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Phytocaps in mine waste landforms: a nature-based approach to sustainable rehabilitation

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Abstract

This paper examines the application of phytocaps as final cover systems for mine waste landforms, presenting them as a robust, sustainable alternative to conventional compacted clay barrier covers. Traditional low-permeability barrier systems have been widely adopted to limit percolation and oxidation; however, long-term field studies in both landfill and mining contexts have demonstrated that their hydraulic performance can deteriorate rapidly due to desiccation cracking, root penetration and soil structure changes.

Drawing on long-term field monitoring and applied rehabilitation experience from multiple sites in Victoria and South Australia, this paper demonstrates that phytocaps – cover systems that rely on soil water balance processes and healthy vegetation communities – can achieve equal or superior hydraulic performance compared with compacted clay barrier systems. Data from a decade-long field trial show that phytocaps achieved lower net percolation than a compacted clay barrier despite being constructed without a low-permeability layer.

The paper further demonstrates that vegetation selection, soil preparation and ongoing vegetation management are critical determinants of phytocap performance and long-term stability. Several case studies illustrate how inappropriate soil material selection, seedbed preparation or species selection can undermine rehabilitation outcomes, while adaptive management and the use of indigenous vegetation communities can result in resilient, self-sustaining landforms.

The findings support a shift in mine closure practice away from reliance on high-specification engineered barriers toward vegetation-led, nature-based systems that reduce long-term closure risk, improve sustainability and better align with post-closure land use objectives.

Keywords: *phytocaps, mine closure, rehabilitation, evapotranspiration covers, vegetation establishment*

1 Introduction

Cover systems have been used as part of waste landform rehabilitation in mining and landfilling for decades. In the mid-70s to 80s, soil covers commonly included a low permeability layer to control percolation and oxidation (Figure 1). The *US Resource Conservation and Recovery Act 1976* was the first legislative instrument to mandate the use of a reduced permeability layer (RPL) as a barrier for hazardous waste landfills. One of the earliest mining applications was the Rum Jungle Mine, where a cover system with a compacted clay barrier was installed in the 1980s (Bennett et al. 1988).

In the 1990s, research on compacted clay barriers or RPLs identified fundamental long-term performance issues. Specifically long-term performance was compromised because compacted layers did not remain intact or impermeable over time. One of the earliest studies reported on changes in conventional compacted clay barrier covers in a German landfill (Melchoir 1997). This eight-year study found the saturated hydraulic

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conductivity of the 600 mm thick compacted clay barrier increased from 1 x 10^{-10} m/s at construction to 1 x 10^{-7} m/s after five years. Dye tracer studies showed the clay initially developed small fissures followed by plant root penetration. Similar results have been reported in the United States (Benson et al. 1995; Joshi and Maulé 1999; Montgomery & Parsons 1990) and in Australia (Williams et al. 2006; Salt 2013).

