Office of the **Queensland Mine Rehabilitation Commissioner**

Evaluating Native Ecosystem Rehabilitation Options in Queensland

Technical paper

Queensland Government

Prepared by: Office of the Queensland Mine Rehabilitation Commissioner

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Contents

Figures

Tables

Executive Summary

In Queensland, the objective of mine rehabilitation is for land disturbed by mining to attain a 'stable condition', which is defined as land that is safe, structurally stable, does not cause environmental harm, and is able to sustain a post-mining land use (PMLU) (*Environmental Protection Act 1994* (Qld) (EP Act), s111A). This paper outlines a framework that supports mine rehabilitation planning in Queensland where the desired PMLU is a native ecosystem. Biophysical limitations that influence the feasibility of native ecosystem outcomes (landform, climate, soil, fauna, regional connectedness) are discussed, and best practice approaches to overcome these limitations outlined.

If biophysical conditions similar to those that existed pre-mining can be restored, then rehabilitation that targets *natural historical* ecosystems (Doley and Audet, 2013, 2016) is best practice. Within Queensland, such ecosystems must resemble a naturally occurring regional ecosystem (RE) (see Neldner et al 2019). Establishing *historical* ecosystems is best practice because it restores local species that have evolved for local conditions and the ecosystem is therefore more likely to be sustainable in the long-term with lower ongoing maintenance burdens (Gould, 2012; Guimarães et al., 2013; Gastauer et al., 2019). *Natural* ecosystems also deliver the ecosystem services such as water supply, carbon sequestration, recreational and cultural areas and genetic resources (Costanza et al. 2017) that are appropriate for and exist within the region naturally (e.g., Rosa et al. 2020).

In some cases, mining disturbances fundamentally alter the abiotic and biotic conditions of sites, which can make restoring *historical* ecosystems impractical and/or unrealistic. We therefore expand this concept of *natural* rehabilitation to include *substitute* ecosystems, which allows for managers to select another RE within the bioregion that more closely resembles the post-mining conditions. Considering *substitute* ecosystems greatly broadens the range of native ecosystems that can be targeted, whilst utilising coevolved suites of species that will be consistent with the climate and landforms of each site's bioregion.

If biophysical limitations remain (or are predicted to remain) after utilisation of best practice rehabilitation methods to mitigate them, alternative native ecosystem rehabilitation targets may be feasible. *Hybrid* ecosystems (Higgs, 2017; Gwenzi, 2021) that are dominated by native species, are one example. *Hybrid* ecosystems represent a significant deviation from *natural* ecosystems, but their key ecosystem functions and attributes remain largely the same (Hobbs et al., 2009; Doley and Audet, 2013; Clement and Standish, 2018). One advantage of targeting *hybrid* ecosystems is that it allows for flexibility in species selection (e.g., a mixture of native species that do not naturally occur together) while still retaining enough naturally occurring attributes that, if desired later, the ecosystem can be manipulated to resemble a more *natural* ecosystem. For example, species can be selected based on traits that best suit post-mining conditions (e.g., that tolerate low nutrients and high salinity in soils) in order to quickly establish vegetation cover and habitat features. Over time, these biophysical limitations can decrease (e.g., through accumulation of organic matter) and through manipulations (e.g., selective thinning) the *hybrid* ecosystem may be able to be managed towards a more *natural* state over the long-term (Hallett et al., 2013; Hobbs et al., 2013).

Another alternative option for native ecosystem rehabilitation is a *planned novel* ecosystem that meets specific human interests but is aligned with native ecosystem rehabilitation targets, such as biodiversity improvement (Higgs, 2017). *Planned novel* ecosystems may be targeted where (a) there are severe biophysical constraints and *natural* or *hybrid* ecosystems cannot be established, and (b) there is sufficient evidence that the *planned novel* ecosystem will deliver beneficial environmental outcomes (Environmental Protection Regulation 2019, Schedule 8A, part 3) above and beyond the fundamental requirement for land to be safe, structurally stable and not cause environmental harm. Examples include specific human-oriented uses, for example carbon sinks (Pietrzykowski and Daniels, 2014; Tripathi et al., 2016) or native seed orchards (Nichols et al., 1985; Gardner and Bell, 2007; Annandale et al., 2021).

There are some cases of established mine rehabilitation in Queensland that may be considered *unplanned novel* ecosystems. *Unplanned novel* ecosystems contain abiotic and biotic characteristics that do not occur in *natural* ecosystems (Doley et al., 2012; Doley and Audet, 2013), and they are self-sustaining and have crossed theoretical ecological thresholds, meaning they cannot be transformed into *natural* ecosystems through management intervention (Erskine and Fletcher, 2013; Gastauer et al., 2018). *Unplanned novel* ecosystems may result from biophysical limitations, or because these limitations were not addressed through best practice rehabilitation methods during rehabilitation. By definition, these ecosystems are the unintended result of human actions (or inactions) and are not considered a best practice rehabilitation target for new rehabilitation sites.

Hybrid, planned novel and *unplanned novel* ecosystems inherently imply greater uncertainty in terms of ecosystem sustainability and resilience, compared to rehabilitated *natural* ecosystems. Furthermore, non-natural rehabilitated ecosystems will likely require increased ongoing management in the future, compared to *natural* (*substitute* or *historical*) ecosystems. Therefore, these alternative native ecosystem PMLUs should be considered with caution.

1. Introduction, scope and definitions

The purpose of this report is to develop a framework to support life-of-mine decision-makers and rehabilitation planning professionals to compare and evaluate native ecosystem rehabilitation options, given the constraints and biophysical limitations imposed by mining disturbance. Rehabilitation of native ecosystems across highly disturbed landscapes may result in *natural* ecosystems (restored to the extent of historic succession trajectory), *hybrid* ecosystems (having some but not all characteristics of the natural/historic landscape, and some novel attributes), or *novel* ecosystems (new assemblies of abiotic and biotic attributes resulting in a stable alternative ecological form that does not resemble natural ecosystems—Doley et al. 2012, Doley and Audet 2013). This report also assesses the value of native ecosystem rehabilitation in the regional context for each of these possible outcomes and provides recommendations as to the adequacy of these outcomes and managing native ecosystem rehabilitation planning on mine sites within Queensland. We define "native species", as those identified as such in the Queensland Plant Census (Brown, 2021), and "native ecosystem rehabilitation" as rehabilitation dominated by native species. For details regarding dominance and composition within Queensland's native ecosystems, refer to Neldner et al. (2019) and Neldner et al. (2022). Exotic species are defined as those identified as 'naturalised' in the Queensland Plant Census (Brown, 2021) and an invasive weed as any species that threatens the achievement of rehabilitation objectives or the biodiversity values of regional ecosystems (REs).

Restoration of *natural* ecosystems is the goal of restoration ecology (Clewell and Aronson, 2013; Standards Reference Group SERA, 2021) and is a common objective of mine rehabilitation in Australia. Completion criteria for mines often require the re-established vegetation to be comparable with remnant local areas or pre-mining baselines (Nichols, 2004; Shackelford et al., 2013; Erskine et al., 2019). This endeavour involves establishing (or setting a trajectory towards) the species composition, structure and function of a specific naturally occurring ecosystem (an RE within the Queensland framework — Neldner et al. 2019). Such restoration is appropriate when soils and other environmental conditions remain suitable (Lamb et al., 2015), or can be made so with targeted techniques and management inputs. Doley and Audet (2013, 2016) define rehabilitation/restoration of *natural* ecosystems as synonymous with establishing the pre-disturbance (i.e., *historical*) ecosystem. However, this definition is too restrictive as it ignores the option of substituting another naturally occurring RE. Here, we expand the definition of *natural* rehabilitated ecosystem targets to also include *substitute* native rehabilitation. A suitable *substitute* ecosystem necessarily must occur within the bioregion of the site, or within a neighbouring bioregion if geographically close e.g., within 30 km to a bioregion boundary as per the consideration for RE outliers (Neldner et al., 2022).

Given that mining disturbances often fundamentally alter landscapes and the geological, pedological, hydrological, and topological conditions (Erskine and Fletcher, 2013; Paradella et al., 2015; McCaffrey et al., 2017; Hancock et al., 2020; Gwenzi, 2021), rehabilitation efforts may not be able to establish *natural* ecosystems that are similar to pre-mining *historical* ecosystems (Humphries and Tibbett, 2015; Gastauer et al., 2019). *Substitute* ecosystem targets may therefore be appropriate when differences between the physicochemical and biological characteristics of the pre-mining versus rehabilitated mine environments preclude restoration (*sensu stricto*) of the *historical* ecosystem (Doley et al., 2012). For example, in mine types such as bauxite, where most of the disturbed material is product and is removed, the resulting landscape is lowered (Taylor et al., 2008); in such circumstances, *substitution* of a new RE vegetation community suitable to the altered soil depth and/or hydrological regime may be appropriate (Gould, 2011, 2012). Similarly, where mining creates waste rock dumps, *substitute* ecosystems that exist in areas topographically analogous to the new landform may be appropriate, such as vegetation typical of a local mesa instead of the rolling plains present prior to mining (Gillespie et al., 2015). Thus, the approach of ecosystem *substitution* greatly broadens the range of native ecosystems that can be targeted for rehabilitation, whilst pragmatically accounting for biophysical limitations that are imposed by the mining process and subsequent landscape reconstruction.

Given the radical changes to almost every component of the landscape that mining disturbance causes, it has been proposed that rehabilitation to a *natural* ecosystem is an unrealistic outcome in some circumstances (Doley et al., 2012) and that *novel* rehabilitated ecosystems are an alternative stable form (Doley et al., 2012; Doley and Audet, 2013). *Novel* ecosystems contain community assemblages that have not existed naturally before; typically including both native and exotic species (Higgs, 2017). *Novel* ecosystems are stable ecosystems that often develop in highly disturbed sites as the **unintended** result of human alteration of the environment (Hobbs et al., 2014). Once established, management intervention cannot drive these ecosystems towards more natural states, therefore they are said to have crossed an irreversible disturbance threshold (Erskine and Fletcher,

2013; Gastauer et al., 2018).

In contrast, intentionally created non-natural ecosystems are known as *designed* (or *designer*) ecosystem (Higgs, 2017). *Designer* ecosystems are goal-oriented, established to meet specific objectives with a primary focus on human interests (Ross et al., 2015), although these interests may relate to goals that align with native ecosystem rehabilitation, such as biodiversity improvement (Higgs, 2017). *Designer* ecosystems are thus the **planned** counterpart to novel ecosystems (Table 1).

