

Canary in the coal mine: lessons from the Jarrah Forest suggest long-term negative effects of phosphorus fertilizer on biodiverse restoration after surface mining

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Canary in the coal mine: Lessons from the Jarrah Forest suggest long-term negative effects of phosphorus fertilizer on biodiverse restoration after surface mining

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Despite nutrient enrichment having widely reported negative impacts on biodiversity, fertilizer is routinely applied in *post* mining restoration to enhance plant growth and establishment. Focusing on surface mine restoration (predominately bauxite and mineral sands), we outline the long-term negative impacts of fertilizer, particularly phosphorus fertilizer, on plant community composition, species richness, fire fuel loads, and belowground impacts on nutrient-cycling. We draw from extensive research in south-western Australia and further afield, noting the geographical coincidence of surface mining, phosphorus impoverished soil and high plant biodiversity. We highlight the trade-offs between rapid plant-growth under fertilisation and the longer-term effects on plant communities and diversity. We note that the initial growth benefits of fertilisation may not persist in water-limited environments: growth of unfertilised forests can eventually match that of fertilised forest, throwing doubt on the premise that fertilisation is necessary at all.

KEYWORDS

diversity, legume, nitrogen, nutrients, rehabilitation

Introduction

Negative impacts of nutrient enrichment on biodiversity in natural and semi-natural ecosystems are widely reported (e.g., [Wheeler and Shaw, 1991](#); [Wassen et al., 2005](#); [Isbell et al., 2013](#)). Consequently, for ecosystem restoration of human-modified and degraded landscapes, one of the first considerations is often how to *reduce* the legacy of nutrient enrichment. Approaches include soil stripping and removal to reduce nitrogen or phosphorus (hereafter N and P, respectively) concentrations ([Diaz et al., 2008](#)),

applying carbon to reduce soil-N availability (Perry et al., 2010) or P-mining (*sensu* Schelfhout et al., 2015).

However, adding fertilizer to stimulate plant growth is often a key aspect of *post* mining restoration and is recommended as best practice in policy guidelines (e.g., EPA, 1995; Minerals Council of South Africa, 2007; Tibbett, 2010; DFAT, 2016). Further, many formal closure criteria and regulatory conditions require outcomes such as minimum levels of plant cover (e.g., Kragt et al., 2019; Manero et al., 2021) – and applying fertilizer can help achieve these goals. While well meaning, there is a risk that these guidelines focus too narrowly on short-term outcomes such as cover, with scant consideration to longer-term impacts on plant community assembly.

Using examples largely drawn from restoring bauxite mines in the Jarrah Forest of Western Australia, we outline why fertilizer has been used in mine restoration and how application rates compare with mining related nutrient losses. We further outline the suboptimal outcomes that result from adding P using examples drawn from bauxite and mineral-sands mine restoration. We focus on surface mining of bauxite and mineral sands because the returned soil profile is often intact (e.g., Audet et al., 2013; Standish et al., 2015; Riviera et al., 2021), compared with hard rock mining where novel growth substrates such as waste rock and process waste may be used. We acknowledge that the use of soil amendments and fertilizer inputs may be essential for overcoming initial barriers to vegetation establishment on novel substrates (e.g., Doley and Audet, 2013; Bateman et al., 2021). We also focus primarily on P, because P (unlike N) cannot be replaced by biological fixation, is often the most limiting nutrient in highly impoverished soils, and the source of P-input to restored soils only comes from fertilizer inputs (e.g., organic and inorganic fertilizers). Regolith weathering contributes P to soils too, but occurs over such impressively long time scales that it is not usually factored into restoration and management plans.

Surface mining often occurs in nutrient-poor, biodiverse landscapes

Bauxite occurs in ancient landscapes that have undergone significant periods of weathering and *in situ* leaching. Consequently, we tested whether bauxite and nutrient-deficient soils co-occur using a GIS analysis of the location of bauxite deposits (Mason and Arndt, 1996) combined with the location of ancient and deeply weathered nutrient-deficient soils downloaded from the FAO Soils Portal (i.e., Acrisols, Ferralsols, Lixisols, and Luvisols; FAO, 2012; Tibbett et al., 2019). A layer showing the world's plant species richness was also added (Barthlott et al., 1999), although it should be noted that to improve visual clarity, we simplified the original 10 zones of increasing richness into three (Figure 1).

