Soil reconstruction after mining fails to restore soil function in an Australian arid woodland

Corrine Duncan^{1*}, Megan K Good², Ian Sluiter¹, Simon Cook¹, and Nick L Schultz¹

¹ School of Health and Life Sciences, Federation University, Mount Helen, VIC 3350, Australia.

² BioSciences, University of Melbourne, Parkville, VIC 3052, Australia.

* Corresponding author email: corrineduncan@students.federation.edu.au

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Abstract

The biogeochemical properties of soils drive ecosystem function and vegetation dynamics, and hence soil restoration after mining should aim to reinstate the soil properties and hydrological dynamics of remnant ecosystems. The aim of this study is to assess soil structure in two vegetation-types in an arid ecosystem, and to understand how these soil properties compare to a reconstructed soil profile after mining. In an arid ecosystem in southeast Australia, soil samples were collected at five depths (to 105 cm) from remnant woodland and shrubland sites, and sites either disturbed or totally reconstructed after mining. We assessed soil physico-chemical properties and microbial activity. Soils in the remnant arid ecosystem had coarse-textured topsoils that overlay clay horizons, which allows water to infiltrate and avoid evaporation, but also slows drainage to deeper horizons. Conversely, reconstructed soils had high sand content at subsoil horizons and high bulk density and compaction at surface layers (0–20 cm). Reconstructed soils had topsoils with higher pH and electrical conductivity. The reconstructed soils did not show increased microbial activity with time since restoration. Overall, the reconstructed soil horizons were not organised in a way that allowed rainfall infiltration and water storage, as is imperative to arid-zone ecosystem function. Future restoration efforts in arid ecosystems should focus on increasing sand content of soils near the surface, to reduce evaporative water loss and improve soil quality and plant health.

Keywords: Arid zone restoration, biochemical properties, inverse texture, microbial biocrusts, soil profile, topsoils.

Implications for Practice

- The aim of soil reconstruction of an arid-zone ecosystem after mining should be to replicate hydrological dynamics of the remnant ecosystem, which is closely linked to the organisation of soil horizons.
- In arid systems, coarse-textured upper horizons overlying clay subsoils may be essential to increase infiltration, reduce evaporation and prevent water loss through deep drainage.
- Practitioners should understand the hydrological dynamics and organisation of soil profiles of their reference ecosystems, and monitor how well this is replicated in their reconstructed soil.

Introduction

Land degradation and desertification are major global socioeconomic and environmental issues (Wang 2004). Restoration of degraded arid zones is particularly challenging due to low and variable rainfall (Pickup & Stafford Smith 1993). Despite enormous efforts, restoration success in arid ecosystems is low (Cortina et al. 2011; Ravi et al. 2010). Restoration is often complicated by nutrient-poor skeletal soils, erosion from wind and water, and a sparse distribution and cover of slow-growing plant species (Kassas 1995) which makes these ecosystems more sensitive to disturbance, and more challenging to restore than ecosystems with higher rainfall (James et al. 2013). As we improve our understanding of the processes governing arid-zone ecosystem functioning, we must ensure that we incorporate this understanding into our restoration efforts.

Water is the major limiting resource for growth and productivity in arid zones (Noy-Meir 1973) because rainfall is low and evaporation is generally higher than rainfall across all months (Houston 2006). As such, the physical properties of arid-zone soils, such as compaction, bulk density, clay content and horizon thickness, directly influence infiltration and evaporation (Simmers 2003). According to the Inverse Texture Hypothesis (Noy-Meir 1973), arid vegetation is more productive in coarse-textured soils than fine-textured soils due to less evaporative water loss (Noy-Meir 1973; Sala et al. 1988). Conversely, high bulk density and fine-particle content at deeper soil horizons prevents water leaching

below the maximum rooting depth of plants. Consequently, any disturbing process that alters soil texture and reorganises soil horizons can impact the soil's ability to resist water loss, and can negatively impact plant growth (Cleverly et al. 2016).