Sitting between *natural* and *novel* ecosystems are *hybrid* ecosystems (Higgs, 2017; Gwenzi, 2021). They represent a significant deviation from *natural* ecosystems, but their key ecosystem functions and attributes remain largely the same (Hobbs et al., 2009; Doley and Audet, 2013; Clement and Standish, 2018). The species assemblages that characterise *hybrid* ecosystems do not naturally occur (Gastauer et al., 2018), although in response to human activity some *natural* ecosystems may seem to be *hybrid* (e.g., REs that have an introduced grass species as the groundcover layer). A feature distinguishing *hybrid* ecosystems from *novel* ecosystems is that they may be manipulated to become a *natural* ecosystem (Hallett et al., 2013; Hobbs et al., 2013). Thus, if it is biophysical limitations (see section 3) that dictated the formation of the *hybrid* ecosystem then these limitations may be reversible, in contrast to the limitations dictating eventuation of *novel* ecosystems. Or if there are introduced species that make ecosystems *hybrid*, weed management can drive the system towards a more *natural* state.

Hybrid, *novel* and *designer* ecosystems are collectively termed *no-analogue* (Seastedt et al., 2008; Hobbs et al., 2013; Evers et al., 2018) ecosystems. Importantly, the distinction between *natural*, *novel* and *hybrid* is somewhat arbitrary and difficult to define precisely. As these definitions were originally developed to explain ecosystems that develop in disturbed unmined landscapes, the concepts should be applied to mine rehabilitation planning with care (Hobbs et al., 2009; Doley and Audet, 2016). Table 1 summarises the definitions relevant to native ecosystem rehabilitation of mine sites in Queensland.

Table 1. Rehabilitated native ecosystem classes, characteristics and management considerations

Native ecosystem class	Subclass	Characteristics of the rehabilitated native ecosystem	Management considerations
Natural	Historic	Abiotic and biotic characteristics of the RE that was present pre-mining.	Post-mining management expected to be similar to management of pre-mining RE.
Natural	Substitute	Abiotic and biotic characteristics that differ from those in the pre-mining RE, but analogous to another RE within the bioregion.	Post-mining management expected to be similar to management of REs in the surrounding bioregion.
Hybrid	n/a	Ecosystem functions are similar to an RE, but ecosystem is characterised by unique attributes that are not found in an RE. These unique aspects can be overcome by management to move the ecosystem towards an RE.	Management action can be taken to manipulate these systems towards an RE.
Novel	Unplanned	The unintentional result of attempts to establish an RE where biophysical limitations or rehabilitation techniques have resulted in a unique and stable assemblage that does not have an analogous RE. These unique aspects cannot be managed to move the ecosystem towards an RE.	Stable ecological form that cannot be manipulated to become an RE via management intervention.
Novel	Planned	A planned native ecosystem that meets specific ecosystem services objectives but has no RE analogue (i.e., it is intentionally novel). Also known as a designer ecosystem.	Self-sustainability unknown, though expected to require management. Cannot be manipulated to become a RE.

2. Framework for decision-makers and rehabilitation planning professionals

Rehabilitation must return disturbed land that is safe, structurally stable, does not cause environmental harm, and is able to sustain a PMLU (*Environmental Protection Act 1994* (Qld) (EP Act), s111A). Where the PMLU is native ecosystem, we have developed a hierarchical flowchart (Figure 1) to support best practice decision-making. The flowchart is intended for use where native ecosystem rehabilitation is the primary or only PMLU. While some rehabilitation will incorporate other PMLUs in addition to native ecosystems (e.g., grazing), this flowchart was designed to support native ecosystem PMLUs.

Figure 1 implies a preference to restore native biodiversity and functionality as faithfully as possible (Lamb et al., 2015). This does not mean that restoration of *historical* ecosystems should be the objective in all cases, given that there are limits to resources and technical capacities. Rather, a case-by-case approach is warranted, involving an evaluation of feasibility based on the biophysical limitations (see section 3 for details), and utilising an evidence-based decision framework (e.g., Murcia et al. 2014).

Native ecosystem rehabilitation favours endemic species, but there is a spectrum of endemicity. Utilising local species is a higher priority than utilising species from the general bioregion or even from further afield (Figure 1). Within our framework, *historical* is at the top of the native ecosystem rehabilitation hierarchy as it achieves the highest level of endemicity, (i.e., restoration of the pre-mining RE; Miller et al. 2016). The next level in the hierarchy is *substitute* as that aims to restore a specific RE, but not the one that was present historically. Ecosystem targets beyond the bioregion in which the mine occurs, or within 30 km (Neldner et al. 2022), would not generally be considered as *substitute* candidates.

Natural ecosystem rehabilitation is regarded as best practice. Since these community assemblages evolved in the landscape, they are pre-adapted to historical climatic conditions and disturbance regimes. In addition, the species associations are known to function together and be self-sustaining (Cooke and Johnson, 2002; Gould, 2012). *Natural* ecosystem rehabilitation restores a local community, thereby achieving native biodiversity values at the level of ecosystem type. Finally, *natural* ecosystem rehabilitation has the highest conservation value and, once established, the lowest requirement for ongoing management (Guimarães et al., 2013; Gastauer et al., 2019). Where rehabilitation to *natural* ecosystems is successful, these ecosystems also deliver the ecosystem services—the processes and functions that benefit people either directly or indirectly such as water supply, carbon sequestration, pollination, biological control of pests, recreational and cultural areas and genetic resources (Costanza et al. 2017) that are appropriate for and exist within the region naturally (Rosa et al., 2020).

It is recommended that post-mining conditions be carefully assessed, then compared to existing conditions of local and bioregional REs, to determine which rehabilitation target is most appropriate. Successful mine rehabilitation to *natural* ecosystems of both the *historical* and *substitution* types has been achieved (e.g., Milton 2003, Grant and Koch 2007, Koch 2007) but has required substantial intervention (e.g., careful recovery and reuse of soils and appropriate reconstruction of landform, water resources and drainage conditions; Gardner and Stoneman 2003, Grant and Koch 2007, Humphries and Tibbet 2015, Lamb et al. 2015). Refer to Table 2 for examples of best practice methods for native ecosystem rehabilitation of mines. Rehabilitating to a *natural* state is likely to require more effort in the short-term (decades—Milton 2003) but requires lower ongoing management into the future (Guimarães et al., 2013).

Lower in the hierarchy are *hybrid* and *novel* (both unplanned and planned) ecosystems (Figure 1). These ecosystems inherently imply greater uncertainty in terms of sustainability and resilience, and less similarity to Queensland's indigenous native ecosystems. All PMLUs in Queensland must be viable with regard to the surrounding land use and either consistent with the pre-mining land use, consistent with a development approval or planning instrument, or aimed at delivering a beneficial environmental outcome (Environmental Protection Regulation 2019, Schedule 8A, part 3). Since *hybrid* and *novel* ecosystems are not consistent with how the land was used prior to mining, it follows that there must be sufficient evidence that they deliver beneficial environmental outcomes for them to be considered PMLUs. Such beneficial environmental outcomes are in addition to the requirement for rehabilitated land to be safe, structurally stable, and not cause environmental harm.

Hybrid rehabilitated native ecosystems may be *native hybrid* or *exotic–native hybrid*; both are common in Australian mine rehabilitation. *Native hybrid* ecosystems are rehabilitated areas

dominated by native species but with a floristic composition that does not occur in nature (e.g., co-dominant native species that do not occur within the same RE). *Exotic–native hybrid* ecosystems are characterised by both exotic and native species and would not be considered native ecosystems if dominated by exotics. Targeting *hybrid* ecosystems as part of rehabilitation allows for flexibility in species selection while still retaining enough naturally occurring attributes that, if desired later, they can be manipulated to resemble a *natural* ecosystem. For example, species can be selected based on traits that best suit post-mining conditions (e.g., that tolerate low nutrients and high salinity in soils) in order to quickly establish vegetation cover and habitat features. Over time, these biophysical limitations can decrease (e.g., through accumulation of organic matter) and through manipulations (e.g., selective thinning) and the *hybrid* ecosystem can become a more *natural* ecosystem. Regular management of *hybrid* rehabilitation is necessary, to ensure that the ecosystem development trajectory does not track towards a *novel* state, risking an irreversible divergence from the desired objective. We do not recommend targeting *exotic–native hybrid* ecosystems, even if exotics are in low proportion, due to the risk of invasive species spread and increased maintenance burden required to manage these (see section 3.4.1.1).

At the bottom of the native ecosystem rehabilitation hierarchy are *novel* ecosystems. Unplanned novel ecosystems are self-assembling and stable, while planned *novel* (i.e. *designer*) ecosystems are intentionally assembled but not necessarily self-sustaining (Figure 1). Whether planned or unplanned, *novel* ecosystems could only be considered a native ecosystem PMLU if dominated by native species. The *novel* class of rehabilitation carries the greatest risk to native ecosystem rehabilitation outcomes, as it cannot be "upgraded" to one of the higher categories (due to the irreversibility that defines these systems). Furthermore, their novelty is associated with inherent uncertainty with regards to their sustainability, ecosystem services, and potential risk to adjoining land uses.

In order to overcome these risks, it has been suggested that *novel* ecosystems should have ecosystem services greater than or equal to the *historical* ecosystem, or have demonstratable intrinsic value as a PMLU (Humphries and Tibbett, 2015). That is, delivery of beneficial environmental outcomes over and above the fundamental requirement for rehabilitated land to be safe, structurally stable and not cause environmental harm (EP Act, s111A). For example, ecosystem services of *novel* ecosystems may include societal or monetary benefits like recreational use or timber production, and/or regulatory services like water purification and erosion regulation (Costanza et al., 2017; Rosa et al., 2020). *Designer* ecosystems may be designed to specific human-oriented uses, for example carbon sinks (Pietrzykowski and Daniels, 2014; Tripathi et al., 2016) or native seed orchards (Nichols et al., 1985; Gardner and Bell, 2007; Annandale et al., 2021). However, in these cases, the PMLU may be better described as the primary service being delivered (e.g., 'seed orchard') rather than 'native ecosystem'.