There was a high degree of overlap between bauxite and nutrient-deficient soils. Specifically, 54 of the 62 (87%) bauxite deposits co-occurred with weathered, nutrient impoverished soils (Figures 1B–D). In addition, 81% occurred in regions of moderate (1,000–3,000 species) while 13% occurred in the regions of highest plant diversity ($\geq 3,000$ species; Figures 1A–D) reinforcing the link between nutrient-deficient soils and plant biodiversity (Isbell et al., 2013). A congruence of high diversity, a range of species adapted to nutrient-deficient soils and bauxite deposits, highlights the importance of a precautionary approach to applying P-fertilizer to achieve diverse ecological restoration after bauxite mining.

Many mineral sands deposits also occur in biodiversity hotspots (Figure 1) with 67% occurring in regions of moderate diversity (1,000–3,000 species) and 16% in the regions with highest diversity ($\geq 3,000$ species; Figures 1A–D). While formed through different processes to bauxite, mineral sands deposits are also likely to be nutrient impoverished. The minerals in mineral sands (i.e., rutile, ilmenite, and zircon) are originally derived from weathered igneous (e.g., granite, basalt) or metamorphic (e.g., schist) rocks, transported in fluvial systems and deposited in beach, lake, or river environments where the minerals are concentrated by wave action. Consequently, mineral sands deposits are often highly permeable, which together with the rapid weathering and leaching of minerals, forms sandy soils of low fertility (e.g., Richard's Bay, South Africa; Lubke et al., 1996).

In regions with weathered nutrient-deficient soils a significant proportion of species can have specialised adaptations for nutrient acquisition, such as cluster roots and exudation of carboxylates (e.g., Lambers et al., 2008), and/or have conservative growth rates (Daws et al., 2021a). These traits likely contribute to the sensitivity of restored ecosystems in these nutrient-deficient environments to applied-P. In addition, a range of species that occur in P-impoverished soils have been shown, at least in controlled conditions, to exhibit symptoms of toxicity to P when is added at concentrations greater than they would normally experience in native soils (Lambers et al., 2002; Shane et al., 2004; Pang et al., 2010; Williams et al., 2019; Tibbett et al., 2021). Further, the results of Holmes (2001) suggest that direct P-toxicity can occur in a field restoration setting. The potential for direct toxicity of applied-P to impact on restoration outcomes in regions with nutrient-deficient soils suggests a cautious approach to using fertilizer in these environments is appropriate and requires further investigation.

Fertilizer inputs to mine restoration

As part of the strip-mining process, topsoil and overburden are removed, stockpiled and replaced during restoration.