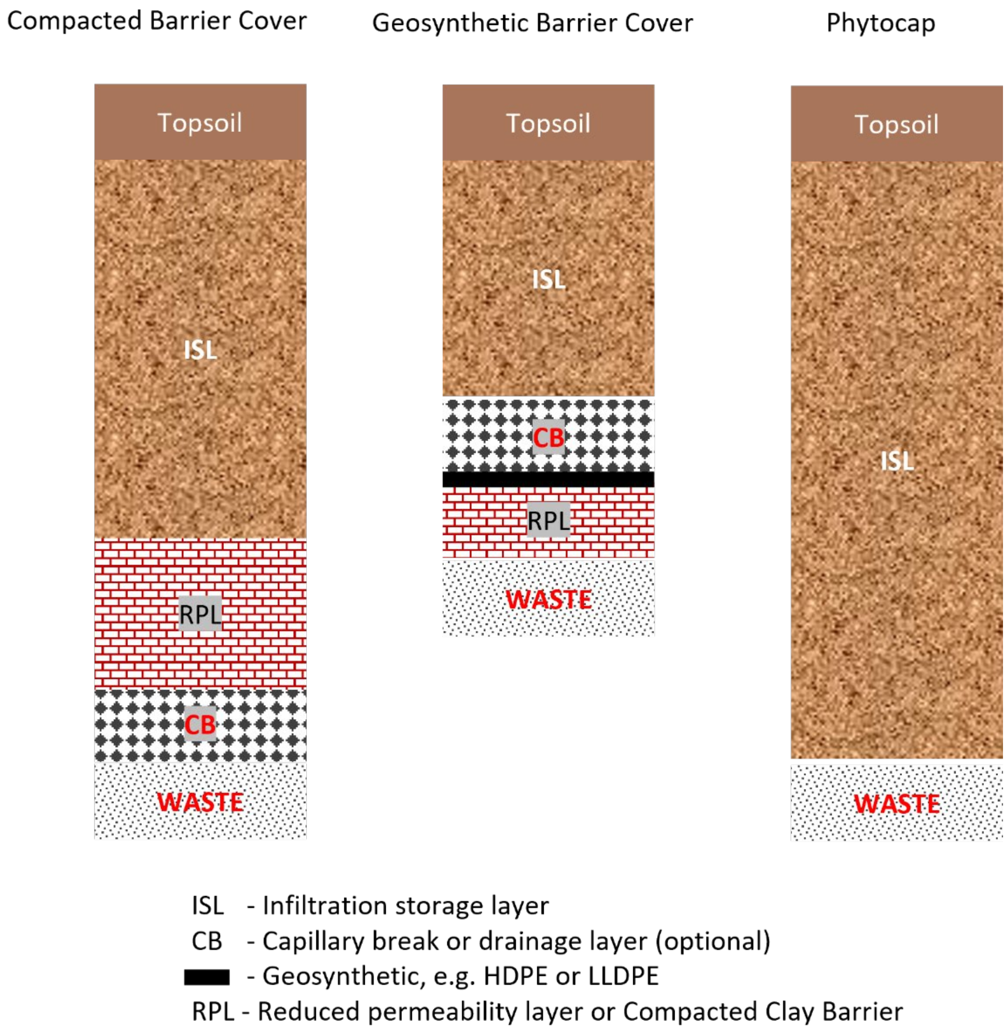


Figure 1 Schematic of typical cover systems, noting that site-specific covers may vary depending on available materials

Due to the high costs and constructability challenges associated with engineered barriers (Figure 1), such as HDPE and LLDPE, research shifted toward alternative cover systems based on natural soil water balance processes. Evapotranspiration (ET) cover systems were pioneered in the United States as part of the Alternative Cover Assessment Program (Albright & Benson 2002) whilst store-and-release cover systems were pioneered in Australia at Kidston Gold Mine (Williams et al. 2006). Although similar, the ET covers used in landfills did not include an RPL, whereas the store-and-release covers often included a deeper RPL to limit oxygen ingress into potentially acid-forming wastes. The ET covers rely on some oxygen ingress to allow soil-borne facultative anaerobes to oxidise methane in landfill gas into carbon dioxide and water.

Building on the work in the US, Australian researchers commenced the Australian Alternative Cover Assessment Program (A-ACAP) in 2006. This program aimed to determine whether ET covers can meet performance criteria for landfills more cost effectively and sustainably than conventional covers by reducing leachate generation and atmospheric release of greenhouse gases (including methane) (Salt 2013). The culmination of this program was the *Guidelines for the Assessment, Design, Construction and Maintenance of Phytocaps as Final Covers for Landfills* (WMAA 2011).

The Australian research team adopted the term to 'phytocaps' to reflect the important role of vegetation ('phyto') in achieving the performance criteria and the long-term sustainability goals of rehabilitation (Figure 1). The change in emphasis from soil water balance to growing suitable vegetation communities is the paradigm shift needed to provide better final cover systems for both mining wastes and landfills. This paper presents monitoring data and applied case studies illustrating the performance of phytocaps and highlights the importance of vegetation selection, soil condition and adaptive management in successful mine closure outcomes.

2 Soil water balance

Effectively, cover systems work by affecting the soil water balance. The simplified equation used in this paper is as shown in Equation 1 with the terms are defined in Table 1.

$$P = ET + R + LF + NP + \Delta S \tag{1}$$

Table 1 Water balance definitions and contributions in Australian context

Abbreviation	Meaning	% of Rainfall in Australia over longer term (Rab et al. 2002)
P	Precipitation (rain, snow, etc.) including any irrigation, if applied	–
ET	Evapotranspiration includes evaporation from the soil and evaporation and transpiration by plants. This includes leaf interception by default	> 75%
R	Surface runoff	0–10%
LF	Lateral flow which is the flow within the soil profile generated by topography and soil pore space changes	0–5%
NP	Net Percolation which is comprised of percolation down through the cover that may become seepage or leachate and capillary rise from underlying water sources, such as shallow groundwater or wet waste (tailings, domestic landfill, etc.).	0–20%
S	Soil moisture storage as calculated for the entire final cover	Variable but usually negligible