The decision tree to support rehabilitation planners evaluate the feasibility of native ecosystem rehabilitation outcomes contains two main branches, one for existing rehabilitation and one for planned rehabilitation (Figure 1). This system recognises that existing mines may contain "legacy" characteristics that were established when stakeholder expectations and/or mining and rehabilitation techniques were different. Where operations have (or elected) a domain for "legacy" rehabilitation, it should have a different set of completion criteria to the operations planned or contemporary native ecosystem rehabilitation. Projects targeting restoration of *natural* ecosystems often do not meet their objectives due to factors such as landscape constraints and legacies of past land use (Suding, 2011). It is therefore important to acknowledge these legacy situations in an effort to allow relinquishment of native ecosystem rehabilitation areas that cannot practically be modified to higher conditions in the hierarchy. However, the requirements for rehabilitated land to be safe, structurally stable and not causing environmental harm must still be met (EP Act, s111A). Furthermore, where the rehabilitated ecosystem is not consistent with the pre-mining use of the land, we recommend an evaluation of the beneficial environmental outcomes that the rehabilitation delivers (Environmental Protection Regulation 2019, Schedule 8A, part 3), over and above the fundamental requirements for land to be safe, structurally stable and not cause environmental harm.

Where rehabilitation is in the planning stage, it is expected that the *natural* ecosystems will be the rehabilitation target and biophysical limitations that support *natural* ecosystems will be addressed through best practice management (Table 2). Only where biophysical limitations are extreme or unable to be managed with currently available technologies, would the hybrid or designer ecosystems be considered as an option for rehabilitation that is yet to be completed. For extant rehabilitation, the flowchart incorporates an upward flowing arrow to symbolise the potential to "upgrade" rehabilitation to a higher status through manipulations. Whilst it is acknowledged that *novel* ecosystems by definition have crossed an irreversible threshold (Doley et al., 2012; Hobbs et al., 2013), thereby negating the possibility of manipulation into a more *natural* state, the "upgrade" arrow is nevertheless

included, so that users can assess whether the rehabilitation is actually *novel*, or indeed *hybrid* and thus could be managed to an improved environmental condition or state. Where no acceptable native ecosystem rehabilitation of any type has been established, the land manager can opt to “retreat” the rehabilitation. Retreating involves corrective actions, and/or management approaches that differ from the approaches attempted earlier, in order to achieve an improved native ecosystem outcome. For example, resetting of the rehabilitation community through combinations of clearing, spraying, burning, ripping, scalping, topsoil supplementation, ploughing, reseeding, and replanting.

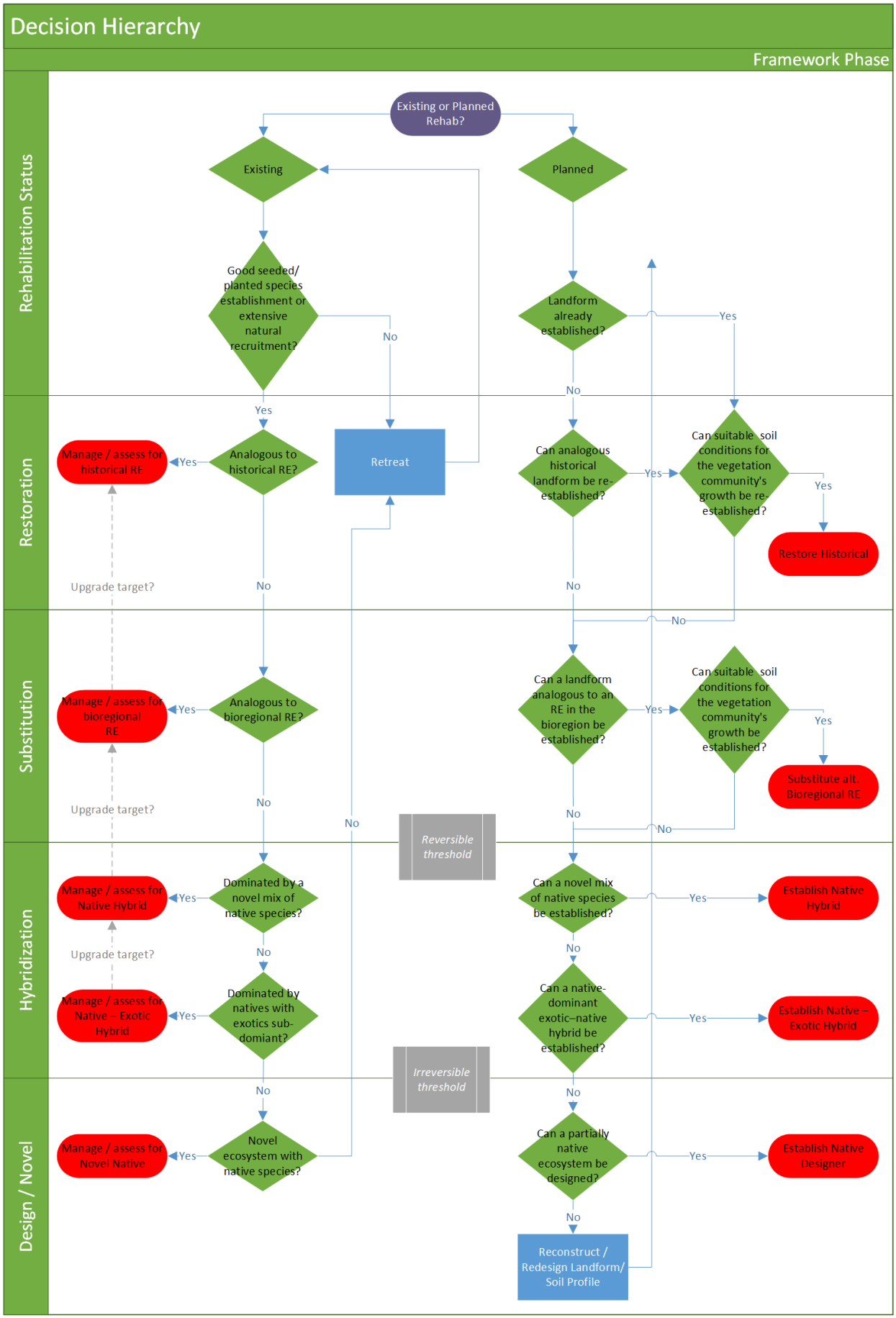

Figure 1. Native ecosystem rehabilitation options decision support flowchart

Guidance on flowchart terminology and use:

We highly recommend using empirical methods of determining target suitability at each step (i.e., determining “yes” or “no” at each green diamond). This is because post-mining conditions are highly site-specific. Example empirical methods for determining suitability include:

- modelling landform design based on measured soil properties, erosion modelling and climate conditions
- undertaking a full suite of chemical analyses to assess nutrient toxicities and limitations soil stockpiles and pre-strip soils and compare to analogous reference sites
- literature reviews (e.g., to help inform best practice soil treatment and amendments, plant species selection and propagation methods, etc.)
- field trials to understand the best combinations of soil treatments and amendments and species combinations
- existing rehabilitation references (nearby mines, or already established rehabilitation at the mine).

Specific examples of best practice rehabilitation methods are given in Table 2.

Good initial establishment—What qualifies as “good seeded/planted species establishment” will be site specific. The internal rehabilitation manual, or annual monitoring reports, used by mines should define the levels of native cover/density/diversity that are acceptable after a certain establishment period. Predefined trigger levels should be used to initiate retreatment or other actions if cover establishment is not proceeding as expected.

Analogousness—The term “analogous” in the context of landform, soil, and ecosystem references is commonly used in the mine rehabilitation (Hollingsworth et al., 2006; Doley and Audet, 2016; Louzeiro, 2019) and broader restoration ecology literature (Lundholm and Richardson, 2010). The term is not defined here as it would not be able to encompass all situations that relate to native ecosystem mine rehabilitation in Queensland. Rather, empirical methods (such as those described above) should be employed to determine if landform, soil, or ecosystem are sufficiently analogous to support the target ecosystem. It is acknowledged that the post-mining landform may not be similar to a natural landform (Hancock et al., 2020), however we suggest that it only needs to be analogous. Where the species composition of existing rehabilitation is analogous to an RE, it can be managed to become floristically similar to that target (e.g., through selective thinning, planting, or prescribed burning).

3. Biophysical limitations imposed on native ecosystem rehabilitation after mining

Every mine site presents unique hydrological, landform and geochemical conditions that influence the desirable native ecosystem that can be achieved from rehabilitation. One aim of this document is to assist practitioners to assess these biophysical limitations that may be present at their specific site, such that realistic and achievable ecosystem rehabilitation objectives can be set. The following review covers each of the biophysical limitations that might influence deciding whether rehabilitation targets of *natural, hybrid* or *planned novel* ecosystems are achievable. This information should be used to inform native ecosystem rehabilitation planning through Figure 1.

The severity of disturbance imposed by mining activities varies considerably among types and scales of operations (Gwenzi, 2021). As mining itself tends to disturb the entire ecosystem, biophysical limitations that may inhibit rehabilitation affect all aspects of abiotic and biotic components of ecosystems. The table below summarises these limitations by category (climate, landform, soil, vegetation, fauna and regional context). Mine wastes are considered separately because the acute toxicity and chemical characteristics of these materials impose unique inhibitions to ecosystem rehabilitation. Examples of best practice methods for native ecosystem rehabilitation methods are given in Table 2, however, this list is not exhaustive. We also welcome new and scientifically tested rehabilitation techniques and technologies.