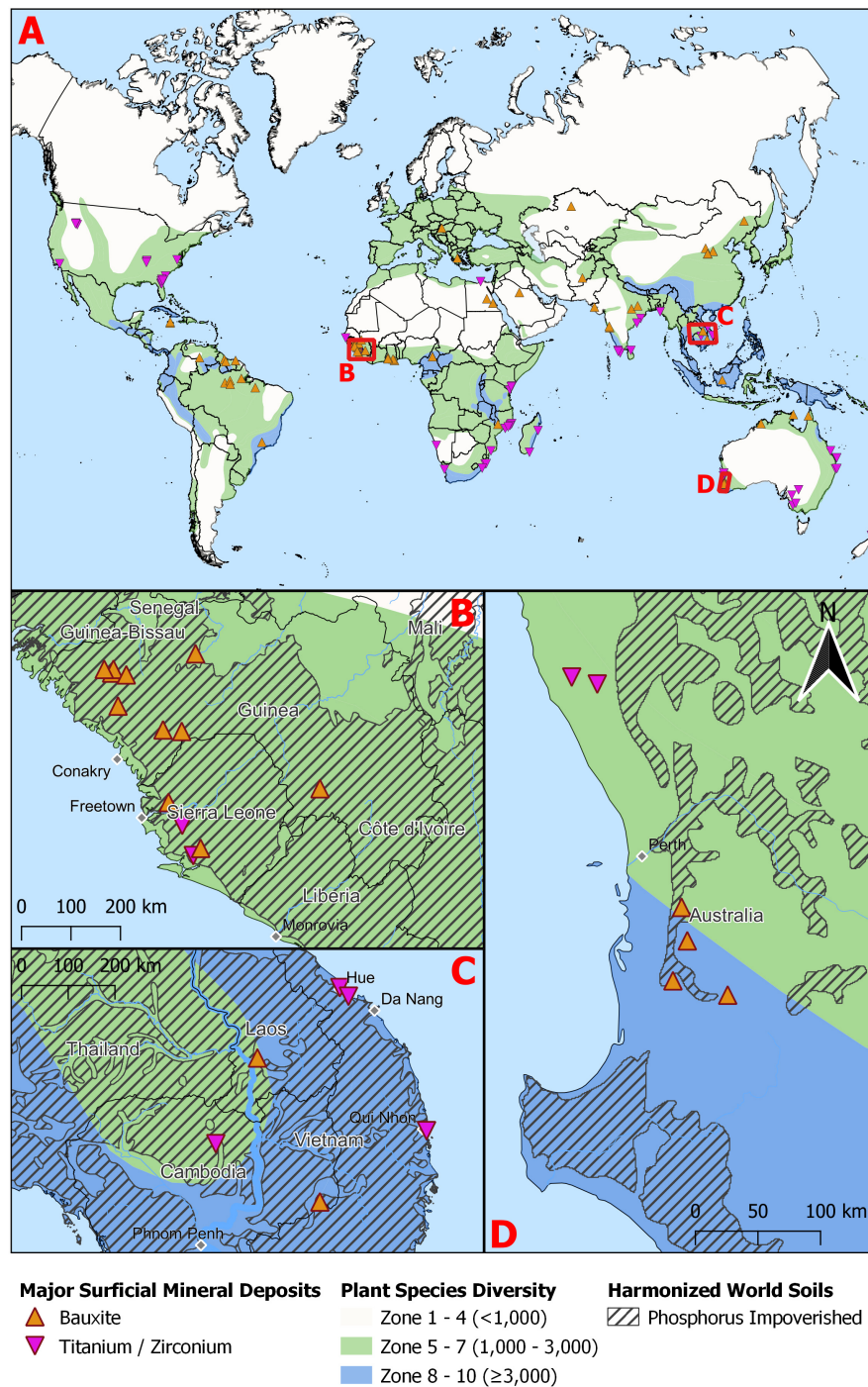


FIGURE 1

(A) The distribution of global biodiversity hotspots overlaid with bauxite and mineral sands (titanium/zirconium) deposits. Insets B–D are close ups of three regions in (B) West Africa, (C) South East Asia and (D) South West Australia to demonstrate the overlap between biodiversity hotspots, four ancient soil types (Acrisols, Ferralsols, Lixisols, and Luvisols) and bauxite and mineral sands (titanium/zirconium) deposits.

Handling large material volumes can result in mixing and redistribution of nutrients across the reconstructed soil profile. Furthermore, removing above-ground vegetation, prior to mining, may reduce the overall nutrient stock leading to the

perception that fertilizer inputs are required to “kick-start” nutrient-cycling (e.g., Grant et al., 2007).

For the Jarrah Forest in western Australia, fertilizer use in mine restoration commenced in the early 1980s and was

motivated by these goals: (1) re-establishing nutrient reserves in the soil, (2) re-establishing vegetation productivity, especially to maximize tree growth and future timber production, and (3) rapidly stabilizing restored surfaces to minimize erosion (Koch, 1985; Grant et al., 2007). These outcomes were reinforced by “Working Arrangements” with the State Government framed around maximizing productivity and timber production. Similar considerations have also been applied to mine sites elsewhere (e.g., Lawrie, 1985; Petersen, 1985).

