) to plant growth. Plant cover at disposal areas also limits the risks of fugitive dust generation, which for red mud have been shown to pose similar public health risks to urban dusts given the fine particle size, with a significant proportion falling below the PM_{2.5} fraction in post-Ajka studies.⁶⁶

Sufficient amendment of surface residue (0 – 20 cm) can successfully address the primary constraints to vegetation establishment.^{67, 68} Application of organic wastes and/or inorganic fertilisers can also overcome the nutrient deficiencies typical of ore processing wastes.⁶⁹ A major concern in mine waste reclamation is the short-term effect of reclamation strategies and the potential for vegetation community collapse due to exhaustion of nutrients and reappearance of residue inhibitory characteristics.

A series of medium to long-term field trials have demonstrated the longevity of direct revegetation approaches for bauxite residue. Although Courtney and Timpson⁷⁰ highlighted the risk of nutrient deficiencies in the second growing season, follow up studies demonstrated adequate values in the rhizosphere and plant content. Further, vegetation diversity increased from initial seed mixes⁷¹ with evidence of soil faunal colonisation and activity.⁷² Microbial community assessments have demonstrated absence of stress indicators and greater

dominance of species more typical of soil communities.⁷²⁻⁷⁴ Evidence of pedogenesis is also encouraging, with older sites exhibiting improved soil physical condition, aggregate stability.⁷⁵ More recently, soil development initiated through residue amendment and promotion of direct revegetation has been evidenced through geochemical assessment.⁷⁶ Although, initial reclamation efforts were confined to the top 20 cm of the residue profile, pit sampling demonstrated sustained rehabilitation after 16 years with improvements in residue chemistry (e.g. reduced alkalinity and solubility of trace elements) found deeper (up to 50 cm) in the profile.

In summary, the environmental impacts associated with alkaline wastes can include the generation of fugitive dusts, challenges associated with revegetation, and most commonly, impacts on ground and surface waters (Table 7.1). The high content of trace metals that form oxyanions (e.g. As, Cr, Mo, Se, V), highly mobile in alkaline water and persistent over decades after site closure is one of the major impacts. Vanadium mobility at high pH is a common problem to both steel slag⁵⁴ and red mud leachates.⁷⁷ Smothering of benthic habitats due to the precipitation of calcite is also commonly associated with steel slag and its leachate.^{9, 59} There are, however, encouraging emerging signs that through relatively low-cost interventions, revegetation of alkaline residues can be directed towards functioning soils with potentially valuable afteruses.

[Table 7.1 near here]

7.3 Opportunities for Resource Recovery

Alkaline residues usually contain high concentrations of elements (La, Li, Co, V, Te, Ga, Se) considered important or critical to green emerging technologies.¹⁶ For example, vanadium can be used in steel manufacture for offshore wind turbines, lithium and cobalt for vehicle fuel cells, and rare earth elements are crucial for next-generation solar power systems. This has led to an increasing interest in the recovery of metals from these waste streams. Table 7.2 summarises the potential amounts that could be recovered from the alkaline residues along with the concentrations typically present in the leachates, emphasising the importance of these wastes as secondary sources.

[Table 7.2 near here]

An inventory of critical raw materials including rare earths (REE) showed that 1 tonne of red mud contains a variety of metal(loid)s valued up to \$64.2.⁷⁸ However, recent reviews on the recovery of metals from red mud highlighted the difficulty of recovery as the metals are in complex mineral phases, and there is a need to improve leaching, extraction selectivity and efficiency, and to develop processes with low energy requirements and cost.^{79, 80} There is an increasing number of recent studies on metal recovery technologies from steel slag and red mud (Table 7.3), also focusing in the combination of technologies, e.g. pyrometallurgical and hydrometallurgical process, solvent extraction and precipitation, and the use of bioleaching as a more environmentally friendly, low energy input and low capital cost alternative.⁸¹ Table 7.3 reviews the maximal recovery of critical metals from steel slag and red mud in these studies. Although in the literature substantial recoveries are achieved in batch and column tests using ion exchange, leaching (bio, acid and alkali), precipitation and combined pyrometallurgical-hydrometallurgical process, no examples were found for pilot or real scale industrial applications. Further developments or incentives are needed to advance

metal recovery from alkaline wastes from bench scale to field systems.

[Table 7.3 near here]

There is also the need to recover metals from waste repositories, both for legacy wastes and for future waste arisings as part of a movement toward a circular economy.⁸² Sapsford et al. reviewed the technologies available for *in situ* direct recovery of metals (Ti, Sc, REE and Al) from red mud repositories.⁸² Most of them are based on acid leaching (sulphuric, nitric, citric/oxalic/sulphuric acids) or bioleaching (*Penicillium tricolor* and *Thiobacilli* cultures).⁸² Acid leaching with sulphuric acid was also the route for recovery of Zn in steelmaking dusts repositories.⁸²

When assessing the Technology Readiness Levels (TRL, adopted by the European Commission) of the metal recovery technologies and the availability (concentration and mobility) in the alkaline leachate, it is clear the elevated potential for vanadium recovery from both steel slag and red mud leachate (Figure 7.2). However, none of the reviewed technologies exceeds TRL 4 – technology validated in lab and TRL 5 – technology validated in a relevant environment, so further efforts have to be made to reach the highest levels TRL 7-9 (commercial scale) and full implementation.

[Figure 7.2 near here]

Given the amounts of steel slag and red mud produced globally yearly (288-427 million tonnes), opportunities for bulk reuse need to be further explored. Table 7.4 summarises potential environmental applications of steel slag and red mud. Most of them are related to water treatment (sorbent, removal of metals, nitrate and phosphate) and catalytic applications. Similarly to the metal recovery technologies, there are limited

applications on full scale, and only laboratory batch scale tests were performed. Currently, the bulk of steel slag is reused as aggregate materials for road construction (Figure 7.2), but bulk usage for red mud is yet to be identified.⁸³ Recent reviews have covered the bulk reuse of red mud as an immobiliser for metal/metalloid-contaminated soil⁸⁴ and potential applications in composite materials, cement, geopolymers, ceramics, and as a catalyst.⁸³ Ferric oxide recovery from red mud is pointed out as the most relevant for bulk utilisation,⁷⁸ and the limiting issue of is the alkalinity of the residue.³⁵

[Table 7.4 near here]