Table 2. **Breakdown of biophysical limitations imposed by mining, the outcomes for native ecosystem rehabilitation and examples of best practice rehabilitation methods**

Biophysical limitation	Consequence	Best Practices for Rehabilitation
Challenging climatic conditions	Extreme temperatures and rainfall conditions limit seed germination and plant establishment or induce topsoil erosion.	Select planting at an optimal time of year, or during years that have optimal conditions for plant growth (Schwenke et al., 2000; Gillespie et al., 2015; Ngugi et al., 2015).
Landform characteristics (slope, stability)	Steep slopes of spoil heaps or deep voids alter hydrologic regimes, with an increased risk of void collapse, surface subsistence, or erosion gullies (Emmerton et al., 2018).	Utilisation of long-term erosion models, informed by on-ground data, will optimise geomorphic landform design so as to integrate with surrounding water catchment and incorporate natural variation (Ayres et al., 2006; Howard et al., 2011; Hancock et al., 2020)**.**
Physicochemical and biological properties of soil and mine wastes	Substrate may have unfavourable pH, be sodic, saline, nutrient-limited and/or have high levels of heavy metals or other toxic substances that limit plant growth (e.g. in mine tailings; Huang et al. 2012, Cross and Lambers 2017). Highly sodic material may be susceptible to erosion, or form hard crusts when dry, potentially limiting plant establishment (Carroll et al., 2004; Emmerton et al., 2018). Poor soil structure may lead to reduced water-holding capacity and water stress for establishing plants (Ngugi et al., 2015). Low microbial activity affects soil nutrient, carbon cycling and plant health (Li, You, et al., 2015; Kumaresan et al., 2017).	Undertake full suite of soil chemical analyses to assess nutrient toxicities and limitations. Amend growth medium with organic matter (e.g. topsoil, compost or woodchips), gypsum and/or any desirable trace metals as informed by soil chemistry data (Nichols, 2004; Huang et al., 2014; Nussbaumer et al., 2016; Spargo and Doley, 2016; Antonelli et al., 2018; Dale et al., 2018; Page-Dumroese et al., 2018; You et al., 2018; Fourrier et al., 2020; Office of the Queensland Mine Rehabilitation Commissioner, 2022a).
Paucity or unviability of native propagules and competition from exotics	Poor establishment of native species. Competition from exotic species limits native establishment and can alter the successional trajectory of desired native species (Cole et al., 2006).	Early effort to promote the successful establishment of species from the topsoil seedbank, broadcast seeds and/or tubestock planting. Use fresh topsoil, or store topsoil for < 6 months, to ensure viability of desired seeds and beneficial microbes. Store and spread topsoil and spoil separately to avoid dilution of topsoil (Rokich et al., 2000; Cole et al., 2006; Office of the Queensland Mine Rehabilitation Commissioner, 2022c). Limit use of fertilisers, as these may encourage weeds and exotics, and inhibit nutrient sensitive natives (Erskine and Fletcher, 2013; Daws et al., 2015). Control weeds by spraying or mowing prior to seed-drop (Cole et al. 2006). Silviculture techniques may be required to alter dominance ratios between grasses, shrubs and trees (Grant and Koch, 2007; Scoles-Sciulla and DeFalco, 2009)**.**
Lack of suitable fauna habitat	Delay in ecological functions and human-benefited services from fauna e.g. pollination, seed and spore dispersal, vegetation population	Plant appropriate feed plants for local fauna. Recreate structural complexity, for example, retain woody debris and hollow trees, and

Biophysical limitation	Consequence	Best Practices for Rehabilitation
	regulation from herbivory; (White et al., 2004; Maher et al., 2010; Costanza et al., 2017; Ismail et al., 2017; Kaiser-Bunbury et al., 2017; Nuske et al., 2017).	construct nest boxes (Nichols and Grant, 2007; Cristescu et al., 2012).
Isolation from surrounding native ecosystems	If connectivity to native ecosystems in surrounding areas is low, there will be limited dispersal and colonisation of species from surrounding region, thus compromising gene flow (Macqueen et al., 2008; Neaves et al., 2009). Long-term population sustainability and species diversity will be at risk (Kuussaari et al., 2009; Streatfeild, 2009). Connectivity may be constrained or dictated by the size and shape of the rehabilitation area (Alharbi and Petrovskii, 2016).	Establish and manage for high genetic and species diversity, especially if there is low connectivity with surrounding native ecosystems (Méndez et al., 2014; Frankham et al., 2017). Plan native ecosystem rehabilitation to adjoin existing native ecosystems to allow for habitat corridors (Cristescu et al., 2012; Macdonald et al., 2015).

3.1 Challenging climatic conditions

Challenging climatic conditions can limit seed germination or plant survival during rehabilitation. Native species have evolved strategies to cope with the naturally challenging climatic conditions in their bioregion. By utilising endemic species and accommodating for site-specific climatic conditions (e.g., seasonality, droughts, rainfall intensity), it is feasible to establish a *natural* ecosystem from rehabilitation. For example, in semi-arid sites of western Queensland, Gillespie et al. (2015) recommended to aim for planting after major rainfall events (> 100 mm), or using selective irrigation for successful plant establishment.

Incorporating long-term climatic patterns into rehabilitation planning will also reduce the risk of native ecosystem rehabilitation failure. For example, taking advantage of wet seasons or years to ensure plant establishment (Halwatura et al., 2015). Refraining from harvesting and stockpiling fresh topsoil when it is wet will prevent the loss of viable nutrients in the stored topsoil. Where species naturally occur across different climatic zones, collecting propagules from provenances with more extreme climatic conditions may promote ecosystem resilience (Prober et al., 2015).

3.2 Landform characteristics

3.2.1 Best practice landform design for native ecosystem rehabilitation

Careful design of post-mining landforms with consideration of rainfall, erosion potential, soil and waste material properties, and subsequent vegetation cover on a site-specific basis, will underpin rehabilitation (Howard et al., 2011). Geomorphic landform design principles and advanced erosion models such as GeoFluv (Bugosh, 2009) ensure that reshaped landforms include natural variation (Ayres et al., 2006), integrate into the surrounding water catchment and are stable over the long term (Hancock et al., 2020). Such landforms also visually integrate the rehabilitation into the surrounding landscape. Geomorphic landform design has been used in coal mines, minerals sand mines, metalliferous mines, bauxite mines and iron ore mines both in Australia and globally (Hancock and Willgoose, 2017). Where space is sufficient, it is feasible to target *historical* ecosystems, underpinned by a geomorphically designed landform. Mineral sands mining lends itself to this type of rehabilitation, since the material volume of ore removed is relatively small and the tailings can easily be reshaped (Bellairs and Davidson, 1999; Cooke and Johnson, 2002). It is also often an option for bauxite mining in cases where, despite the landscape profile being lowered by a few metres (Taylor et al., 2008), the hydrology remains similar to the pre-disturbance regime (Grant and Koch, 2007).

However, reshaping of landforms to strictly resemble the original landform rarely occurs. Often the reshaped landform differs from the pre-mining landscape due to changes in lithology and the volume of material available to extend or flatten slopes (Hancock et al., 2020). For example, where bauxite

mining removes material to the extent that the new landform is at or close to the level of the water table or subject to seasonal inundation. In such circumstances, *substitution* of a new RE, suitable to the altered hydrological regime, may be appropriate (Gould, 2012), e.g., establishing paperbark wetlands where previously eucalypt forest grew. In contrast, open cut coal and metal mining invariably create waste dumps of greater volume than the material that was *in situ*, due to breaking up of the formerly coherent geological structure (Emmerton et al., 2018; Hancock et al., 2020). If an undulating landscape exists within the bioregion, using geomorphic principles to recreate a similar topography from within the regional landscape will underpin establishment of a *substitute* native ecosystem in waste rock dump areas (Howard et al., 2011).

Where space does not allow for sufficient re-profiling of slopes, finding local analogue ecosystems that exist on landforms resembling post-mining landscapes may be challenging. In the economic mining areas of the Bowen Basin, for example, natural slopes are typically 3–5% in grade, compared to coal spoils, which vary from 5–35% (Emmerton et al., 2018). The search for habitat analogues is an important principle in efforts to encourage native biodiversity in anthropogenic landscapes (Lundholm and Richardson, 2010). However, analogues should be within a bioregion, or geographically close, consistent with the RE framework of Queensland (Neldner et al., 2022). Depending on the bioregion, climatic considerations (see section 3.1) may intersect; for example on steeper slopes in arid and semi-arid environments the reestablishment of vegetation alone may not suffice to control erosion (Gillespie et al., 2015; Emmerton et al., 2018). Management actions (such as reinforcing slopes with stable rock fragments) can reduce erosion, however the resulting landform may then be without natural analogue in the bioregion because reconstructed landforms using fractured rocks will have different hydrology to existing slopes (Howard et al., 2011). Therefore, it may be necessary to target a *novel* or *hybrid* ecosystem to manage erosion in some areas, whilst establishing the *substitute* ecosystem in areas less prone to erosion.

3.3 Physicochemical and biological properties of soils and mine wastes

3.3.1 Best practice rehabilitation approaches to overcome substrate limitations

Soil substrates used in mine rehabilitation often have poor physical structure and hydrological functioning, lack organic material and biologically available nutrients, and have unfavourable osmotic conditions and geochemistry. These physical and biological constraints limit establishment of vegetation, and therefore impede rehabilitation to a self-sustaining ecosystem (Cooke and Johnson, 2002), unless intervention is implemented (see Table 2).

3.3.1.1 Topsoil management

The quality and quantity of topsoil that is spread during rehabilitation, greatly influences rehabilitation success (Ward and Koch, 1996; Loch and Orange, 1997; Strohmayer, 1999; Cristescu et al., 2012; Lamb et al., 2015). Best practice is to segregate soil layers during stripping (i.e., no mixing of topsoil with underlying layers) and to reconstruct soil profile layers during rehabilitation. Direct placement of topsoil from one area of the mine onto a prepared rehabilitation area will maximise native ecosystem outcomes compared to using stockpiled soil (Rokich et al., 2000; Van Gorp and Erskine, 2011; Vickers et al., 2012; Golos and Dixon, 2014; Golos et al., 2016). Stockpiling topsoil results in a lack of viable seeds (Bellairs and Bell, 1993), because of early seed germination and mortality (Rokich et al., 2000; Rivera et al., 2012). Seeds or other propagules that do survive the stockpiling period, as well as broadcast seed, then face potentially adverse soil conditions, due to chemical, biological and structural changes that have taken place in the soil prior to respreading (Paterson et al., 2019). Topsoil stockpiling may also increase the incidence of weedy species invasion, as weed seed banks tend to build up in the surface layer of the stockpiles themselves (Rokich et al., 2000). Where direct placement of stripped topsoil is not an option, rehabilitation as soon as possible after topsoil is harvested is best practice, to leverage the natural seedbank and available nutrients (Rokich et al., 2000; Ngugi et al., 2018; Paterson et al., 2019). Mining operations that allow for progressive rehabilitation (e.g., shallow strip mining, coal mining) are well suited for this effective use of topsoil. It is critical that rehabilitation activities are integrated into life-of-mine planning at these operations, to ensure that the opportunity to use topsoil resources as soon as practicable after harvest, is not missed.

Longer topsoil storage times (e.g., at open cut metal mines) or limited availability of any topsoil, makes vegetation re-establishment more difficult (for more information regarding best practice

management methods on handling topsoil deficiency and improving soil physiochemical properties see Machado et al. 2013, Dale et al. 2018).