ET is the largest loss of rainfall in natural systems in Australia commonly exceeding 75% of long-term average annual rainfall (Rab et al. 2002). Consequently, even small increases in ET can significantly reduce net percolation, runoff and lateral flow. Although ET is often assumed to be higher under tree-dominated systems, Zhang et al. (2001) developed a relationship between average annual ET and average annual rainfall that showed ET from grasslands was similar to treed communities in rainfall areas < 500 mm/yr. Hence, in semi-arid and arid climates there would be little difference in ET between grasses and trees due to the limitation imposed by soil moisture.

Regardless of the importance of ET on the water balance, the selection of vegetation and the soil conditions to improve vegetation establishment are often overlooked in cover system design. This omission can significantly limit the long-term effectiveness and sustainability of the cover system. Field trials focus on the measurement of soil moisture and percolation but may not include trials on vegetation type, establishment, suitable soil amelioration, etc. From the authors' own experiences, the establishment of vegetation can be difficult but the rewards in the water balance and sustainability are significant.

3 Hydraulic performance

The cover system's hydraulic performance is fundamental to achieving regulatory requirements and environmental protection. This performance is influenced by climate, soil and vegetation interactions. While climate is an external factor, soil and vegetation are design elements that can be controlled and optimised

The A-ACAP trial site in Adelaide was monitored for 10 years from construction in 2007 until 2017. Data were collected to compare the performance of a phytocap against a compacted clay barrier cover system using field-scale non-weighing lysimeters. The Adelaide trial site is located in a Mediterranean climate, characterized by hot, dry summers and mild, wet winters.

The phytocap was comprised of 0.3 m sandy loam topsoil over 1.2 m loamy sand subsoil with compaction limited to < 8% maximum dry density. The compacted clay barrier cover consisted of 0.1 m sandy loam topsoil over 0.8 m loamy sand subsoil overlying a 0.6 m compacted clay barrier. The compacted clay layer was placed in a minimum of three lifts, each moisture conditioned and compacted to a maximum 0.2 m. Compaction was achieved using a vibratory sheepsfoot roller, resulting in a hydraulic conductivity of less than 1×10^{-9} m/s. Both cover systems were seeded with a mixed native grass stand. Details of the cover systems and the first four years of data are provided by Salt (2013). On-going monitoring of the trial sites for an additional six years has provided further insight into their relative performance.

The phytocap initially performed well. However, in early 2009, a broad-spectrum herbicide was accidentally applied to the phytocap area, which killed the native grasses and impacted water extraction from the cover system. Figure 2 shows the soil profile moisture content, which is the volume of moisture stored in the cover system throughout its depth during this period. Also shown for comparison are the field capacity, which is the amount of water held in a soil when it has been allowed to drain under gravity after saturation, and the permanent wilting point (PWP), which is the amount of water held in the soil at high negative pressures so that plants can no longer extract the water (Hazelton & Murphy 2016). Following herbicide application, 20 mm of percolation was reported in winter 2009, and a further 11 mm in winter 2010. In the summer of 2009/2010, the soil profile moisture content did not dry to earlier or subsequent lows which reduced the available capacity to store moisture and caused the repeat of a percolation event in 2010. This unplanned disturbance clearly demonstrated the critical role of vegetation in maintain phytocap performance.