Further opportunities for valorisation of steel slag and red mud focus on mineral carbonation, a promising carbon capture and storage option, which can store carbon permanently as carbonates, thermodynamically the most stable form of carbon. Carbonation may play a significant role in management strategies, and climate change mitigation options,⁸⁵ and extensive reviews on the mechanisms and techniques for carbonation of alkaline wastes can be found in the literature.^{86, 87} The global theoretical CO₂ emissions reduction potential of steelmaking slag carbonation is 170 Mt CO₂ a⁻¹,³⁴ approximately 10 % of the global steel industry emissions.⁸⁸ There is also potential for carbonation in legacy sites, as 2,800 tCO₂ atmospheric carbon has been sequestered in stable secondary carbonate minerals due to ambient weathering of an iron and steel slag deposit in Consett, UK.⁸⁹ This slag deposit may be able to capture 4 to 8 million tonnes of CO₂.⁸⁹ The carbon capture potential for the global red mud inventory is 572 Mt CO₂.³³ Yearly, 3–4% of the CO₂ emissions could be carbonated, with considerable benefits for the aluminium industry.³³ Currently, red mud repositories are not optimised for carbon capture, but estimates point that natural weathering unintentionally sequestered 100 Mt CO₂ worldwide from the late 19th century to 2008.⁹⁰ If appropriate carbonation technologies are applied, an additional 6 Mt CO₂ can be potentially captured and stored annually, simultaneously reducing the hazardousness of the red mud.⁹⁰

Another ‘hidden value’ that can be provided by alkaline wastes is through ecosystem services. Historical abandoned industrial sites with highly alkaline substrates have been shown to be spontaneously colonised with rare vegetation communities in orchid-rich habitats, particularly rare taxa *Dactylorhiza purpurella* (Northern Marsh-orchid) and *D. praetermissa* (and associated hybrids).⁹¹ Coatham Marsh in Redcar, England is a nature reserve owned by the steel mill operator and has been leased and managed by the Tees Valley Wildlife Trust since 1982. It is an example how the extreme alkaline, CaO-rich conditions and low nutrient substrate prevent the establishment of many common ruderal species, but enhances floristic diversity with the low sward height encouraging colonisation by a range of attractive and rare species.^{46, 91} It is located in mouth of the Tees estuary that was designated as a Site of Special Scientific Interest (SSSI) in 1988 because of the presence of rare species including the yellow wort (*Blackstonia perfoliata*), Northern Marsh Orchid (*D. majalis*) and the nationally rare grass, rush-leaved fescue (*Festuca juncifolia*).⁴⁶ There are at least five other designated conservation sites in the UK over formerly highly alkaline substrates where similar assemblages of orchid-rich calcareous grassland have spontaneously established. This raises the additional possibility of explicitly directing surface rehabilitation towards these communities, offering a significant opportunity for high economic value environmental enhancement, beyond simple cap-and-cover remediation. However, in the short-term accelerated weathering could give rise to leachates with higher metal(loid) ion concentrations so the approach must be coupled with advanced remediation (and value-recovery) so that unacceptable water discharges are avoided.

Alkaline residues can be a relevant scientific resource for different areas. Alkaline waters associated with ophiolites and serpentinization activity were considered to be a habitat analogue of Mars.⁹² Also, these extreme environments are useful analogues to primitive Earth-like environments. The study of bacterial populations in a soil that has been in contact with highly alkaline groundwater allowed to assess how they evolved to tolerate

the high pH and are now undertaking geomicrobiological processes similar to those observed at neutral pH values.⁹³ Microbes in red mud have unique adaptation mechanisms for survival in multiple stressors at a time (alkaline, saline, high electrical conductivity, sodic condition, heavy metals and low organic carbon and nitrogen).⁹⁴ There is also an enduring interest in hyperalkaline systems with regard long-term disposal of nuclear industry wastes. Hyperalkaline conditions can occur due to the presence of concrete used as immobilisation matrix or in engineering structures of the nuclear waste repositories.^{95, 96}

7.4 Challenges to Resource Recovery

Several challenges have to be overcome to fully implement resource recovery from waste streams such as steel slag and red mud. These challenges range from technological, economic, and socio-political, needing integrated approaches to underpin increased valorisation of these increasingly abundant global residues streams to both minimise the documented environmental risks and maximise the efficiency of mineral resource use.

Most existing studies on metal recovery and residue after-use have been conducted at the laboratory scale, so up-scaling to pilot field systems is critical to assessing the scale-dependency of metal leaching and recovery. The existing technologies have not yet reached Technology Readiness Levels (TRL) higher than 5 (Figure 7.2). Piloting new environmental technologies under field conditions is crucial for, not only formulating detailed engineering guidance for full-scale application, but also for undertaking integrated Life Cycle Assessment (accounting for environmental benefits of recovery and reduced environmental impacts) to assess feasibility and obtain regulatory scrutiny. However, the fact that few of these technologies have proceeded to full scale thus far also suggests other factors are preventing commercial uptake, given the tangible win-win

that would be offered by diversion of material from landfill for new environmental applications or valorisation through carbon capture and metal recovery. Such factors could include industrial inertia (e.g. slow investment decisions in large multinationals), volatility of metal commodity markets (especially in the case of new recovery technologies), and the complexities of waste policy at national and international level.⁹⁷⁻¹⁰⁰ In some cases, viable technologies may get promising results in full-scale demonstrations, but commercial uptake can be limited due to competitive advantages held by traditional technologies. For example, modified red mud has been successfully trialled in wastewater and contaminated land applications for metal and nutrient removal,¹⁰¹¹⁰² but engendering change in industries where long-established, high volume (and relatively low-cost), approved and accredited treatment options exist (e.g. ferric and alumina salts in water treatment) may mean stakeholders are reluctant to invest in alternative technologies without regulatory prompting. Thus, identifying institutional structures and policy constraints that govern resource management and recovery processes specific to different residues in different geographical areas is required in tandem with the scientific advances in recovery and recycling technologies.