3.3.1.2 Soil quality assessment and monitoring

If best practice methods have been implemented during initial rehabilitation (see Table 2), some, if not most, aspects of rehabilitated soil can return to values similar to reference sites over time. For example, soil nitrogen and carbon concentrations increase with rehabilitation age and vegetation coverage (Ahirwal et al., 2017) and can reach levels similar to reference sites within at least four years (Muñoz-Rojas et al., 2016), or up to several decades (Grant et al., 2007; Banning et al., 2008; Frouz et al., 2008; van Soest et al., 2011; Kumar et al., 2015; Yuan et al., 2017; Ngugi et al., 2018). Other adverse soil conditions, such as high salt concentrations, can also decrease and reach reference site levels over time (Ngugi et al., 2018). Soil microbial richness, abundance and activity has also shown to recover with rehabilitation age (Banning et al., 2008; Glen et al., 2008; Kumar et al., 2015). Even though microbial community assembly may be dissimilar to reference sites (Ngugi et al., 2018), high functional redundancy in many microbial communities means that microbial functional recovery is possible (Kumaresan et al., 2017). The abundance of soil fauna, and the similarity of their communities to reference sites, has also been shown to increase as rehabilitation ages (Frouz et al., 2008; Menta et al., 2014).

It is recommended that rehabilitators undertake a full suite of soil chemical and microbial analyses to assess nutrient and biotic limitations prior to commencement of rehabilitation. Pedogenesis (soil formation) can be accelerated by establishing pioneer microbial and vegetation communities (Cross et al., 2017), and nutrient limitations can be overcome by amending mine waste with topsoil or by adding fertiliser (Mulligan et al., 2006; Cross, Ivanov, et al., 2021; Cross, Stevens, et al., 2021) or treatments such as lime and gypsum (Fourrier et al., 2020) at the start of the rehabilitation process (Cross and Lambers, 2017). However, soil amendments must be applied with care, given that many native species are adapted to low nutrient levels, and the addition of fertilisers can promote the growth of non-natives (Cole et al., 2006; Erskine and Fletcher, 2013). Where substrate nutritional levels can be amended to be similar to the pre-mining RE or are inherently similar to those in an RE within the bioregion, *natural* ecosystems are a feasible rehabilitation target.

3.3.1.3 Soil amendments and native ecosystem rehabilitation outcomes

The inherent properties of some soils and subsoils, along with disturbances from mining, can add challenges to achieving *natural* ecosystem rehabilitation. For example, the saline and sodic nature of many mine spoils in Queensland makes them dispersive and prone to erosion. In other cases, high alkalinity or acidity inhibits microbial activity (Fourrier et al., 2020) and provides challenging osmotic conditions for plant establishment (Bell, 2001; Erskine and Fletcher, 2013; Gillespie et al., 2015; Lamb et al., 2015; Di Carlo et al., 2020). Sodicity of soil can decrease with time (Wehr et al., 2006), especially when amended with gypsum, and addition of organic matter can help reduce rising salts, enhance soil-water-microbe interactions and improve plant establishment (Grigg et al., 2006; Courtney et al., 2014; Di Carlo et al., 2020). Further research is needed to understand whether such initial amendments to sodic and saline soils enable long-term self-sustainability of native ecosystems without further inputs, or whether salt-tolerant *substitute* ecosystems are more feasible targets.

In certain circumstances it will be appropriate to plan for *hybrid* native ecosystem rehabilitation to overcome potential hydrological limitations on the soil. If the hydrology of the recipient site is marginal or uncertain, this may warrant the selection of native species from a mix of different habitats and landscape positions, including the *historical* RE. Such selection can result in a *hybrid* ecosystem that is a mix of floristic elements from both the pre-mining RE and other REs within the region. This approach may be particularly useful for recalcitrant sites, where rehabilitation needs to be retreated due to poor initial establishment, and there is uncertainty regarding which species will establish in the harsh conditions. For example, a novel mix of *Melaleuca* and *Eucalyptus* species may increase the chances of establishing vegetation that can tolerate the prevalent soil moisture conditions (Setyawan et al., 2002). That said, it is not recommended that hybrid combinations of native species be used for rehabilitation where the soil and hydrology is likely to support *natural* ecosystem rehabilitation. Generic or broad vegetation group seed mixes that are likely to result in *hybrid* native ecosystems, should only be used in limited, pressured circumstances. It is recommended that rehabilitators establish field trials of different seed mixes to assess whether a *hybrid* ecosystem target is warranted and more viable than a *natural* ecosystem target.

There are many examples where existing established native ecosystem rehabilitation sites have not reached their *natural* ecosystem target, and a likely cause or contributor to this failure is limitations imposed by soil properties (e.g., Audet et al. 2013, Erskine and Fletcher 2013, Nussbaumer et al.

2016). It remains unclear whether these ecosystems are potentially *hybrid,* and the soil limitations can be overcome with time. In this case also it is recommended that rehabilitators conduct field trials, using various mixes of soil amendments, to test the viability of the *hybrid* status, or to confirm whether the ecosystems are stable *novel* forms.

3.3.1.4 Unique considerations where the substrate is mine waste

Mine waste and tailings from processing have unique hydrogeochemical properties not found in natural soil and they present challenges to biota (Cross et al., 2017). Mine waste may not be able to be rehabilitated by natural successional processes, for example where high concentrations of heavy metals or radiation are present (Huang et al., 2014; Cross, Ivanov, et al., 2021; Cross, Stevens, et al., 2021). Another example, in sulphide-hosted metal deposits, is the generation of acid and associated release of metals from sulphide-rich mine waste. Pyrite weathering and acid generation can continue for hundreds of years, rendering phytostabilisation (use of vegetation for stabilisation) of tailings impractical (Li, Bond, et al., 2015). Therefore, the removal, containment or remediation of toxic substances and stabilisation of unfavourable physio-chemical conditions of tailings is needed before the rehabilitation stage (Huang et al., 2012). For more information on the treatment of mine waste see QMRC research on 'Mine Waste Cover Systems' (Office of the Queensland Mine Rehabilitation Commissioner, 2022b)**.**

Native ecosystem rehabilitation, upon mine waste or tailings, will depend on the effectiveness of the removal, remediation or containment of any toxic substances and the stabilisation of other unfavourable physical and hydrological aspects of the waste. Theoretically, in some circumstances the hydrogeochemistry of tailings can be stabilised by phytoremediation; through formation of soil that is biogeochemically functional, under a correct treatment by specialist flora and fauna that, in themselves, constitute a *natural* ecosystem albeit (at least temporarily) one different to the *historic* ecosystems in the locality (Huang et al., 2014; Cross and Lambers, 2017). However, successful reestablishment of *natural* ecosystems on mine tailings has not yet been achieved anywhere in the world (Cross and Lambers, 2017).

It is recommended that rehabilitators undertake a full suite of soil chemical analyses to assess toxicities and limitations of mine waste before considering tailings or mine waste as a substrate for ecosystem establishment. Tailings can be treated by (for instance) altering pH to suit target vegetation (for example, by using acidifying fertiliser or flushing with seawater), inoculating with appropriate microbial communities, and encouraging early establishment of pioneer plants (Wehr et al., 2006; Cross and Lambers, 2017). Adding topsoil, organic matter or fertiliser can alleviate some of the nutritional constraints imposed on mine wastes as a growth medium (Mulligan et al., 2006; Asensio et al., 2013; Li, Bond, et al., 2015; Li, You, et al., 2015; Robson et al., 2018; You et al., 2018; Cross, Ivanov, et al., 2021; Cross, Stevens, et al., 2021), provided no toxic material or other physical and hydrological constraint remains (Cross and Lambers, 2017).

Cover systems over mine waste structures are an important part of rehabilitation (Office of the Queensland Mine Rehabilitation Commissioner, 2022b). Vegetation on the cover system promotes evapotranspiration and reduces the amount of water that reaches reactive underlying waste. However, whether native ecosystems are suitable PMLUs on mine waste structures requires detailed evaluation, as tree roots can penetrate cover system layers. Where native ecosystem is the desired PMLU for a mine waste structure with reactive waste, the species mix chosen as part of cover system design needs to promote evapotranspiration without undermining the integrity of low permeability components of the cover (Mulligan et al., 2008). Such a species mix may not always reflect composition of a *natural* ecosystem.

If there is insufficient inert material to form a long-term stable cover system over reactive mine waste, and/or removal or remediation of toxic material is unrealistic, then the establishment of a *natural* ecosystem will be limited by nutritional toxicity of the bioavailable elements in the mine waste. Potentially, natural ecosystems thriving in soils with elevated levels of metals (Batty and Hallberg, 2010; Tang et al., 2021), may exist within the bioregion and could be utilised as valuable *substitute* targets (Cooke and Johnson, 2002). Alternatively, *designer* systems, with tolerant native and exotic species, may be appropriate targets (Whiting et al., 2004; Wehr et al., 2006; Batty and Hallberg, 2010; Corzo Remigio et al., 2020). This was the philosophy of the mine at Gove, NT for the (difficult to revegetate) bauxite refinery residue. A mix of exotic and native salt- and alkali-tolerant plant species were used to establish a vegetation community that approaches a native woodland (Wehr et al., 2006). This is an example of a functioning *planned novel* ecosystem that prioritises the use of suitable local native species whilst acknowledging the inherent limitations of the post-mining growth media. Further research is needed on the requirements for rehabilitation, and mechanisms for ameliorating ecological hostility of toxic mine tailings, in order to successfully create sustainable *planned novel*

ecosystems in other settings (Cross et al., 2017). There are other cases of established ecosystem rehabilitation where some Australian native flora have grown in unamended tailings and the resultant ecosystem may be considered *unplanned novel* (Mulligan et al., 2006; Cross, Ivanov, et al., 2021). However, it remains unclear whether *unplanned novel* communities on tailings possess longevity, resilience and the functional capacity to support biodiverse and self-sustaining ecosystems (Cross et al., 2017) that deliver beneficial environmental outcomes above and beyond the fundamental requirement for rehabilitated land to be safe, structurally stable and not cause environmental harm.

3.4 Paucity or unviability of native propagules and competition from exotics

3.4.1 Planning native ecosystem rehabilitation and overcoming vegetation establishment limitations

Through best practice methods of species selection, soil handling, germination and planting methods, the biophysical limitations that pertain to establishing native vegetation can be overcome. Therefore, where landform and soil conditions are suitable, targeting community assemblages that represent *natural* ecosystems is feasible. There are many advanced techniques that facilitate germination and growth of native vegetation (Bell, 2001; Khurana and Singh, 2002; Prober et al., 2005; Probert et al., 2007; Fowler et al., 2015; Nativel et al., 2015; Dobrowolski, 2019; Erickson et al., 2019; Golos et al., 2019). Therefore, there is no need to target *substitute* or *planned novel* ecosystems solely because of limitations that pertain to seed preservation, sourcing and germination of native vegetation. Initial community assemblages used in rehabilitation are a primary determinant of later rehabilitation success (Jefferson, 2004; Grant, 2006). Therefore, best practice rehabilitation planning includes early effort to ensure there is adequate quantity and viability of native seeds and seedlings necessary for the target *natural* ecosystem.