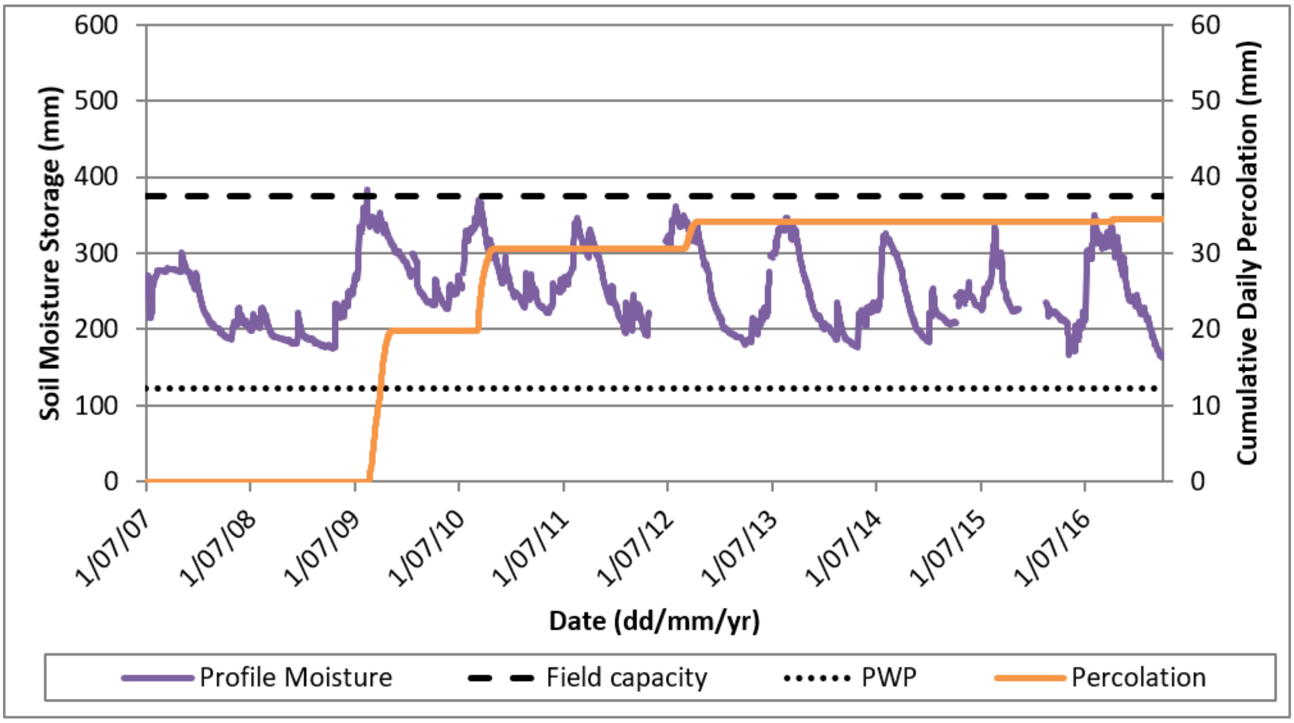


Figure 2 Phytocap profile soil moisture content and cumulative percolation. Red circle highlights timeframe for native grass to recover after accidentally spraying with herbicide

A third percolation event of 3.6 mm occurred in winter 2012 (Figure 3) after the grass had naturally re-established and was actively growing. Although there are missing data due to a datalogger failure, the moisture content increases throughout the profile with all sensors reporting maximum soil moisture content (SMC) by August. This is in response to the high rainfall in June (147 mm), which is within the top 10% of highest recorded June monthly totals.