The implementation of resource recovery technologies is often driven by the potential economic benefits, which means that technologies must be cheap, practical and robust.¹⁰³ However, the economic potential can fluctuate due to the market demand for the metals potentially recovered from steel slag and red mud, but can also advance with statutory requirements created by environmental regulations, especially for V. The use of techno-economic guidelines in conjunction with net present value analysis (NPV), risk assessment and layer of protection analysis (LOPA) could accelerate the implementation of resource recovery technologies.¹⁰³ The performance of mineral carbonation and the associated energy inputs also still require optimisation to be economically feasible.¹⁰⁴

The extensive regulation and policies on waste management often fail, due to poor enforcement and

application, disparities in regulations between separated jurisdictions, and illegal movements of waste.¹⁰⁵ Several studies highlight the need for mandatory and comprehensive waste management guidelines and policies for economic blocks, countries and municipalities, allied to a strong policy maker's commitment and support to environmental sustainability issues.¹⁰⁶ Currently, in the European Union, there is a complex array of relevant regulation (with implementation varying within the EU), such as definitions of waste (by-product, end of waste). The legal status of slag (their classification as either waste, product, or by-product) has been discussed worldwide for more than 25 years. Even though, according to the Waste Framework Directive 2008/98/CE–Article 5, steel slag may be currently classified as a by-product because it is produced as an integral part of the production process and its use is certain without any further processing.⁴⁹ Also, although the strategy is aiming towards the implementation and development of a circular economy, present regulatory frameworks have not been designed primarily with that in mind, particularly for high volume residues in the early stages of production cycles.¹⁰⁷

Additional issues relating to remediation at repositories and legacy sites are land ownership and liabilities. In their case study, Deutz et al.¹⁰⁷ demonstrated how ownership of the materials and liability for environmental protection could be obstacles to resource recovery. The slag producer is not the owner of current and recently produced steel slag, though retains an environmental liability for the slag, which cannot easily be disentangled from the environmental risks associated with older legacy slag.¹⁰⁷

7.5 Conclusions and Future Directions

Research into alkaline residues addresses an increasingly important arena of environmental science given the increasing volume of caustic industrial residues being generated globally and the growing quantities destined for landfill. These materials have been relatively under-studied compared to acid-generating mine wastes, yet

there is a strategic potential for both industry and governments from reuse of these residues, as well as from carbon sequestration and recovery of critical and challenging metals.

This chapter surveyed current and past practices used in processing waste streams from the steel and alumina industries and discussed recovery technologies and challenges to their implementation. Based on the availability and development of the technologies reviewed here, further incentives are needed to scale up and test at commercial scale metal resource recovery units for steel slag and red mud. There is also a need to integrate economic evaluation and technology readiness, which is key to successfully developing resource recovery projects for industry.

Where a contaminant of concern is also a critical raw materials (e.g. Co, Se, V), remedial efforts should have metal recovery and recycling as inherent goals, and an improved understanding of leaching mechanisms will facilitate accelerated leaching and recovery. An allied area of research focus should be on remedial technologies for metal recovery, residue stabilisation and leachate buffering. Given the long-term legacy of leachate generation at some sites it is likely that passive approaches to residue management will be economically favourable in many cases.

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References*

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1. H. I. Gomes, W. M. Mayes, M. Rogerson, D. I. Stewart and I. T. Burke, *Journal of Cleaner Production*, 2016, **112**, Part 4, 3571-3582.
2. USGS, *Iron and Steel Slag Statistics and Information*, U.S. Department of the Interior. U.S. Geological Survey, 2017.
3. USGS, *Mineral Commodity Summaries 2016*, U.S. Department of the Interior. U.S. Geological Survey, Reston, Virginia, USA, 2017.
4. K. Evans, *J. Sustain. Metall.*, 2016, **2**, 1-16.
5. H. Motz and J. Geiseler, *Waste Management*, 2001, **21**, 285-293.
6. L. Reijnders, *Journal of Cleaner Production*, 2016, **131**, 76-96.
7. IAI and EAA, *World Aluminium and the European Aluminium Association ‘Bauxite Residue Management: Best Practice’*, available from the International Aluminium Institute, 10 King Charles II Street, London, SW1Y 4AA, UK and on line from <http://bauxite.worldaluminium.org/refining/bauxite-residue-management.html>, 2015.
8. W. M. Mayes, I. T. Burke, H. I. Gomes, Á. D. Anton, M. Molnár, V. Feigl and É. Ujaczki, *J. Sustain. Metall.*, 2016, DOI: 10.1007/s40831-016-0050-z, 1-12.
9. W. M. Mayes, P. L. Younger and J. Aumônier, *Water Air Soil Pollut*, 2008, **195**, 35-50.
10. A. L. Riley and W. M. Mayes, *Environ Monit Assess*, 2015, **187**, 1-16.
11. J. P. Olszewska, K. V. Heal, I. J. Winfield, L. J. Eades and B. M. Spears, *Water Research*, 2017, **123**, 569-577.
12. J. P. Sykes, J. P. Wright and A. Trench, *Applied Earth Science*, 2016, **125**, 3-20.
13. Á. Cunha, J. Martins, N. Rodrigues and F. Brito, *International Journal of Energy Research*, 2015, **39**, 889-918.
14. W. M. Mayes, A. P. Jarvis, I. T. Burke, M. Walton, V. Feigl, O. Klebercz and K. Gruiz, *Environmental Science & Technology*, 2011, **45**, 5147-5155.
15. Y. Qu and B. Lian, *Bioresource Technology*, 2013, **136**, 16-23.
16. J. Naden, *Science and Implementation Plan. Security of Supply of Mineral Resources (SoS Minerals) Research Programme 2012–2017*, Natural Environment Research Council, 2013.
17. U. D. o. Energy, *Critical Materials Strategy*, 2011.
18. E. Commission, *Journal*, 2017.
19. N. L. Ukwattage, P. G. Ranjith and X. Li, *Measurement*, 2017, **97**, 15-22.
20. S.-Y. Pan, K. J. Shah, Y.-H. Chen, M.-H. Wang and P.-C. Chiang, *ACS Sustainable Chemistry & Engineering*, 2017, DOI: 10.1021/acssuschemeng.7b00291.
21. Y.-S. Han, S. Ji, P.-K. Lee and C. Oh, *Journal of Hazardous Materials*, 2017, **326**, 87-93.
22. A. Said, T. Laukkanen and M. Järvinen, *Applied Energy*, 2016, **177**, 602-611.
23. A. Polettoni, R. Pomi and A. Stramazzo, *Chemical Engineering Journal*, 2016, **298**, 26-35.
24. A. Polettoni, R. Pomi and A. Stramazzo, *Journal of Environmental Management*, 2016, **167**, 185-195.
25. S.-Y. Pan, R. Adhikari, Y.-H. Chen, P. Li and P.-C. Chiang, *Journal of Cleaner Production*, 2016, **137**, 617-631.
26. S. Y. Pan, H. L. Liu, E. E. Chang, H. Kim, Y. H. Chen and P. C. Chiang, *Chemosphere*, 2016, **154**, 63-71.
27. R. Li, T. Zhang, Y. Liu, G. Lv and L. Xie, *J Hazard Mater*, 2016, **316**, 94-101.
28. S. Lee, J.-W. Kim, S. Chae, J.-H. Bang and S.-W. Lee, *Journal of CO2 Utilization*, 2016, **16**, 336-345.