In mining landscapes, soils that are reconstructed after mining often have transformed soil chemical and physical profiles (Sheoran et al. 2010) with lower nutrient availability and higher bulk density, pH and electrical conductivity (Shrestha & Lal 2011). These soil properties are crucial to soil–water balance and distribution of plant species in arid zones (Huang et al. 2017). Poor storage of topsoil can cause significant erosion and soil loss (Golos & Dixon 2014). Additionally, significantly altered soil physico-chemical properties in reconstructed soils can further influence resource dynamics, ecosystem productivity (Muñoz-Rojas et al. 2016), and the ability of native species to adapt to local conditions (Griscom & Ashton 2011). Given the importance of soil properties to the establishment and survival of arid plants (Austin et al. 2004), quantifying the effects of soil reconstruction may provide insights to enhance restoration outcomes in arid zones.

Clearing vegetation and stripping topsoils changes the soil microbiome, a critical factor for plant fitness and diversity in arid zones (Lau & Lennon 2012; Moreira-Grez et al. 2019). Microbial activity in biocrusts is important for soil stability, nutrient cycling and moisture retention

(Lau & Lennon 2012). Recovery of microbial crusts on bare soil is incredibly slow and may require hundreds of years to fully recover (Belnap & Eldridge 2001). Microorganisms have varied responses to soil amendments, and some taxa exhibit significant positive changes with amendments of water and organic matter, while other taxa decline (Buelow et al. 2016). Likewise, microbial responses to disturbance vary widely amongst taxa and further studies are required to understand how the scale of disturbance can impact the distribution and recovery of microbial populations in arid soils.

There is a growing recognition of how biotic components of arid ecosystems influence ecosystem function; for example, a high cover of vascular plants can improve ecosystem productivity and soil fertility, creating positive feedback loops (Maestre et al. 2016). Furthermore, the importance of biotic attributes to soil properties tends to increase as environmental stress or aridity increases (Maestre et al. 2010). Considering vegetation cover can exert significant influences on the biogeochemistry of arid soils, we might expect correlations between above ground cover, soil properties and ecosystem function (Johnson et al. 2016). Feedbacks between vegetation cover, soil fertility and ecosystem function can be positive or negative (Johnson et al. 2016) and are still poorly understood (van der Putten et al. 2013).

The aim of the study was twofold: (1) to assess soil properties in a natural arid ecosystem, and (2) to understand how soil properties important for plant growth are impacted by severe disturbance and soil reconstruction. Based on the prediction that changes to soil physical, chemical and biological structure can impede plant recovery and inhibit growth of trees and shrubs (Schwinning & Sala 2004; Sheoran et al. 2010), we test the hypotheses that reconstructed soils differ from the undisturbed remnant soils in properties important for plant growth, specifically in the distribution of particle-size content, nutrients and microbial activity.

Methods

Study area

The study site was 30 km west of Pooncarie in south-eastern Australia's arid zone (33°22'S, 142°13'E) at the Ginkgo Mine, a mineral sands mine. Mining commenced in 2006 and will conclude in 2020, although rehabilitation will continue beyond 2020. The study area is a hot desert climate, according to the Koppen Classification System. Mean annual rainfall was 250 mm (BOM 2018), although annual rainfall often falls below 200 mm for consecutive years (Envirodata 2019). Mean annual temperature is 18°C, although temperatures can range from 48°C to -3°C (BOM 2018). Evaporation rates (mean 5.6 mm.hr⁻¹) are higher than rainfall across all months (Envirodata 2019). Soils of the study area are

Pleistocene age aeolian deposits of the Woorinen Formation and arranged mostly as plains (Lawrence 1966), but also occur as east-west aligned dunes and swales. Prior to mining, soils are predominantly Calcarosols (Isbell & NCST 2016) with surface soil textures of plains dominated by sandy clay loam and clay loam, with subordinate dune areas often characterized by light sandy clay loam and sandy loam. Subsoil B-horizons are ubiquitously characterized by carbonate pedoderms with elevated calcium carbonate content. Soils of rehabilitation areas are Anthrosols (Isbell & NCST 2016). Native shrubs have been successfully established in rehabilitation areas, but successful establishment of tree species has been negligible (Sluiter & Schultz 2017).