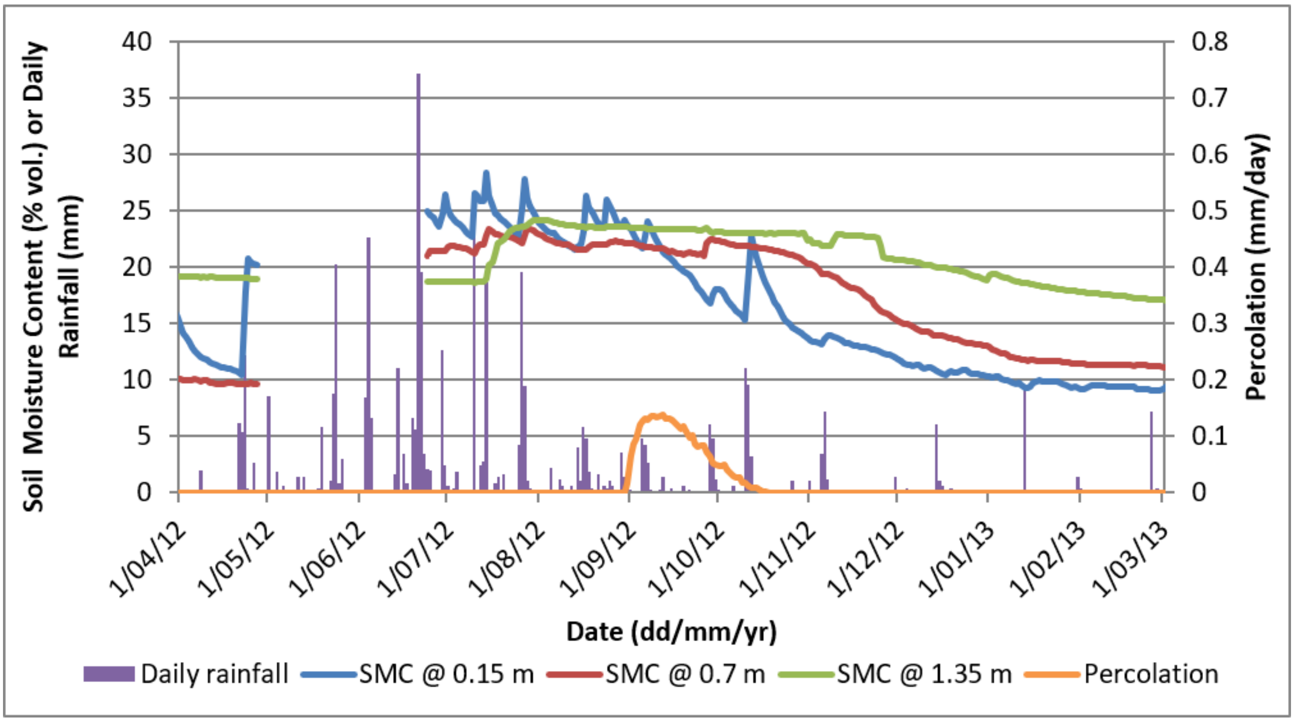


Figure 3 Soil moisture content (SMC) at three depths in the soil profile and measured daily rainfall and percolation in the phytocap from April 2012 to February 2013

The compacted clay barrier cover also reported three percolation events over the monitoring period (Figure 4). Lateral flow was also measured, which may report as percolation, depending on the design and integrity of the compacted clay barrier. Similar to the phytocap, percolation occurred in 2009, with a total of 1.5 mm; however, in this case the vegetation was not affected by herbicide. Notably, in 2009, the percolation commenced prior to lateral flow (which totalled 6.7 mm) suggesting desiccation cracks had formed in the clay layer. In 2010, the soil profile dried fully over summer with a total lateral flow of 1 mm and no percolation was recorded. The relatively slower rate of moisture increase in this year likely resulted in the clay layer rehydrating, and therefore reducing the extent of the desiccation cracks. Salt (2013) provides more details on this phenomenon.