29. S. Gopinath and A. Mehra, *Chemical Engineering Research and Design*, 2016, DOI: <http://dx.doi.org/10.1016/j.cherd.2016.09.010>.
30. A. B. Ghacham, L.-C. Pasquier, E. Cecchi, J.-F. Blais and G. Mercier, *Environ Sci Pollut Res*, 2016, DOI: 10.1007/s11356-016-6926-4, 1-12.
31. M. Tu, H. Zhao, Z. Lei, L. Wang, D. Chen, H. Yu and T. Qi, *ISIJ International*, 2015, **55**, 2509-2514.
32. T. D. Rushendra Revathy, K. Palanivelu and A. Ramachandran, *Environ Sci Pollut Res*, 2015, DOI: 10.1007/s11356-015-5893-5, 1-11.
33. P. Renforth, W. M. Mayes, A. P. Jarvis, I. T. Burke, D. A. C. Manning and K. Gruiz, *Science of The Total Environment*, 2012, **421-422**, 253-259.
34. S. Eloneva, S. Teir, J. Salminen, C.-J. Fogelholm and R. Zevenhoven, *Energy*, 2008, **33**, 1461-1467.
35. I. Panda, S. Jain, S. K. Das and R. Jayabalan, *International Biodeterioration & Biodegradation*, 2017, **119**, 368-376.
36. T. Mishra, V. C. Pandey, P. Singh, N. B. Singh and N. Singh, *Journal of Geochemical Exploration*, 2017, **182**, 206-209.
37. É. Ujaczki, V. Feigl, M. Molnár, E. Vaszita, N. Uzinger, A. Erdélyi and K. Gruiz, *Journal of Environmental Sciences*, 2016, DOI: 10.1016/j.jes.2015.12.014.
38. S. Xue, F. Zhu, X. Kong, C. Wu, L. Huang, N. Huang and W. Hartley, *Environ Sci Pollut Res*, 2015, DOI: 10.1007/s11356-015-4558-8, 1-13.
39. T. C. Santini, L. I. Malcolm, G. W. Tyson and L. A. Warren, *Environmental Science & Technology*, 2016, DOI: 10.1021/acs.est.6b01973.
40. T. C. Santini and N. C. Banning, *Hydrometallurgy*, 2016, **164**, 38-47.
41. T. C. Santini and M. V. Fey, *Catena*, 2015, DOI: 10.1016/j.catena.2015.08.006.
42. H. J. Ash, R. P. Gemmill and A. D. Bradshaw, *Journal of Applied Ecology*, 1994, **31**, 74-84.
43. S. Tischew, A. Baasch, H. Grunert and A. Kirmer, *Applied Vegetation Science*, 2014, **17**, 288-301.
44. K. Kiehl, A. Kirmer, T. W. Donath, L. Rasran and N. Hölzel, *Basic and Applied Ecology*, 2010, **11**, 285-299.
45. P. J. Shaw, *Botanical journal of the Linnean Society*, 1998, **128**, 385-401.
46. E. Raper, S. Davies, B. Perkins, H. Lamb, M. Hermanson, A. Soares and T. Stephenson, *Environ Geochem Health*, 2014, DOI: 10.1007/s10653-014-9672-x, 1-12.
47. N. M. Piatak, M. B. Parsons and R. R. Seal, *Applied Geochemistry*, 2015, **57**, 236-266.
48. Euroslag, 2017.
49. A. Gutierrez, L. Miró, A. Gil, J. Rodríguez-Aseguinolaza, C. Barreneche, N. Calvet, X. Py, A. Inés Fernández, M. Grágeda, S. Ushak and L. F. Cabeza, *Renewable and Sustainable Energy Reviews*, 2016, **59**, 763-783.
50. N. C. C. Lobato, E. A. Villegas and M. B. Mansur, *Resources, Conservation and Recycling*, 2015, **102**, 49-57.
51. N. Palankar, A. U. Ravi Shankar and B. M. Mithun, *Journal of Cleaner Production*, 2016, **129**, 437-448.
52. R. J. Galán-Arboledas, J. Álvarez de Diego, M. Dondi and S. Bueno, *Journal of Cleaner Production*, 2017, **142**, 1778-1788.
53. G. S. Roadcap, W. R. Kelly and C. M. Bethke, *Ground Water*, 2005, **43**, 806-816.
54. P. Chaurand, J. Rose, V. Briois, L. Olivi, J.-L. Hazemann, O. Proux, J. Domas and J.-Y. Bottero, *Journal of Hazardous Materials*, 2007, **139**, 537-542.
55. K. Matern, T. Rennert and T. Mansfeldt, *Chemosphere*, 2013, **93**, 2108-2115.
56. A. J. Hobson, D. I. Stewart, A. W. Bray, R. J. G. Mortimer, W. M. Mayes, M. Rogerson and I. T. Burke, *Environ Sci Technol*, 2017, **51**, 7823-7830.