Best practice includes direct transfer of topsoil from one area of the mine footprint to a rehabilitation area, with limited mixing of layers (Ward and Koch, 1996; Loch and Orange, 1997; Strohmayer, 1999). This ensures that soils maintain a healthy, viable native species seedbank (Rokich et al., 2000; Vickers et al., 2012; Golos et al., 2016). Fresh topsoil also has the advantage of biologically active soil fauna and microbes (e.g. mycorrhizal fungi or nitrogen-fixing bacteria) critical for the establishment and growth of some plant species (Gardner and Malajczuk, 1988; Reddell and Milnes, 1992; Bowman and Panton, 1993; Reddell et al., 1999; Jasper, 2007). In addition to seeds, resprouting from root fragments within topsoil has been shown to be of primary importance when re-establishing woody vegetation in respread soil, especially for seasonal tropical dry forest–woodland environments (Ferreira and Vieira, 2017). This “bud bank” is conceptually similar to the familiar seedbank, but depends upon buds which reproduce vegetatively from lignotubers, rhizomes, corms, bulbs, roots, and fragments (both aboveground and belowground) that have the potential to resprout (Klimešová and Klimeš, 2007). Topsoil transfer trials in a dry tropical environment in Brazil resulted in dense native ecosystem regeneration using just the bud bank and inherent seedbank, with no artificially broadcast seed at all (Ferreira et al., 2015; Ferreira and Vieira, 2017). For some species, plant fragments can also be specifically sourced and used in mine rehabilitation, instead of seed. In Central Queensland, Brigalow (*Acacia harpophylla*) has been successfully transplanted using whole soil-root compartments (Arnold et al., 2014).

Best practice also includes accounting for differences in germination or colonisation requirements for each species. For example, knowing which species are not well represented in seed banks of topsoil, or are not likely to recolonise the site spontaneously from the surrounding landscape, will inform which species need to be sourced from nurseries or field collected (Grant and Koch, 2007; Meers et al., 2012; Van Etten et al., 2014; Erickson et al., 2017, 2019). When fresh topsoil is not available, or is in limited supply, more emphasis on sourcing of seeds and direct planting (as well as soil amendments, see above) will be needed to compensate for less establishment from native bud banks and seedbanks (Rokich et al., 2000; Van Gorp and Erskine, 2011; Rivera et al., 2012; Golos and Dixon, 2014). A useful strategy to allow for the best possible seedbank mix is clearing overstory vegetation soon after most trees have developed ripe seed and to strip the soil without delay in the following season, thereby also enabling understory species to set seed under heightened light conditions (Spain et al., 2006).

To improve similarity to specific REs (and even landscape positions therein) and to avoid inefficiencies in cost and effort, it is recommended that rehabilitators formulate specific seed mixes, which can then be applied to the appropriate recipient site within the rehabilitation (Ngugi et al., 2015; Miller et al., 2016; Erickson et al., 2017). For example, to improve similarity to target shrublands at

Eneabba mineral sands operations, WA, Iluka Resources Ltd formulated individual swale, dune, laterite, and wetland seed species mixes that had previously been combined. This attention to detail produced species combinations closer to target *natural* community types in comparison with using generic seed mixes (Herath et al., 2009). Best practice methods also include taking into account any treatments needed to break dormancy of seeds (Baskin and Baskin, 2004). For example many *Acacia* species require acid or mechanical scarification, or hot water treatments, to break dormancy and germinate (Khurana and Singh, 2002). To establish some native species, adoption of more direct propagation protocols may be required, such as tissue culture or cuttings. For example, in WA Alcoa established a tissue culture laboratory for the purposes of targeting 'recalcitrant' species that could not be established from seedbank, planting or natural dispersal (Grant and Koch, 2007).

In some cases, achieving native species mixes representative of a specific RE can be challenging, especially in diverse or little studied settings. For example, the Ranger mine lease currently lists more than 500 species, and it is likely that for the majority of these there is no information on propagation requirements or on their ability to establish and persist in a waste rock substrate with limited soil development (Erskine et al., 2019). Targeted research on the autecology of priority species is therefore critical (Emilia, 2015; Navarro-Cano et al., 2019), and should be supported within the native seed and restoration sectors (Hancock et al., 2020). Effective collection of seed for rehabilitation purposes poses a number of challenges in Australia (Broadhurst et al., 2015). For example, there are often practical issues for seed collection due to spatial and temporal variability in seeding of target species (Vickers et al., 2012; Broadhurst et al., 2015). Rehabilitation practitioners can partially overcome this issue through forward planning and obtaining seed of key species when it is available (Norman et al., 2007; Probert et al., 2007; Nguyen et al., 2015).

Careful selection of species and seeding rate within the seed mix will also avoid inefficiencies in cost and effort as different native species will have varying germination and establishment rates. For example, in the sub-humid Brigalow Belt region of south-eastern Queensland, the dominant tree species that established on rehabilitation plots were *Corymbia citriodora* and some *Acacia* species (especially *Acacia leiocalyx*), despite *Eucalyptus crebra* and *E. siderophloia* comprising >60% of the seed mix (Ngugi et al., 2015). Examination of species traits, such as seed size, can help in selecting species for qualities such as drought tolerance. Species with small seeds have little of their own resources and need more prolonged rainfall events for successful establishment. Large seeds, on the other hand, can establish in more water and nutrient limiting environments. For example, the relatively large seeds of *C. citriodora* (compared with e.g. *Eucalyptus crebra* and *E. siderophloia*) means this species has more resources to establish in more water-limited areas (Ngugi et al., 2015).

3.4.1.1 Exotic and invasive species control

Exotic and invasive weedy species can threaten rehabilitation targets by preventing the establishment of native species through competition (Parrotta et al., 1997; Tucker and Murphy, 1997; Umwelt (Australia) Pty Limited, 2017; Antonelli, 2018; Peake et al., 2021). For example, buffel grass (*Cenchrus ciliaris*) is a common invasive pasture species in the many areas of Queensland and readily grows on mine rehabilitation sites (e.g., Morrison et al. 2005, Erskine and Fletcher 2013). Its fast growing properties can minimise initial erosion, but can also compete with the growth of tree species (Spargo and Doley, 2016; Emmerton et al., 2018), resulting in significant declines in native ground flora diversity (Jackson, 2005), constraining the feeding and habitat prospects for local birds (Wright et al., 2021), and elevating the risk of high intensity fires (McKenna et al., 2017).

All rehabilitation must achieve land in a stable condition, which includes the requirement that nothing in or on the land cause environmental harm (EP Act, s111A), for example as a source of weeds to the surrounding area. Initial methods to control exotic species in rehabilitation include avoiding the use of exotic species in seed mixes, minimising the use of topsoil if it contains a substantial exotic seedbank (weedy stockpiles may need to be repeat sprayed, or scalped before being applied), and promoting the fast establishment of native species to outcompete exotic species that might colonise from surrounding areas (Bayliss et al., 2006; Ewel et al., 2013). Continual monitoring of vegetation communities and incorporating adaptive practices that allow for control measures when exotic species are found in rehabilitation is also important.

Native groundcover species can be used instead of exotic invasive species (e.g., buffel grass) to control initial erosion (Rocha-Nicoleite et al., 2017). For example, members of the Chenopodiaceae have been observed to establish as pioneer species on heterogenous mine tailings (Cross and Lambers, 2017) and saltbushes (*Einadia* species) have potential to establish rapidly with high levels of cover in native ecosystem rehabilitation. The presence of exotic species in the seed bank of topsoil and/or surrounding ecosystems (see section 3.6) may make control strategies hard to implement (Gillespie et al., 2015). Indeed, even when the appropriate "best on offer" site selection protocols are

followed (Eyre et al., 2017), given that few REs are totally free of anthropogenic impacts (Landsberg and Crowley, 2004; Thackway and Freudenberger, 2016) exotic species may be present at reference sites. In cases of *hybrid* and *planned novel* ecosystems, the possible intentional inclusion of some exotic species to achieve a certain ecosystem service does not remove the rehabilitation obligation to achieve land in a stable condition, including that the land does not cause environmental harm as a source of weeds.

3.4.2 Considerations at already established native ecosystem rehabilitation sites

There are many examples of existing established mine rehabilitation where the vegetation community does not resemble *natural* ecosystems. The resulting communities may be *hybrid* ecosystems due to initial species selection. Hybrid ecosystems may comprise native species from a homogenised mix of different (and perhaps not even local or regional) habitats and landscape positions. For example, in Weipa, North Queensland, species not native to the local RE were included in the original seeding and later contributed to the difference between rehabilitation and reference RE sites (Gould, 2012). Erksine and Fletcher (2013) reported that vegetation communities from five coal mine rehabilitation sites in the Bowen Basin were significantly different to reference communities. This was due, in part, to the presence of species native to the broader region but not at the selected reference sites (e.g., novel mixtures/assemblages of species such as *Acacia bancroftiorum, A. victoriae, Corymbia citriodora,* and *Eucalyptus camaldulensis*), and common species (e.g., *Eremophila mitchellii*) present at reference sites that were absent from the rehabilitation sites. Other examples include mines in the Hunter Valley, NSW, where species that are not native to the target ecosystems (including WA endemic species) were present in rehabilitation sites (for example, *Corymbia maculata*, *Acacia saligna, Eucalyptus cladocalyx*) (Peake et al., 2021).

Alternatively, existing rehabilitation areas may be *hybrid* ecosystems dominated by exotics, or *unplanned novel* ecosystems due to the choice of species used during initial establishment. The use of exotics in the early stages of mine rehabilitation may result in persistent unproductive landscapes of legacy rehabilitation with arrested succession (Bauman et al., 2015). Invasive exotic plants can prevent the establishment of newly dispersed native species (Yurkonis et al., 2005), for example, by occupying germination niches (Davis et al., 2000). Weeds can have synergistic effects that promote the growth of other weeds by transforming ecosystem processes, such as nutrient cycles, or by altering fire regimes and promoting other species with similar fire and nutrient tolerances (Prober et al., 2005). This can create a legacy effect that persists long after eradication of the actual original exotics, for example, as a result of altered soil nitrogen pools (Corbin and D'Antonio, 2004; Dickie et al., 2014).