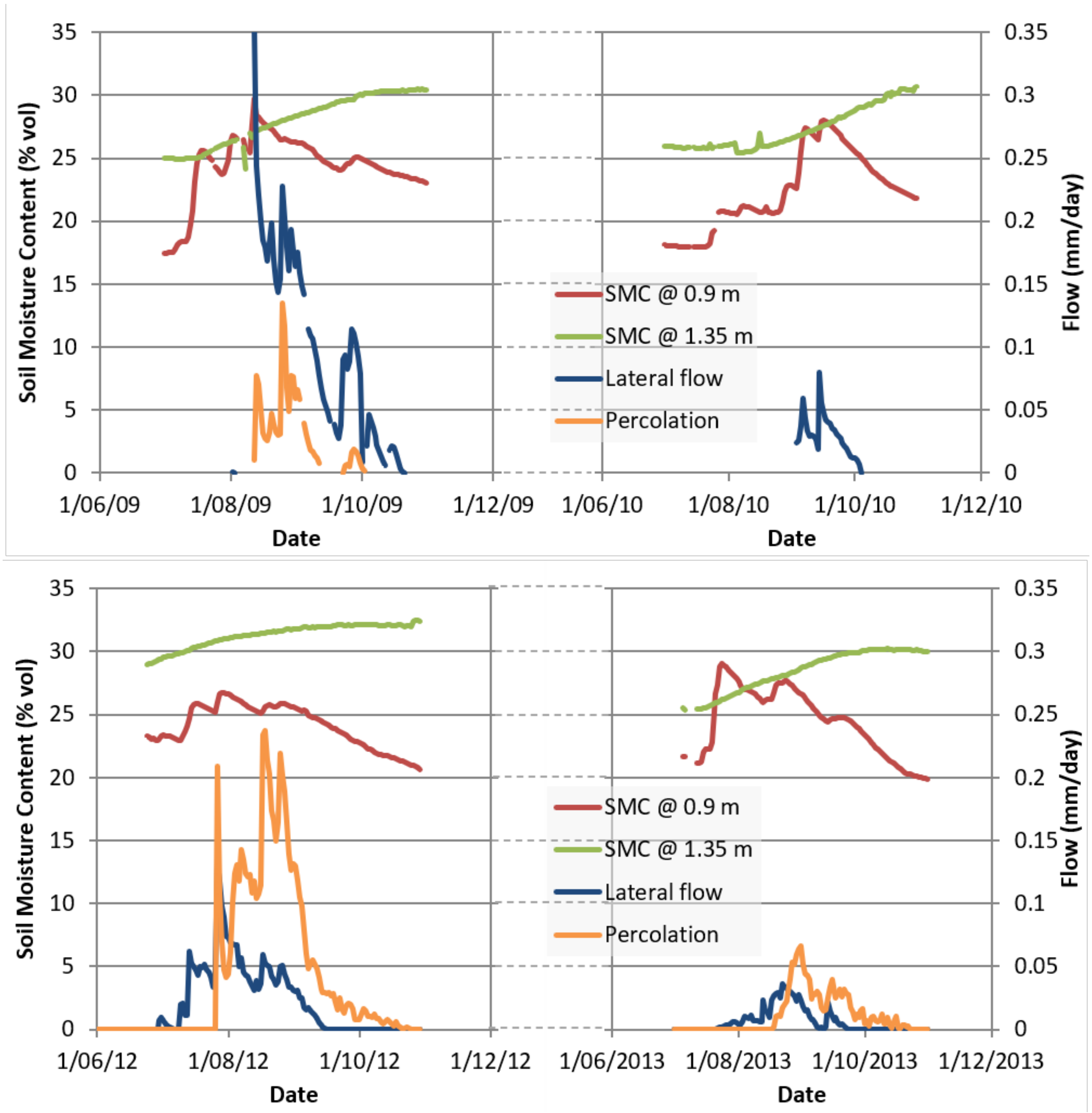


Figure 4 **Compacted clay barrier cover system soil moisture on the clay barrier surface (0.9 m depth) and within the barrier (1.35 m) and measured daily lateral flow and percolation**

Continued monitoring of the compacted clay barrier cover system has shown that the barriers performance has continued to deteriorate. In 2012, 6.4 mm of percolation and 2.9 mm lateral flow were measured from the compacted clay barrier cover system compared with 3.6 mm from the phytocap in 2012 (Figure 4). The 2012 data show the clay layer (measured by the sensor at 1.35 m depth) was wetter than in 2009 and 2010 but still transmitted more moisture through it than shed off the surface. In 2013, the compacted clay barrier cover system reported percolation of 1.4 mm, exceeding the lateral flow of 0.8 mm, and in a year when the phytocap reported no percolation.

Overall, long-term monitoring demonstrates that phytocaps can achieve equal or lower percolation rates than a cover system with a compacted clay barrier. Further, the compacted clay barrier performance was affected within 2 years of construction, even at 0.9 m below the surface and did not return to constructed permeability following rewetting. Although a clay layer can be beneficial in a capping system to store moisture, compaction to high standard, such as 95% maximum dry density , is only a very short-term benefit to percolation and this benefit should be weighed against the cost and practicality of compaction.

4 Vegetation selection

Vegetation selection is critical to the success of these types of systems. Without vegetation, even in arid and semi-arid environments, the surface is vulnerable to erosion and the moisture removal is limited to relatively shallow depths. For successful vegetation establishment, it is essential to consider soil properties, species selection and site constraints.