57. M. Koryak, L. J. Stafford, R. J. Reilly and M. P. Magnuson, *Journal of Freshwater Ecology*, 2002, **17**, 461-465.
58. W. M. Mayes, J. Aumonier and A. P. Jarvis, *Water Sci Technol*, 2009, **59**, 2253-2263.
59. S. L. Hull, U. V. Oty and W. M. Mayes, *Hydrobiologia*, 2014, **736**, 83-97.
60. EA, *Steel slag quality protocol: Chemical risk assessment on BOS and EAF slags*. Environmental Agency. Bristol, pp. 179 2013.
61. Y. Pontikes and A. Malfliet, *J. Sustain. Metall.*, 2016, **2**, 1-2.
62. J. P. Olszewska, A. A. Meharg, K. V. Heal, M. Carey, I. D. M. Gunn, K. R. Searle, I. J. Winfield and B. M. Spears, *Environmental Science & Technology*, 2016, **50**, 9044-9052.
63. C. Lockwood, D. Stewart, R. G. Mortimer, W. Mayes, A. Jarvis, K. Gruiz and I. Burke, *Environ Sci Pollut Res*, 2015, DOI: 10.1007/s11356-015-4282-4, 1-11.
64. S. Ruyters, J. Mertens, E. Vassilieva, B. Dehandschutter, A. Poffijn and E. Smolders, *Environmental Science & Technology*, 2011, **45**, 1616-1622.
65. M. Mišík, I. T. Burke, M. Reismüller, C. Pichler, B. Rainer, K. Mišíková, W. M. Mayes and S. Knasmueller, *Science of The Total Environment*, 2014, **493**, 883-890.
66. A. Gelencsér, N. Kováts, B. Turóczy, Á. Rostási, A. Hoffer, K. Imre, I. Nyirő-Kósa, D. Csákberényi-Malasics, Á. Tóth, A. Czitrovsky, A. Nagy, S. Nagy, A. Ács, A. Kovács, Á. Ferincz, Z. Hartyáni and M. Pósfai, *Environmental Science & Technology*, 2011, **45**, 1608-1615.
67. R. Courtney and G. Mullen, *Water Air Soil Pollut*, 2009, **197**, 15-22.
68. B. E. H. Jones and R. J. Haynes, *Critical Reviews in Environmental Science and Technology*, 2011, **41**, 271-315.
69. R. Courtney and T. Harrington, *Land Degradation & Development*, 2012, **23**, 144-149.
70. R. Courtney and J. Timpson, *Water Air Soil Pollut*, 2005, **164**, 91-102.
71. R. Courtney, S. Jordan and T. Harrington, *Land degradation & development*, 2009, **20**, 572-581.
72. R. Courtney, J. A. Harris and M. Pawlett, *Restoration Ecology*, 2014, **22**, 798-805.
73. N. C. Banning, D. B. Gleeson, A. H. Grigg, C. D. Grant, G. L. Andersen, E. L. Brodie and D. Murphy, *Applied and environmental microbiology*, 2011, **77**, 6158-6164.
74. A. Schmalenberger, O. O'Sullivan, J. Gahan, P. D. Cotter and R. Courtney, *Environmental science & technology*, 2013, **47**, 7110-7119.
75. R. Courtney, T. Harrington and K. A. Byrne, *Ecological Engineering*, 2013, **58**, 63-68.
76. A. W. Bray, D. I. Stewart, R. Courtney, S. P. Rout, P. N. Humphreys, W. M. Mayes and I. T. Burke, *Environmental Science & Technology*, 2018, **52**, 152-161.
77. I. T. Burke, C. L. Peacock, C. L. Lockwood, D. I. Stewart, R. J. G. Mortimer, M. B. Ward, P. Renforth, K. Gruiz and W. M. Mayes, *Environmental Science & Technology*, 2013, **47**, 6527-6535.
78. É. Ujaczki, Y. S. Zimmermann, C. A. Gasser, M. Molnár, V. Feigl and M. Lenz, *Journal of Chemical Technology & Biotechnology*, 2017, **92**, 2835-2844.
79. Y. Liu and R. Naidu, *Waste Management*, 2014, **34**, 2662-2673.
80. Z. Liu and H. Li, *Hydrometallurgy*, 2015, **155**, 29-43.
81. H. I. Gomes, V. Funari, W. M. Mayes, M. Rogerson and T. J. Prior, *Journal of Environmental Management*, 2018, **222**, 30-36.
82. D. Sapsford, P. Cleall and M. Harbottle, *J. Sustain. Metall.*, 2016, DOI: 10.1007/s40831-016-0102-4, 1-18.
83. A. S. Verma, N. M. Suri and S. Kant, *Waste Management & Research*, 2017, DOI: 10.1177/0734242X17720290, 0734242X17720290.
84. Y. Hua, K. V. Heal and W. Friesl-Hanl, *Journal of Hazardous Materials*, 2017, **325**, 17-30.
85. F. J. Doucet, *Minerals Engineering*, 2010, **23**, 262-269.

86. E. R. Bobicki, Q. Liu, Z. Xu and H. Zeng, *Progress in Energy and Combustion Science*, 2012, **38**, 302-320.
87. M. Bodor, R. Santos, T. Van Gerven and M. Vlad, *cent.eur.j.eng*, 2013, **3**, 566-584.
88. IPCC, *Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, 2007.
89. W. M. Mayes, A. L. Riley, H. I. Gomes, P. Brabham, J. Hamlyn, H. Pullin and P. Renforth, *Environmental Science & Technology*. <https://pubs.acs.org/doi/10.1021/acs.est.8b01883>, 2018, DOI: 10.1021/acs.est.8b01883.
90. C. Si, Y. Ma and C. Lin, *Journal of Hazardous Materials*, 2013, **244-245**, 54-59.
91. W. Mayes, J. Aumônier and C. J. Lawson, *High pH aquatic environments in the UK: an assessment of sources, hydro-geochemistry and vegetation as a prelude to the development of passive remediation technologies*, Passive Remediation of Alkaline Water to Neutral pH (PRAWN). MIRO Project RC174., School of Civil Engineering and Geosciences. Newcastle University, 2005.
92. I. Tiago, A. P. Chung and A. Verissimo, *Applied and environmental microbiology*, 2004, **70**, 7378-7387.
93. I. T. Burke, R. J. G. Mortimer, S. Palaniyandi, R. A. Whittleston, C. L. Lockwood, D. J. Ashley and D. I. Stewart, *Geomicrobiology Journal*, 2012, **29**, 769-779.
94. P. Krishna, A. G. Babu and M. S. Reddy, *Extremophiles*, 2014, **18**, 665-676.
95. L. Duro, J. Bruno, M. Grivé, V. Montoya, B. Kienzler, M. Altmaier and G. Buckau, *Applied Geochemistry*, 2014, **49**, 206-217.
96. A. Pallagi, É. G. Bajnóczi, S. E. Canton, T. Bolin, G. Peintler, B. Kutus, Z. Kele, I. Pálinkó and P. Sipos, *Environmental Science & Technology*, 2014, **48**, 6604-6611.
97. Z. Bian, X. Miao, S. Lei, S.-e. Chen, W. Wang and S. Struthers, *Science*, 2012, **337**, 702-703.
98. P. Deutz, in *Pathways to Environmental Sustainability*, ed. S. R. S. G., Springer, Dordrecht., 2014, pp. 3-11.
99. A. J. Gregory, J. P. Atkins, D. Burdon and M. Elliott, *European Journal of Operational Research*, 2013, **227**, 558-569.
100. D. R. Lombardi, D. Lyons, H. Shi and A. Agarwal, *Journal of Industrial Ecology*, 2012, **16**, 2-7.
101. M. Taneez, C. Hurel, N. Marmier and G. Lefèvre, *Applied geochemistry*, 2017, **78**, 272-278.
102. P. B. Cusack, M. G. Healy, P. C. Ryan, I. T. Burke, L. M. O'Donoghue, É. Ujaczki and R. Courtney, *Journal of Cleaner Production*, 2018, **179**, 217-224.
103. S. S. Mansouri, I. A. Udugama, S. Cignitti, A. Mitic, X. Flores-Alsina and K. V. Gernaey, *Current Opinion in Chemical Engineering*, 2017, **18**, 1-9.
104. G. Costa, A. Polettini, R. Pomi, A. Stramazzo and D. Zingaretti, *Greenhouse Gases: Science and Technology*, 2017, **7**, 530-541.
105. E. Iacovidou, J. Millward-Hopkins, J. Busch, P. Purnell, C. A. Velis, J. N. Hahladakis, O. Zwirner and A. Brown, *Journal of Cleaner Production*, 2017, **168**, 1279-1288.
106. N. Pietzsch, J. L. D. Ribeiro and J. F. de Medeiros, *Waste Management*, 2017, **67**, 324-353.
107. P. Deutz, H. Baxter, D. Gibbs, W. M. Mayes and H. I. Gomes, *Geoforum*, 2017, **85**, 336-344.
108. U. V. Oty, 2015.
109. K. Binnemans, Y. Pontikes, P. T. Jones, T. V. Gerven and B. Blanpain, Leuven, Belgium, 2013.
110. S. M. Mirazimi, Z. Abbasalipour and F. Rashchi, *J Environ Manage*, 2015, **153**, 144-151.
111. H. I. Gomes, A. Jones, M. Rogerson, G. M. Greenway, D. F. Lisbona, I. T. Burke and W. M. Mayes, *J Environ Manage*, 2017, **187**, 384-392.
112. I. Nikolić, A. Drinčić, D. Djurović, L. Karanović, V. V. Radmilović and V. R. Radmilović, *Construction and Building Materials*, 2016, **108**, 1-9.
113. P. Ning, X. Lin, X. Wang and H. Cao, *Chemical Engineering Journal*, 2016, **301**, 132-138.