Where ecosystem trajectories have deviated from the target trajectory, substantial management intervention may be needed to re-direct the rehabilitation process. For example, supplementary seeding and planting, weed control, prescribed burning, or thinning, may be required (Grant, 2006; Grigg and Grant, 2009; Gould, 2012; Humphries, 2016). Often multiple management techniques will need to be executed in combination, for example, in fire-prone and fire-adapted landscapes it is preferable that thinning and burning are undertaken together; thinning alone is an undesirable management option due to elevated fire risk caused by the increased ground fuel (Craig et al., 2010). In other circumstances, mulching of thinned trees could be advantageous as this increases fuel load only temporarily whilst aiding future controlled burns to further reduce excessively dense species.

As a tool of ecological restoration, prescribed burning can be utilised to recreate reference conditions of surrounding vegetation (Freeman et al., 2017). However, factors such as the rehabilitation age and resilience of the rehabilitation must be taken into account. Fire may be used to re-direct *hybrid* ecosystem rehabilitation towards a desired *natural* ecosystem, if it is not currently meeting specified targets. Or in *unplanned novel* ecosystem situations, fire may be used to promote local flora and fauna colonisation while not targeting a particular RE outcome. In other cases, fire may be excluded intentionally, so as to facilitate a mesic shift toward a fire sensitive *natural* ecosystem community, such as a vine thicket or monsoon forest.

3.5 Lack of suitable fauna habitat

3.5.1 Overcoming limitations in promoting fauna recolonisation

Fauna perform important ecosystem functions (seed dispersal, pollination, etc.) that are essential to achieving a self-sustaining ecosystem post-mining (Cross et al., 2020). Therefore, it is recommended

that rehabilitators consider fauna during the assessment of rehabilitation success. Best practice techniques for promoting rapid recolonisation by fauna from surrounding ecosystems depend upon ensuring that key habitat features return. This can happen more quickly if fresh topsoil, with intact bud- and seedbanks is used, with additional planting of native seeds and seedlings, paying special attention to food plants and promoting compositional and structural diversity (Tucker and Murphy, 1997; Holland and Bennett, 2007; Gould, 2011; Cristescu et al., 2012; Nielsen and Kelly, 2016). Some requirements for fauna habitat in rehabilitation sites will not naturally eventuate for decades; these include development of tree hollows and large fallen logs. Therefore, best practice techniques also include respreading of coarse woody debris, and the placement of nest boxes and salvaged logs and stag trees (Nichols and Grant, 2007; Craig et al., 2012).

Exotic fauna, such as foxes and cats, can threaten the populations of native fauna in rehabilitation sites, so integrating control programs, like baiting or trapping these predators, is recommended, to promote small-to-medium sized mammal numbers (Nichols and Nichols, 2003; Grant and Koch, 2007; Nichols and Grant, 2007), especially if these measures are already being implemented in the surrounding landscape. Theoretically, with time and the implementation of best practice rehabilitation techniques, it is possible to achieve fauna communities that resemble those in *natural* ecosystems, especially if the immediate surrounding landscape is also the target RE ecosystem and habitat corridors are created (Macdonald et al., 2015).

As fauna recolonise spontaneously from the surrounding landscape, the regional context of the rehabilitation site is an important consideration when assessing the risk threatening rehabilitation. If the surrounding landscape is not the same target RE, or does not contain the same species and does not create similar habitat requirements as the target ecosystem, then rehabilitation sites may be isolated from source populations (Cox et al., 2004). This is especially challenging for terrestrial species that have limited dispersal ability and/or cannot utilise or transport through intervening matrix areas (Bennett, 1990; Eycott et al., 2012). Rehabilitators should therefore take the landscape context into account when assessing factors potentially impacting success of fauna colonisation across rehabilitation sites.

3.5.2 Fauna considerations at existing ecosystem rehabilitation sites

Fauna community assemblages on mined sites often do not resemble target REs or unmined reference sites (Andersen et al., 2003; Nichols and Nichols, 2003; Brady, 2005; Craig et al., 2012), even for sites that have a long history (> 2 decades) of rehabilitation (Majer et al., 2007; Nichols and Grant, 2007; Gould, 2011). However, for some groups of fauna (birds, mammals), abundance and community composition can be comparable to unmined sites (Craig et al., 2012; Cristescu et al., 2012), while others are less than unmined sites (reptiles, amphibians, some groups of arthropods; Cristescu et al. 2012). Differences in fauna populations between rehabilitation and reference sites may contribute to the development of *hybrid* or *unplanned novel* ecosystems (Doley and Audet, 2013). For example, the presence of some species may promote positive feedbacks (Wood et al., 2015), including those related to exotic fauna, which can be at a higher density and richness at rehabilitation sites compared to unmined sites (Cristescu et al., 2012). Management of fauna in established *unplanned novel* or *hybrid* ecosystems to ensure that the ecosystem does not cause environmental harm (i.e., become a source population for invasive exotic fauna) is an area in need of more research. In some circumstances it has been found that no-analogue ecosystems provide habitat for threatened fauna species. For example, the high density of *Allocasuarina* species in rehabilitation sites on Minjerribah (North Stradbroke Island) provides additional feeding habitat for the threatened Glossy Black Cockatoos (Clout and Clout, 1989; Crowley and Garnett, 2001), despite the target ecosystem being mixed *Eucalyptus* (Audet et al., 2013).

3.6 Isolation from the surrounding native ecosystems

3.6.1 Connectivity for maximised ecosystem rehabilitation outcomes

An important, and understudied, aspect of native ecosystem rehabilitation is its role in the local and regional landscape context. A landscape approach provides a new foundation and integrated perspective for mine rehabilitation ecology (Lei et al., 2016). *Natural*, *hybrid* or *novel* ecosystem rehabilitation will have different influences on their surroundings, dependent on their characteristics and that of the landscape matrix. Simultaneously, unmined native ecosystems adjacent or proximate to rehabilitation will also exert an influence on rehabilitated ecosystem development. The influences in both directions may be positive or negative.

Where rehabilitation areas are surrounded by and connected with remnant native ecosystems, the colonisation of local native species into the rehabilitation promotes genetic flow that is vital for resilient, self-sustaining *natural* ecosystems (Aavik and Helm, 2018). Ecosystem connection can also promote turnover in taxonomic composition that may shift no-analogue ecosystems closer to a *historical* state (Gould, 2012; Humphries and Tibbett, 2015). Some animal species can disperse across hostile landscape matrices and facilitate native plant and fungal species establishment in rehabilitation (Brady et al., 2009). For example, seed dispersal by frugivorous birds and bats can occur (Keenan et al., 1997; White et al., 2004), including those not planted in the rehabilitation or present in its bud and seedbank. Distance to remnant woodland will influence dispersal, with higher and more diverse seed dispersal likely to occur if remnant woodland occurs nearby (Tucker, 2000; White et al., 2004).

However, colonisation of native species into rehabilitation areas is not always guaranteed in cases of connected *hybrid* or *unplanned novel* ecosystems. For example, exotic plants in rehabilitated ecosystems may prevent the establishment of newly-dispersed native species (Yurkonis et al., 2005), by occupying germination niches (Davis et al., 2000). In *hybrid* rehabilitation dominated by native (but not local) *Acacia* or *Allocasuarina* species (Gould, 2012; Audet et al., 2013; Annandale et al., 2021), which alter nutrient cycles via nitrogen fixation, local low-nutrient-adapted native plants can be excluded (Specht and Specht, 1989). Furthermore, colonisation rates for groundcover and mid-canopy vegetation may be inhibited by the dense foliage cover of these species and their alterations in soil properties (Gould, 2012).

The benefits of connecting rehabilitated ecosystems with *natural* ecosystems can also be understood by considering the challenges presented in disconnected ecosystems. Natural dispersion processes will be slower in rehabilitated ecosystems that have no boundary with remnant vegetation (Cooke and Johnson, 2002; Gould, 2012). Species would need to rely on long distance dispersal mechanisms and species immigration will be mostly driven by chance (Tischew et al., 2014). Over the long-term, gene flow is compromised in isolated populations, which also creates a risk of extinction, lowering species diversity (Kuussaari et al., 2009; Méndez et al., 2014; Frankham et al., 2017). This emphasises the need to create appropriate linkages to surrounding native ecosystems and to maintain the habitat quality across the mining lease wherever possible.

The colonisation of less mobile fauna species into rehabilitated areas relies upon the presence of these fauna in undisturbed surrounding areas (Cristescu et al., 2012) and suitable linking habitat for them to disperse through (i.e., habitat corridors). Rehabilitation that has strong connections to other habitat patches is more likely to be become populated by fauna than poorly connected rehabilitation, even when the ecosystem type is quite different to the adjacent area (Gilby et al., 2018). Therefore, regardless of whether the rehabilitation is *natural*, *hybrid* or *novel*, local fauna will utilise it if they can disperse there and suitable habitat characteristics are available.

3.6.2 Regional scale benefits and risks from connectivity of rehabilitation with native ecosystems

Native ecosystem rehabilitation also provides an opportunity for landscape-level connectivity and regional benefits. The most positive impacts on the broader landscape from native ecosystem rehabilitation establishment may be in biogeographic regions dominated by modified ecosystems (i.e., very little indigenous ecosystems remain). An example is the Brigalow Belt, where native ecosystems were cleared for agriculture prior to mining. In these fragmented landscapes, connectivity of historical areas through corridors ultimately ensure landscape functionality (Hobbs et al., 2014).

The role of developing (i.e., young) rehabilitation, or that which is structurally distinct from other vegetation in the landscape, offers a unique regional benefit, important for supporting a diversity of fauna. For example, Tudor (2021) found that, by acting as a thermal refuge, early successional rehabilitation provides valuable habitat for the insect pollinator community, compared to the suboptimal conditions of the closed-canopy forest. By extension, it may be that physiognomic diversity of *hybrid* and *novel* ecosystems can offer similar benefits. At the least, rehabilitation adjacent to remnant areas, like unmined regrowth REs, contributes to protecting forest and woodland dependent species in small mature remnant fragments, by providing a buffer from edge effects and enhancing faunal dispersal (Bowen et al., 2009).