Fertilizer-P inputs to Jarrah Forest restoration have ranged from 80 to currently between 20 and 40 kg ha⁻¹ (Standish et al., 2015; Tibbett et al., 2020), exceeding the estimated loss of ca. 14 kg P ha⁻¹ through removing 340–430 tonnes of aboveground vegetation biomass ha⁻¹ (Hingston et al., 1981; John Koch unpublished data). The Jarrah Forest is a tall, closed canopy eucalypt forest. For other restored vegetation types, such as open eucalypt woodland at Gove in Australia’s Northern Territory, the excess P added may be even greater since 25 kg P ha⁻¹ is applied (Spain et al., 2015) while the aboveground biomass removed is only ca. 108 tonnes ha⁻¹ (Chen et al., 2003), although P concentrations in wood are unknown for this site. Bauters et al. (2022) demonstrated that total above-ground P stocks in several neotropical forests range from approx. 20–50 kg ha⁻¹. While these data are rare, and fertilizer rates are often not reported, in some cases it appears excess P is applied for restoration, above what is lost as vegetation cleared for mining (Table 1). This could be especially true of sites where fertilizers are applied despite minimizing P loss by returning cleared vegetation (trunks and branches) to restoration sites (e.g., Barbosa et al., 2022).

Fertilizer inputs to restored Jarrah Forest and at Gove not only represent the likely addition of excess P, but also the addition of P in readily available forms (e.g., diammonium phosphate or superphosphate). In contrast, the P removed pre-mining in biomass has often been sequestered, in wood, for decades to centuries. The conundrum is how to replace this P in a form that is less readily available and “slow release.” One solution may be to use rock phosphate, however, even this can have short-term negative impacts on plant community reestablishment (Tibbett et al., 2020), potentially because a range of species from nutrient-impooverished environments release carboxylates to release P from strongly sorbed forms (Lambers et al., 2002).

Nutrient addition alters soil geochemistry and vegetation responses

A single initial fertilizer application to bauxite mining restoration in the Jarrah Forest can elevate available

TABLE 1 Available information on fertilizer-P application rates applied to restored bauxite and mineral sands mines.

Habitat type	Country	Elemental P-application rates
Bauxite mines		
Eucalyptus (Jarrah) forest	Darling Range, WA, Australia	20–40 kg ha ⁻¹ [20, 22]
Open Eucalyptus woodland	Gove, NT, Australia	25 kg ha ⁻¹ [19]
Eucalyptus woodland	Weipa, Queensland, Australia	26 kg ha ⁻¹ [17]
Semideciduous forest	Poços de Calda, Minas Gerais, Brazil	338 kg ha ⁻¹ [4]
Montane forest	Ouro Preto, Minas Gerais, Brazil	0 kg ha ⁻¹ [11]
Semideciduous montane forest	Descoberto, Minas Gerais State, Brazil	Applied but P-application rate not specified [2,10]
Tropical evergreen forest	Trombetas, Pará State, Brazil	Unclear if fertilizer was applied [13]; 69 kg ha ⁻¹ [9]
Tropical evergreen forest	Paragominas, Pará State, Brazil	29 kg ha ⁻¹ applied at planting, but total rate unclear as P was also applied prior to planting [3, 8]; Unclear if fertilizer was applied [15];
Mineral sands mines		
Kwongan	Eneabba, WA, Australia	Applied but P-application rate not specified [12, 16]
Kwongan	Cooljarloo, WA, Australia	Applied but P-application rate not specified [16]
Banksia woodland	Perth, WA, Australia	Applied but P-application rate not specified [21]
Open woodland	Wemen, Victoria, Australia	Unclear if fertilizer was applied [18]
Coastal rainforest	Coffs Harbour, NSW, Australia	20 kg ha ⁻¹ [6]
Open forest/woodland	Stradbroke Island, Queensland, Australia	Unclear if fertilizer was applied [24]; Applied but P-application rate not specified [1]
Coastal dune forest	Richard’s bay, KwaZulu Natal, South Africa	Unclear if fertilizer was applied [7, 23]
Succulent Karoo	Brand-se-Baai, Namaqualand, South Africa	Unclear if fertilizer was applied [5, 14]

Numbers in parentheses relate to the references below the table.