114. W. Wang, Y. Pranolo and C. Y. Cheng, *Separation and Purification Technology*, 2013, **108**, 96-102.
115. Y. Qu, H. Li, W. Tian, X. Wang, X. Wang, X. Jia, B. Shi, G. Song and Y. Tang, *Minerals Engineering*, 2015, **81**, 1-4.
116. C. R. Borra, Y. Pontikes, K. Binnemans and T. Van Gerven, *Minerals Engineering*, 2015, **76**, 20-27.
117. M. Urik, M. Bujdos, B. Milova-Ziakova, P. Mikusova, M. Slovak and P. Matus, *J Inorg Biochem*, 2015, **152**, 154-159.
118. J. Roosen, S. Van Rosendael, C. R. Borra, T. Van Gerven, S. Mullens and K. Binnemans, *Green Chemistry*, 2016, **18**, 2005-2013.
119. H. I. Gomes, A. Jones, M. Rogerson, I. T. Burke and W. M. Mayes, *Environ Sci Pollut Res*, 2016, **23**, 23034-23042.
120. P. Davris, E. Balomenos, D. Panias and I. Paspaliaris, *Hydrometallurgy*, 2016, **164**, 125-135.
121. X. Zhu, W. Li, S. Tang, M. Zeng, P. Bai and L. Chen, *Chemosphere*, 2017, **175**, 365-372.
122. W. Zhang, R. Koivula, E. Wiikinkoski, J. Xu, S. Hietala, J. Lehto and R. Harjula, *ACS Sustainable Chemistry & Engineering*, 2017, DOI: 10.1021/acssuschemeng.6b02870.
123. B. Onghena, C. R. Borra, T. Van Gerven and K. Binnemans, *Separation and Purification Technology*, 2017, **176**, 208-219.
124. B. Yagmurlu, C. Dittrich and B. Friedrich, *J. Sustain. Metall.*, 2016, DOI: 10.1007/s40831-016-0098-9, 1-9.
125. Y. Huang, W. Chai, G. Han, W. Wang, S. Yang and J. Liu, *J Hazard Mater*, 2016, **307**, 318-327.
126. S. Nayak and N. Devi, *Hydrometallurgy*, 2017, **171**, 191-197.
127. C. Oh, S. Rhee, M. Oh and J. Park, *Journal of Hazardous Materials*, 2012, **213**, 147-155.
128. Y. Liyun, X. Ping, Y. Maomao and B. Hao, *Environ Sci Pollut Res*, 2016, DOI: 10.1007/s11356-016-8171-2, 1-12.
129. J. S. Ahn, C.-M. Chon, H.-S. Moon and K.-W. Kim, *Water Research*, 2003, **37**, 2478-2488.
130. V. K. Jha, Y. Kameshima, A. Nakajima and K. Okada, *Journal of Hazardous Materials*, 2004, **114**, 139-144.
131. F. Luan, L. Xie, J. Sheng, J. Li, Q. Zhou and G. Zhai, *Journal of Hazardous Materials*, 2012, **217-218**, 416-421.
132. Y. J. Zhang, L. C. Liu, Y. Xu, Y. C. Wang and D. L. Xu, *Journal of Hazardous Materials*, 2012, **209-210**, 146-150.
133. I. Blanco, P. Molle, L. E. Sáenz de Miera and G. Ansola, *Water Research*, 2016, **89**, 355-365.
134. F. M. Francisca and D. A. Glatstein, *Desalination and Water Treatment*, 2016, **57**, 21610-21618.
135. M. S. M. Zahar, F. M. Kusin and S. N. Muhammad, *Procedia Environmental Sciences*, 2015, **30**, 145-150.
136. C. Han, Z. Wang, W. Yang, Q. Wu, H. Yang and X. Xue, *Ecological Engineering*, 2016, **89**, 1-6.
137. G. Hua, M. W. Salo, C. G. Schmit and C. H. Hay, *Water Research*, 2016, **102**, 180-189.
138. T. Park, V. Ampunan, S. Maeng and E. Chung, *Chemosphere*, 2017, **167**, 91-97.
139. E. R. Goetz and R. G. Riefler, *Chemical Engineering Journal*, 2014, **240**, 579-588.
140. W. Nishijima, A. Tsukasaki, T. Tanimoto, M. Nagao, N. Tsurushima and M. Suzumura, *Ecological Engineering*, 2015, **81**, 418-427.
141. S.-Y. Arzate-Salgado, A.-A. Morales-Pérez, M. Solís-López and R.-M. Ramírez-Zamora, *Catalysis Today*, 2015, DOI: 10.1016/j.cattod.2015.09.026.
142. K. Komnitsas, G. Bartzas and I. Paspaliaris, *Minerals Engineering*, 2004, **17**, 183-194.
143. Y. Cengeloglu, A. Tor, M. Ersoz and G. Arslan, *Separation and Purification Technology*, 2006, **51**, 374-378.
144. H. Nadaroglu, E. Kalkan and N. Demir, *Desalination*, 2010, **251**, 90-95.