Site selection

In this study, we selected sites in three types of soil profiles that remain after mineral sand mining, each of which is subject to ecological restoration: (1) mine path sites, where soils have been removed, stored and replaced with reconstructed subsoil and topsoil; (2) overburden sites; raised structures where subsoil and topsoils were reconstructed over excess rock fragments; and (3) soil stockpile sites where soils were temporarily stored, and then removed, leaving a degraded area requiring restoration (Fig. 1). The mine path and overburden sites were considered as reconstructed soil profiles herein. In the soil stockpile sites, only the surface layer has been disturbed and hence these sites will be

considered separately. Mine path and overburden sites were reconstructed by spreading a minimum of 20 cm subsoil and 20 cm of topsoil prior to seeding (Fig. 1). Restoration activities (including direct-seedling with a mix of native plant species) occurred in four years in overburden sites (2009, 2011, 2012 and 2013), and in three years in mine path and soil stockpile sites (2014, 2015, and 2017). Ten sites were selected in the mined area (three mine path, four overburden and three soil stockpile). Five remnant shrubland sites and five remnant woodland sites were also selected from around the study area for comparison to the mine sites, providing a total of 20 sites.

Vegetation of the study area

The two dominant broad vegetation community types are (a) woodlands dominated by *belah* (*Casuarina pauper*) with a bluebush (*Maireana* spp.) understorey shrub layer; and (b) chenopod shrublands primarily dominated by *Maireana* species. Shrubs were observed across all sites, although shrubs were typically smaller at woodland sites. The shrub layer of the vegetation is dominated by either *Maireana pyramidata* or *M. sedifolia*, and seldom both. These species are long-lived perennials which grow to 1.5 m (Walsh 1996) and in the case of *M. sedifolia*, may live for centuries (Crisp 1978). Other common perennial chenopod shrubs present include *Maireana georgei*, *M. turbinata* and *M. trichoptera* and *Atriplex* spp., including *A. vesicaria* and *A. stipitata*. Only the woodlands featured tree and grass species, and these sites were dominated by the

tree *Casuarina pauper* and the perennial grass *Austrostipa nitida*. Both of these species were absent from rehabilitation sites. The most common ground flora species across all sites was the short-lived perennial herb, *Sclerolaena obliquicuspis*. The most prolific weed across all sites was Ward's weed (*Carrichtera annua*) which was more prevalent in the reconstructed ecosystems. Current vegetation structure at the reconstructed sites was more similar to shrubland than woodland sites, despite broadcasting significant quantities of tree and grass species.

Soil description

Soil profiles of remnant and rehabilitation sites were described using the Australian Soil Classification (Isbell & NCST 2016), by excavating a pit with a backhoe to 1.5 m depth. At each pit site, all soil horizons were identified using the following criteria: soil texture, colour, pedality (type and grade), field pH, readily available water at maximum root zone depths (herein referred to as available water) and carbonate reaction type. Topsoil depth at all sites was determined as the depth of the A horizon. Soil samples were collected from all sites by scraping soil from the pit face with a small pick hammer at depths of 0–10 cm, 10–20 cm, 25–45 cm, 55–75 cm and 85–105 cm. Soil samples were stored at 4°C upon collection, and until processed and analysed. All soil samples were collected during the 21st – 23rd of February 2018.

Biophysical and chemical properties of soils

Samples were collected using a 10-cm corer hammered into the pit face at five depths (5, 15, 35, 65 and 95 cm). To measure bulk density, gravimetric soil-moisture and soil porosity according to Carter & Gregorich (2007), soil samples were oven dried at 105°C until soil mass remained constant. The structural stability of soil aggregates was measured using the Emerson (1967) test. Slaking was noted after 1 minute and 1 hour, and dispersion was noted after 1 hour and 20 hours. All samples that showed no dispersion as aggregates were re-worked and a sample placed in de-ionised water. The plates were then examined after 1 hour. Soil compaction along each pit face was measured using a pocket penetrometer, with three measurements taken at each of the five depths. Surface soil compaction was also measured with a pocket penetrometer, and six soil surface penetrations were performed at every 50 cm along a 3-m transect.

Soil physical and chemical properties were measured at all 20 sites, and at each of the five depths, by APAL Agricultural Laboratory (Adelaide, Australia) who analysed all soil properties according to Rayment & Lyons (2011), as follows: soil pH was measured via the 1:5 soil water suspension method, determined after shaking for one hour at 25°C, and the 1:5 soil/0.01 M calcium chloride extract method with stirring during measurement. Electrical conductivity (EC) was determined using the 1:5 soil/water extract method, using air dried (40°C) soils shaken for 1 hour