However, not all connectivity is positive: there is a risk that connecting *natural* or *hybrid* ecosystems (either rehabilitated or unmined) with degraded ecosystems could result in shifting towards an irreversibly *novel* state (Hobbs et al., 2014). Positive feedback loops are frequently associated with *unplanned novel* ecosystems, and these can sometimes facilitate the spread of the system into surrounding areas (Hobbs et al., 2006). For example, increasing populations of exotic weeds within

unplanned novel ecosystems may result in their spread to adjoining areas (Lloyd et al., 2002). Consequently, broader biodiversity management efforts may be undermined near national parks or conservation zones (Hallett et al., 2013) or in REs generally, within unfragmented bioregions (e.g., Cape York and Northwest Highlands) potentially being most vulnerable. Planned and unplanned *novel* ecosystems in these bioregions will have a higher ongoing maintenance burden and require more rehabilitation effort to attain a stable condition (i.e., not causing environmental harm to the surrounding ecosystems). From a natural resource management perspective, *novel* ecosystems are not recommended if the surrounding area has a high conservation value (Backstrom et al., 2018). In contrast, rehabilitating to *natural* ecosystems when surrounding areas have high conservation value will benefit from connection with the local populations and dispersion processes as outlined above (see section 3.6.1). Therefore, it is recommended that landscape factors that influence the net value of rehabilitated ecosystems (Hallett et al., 2013) be taken into account in planning and management of native ecosystem rehabilitation.

4. Limitations and opportunities with the *natural–hybrid–novel* ecosystem concept

4.1 Lack of clear objectives and risks involved with *novel* ecosystems

A particularly important consideration, where the proposed PMLU is a native ecosystem, is that not all *novel* or *hybrid* ecosystems can be considered "native". Further, while these alternative ecosystems have the potential to provide safe, structurally stable and manageable environments with acceptable ecological functions (Doley and Audet, 2016), where the objective is native ecosystem rehabilitation, it is not clear that they can necessarily provide beneficial environmental outcomes. Generally, there is limited knowledge of function and recovery from disturbances of degraded *novel* ecosystems (Milton, 2003; Harris et al., 2006; Sasaki et al., 2015). Adopting *novel* or *hybrid* ecosystems as targets for severely degraded post-mining landscapes is problematic since, by definition, such systems lack analogues to provide the baseline or reference conditions (Gwenzi, 2021). Nor is there adequate knowledge about the development trajectories and compatibility of these systems with broader landscape disturbance regimes. This renders difficult the formulation of specific completion criteria for *novel* or *hybrid* ecosystem outcomes.

It is notable that the term 'novel ecosystem' is primarily considered in relation to invasive species (alongside climate change; Hobbs et al. 2013b). In this context, the adoption of *novel* or *hybrid* ecosystem concepts within the scope of a preferred native ecosystem rehabilitation needs appropriate care and scrutiny. The dominance of exotic species in mine land rehabilitation is typically accompanied by low abundances of native species (Gastauer et al., 2018). *Novel* and *hybrid* ecosystem rehabilitation must be dominated by native species (Figure 1) to be considered native ecosystem rehabilitation. Furthermore, for *novel* native ecosystem rehabilitation to be considered a viable PMLU, we recommend that it must deliver demonstrable ecosystem services (e.g. carbon sequestration, habitat, cultural values; see others in Costanza et al., 2017) above and beyond those ecosystem services that are inherently required for rehabilitated land to achieve a safe, structurally stable condition and not cause environmental harm (EP Act, s111A).

For the mining industry to make best use of the *natural–hybrid–novel* ecosystem concept, there is a need for clearer objectives at each mine operation, both in terms of what is meant by 'native' and in relation to the robustness and specificity of completion criteria. Stated objectives for native ecosystem rehabilitation are often ambiguous or vague, with words such a 'native', 'local' and 'similar' being undefined in mine documentation, for example, simply specifying 'some bushland characteristics'. Consequently, objectives may be interpreted differently by different stakeholders over time. Many mines lack quantitative completion criteria, or, where present, there may be a disconnect between the criteria and the stated native ecosystem target.

In Australia, notions of 'nativeness' in relation to ecological restoration are complex (Trigger et al., 2008). Whilst in the broadest sense 'native species' include all non-introduced species in Australia, rehabilitation using species from far-flung regions within Australia (e.g., using WA endemics in Qld rehabilitation—Peake et al. 2021) is no longer common practice. Therefore, we recommend that mine plans have 1) a detailed outline and justification for the native ecosystem target(s), 2) quantifiable completion criteria that align with the target(s), 3) incorporation of current ecological knowledge about appropriate ecosystem trajectories, and 4) an outline of the predicted beneficial environmental

outcome(s) that can be measured. Additionally, best practice should focus on local species and we recommend species combinations to be specifically those from within the *natural* REs within the bioregion where post-mining biophysical conditions support these.

4.2 Potential to "lower the bar"

Concerns have been voiced that the concept of *novel* ecosystems may promote *laissez-faire* attitudes to conservation and restoration (Murcia et al., 2014; Higgs, 2017). Some researchers and rehabilitators contend that acceptance will facilitate the degradation of land through less stringent regulation (Clewell and Aronson, 2013; Aronson et al., 2014), although this is debated (Hobbs et al., 2014). At a minimum, the formal incorporation of *novel* ecosystems as a target in mine rehabilitation guidelines would acknowledge that, in some cases, impacts are severe enough to result in permanent loss of fidelity to unmined native ecosystems (including even the ability to restore fidelity). This may be seen as problematic (Perring et al., 2013), in that "settling" on *hybrid* or *novel* ecosystems as a target may stifle innovation and the development of new techniques for establishment of *natural* rehabilitation. Thus, managers should remain cognisant of the distinctions between *planned novel* (i.e., *designer*), *unplanned novel* and *hybrid* ecosystems (Table 2) and when each of these may be necessitated due to inherent biophysical limitations (Figure 1). Furthermore, operators should consider the implications of future management requirements for the final rehabilitated ecosystem. This is important as no-analogue rehabilitated ecosystems will likely require increased ongoing management compared to rehabilitated *natural* ecosystems.

4.3 Potential benefits of adopting the *hybrid–novel* concept

PMLUs need to minimise the maintenance burden of rehabilitated areas into the future (Doley et al., 2012). Where both the abiotic and biotic post-mining systems have been significantly and irreversibly affected by the mining, *hybrid* or *novel* ecosystems may represent the only achievable self-sustaining option (Doley and Audet, 2013; Higgs, 2017). Incorporating *hybrid* and *novel* ecosystem outcomes into the native ecosystem rehabilitation framework assists land managers in ecological goal setting, to identify what can realistically be accomplished and the magnitude of intervention required to achieve a desired ecological state (Doley and Audet, 2016; Wagner et al., 2016). Therefore, we recommend the adoption of *planned novel* and *hybrid* ecosystems as valid considerations within the scope of native ecosystem rehabilitation, but only with caveats lest unnecessary land degradation eventuates. Establishing or retaining *hybrid* ecosystem rehabilitation may be beneficial since this state may deviate little from *natural* rehabilitation and is therefore a less ambiguous native ecosystem rehabilitation objective. Simultaneously, as *hybrid* ecosystems, by definition, are able to be managed or modified to become fully *natural* ecosystems (Table 2), it could be argued that such work *should* be done. Where the rehabilitation is quite young, or the elements that are resulting in the *hybrid* status are few, then interventions to target a *natural* ecosystem (e.g., selective removal of some tree species) are likely justified. In contrast, where the rehabilitation's floristics are not closely aligned with any candidate *natural* ecosystem, or where trees are large and offer high levels of habitat value, then retaining the *hybrid* state is likely to be preferable.

The situation is somewhat different for *novel* ecosystems where, by definition, the rehabilitation cannot be managed into a *natural* state. Sometimes *novel* ecosystem rehabilitation has developed into a self-sustaining ecosystem that has valuable native elements but cannot practically be restored to a more *natural* state. In such cases the concept of *novel* ecosystems may be validly useful as a category and retaining *novel* native ecosystems may be desirable. Incorporating, where appropriate, *novel* ecosystems under the native ecosystem rehabilitation umbrella, allows for this flexibility (Doley et al., 2012). By not requiring strict fidelity to the species composition of a reference ecosystem, this concept recognises that there are alternative opportunities for biodiversity conservation and ecosystem service provisioning (Perring et al., 2013). *Novel* ecosystems are also useful for emphasising ecosystem function as a goal in itself (Miller and Bestelmeyer, 2017). Nevertheless, best practice rehabilitation methods include managing *novel* ecosystems to deliver beneficial environmental outcomes and ecosystem services above and beyond those required for land in stable condition as defined in the EP Act (s111A). These may include recreational services or timber production, or regulatory services like water purification and maintaining populations of pollinators (Costanza et al., 2017; Rosa et al., 2020). However, in these cases, the PMLU designation and thus rehabilitation efforts on the *novel* ecosystem itself, may be more appropriately directed at the service being delivered (e.g., forestry, recreation), rather than 'native ecosystem' (Department of Environment and Science, 2021) which inherently suggests development of a *natural* ecosystem.

This is particularly relevant for *planned novel* (i.e. *designer*) ecosystems which may facilitate multi-use

outcomes while simultaneously bridging the conceptual divide separating the ecological function of re-instated *natural* landscapes versus derelict and (or) unusable landscapes (Doley et al., 2012; Higgs, 2017). Examples of *planned novel* ecosystems that deliver beneficial environmental outcomes include:

- Native seed orchard (Nichols et al., 1985; Gardner and Bell, 2007; Annandale et al., 2021)
  - It is anticipated that wild harvest will not meet future demands for native seed (Hancock et al. 2020b)
  - These may facilitate climate adaptive provenancing (Prober et al., 2015)
- Bush tucker gardens (Annandale et al., 2021)
- Carbon sequestration (Ntshotsho, 2006; Ahirwal and Maiti, 2017).

As they are planned to achieve specific outcomes, *planned novel* ecosystems can place an even greater emphasis on delivery of certain ecosystem functions than their *unplanned novel* counterparts including new functions not generally provided by native ecosystems, or optimised functions, or both (Light et al., 2013). In this regard, *planned novel* ecosystems may be considered 'native ecosystem' where the services they deliver restore those lost from the disturbed native ecosystem.

5. Conclusions

Where the proposed PMLU is a native ecosystem, there is ambiguity regarding what constitutes best practice rehabilitation. We consider the best practice for native ecosystem rehabilitation is one that aims to restore native ecosystems with as much fidelity as possible to existing *natural* ecosystems. This may be rehabilitation to *historical* ecosystems, or to *substitute* ecosystems where post-mining landforms and soil properties better suit a substitute RE from within the mine's bioregion. In some cases, *hybrid* and *novel* ecosystems may be warranted if biophysical limitations caused by mining disturbances dictate that *natural* ecosystem types (*historical* and *substituted*) are not achievable. However, we caution managers when incorporating *hybrid* or *novel* targets in rehabilitation plans, that their novelty means there is uncertainty in ecosystem sustainability and future maintenance burden.

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