145. R. C. C. Costa, F. C. C. Moura, P. E. F. Oliveira, F. Magalhães, J. D. Ardisson and R. M. Lago, *Chemosphere*, 2010, **78**, 1116-1120.
146. E. López, B. Soto, M. Arias, A. Núñez, D. Rubinos and M. T. Barral, *Water Research*, 1998, **32**, 1314-1322.
147. V. K. Gupta and S. Sharma, *Environmental Science & Technology*, 2002, **36**, 3612-3617.
148. V. K. Gupta, I. Ali and V. K. Saini, *Environmental Science & Technology*, 2004, **38**, 4012-4018.
149. M. W. Clark, J. Harrison and T. Payne, *Journal of colloid and interface science*, 2011, **356**, 699-705.
150. I. D. Pulford, J. S. J. Hargreaves, J. Ďurišová, B. Kramulova, C. Girard, M. Balakrishnan, V. S. Batra and J. L. Rico, *Journal of Environmental Management*, 2012, **100**, 59-64.
151. J. Ye, X. Cong, P. Zhang, E. Hoffmann, G. Zeng, Y. Wu, H. Zhang and W. Fan, *Water, Air, & Soil Pollution*, 2015, **226**, 306.
152. Z. Liang, X. Peng and Z. Luan, *Environ Earth Sci*, 2012, **66**, 1321-1328.
153. S. C. Kim, S. W. Nahm and Y. K. Park, *J Hazard Mater*, 2015, **300**, 104-113.
154. A. S. Milenković, I. D. Smičiklas, M. Z. Šljivić-Ivanović, L. S. Živković and N. S. Vukelić, *Journal of Environmental Science and Health, Part A*, 2016, **51**, 679-690.
155. M. Taneez, N. Marmier and C. Hurel, *Chemosphere*, 2016, **150**, 116-122.

Table Captions

Table 7.1 Summary of key environmental risks associated with alkaline wastes with an overview of estimated timescales of potential impact and current level of scientific understanding (H: high, M: medium, L: low).

Adapted from Mayes et al. (2016)⁸

Table 7.2 Production of alkaline residues and typical e-tech element composition (mg kg^{-1}), percentage of annual global production in the respective residue assuming a 5% metal recovery rate and dissolved concentrations (passed a $< 0.45\mu\text{m}$ filter) in leachates ($\mu\text{g L}^{-1}$). Modified and updated from Gomes et al. (2016).¹

Table 7.3 Maximal recovery of critical metals from alkaline wastes – steel slag and red mud using different technologies.

Table 7.4 Summary of the use of alkaline residues for environmental applications.

Figure Captions

Figure 7.1 Bulk reuse^{4, 49}, environmental impact and opportunities for the management of steel slag and red mud (bauxite residue).

Figure 7.2 Availability (concentration in $\mu\text{g L}^{-1}$ in the leachates, logarithmic scale) of critical metals at high pH vs recovery technology readiness levels (TRL as defined by the European Commission). The difference in sizes is related with the metal concentration in the leachates.

Table 7.1

Source	Key pathway(s)	Key receptor(s)	Timescale of potential impact				Level of understanding	Comment
			Days	Weeks	Months	Years		
Fine particle size distribution (fugitive dusts)	Air	Humans	██████████				M	Short term risks similar to urban dusts
	Water	Aquatic biota (e.g. benthic smothering)	██████████				L	Particle size a possible contributor to negative response in aquatic bioassays. Not well quantified.
Salinity (red mud) / High ionic strength	Water	Aquatic biota	██				L-M	Longevity of issues in water minimised by dilution / remediation.
	Soil	Terrestrial plants	██████████				M-H	Long term salinization of soils identified as key risk in soil studies and of detriment to plant growth.
High alkalinity	Water	Aquatic biota Humans (dermal contact)	██				M	Associated issues for metal(loid) mobility at high pH; buffering by dilution (water) or carbonation (soils/waters) minimises timescale of issue.
	Soil	Terrestrial ecosystems	██████				M	Carbonation (via atmospheric CO2 and respiration of soil organisms) likely to limit in surface layers within weeks.
Metal(loid) enrichment	Water	Aquatic biota	██████████				L-M	Short term issues coupled with pH. Key risks for those not efficiently removed during dilution (e.g. V and Mo).
	Soil	Terrestrial plants	██████████				L-M	Possible genotoxic effects on plants in high dose rates. Residual issues surround anion exchange in soil-water systems.
Phosphorus enrichment (red mud)	Water Soil	Aquatic / terrestrial primary producers / ecosystems	██████████				L	Residual issues for long term cycling in soil-water environments and eutrophication risk. Largely unquantified.

Source	Key pathway(s)	Key receptor(s)	Timescale of potential impact				Level of understanding	Comment
			Days	Weeks	Months	Years		
Calcium enrichment (steel slag)	Water	Aquatic biota (e.g. benthic smothering)	██████████				M	Rapid recovery of invertebrate communities downstream of the discharges
Radionuclide presence (red mud)	Soils / Air	Terrestrial ecosystems / humans	██████████				M-H	Risks well characterised for red mud.

Table 7.2

Residue	Production (Mt a⁻¹)	Cobalt	Gallium	Lanthanum	Lithium	Scandium	Vanadium
Global production (t a ⁻¹) ³		123,000 (in 2016)	375 (in 2016)	126,00 (rare earths in 2016)	35,000 (in 2015)	10-15 (in 2016)	76,000 (in 2016)
Steel slag¹⁰⁸ (mg kg⁻¹)	170-250 (in 2014)¹	40-700²	30-60²	60-75²	1-140²	na	100-1500²
Percentage of annual global production assuming a 5% recovery rate		0.03-0.71%	7-20%	0.04-0.07%	0.002-0.5%	na	0.1-2.5%
Dissolved concentrations (µg L ⁻¹) in leachates (<0.45µm filter)		0.20 ³	na	na	4.4-822 ⁴	na	6-120 ⁵
Red mud (mg kg⁻¹)	118-177 (in 2016)^{3,4}	90-110	70-80	141-282	50-60	130-390¹⁰⁹	860-1100
Percentage of annual global production assuming a 5% recovery rate		0.4-0.8%	>110%	0.7-2%	0.8-1.5%	>7670%	0.9-10%
Dissolved concentrations (µg L ⁻¹) in leachates (<0.45µm filter)		17	12-2350	na	4-303	na	35-6398