at 25⁰C. Effective cation exchange capacity (ECEC) is the sum of exchangeable bases plus exchange acidity and was calculated as; ECEC [cmol_c/kg] = Exchangeable bases ($\text{Ca}^{2+} + \text{Mg}^{2+} + \text{Na}^{+} + \text{K}^{+}$) + exchange acidity ($\text{Al}^{3+} + \text{H}^{+}$). Chemical components of exchangeable bases were measured using the 1 M ammonium acetate method and exchange acidity components by the 1 M potassium chloride method. Results were determined by atomic absorption spectrometry and reported as exchangeable cations (c.mol/kg) on an oven-dry basis. Colwell phosphorus was determined by the bicarbonate, automated colour, continuous segmented flow method (Colwell 1963). Bicarbonate-extractable K^{+} tests were applied to measure Colwell potassium and both tests were reported as mg/kg (Rayment & Higginson 1992). Results were determined through atomic absorption spectrometry and reported as extractable K and P (mg kg⁻¹) on an air-dry basis. Total nitrogen was calculated using the Dumas high temperature combustion method (Dumas 1831), using high-grade EDTA ($\text{C}_{10}\text{H}_{16}\text{N}_2\text{O}_8$) dried at 105⁰C for 2 hours. Mineral nitrogen, including nitrate and ammonium, was measured by the 2 M KCl, automated colour, continuous segmented flow method where, and reported as $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ (mg N/kg). Total organic carbon (TOC) was determined through Dumas high-temperature combustion, which analyses the amount of CO_2 liberated from combustion at 1050-1200⁰C. Calcium carbonate equivalent was calculated through rapid titration, where soils treated with 1 M Hydrochloric Acid were titrated with 0.5 M sodium hydroxide.

Soil samples for analysis of microbial activity were collected from the top 15 cm of soil at each site using a 26-mm soil auger. The auger was moved out in a spiral pattern, until 25 samples were collected at each site. Sub-samples were thoroughly mixed in buckets. Samples were sent to Microbiology Laboratories Australia Pty Ltd for a microbe activity wise (MAWS) test (Franzluebbers et al. 1996).

Statistical analysis

Differences in soil properties were tested at two levels: (1) among site types (mine path, overburden, soil stockpile, shrublands and woodlands) and (2) at a broader ecosystem level, to test for differences between reconstructed soils (mine path and overburden sites grouped) and remnant soils (shrublands and woodlands grouped). Soil stockpile sites were excluded from the ecosystem level analysis, as they did not fit either the remnant or reconstructed categories. A Kruskal-Wallis test was used to test differences among site types, and the Mann-Whitney test was used for the ecosystem-level test. To investigate the correlation between soil properties and sites, we performed correlation-based Principal Components Analyses (PCA), based on normalised data and Euclidean distance. We tested for differences between woodland and shrubland sites in mean topsoil depth, observed root zone depths, and available water using t-tests. Correlations between available water, topsoil depth and observed root zone depth, and correlations between microbial activity and time since restoration, were assessed using Pearson tests. All

statistical analyses were performed using SPSS (IBM Corp 2017), except for PCA which was performed using CANOCO 5.0 (ter Braak & Šmilauer 2012).

Results

Physical properties of topsoils

Fine particle content at all remnant sites showed coarse topsoil overlying finer-textured subsoil, demonstrating a highly organised soil profile (Figure S1); topsoils at these remnant sites ranged from sandy loam to sandy clay loam (10–30% clay content) and, at depths below 85 cm, subsoils were classified as light clay or light medium clays with 35–40% clay content. In contrast, the reconstructed soils did not demonstrate this organisation of the soil horizons; two mine path sites and two overburden sites had the largest soil particles in the profile at subsoils below 85 cm, and all overburden sites (excluding the overburden site reconstructed in 2009) had a coarse soil texture throughout the entire profile (Figure S1). The only reconstructed site to exhibit similar distribution of particle-size content to the remnant soils was the oldest overburden site, which was reconstructed in 2009.