1 Audet et al. (2013); 2 Balestrin et al. (2020); 3 Barbosa et al. (2022); 4 Bizuti et al. (2020); 5 Blood (2006); 6 Cummings et al. (2005); 7 Lubke et al. (1996); 8 Martins et al. (2021); 9 Melo et al. (2018); 10 Neto et al. (2014); 11 Onésimo et al. (2021); 12 Osborne and Schatral (1997); 13 Parrotta and Knowles (2001); 14 Pauw et al. (2018); 15 Ribeiro et al. (2019); 16 Riviera et al. (2021); 17 Short et al. (2000); 18 Sluiter et al. (2016); 19 Spain et al. (2018); 20 Standish et al. (2015); 21 Stevens et al. (2016); 22 Tibbett et al. (2020); 23 Van Aarde et al. (1998); 24 Van Gorp and Erskine (2011).

(bicarbonate extractable) soil P concentrations for more than 20 years (Banning et al., 2008; Daws et al., 2021a) suggesting that applying P at the onset of restoration, may have long-term ongoing impacts on vegetation responses. In addition, at Gove bauxite mine, Spain et al. (2018) demonstrated that labile-P and long-term P concentrations in near-surface soils increase over

a 26-year period following an initial P-application suggesting fertilizer-induced alterations to geochemical processes and nutrient-cycling.

Fertilizer addition can benefit weeds in restored systems (Daws et al., 2021a; Holmes, 2001). P can also preferentially benefit species such as N₂-fixing legumes (e.g., *Acacia* spp.). N₂-fixing species may benefit when fertilizers containing only P are used (since they themselves are not limited by N-availability) and because of a mismatch in the timeframes of availability of applied N and P. Specifically, applied-N can be lost rapidly from the soil due to leaching or volatilisation (depending on the form of the applied N and soil pH). Therefore, any potential growth benefit of applying N is short-term while P may be available over the longer term. A vigorous growth response of weeds and/or legumes can result in other species being outcompeted leading to reduced understorey diversity (Daws et al., 2015, 2019a, 2021a). These reductions in species richness can be significant. For example, species richness in 20-year-old restored Jarrah Forest sites was reduced between 20 and 25% as the initial fertilizer-P application rate increased from 0 to 80 kg ha⁻¹ (Daws et al., 2019a) and slow-growing resprouter species, that are important for post-fire resilience, were particularly impacted (Daws et al., 2019a,b, 2021a). While these negative impacts of a single, initial application of P-fertilizer on species richness and community composition persist for at least 20 years we are still learning about fertilizer impacts on long(er)-term vegetation trajectories.

The establishment of a dominant legume layer may also result in restored sites being locked into alternate successional pathways (Grant, 2006) at least partly because legumes may produce large quantities of leaf litter and debris that limits seedling emergence and establishment of other species (Tibbett, 2010; Boyes et al., 2011; Le Stradic et al., 2014; Daws et al., 2019a,b). In the Jarrah Forest, vigorous legume establishment, combined with the formation of a prolific soil seed bank can result in the establishment of a “legume-cycle” where legumes rapidly germinate from the soil seed bank, and re-establish dominance, *post* fire (Grigg and Grant, 2009). Legume dominance has also been reported in restored sites on Stradbroke Island, Australia (Rogers and Mokrzecki, 1984) and at Richard’s Bay in South Africa. At Richard’s Bay, *Vachellia karoo* (Hayne) Banfi & Galasso dominates some restored sites and can arrest succession for 50 or more years (Boyes et al., 2011). While it is unclear whether the initial establishment and dominance of *V. karoo* was facilitated by P-addition (Table 1), Van Aarde et al. (1998) reported significantly higher soil-P concentrations in younger restored sites than either older restored sites or unmined reference forest suggesting that fertilizer was applied. Interestingly on Stradbroke Island, a similar pattern has been observed with the N₂-fixing species *Allocasuarina littoralis* (Salisb.) L. A. S. Johnson (Casuarinaceae) dominating some restored sites and suppressing other species

by producing a dense layer of “needles” (leaf litter) on the soil surface (Audet et al., 2013). Further studies into the dominance of N₂-fixing species in restored sites and interactions with fertilizer application would be of value.