na – not available

Table 7.3

Waste	Target metal	Recovery technology	Maximal recovery	Ref.
Steel slag	V	Bioleaching	92%	110
	V	Ion exchange	>99% (batch tests), 72% (column tests)	111
	Al and Si	Alkaline leaching	40%	112
	V	Extraction with primary amine	90%	113
	REE	Bioleaching	77%	15
Red mud	Sc	Solvent extraction	99%	114
	Ga, Ge, V, Sc, La, Eu, Yb	Bioleaching	63%	115
	REEE	Acid leaching	70%	116
	Al	Bioleaching	70%	117
	Sc	Chitosan-silica hybrid materials	100%	118
	V	Ion exchange	76% (column tests)	119
	REE	Ionic liquids	85%	120
	Sc and V	Ion exchange and solvent extraction	99%	121
	Sc	Ion exchange	91%	122
	Ga and REEE	Acid leaching	59% for Nd, 60% for Ga, 51% for La	78
	Sc	Combined pyrometallurgical-hydrometallurgical process with ionic liquid and solvent extraction	98%	123
	Sc	3 stage precipitation	80%	124
	Ti	Acid leaching and precipitate flotation	92%	125
	Ga	Acid leaching and ionic leaching	99%	126

Table 7.4

Material	Environmental purpose	Development stage	Efficiency range Main results	Ref.
Steel slag	Water treatment: removal of metals, nitrate, NH_4^+ and PO_4^{3-}	Batch tests	53–100%	127, 128, 129, 130, 131, 132-138
	Acid mine drainage treatment	Full scale	Steel slag leach beds lost more than 75% of peak alkalinity production within 50 empty bed volumes	139
	Substrate for eelgrass bed restoration	Mesocosm	Slag mixed with dredged material can be used for eelgrass beds	140
	Fenton-type photocatalyst for the degradation of an emerging pollutant (Diclofenac)	Batch tests	87%	141
Bauxite residue (red mud)	Water treatment: removal of metals, nitrate, chlorophenols, phosphate	Batch and column tests	20-100%	142,143, 144, 145,146, 147,148, 149, 150, 151
	Immobilization of Cd, Zn and Pb in sewage sludge	Batch tests	82-100%	152
	Catalytic oxidation of volatile organic compounds	Batch tests	100%	153
	Sorbent for Co^{2+} and Sr^{2+} immobilization	Batch tests	62-100%	154
	Amendment to stabilize sediments	Batch tests	40-72%	155

THE FOLLOWING IMAGES SHOULD BE SUPPLIED AS SEPARATE FILES in one of the following formats: TIFF/PDF/EPS/DOC/XLS/PPT/JPEG/CDX

Figure 7.1

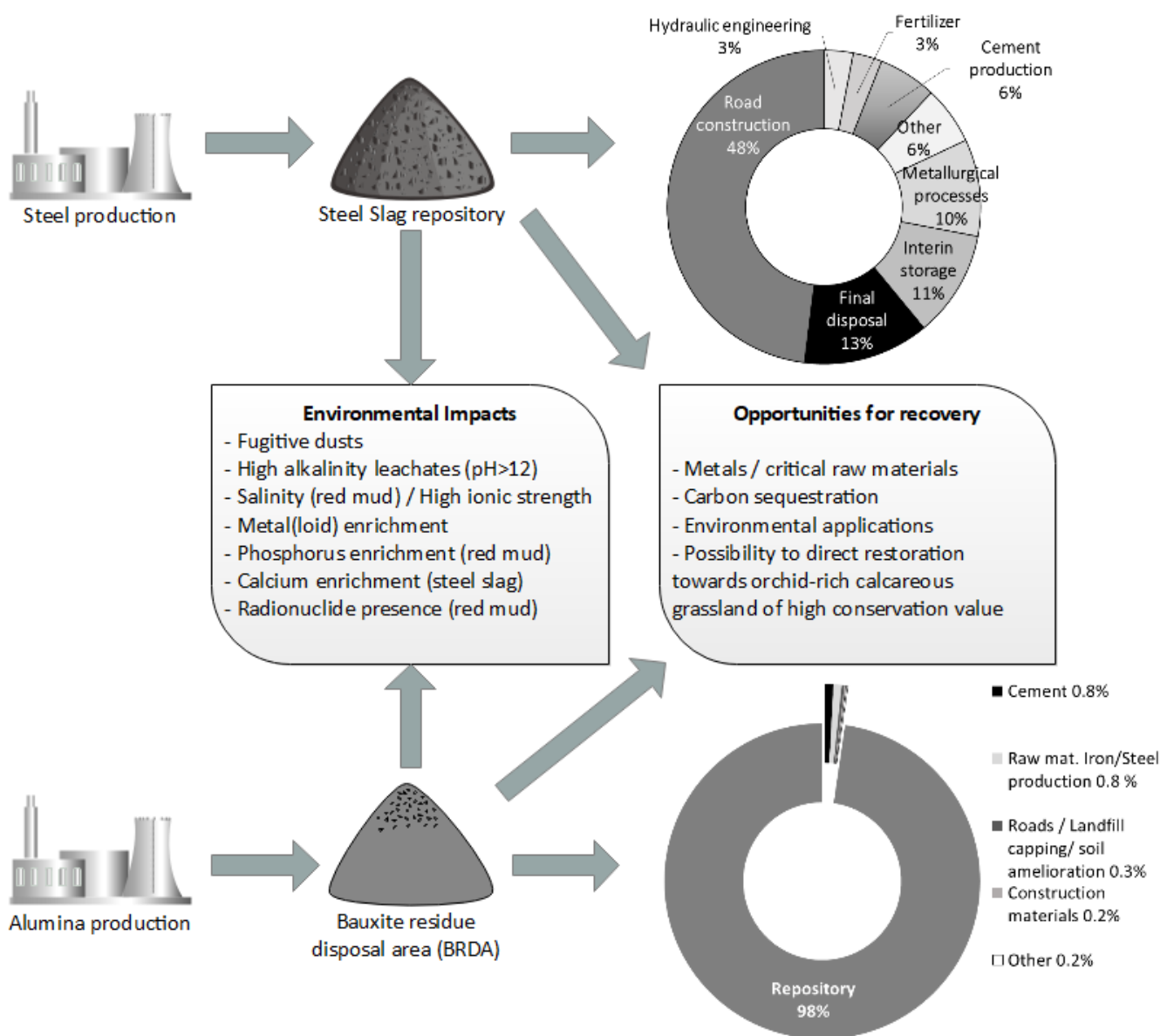


Figure 7.2