Overall, soils were classified as clay loam, with a subangular blocky pedality and were prone to slaking. Across all depths, soils at woodlands, shrublands and soil stockpiles were non-dispersive or showed < 10% dispersion after 20 hours, except at two shrubland sites which had Emerson class numbers of 2 (10–50% dispersion) and 3 (50–75% dispersion) at 85 – 105 cm (Table S2). The mine path and overburden sites were non-dispersive or showed < 10% dispersion after 20 hours at depths below 20 cm, but dispersion was more variable below 20 cm; some sites were non-dispersive, but four of the seven sites showed >50% dispersion at one or more depths (Table S2). Subsoil horizons were classified as Bk (indicating the accumulation of calcium carbonates). Reconstructed sites contained a relatively shallow C horizon, with weakly consolidated sediments and a lack of aggregation. Topsoil depth ranged from 20–60 cm in woodlands, and from 15–50 cm in shrubland sites. Reconstructed sites had highly variable topsoil depths. Soil stockpile sites had the lowest topsoil depth, with only 10–15 cm of topsoil, which was significantly lower than observed in overburden sites ($p = 0.041$).

Compaction and bulk density

Surface compaction was higher in remnant soils than in reconstructed soils (Fig. 2a). Compaction throughout the soil profile, at depths below 5 cm, generally increased progressively with soil depth at all sites. Mean bulk density along the soil profile was generally consistent at remnant

sites, but not at reconstructed sites or soil stockpiles (Fig. 2b). At overburden and mine path sites at 0 – 20 cm, bulk density ranged from 1.64 - 2.28 Mg.m⁻³ and 1.53 - 2.05 Mg.m⁻³ respectively. Bulk density was higher in mine path soils at depths below 35 cm than at all other sites.

Soil chemical properties

Soils were low in nutrients at all sites, although differences were observed among sites in the distribution of chemical properties through the soil profile (Table 1 and Table S1). All sites were very low in total organic carbon and nitrogen, and in ammonium nitrogen which was generally less <2 mg/kg. Total nitrogen, potassium and Ca:Mg generally decreased with soil depth, the latter due to an increase in magnesium with soil depth. Soils were non-saline, indicated by electrical conductivity values below 1.5 dS/m. Soil pH increased with depth and ranged from 7.6–8.4 across all sites, with topsoils and subsoils considered moderately to strongly alkaline (Wetherby 1994). Remnant and reconstructed soils varied significantly in terms of their soil chemical composition (Table 1 and Table S1). At 10–20 cm, bulk density, total organic carbon, pH, electrical conductivity, total nitrogen and calcium carbonate were significantly different among treatments (Table 1). Electrical conductivity and cation exchange capacity were generally significantly higher in mine path than woodland sites at depths between 0–45 cm. Electrical conductivity

(salinity; dS/m) was significantly higher in reconstructed soils at all depths from 0–75 cm (Table S1), and pH was significantly higher in mine path and overburden sites at 0–20 cm (Table 1).

The reconstructed soils were most easily differentiated from the remnant soils based on variables associated with salts; soil properties that were the most influential in explaining variation among treatments, as shown in the PCA diagrams (Figure S2), include cation exchange capacity (eCEC), electrical conductivity (EC), calcium, magnesium, potassium, calcium carbonate, and pH. Remnant woodland and shrubland soils had similar soil properties, and were different from reconstructed soils, particularly at surface (0–20 cm) and lowest (55–105 cm) soil depths; remnant soils in the PCA diagrams were clustered at these depths showing greater homogeneity in soil properties compared to the reconstructed soils, which were more-widely spread across the axes at each depth.

Microbial activity

Microbial activity among the site types varied from a mean of 138.1 ± 43.2 mg C.kg⁻¹ at the overburden sites to 210.0 ± 42.0 mg C.kg⁻¹ in the shrublands, but there were no significant differences in microbial activity among the site types ($F = 1.03$, $p = 0.42$). There was no significant

correlation between microbial activity and year since restoration for the mine path sites ($F = 0.28$, $p = 0.69$) or the overburden sites ($F = 0.09$, $p = 0.81$). The oldest overburden site had high microbial activity ($261.6 \text{ mg C.kg}^{-1}$), but was removed from the correlation analysis as it had inoculates added during reconstruction.

Discussion

The aim of the study was to assess soil structure and properties in a remnant arid ecosystem, and to understand if reconstructed soil profiles after mining exhibit similar structure and properties to the remnant systems. Reconstructed sites had higher bulk density, compaction and clay content than remnant sites at topsoil depths, but this declines with soil depth. Soils from the remnant ecosystem exhibit topsoils that are coarse in texture, with clay content and compaction increasing with soil depth. Reconstructed soils differed from the remnant ecosystem in soil properties, particularly in the distribution of clay content and variables associated to the distribution of ions. Reconstructed soils were more likely to be dispersive at soil depths below 20 cm. These results suggest that despite enormous restoration effort, the reconstructed soils have reduced ecohydrological function compared to the remnant ecosystems.