The increased growth of P-responsive legumes and production of leaf litter and debris can also increase fire fuel load accumulation and fire risk. For example, an initial application of 80 kg P ha⁻¹ at the outset of restoration resulted in the litter depth (and hence fine fuel loads) in 15-year-old restored sites being increased 3-times compared to unfertilized sites (Daws et al., 2019a). Similarly, at Gove, adding an initial 25 kg P ha⁻¹ resulted in litter depth being elevated by about 3-times in 25-year-old restored sites compared to non-mined sites. This was due to a combination of not only increased litter inputs, but also reduced litter decomposition rates suggesting ongoing impacts on microbial and decomposer communities (Spain et al., 2015).

A vigorous response of legumes to applied-P also increases atmospheric N₂-fixation, thereby increasing soil-N pools, which can be a goal of restoration where the intent is to increase N-availability for maximizing tree growth (Koch, 1985; Ward and Koch, 1995). For example, following application of an initial 80 kg P ha⁻¹, N₂ fixation rates of up to 207 kg N ha⁻¹ year⁻¹ have been reported in Jarrah Forest restoration (Koch, 1987). However, these rates far exceed fixation rates of 7 kg N ha⁻¹ year⁻¹ reported in unmined forest (Hingston, 1980). In addition, the high rates of N-fixation are associated with a drop of up to one unit in soil pH compared with soils in unmined forest (Ward, 2000). The implications of these changes for soil processes and above-ground vegetation responses are unknown but could be significant and require further research.

In the short-term, using fertilizer in restoration appears beneficial due to increases in both plant cover and growth of individual plants (Daws et al., 2013). Rapid re-establishment of cover may both minimize soil erosion (Ward, 2000) and give the visual impression that restoration efforts have been successful. However, the growth benefits may not persist. For example, in restored Jarrah Forest height and diameter growth of jarrah trees (*Eucalyptus marginata* Sm.) from 0 to 9 years-of-age increases with applied N or P, but by 20 years-of-age these differences have disappeared (Walters et al., 2021). Similarly, Grigg and Grant (2009) found that restored jarrah stands aged 10–13 years only exhibit a growth response to applied fertilizer when water limitation is temporarily reduced by stand thinning, a response also observed in unmined forest (Stoneman et al., 1997; Daws et al., 2021b). This observation is not unique to jarrah. For example, Campion et al. (2006) demonstrated that 4-year-old *Eucalyptus grandis* in South Africa was constrained more by water availability than soil nutrient supply. Hence, like the tortoise and the hare, unfertilized jarrah stands catch up eventually—questioning the premise of fertilizing these stands, particularly since they will not be logged until at least 70 years of age (Conservation Commission of Western Australia, 2013).

Discussion

Fertilizer addition may help achieve individual restoration goals, such as maximizing plant cover, but applying fertilizer conflicts with returning biodiverse ecosystems (see [Supplementary Figure 1](#)). From a policy perspective reframing targets away from short-term aspects such as growth rates, productivity and cover to focus on longer-term diversity, resilience and community composition may be more appropriate. Since diversity, resilience and productivity are linked in natural ecosystems ([Isbell et al., 2015](#)), the increased diversity associated with reducing fertilizer rates may have wider benefits for ecosystem function.