Higher sand content in upper soil layers in arid zones is also linked to more stable soil-moisture dynamics (Zhang & Shao 2013). Remnant soils had higher surface compaction than expected, which may increase evaporative water loss from the compacted soil surface and inhibit seedling establishment (Fan et al. 2018). Higher surface compaction in remnant soils may represent degradation from drought, the trampling effects from wildlife or, conversely, natural hydrological processes of surface soils to enhance water run-off towards vegetated patches (Tongway & Ludwig 1990). Physical, chemical and biological crusts can influence surface compaction and subsequent hydrological processes, and are necessary to reduce soil erosion and facilitate water flows (Eldridge et al. 2002). Crust formation has been linked to soil-water retention and repellence, which drives seedling emergence in arid zones (Merino-Martínez et al. 2017), however the role crusts play in controlling soil-moisture dynamics, including in a restoration setting, is complex and warrants further studies (Gypser et al. 2016).

Soil reconstruction affects the distribution of properties important for plant growth

Soil properties in the reconstructed system tended to differ to soils from remnant sites. In general, soil reconstruction increased bulk density at 10–20 cm depths. Bulk density of productive natural soils generally range from 1.1 to 1.5 Mg.m⁻³, and values greater than this directly limit plant growth as most species are unable to grow roots effectively through high bulk density soils (Sheoran et al. 2010; Singh et al. 2015). The

increased bulk density at reconstructed sites at 10–20 cm is likely to increase evaporative water loss and erosion potential, and will inevitably have negative impacts on establishment of woodland species (Sheoran et al. 2010). High bulk density in soils is commonly associated with disturbance and increased likelihood of desertification (Su et al. 2004). Shrestha & Lal (2011) also showed that reclaimed soils after mining had higher bulk density than remnant systems; in their study, bulk density in the top 15 cm the remnant systems ranged from 0.98–1.41 Mg.m⁻³, and 1.11–1.69 Mg.m⁻³ in the reclaimed soils. Bulk density in the reconstructed soils in our study was >2 Mg.m⁻³ at 15 cm at some sites, suggesting a major impediment to root penetration. These properties are crucial factors regulating soil hydrological function, root development and plant-water uptake (Cross et al. 2018; Huang et al. 2012).

Sites containing reconstructed soils lacked a deep clay layer that is integral to reducing drainage in arid regions, and instead had high sand content at depths below 85 cm. There is strong evidence that a clay layer at lower root zone depths strongly influences plant productivity (Macinnis et al. 2010). Other important ecosystem components that reduce deep drainage include vegetation cover and plant roots (Scanlon et al. 2005), and hence recently reconstructed soils with little vegetation structure are likely further prone to water loss from deep drainage. Many woody species in the arid zone rely on large rainfall events to recruit and grow (Wiegand et al. 2004), therefore understanding the fate of water

following large rainfall events is important. High sand content at depths below 85 cm may effectively reduce the ecological impact of large rainfall events, and potentially contributes to water loss in reconstructed soils at the study site. We propose that the loss of profile organisation limits plant function and establishment at the study site.

Reconstructed soils had different chemical properties to remnant soils and these differences may impact the soil-water balance and plant soil interactions (Sheoran et al. 2010). Soil pH in reconstructed soils in this study were slightly alkaline ($\text{pH} = 8.0 - 8.4$) at nearly all depths, because little leaching and high evaporation causes ions to concentrate in the soil (Hall et al. 2009). These thresholds are known to impede plant growth (Kopittke & Menzies 2004), with most plants species favouring lower ranges ($\text{pH} = 5.5 - 8.0$) (Gazey 2018). At soil pH above 8.5, previous studies have reported soil structure degradation, reduced infiltration rates and decreased plant productivity (McCauley et al. 2017). These characteristics likely impact restoration performance in the study site, and further research should investigate ecophysiological performance of young plants. The high soil pH thresholds at reconstructed sites are likely caused by topsoil losses at all stages of soil movement around the mine. This includes initial topsoil-stripping, storage and redistribution onto rehabilitation surfaces. Topsoils are particularly susceptible to mixing with higher-pH subsoils during initial topsoil stripping prior to mining activities. Similar increases in pH and electrical conductivity were

observed after mining in sites in Northeast America (Shrestha & Lal 2011), and in the arid regions of China (Huang et al. 2018) and Western Australia (Cross et al. 2018), causing poor hydraulic conductivity and inhibiting plant growth.

Microbial activity in the reconstructed soils did not increase with time since reconstruction. We expected low microbial activity immediately after soil reconstruction, because when topsoil layers are removed and stockpiled during mining operations, microbial communities are buried underneath upper soil layers (Sheoran et al. 2010), which can limit soil aeration and induce anaerobic conditions (Ghose 2001). However, most studies report biocrusts – which account for much of the microbial biomass in arid soils – slowly recovering during the first 15 years of restoration (Hou et al. 2018; Zhang et al. 2016), although successional times are likely species-specific (Banning et al. 2011) and dependent on soil quality. As the age of the restoration sites increases, plant biomass and subsequent leaf litter generally increase on the soil surface. Thus, increased soil health and microbial activity are expected of older reconstructed soils (Orozco-Aceves et al. 2017). Soil moisture and nutrient content (Chandler et al. 2019), clay content and particularly soil pH (Rousk et al. 2010), are the primary factors affecting growth of biocrusts. Higher pH and salinity in the reconstructed soils may explain the poor microbial responses to time since restoration. Certainly, reducing runoff and increasing infiltration has been shown to facilitate the growth and restoration of important cyanobacterial crusts (Bu et al. 2014), which are

dominant phyla across the study region (Chilton et al. 2015). The oldest overburden site (nine years) had the highest microbial activity, but was removed from the analysis as it had inoculates added during reconstruction. This may suggest there are long-term benefits to treating reconstructed soils with microbial inoculates (Muñoz-Rojas et al. 2018).

Disturbance through mining causes anomalies in soil properties and specifically, increases soil bulk density, compaction, pH and variables associated with salts, and reduces microbial activity. These soil properties directly affect water availability and plant-water uptake, and are key barriers to arid ecosystem restoration.

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Tables (1)

Table 1: Soil physico-chemical properties at different soil depths ($\pm$ standard errors). BD = bulk density (Mg.m^{-3}), Moisture = (%), TOC = total organic carbon (%), pH = pH level following CaCl_2 , EC = electrical conductivity (dS/m), eCEC = effective cation exchange capacity (c.mol/kg), Na = Sodium (mg/kg), Nitrate N = nitrate nitrogen (mg/kg), Total N (%), K = Colwell potassium (mg/kg), Ca:Mg = calcium to magnesium ratio (%) and CaCO_3 = calcium carbonate (%). Significance levels of differences in soil properties at the site level, and at the ecosystem level, are indicated by p values; n.s. when $p > 0.05$, * when $0.05 > p > 0.01$, ** when $0.01 > p > 0.001$, *** when $p < 0.001$. Sites that share a letter (*a* or *b*) share similar soil properties. Soil properties at depths below 20 cm are shown in Table S1.