When assessing restoration success long-term outcomes matter. In the short-term fertilizer application to restored Jarrah Forest increases native diversity (by increasing abundance of short-lived ephemerals), plant growth rates and total cover ([Daws et al., 2013](#); [Tibbett et al., 2020](#)). While over the longer term, self-sustaining and biodiverse Jarrah Forest ecosystems have been successfully restored ([Koch and Hobbs, 2007](#)) there can also be negative impacts on soil biogeochemistry and species composition. Further, any initial growth benefits of applying P – but not the legacy – disappear over time ([Spain et al., 2018](#); [Daws et al., 2019a, 2021a](#)). In short, increased initial growth does not mean improved long-term outcomes, not least by upsetting the balance of competitive interactions between species. Consequently, there are further improvements that can be made when pursuing biodiverse outcomes from restoration in nutrient-impooverished environments.

A common theme from the literature is that while fertilizer is typically applied to mine restoration, the precise rates applied are often unclear ([Table 1](#)). To understand impacts, it is necessary to know both fertilizer rates and type (e.g., NPK versus P-only and formulation e.g., 9.1% P), as these aspects may affect vegetation reestablishment ([Tibbett et al., 2020](#); [Daws et al., 2021a](#)). While we have focused on impacts of P- fertilizer, impacts of N and K fertilizers are likely to be significant in some cases. For example, [Tibbett et al. \(2020\)](#) reported that NPK fertilizers resulted in higher plant diversity than using just P-based fertilizers. While the mechanism(s) behind this response to including N in the fertilizer mix is unclear, applying N may, at least initially, maintain a N:P ratio in the soil more akin to reference forest. It is also possible that including N limits the establishment and competitiveness of N₂ fixing species, since nitrate and ammonium addition can depress nodule production in *Acacia* seedlings (e.g., *A. auriculiformis* A. Cunn. ex Benth.; [Goi et al., 1992](#)). Notably, despite containing the same quantity of P as the P only treatment, NPK fertilizer treatment resulted in significantly lower soil ammonium concentrations, suggesting an impact on atmospheric N₂-fixation by legumes ([Tibbett et al., 2020](#)). The impact of different fertilizer types on N:P stoichiometry and ecosystem responses requires further research.

These findings are likely applicable to a broad range of bauxite mine restoration beyond the Jarrah Forest because bauxite largely overlaps with nutrient-deficient soils often in biodiverse environments. Since research on Jarrah Forest restoration after bauxite mining is more advanced than restoration after bauxite mining in other regions further studies of fertilizer impacts on responses in a broader range of plant community-types are warranted. In a recent study of bauxite mine restoration in Brazil, [Onésimo et al. \(2021\)](#) ascribed their relative success, compared with earlier studies in the region, to not applying fertilizer, although this remains to be tested experimentally. Mineral sands mining also often occurs in regions with nutrient-poor soils (Section “Surface mining often occurs in nutrient-poor, biodiverse landscapes”) and high plant diversity, and there is emerging evidence that restoration of these environments may also be sensitive to P-application. For kwongan restoration after sand mining in Western Australia, [Riviera et al. \(2021\)](#) showed that P-addition was the management intervention with the largest impact on community composition. More research is needed to understand the generality of these findings.

We propose that P-limitation and moderation of fertilizer-P inputs are important for restoring naturally P-impooverished environments. While the majority of available research comes from the Jarrah Forest of south-western Australia, we view it as the “canary in the coal mine.” The considerable overlap worldwide, among areas of high plant diversity, nutrient-deficient soils and bauxite and mineral sands mining, suggest these findings will not be unique. Ongoing research from elsewhere in the world, such as that emerging from Brazil and South Africa, will be an important test of the generality of the findings for the Jarrah Forest.

Data availability statement

The original contributions presented in the study are included in the article/[Supplementary Material](#), further inquiries can be directed to the corresponding author.

Author contributions

MD wrote the manuscript. All authors contributed to the conceptual development of this manuscript.

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Conflict of interest

MD and CB are paid employees of Alcoa of Australia Ltd.

The remaining authors declare that the research was conducted in the absence of any commercial or financial

relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/ffgc.2022.786305/full#supplementary-material>

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