	BD (Mg.m^{-3})	Moisture (%)	TOC (%)	pH	EC (dS/m)	eCEC (c.mol/kg)	Na (mg/kg)	Nitrate N (mg/kg)	Total N (%)	K (mg/kg)	Ca:Mg (%)	CaCO_3 (%)
0 - 10 cm												
Woodland	1.62 ± 0.09	1.7 ± 0.9	0.49 ± 0.14	7.6 ± 0.1 <i>a</i>	0.13 ± 0.03	14.7 ± 3.2 <i>a</i>	44 ± 6	11.6 ± 3.1	0.06 ± 0.01	342 ± 41	5.57 ± 1.26	2.4 ± 1.4
Shrubland	1.56 ± 0.08	1.2 ± 0.1	0.48 ± 0.04	7.9 ± 0.1 <i>ab</i>	0.12 ± 0.01	20.7 ± 2.3 <i>ab</i>	65 ± 25	13.8 ± 0.6	0.11 ± 0.01	496 ± 45	8.07 ± 1.62	2.9 ± 1.1
Mine path	1.58 ± 0.03	1.5 ± 0.4	0.53 ± 0.16	8.0 ± 0.1 <i>ab</i>	0.33 ± 0.05	29.9 ± 1.76 <i>b</i>	752 ± 378	22.3 ± 3.8	0.04 ± 0	317 ± 52	8.03 ± 1.84	5.1 ± 2.5
Overburden	1.8 ± 0.06	1.7 ± 0.6	0.22 ± 0.06	8.0 ± 0.03 <i>b</i>	0.27 ± 0.12	20.2 ± 1.3 <i>ab</i>	510 ± 367	22.8 ± 7.8	0.08 ± 0.01	473 ± 102	11.18 ± 2.36	2.9 ± 1.5
Soil stockpile	1.57 ± 0.08	1.8 ± 0.4	0.42 ± 0.11	8.0 ± 0.1 <i>ab</i>	0.23 ± 0.04	27.7 ± 0.9 <i>ab</i>	362 ± 137	17.7 ± 2.8	0.04 ± 0.01	287 ± 26	8.61 ± 1.9	11.3 ± 1.9
ANOVA												
Site level	n.s.	n.s.	n.s.	*	n.s.	*	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Ecosystem level	n.s.	n.s.	n.s.	*	*	n.s.	*	*	n.s.	n.s.	n.s.	n.s.
10 - 20 cm												
Woodland	1.63 ± 0.06 <i>ab</i>	1.3 ± 0.2	0.52 ± 0.1 <i>a</i>	7.76 ± 0.09 <i>a</i>	0.11 ± 0.01 <i>a</i>	16.17 ± 4.09	88 ± 23	11 ± 2.7	0.05 ± 0.013 <i>ab</i>	260 ± 28	6.76 ± 1.67	3 ± 2.3 <i>a</i>
Shrubland	1.52 ± 0.02 <i>ab</i>	2.5 ± 0.5	0.29 ± 0.09 <i>ab</i>	7.97 ± 0.04 <i>ab</i>	0.12 ± 0.01 <i>ab</i>	24.62 ± 1.46	947 ± 518	13 ± 0.6	0.1 ± 0.007 <i>a</i>	330 ± 73	8.51 ± 1.95	7.8 ± 2.3 <i>ab</i>
Mine path	1.81 ± 0.13 <i>ab</i>	2.6 ± 0.1	0.31 ± 0.02 <i>ab</i>	8.1 ± 0.08 <i>ab</i>	0.36 ± 0.05 <i>b</i>	29.96 ± 0.89	1308 ± 478	17.7 ± 1.3	0.03 ± 0.003 <i>b</i>	257 ± 13	6.27 ± 0.82	11 ± 0.6 <i>ab</i>

Overburden	1.97 ± 0.14 <i>a</i>	1.3 ± 0.4	0.05 ± 0.01 <i>b</i>	8.12 ± 0.03 <i>b</i>	0.16 ± 0.04 <i>ab</i>	20.27 ± 0.61	558 ± 471	13.5 ± 2.3	0.08 ± 0.003 <i>ab</i>	333 ± 20	9.06 ± 0.83	2.3 ± 1.4 <i>ab</i>
Soil stockpile	1.31 ± 0.04 <i>b</i>	3.5 ± 1.5	0.86 ± 0.27 <i>a</i>	8.07 ± 0.06 <i>ab</i>	0.29 ± 0.02 <i>ab</i>	28.7 ± 1.58	832 ± 312	24 ± 11.1	0.05 ± 0.006 <i>ab</i>	197 ± 54	6.27 ± 2.11	20 ± 1.7 <i>b</i>
ANOVA												
Site level	**	n.s.	*	*	*	n.s.	n.s.	n.s.	*	n.s.	n.s.	*
Ecosystem level	**	n.s.	*	**	*	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.

Figures

Fig. 1: Soil profile types at the study site. Mine path sites were excavated then filled with rock fragments and topped with subsoil and topsoil. Overburden sites are mountains of excess rock fragments topped with subsoil and topsoil. Soil stockpiles are where top soils is temporarily stored until they are returned to mine path and overburden sites, which disturbs the original vegetation and topsoil.

Fig. 2: (a) mean compaction and (b) mean bulk density across all sites. Compaction below 65 cm exceeded the maximum reading of the penetrometer at most sites. Error bars have been omitted to improve visualisation; standard errors for mean bulk density are presented in Table 1 and Table S1.

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Author/s:

Duncan, C;Good, MK;Sluiter, I;Cook, S;Schultz, NL

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