4.1 Soil properties

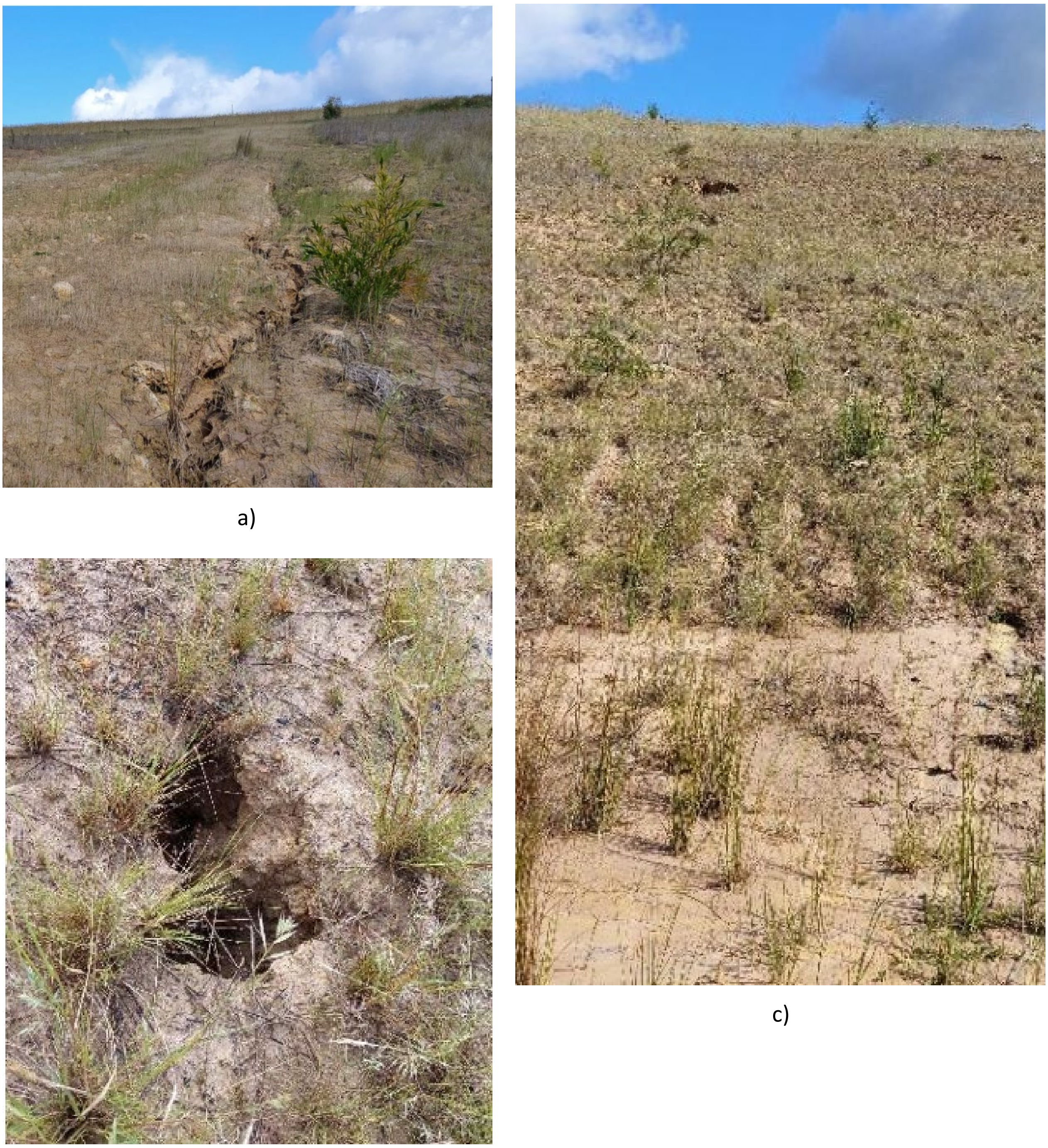
The soil properties need to be conducive to facilitate germination and to sustain vegetative growth and that is often species specific. The soil chemical and physical properties, particularly acidity, salinity and pore-space relations, have a major influence on the plant selection and successful establishment. However, different plant species have different requirements for germination and establishment so a broad range of soil properties must be considered. An obvious fact that we have learned the hard way.

A site located in the Gippsland region of Victoria sourced overburden from local quarry to use for final cover. The cover system design specified a 0.5 m sandy soil over 1 m clay soil with an organic mulch incorporated into the surface. The thick topsoil layer and applied organic mulch were specified to ameliorate the highly dispersive subsoil material available (Table 2).

Table 2 Soil properties as specified from material characterisation and materials placed in the final cover

Property	Specified topsoil (0.5 m thick)	Specified subsoil (1 m thick)	Material used on-site (1.5 m thick)
Sand content (%)	> 60	7–15	3–34
Clay content (%)	15–25	>25	34–59
Emerson Class	≥ 3	≥ 2	2
Exchangeable Sodium % (ESP)	< 6	–	Not tested
Total Organic Carbon (%)	0.6–3	–	Not tested

Construction did not follow the approved specification and instead the high clay subsoil was placed for the full 1.5 m depth and no organic mulch was incorporated. As a result, within two years the site was badly eroded with tunnel erosion evident throughout the site and vegetation struggled to establish and survive (Figure 5). Tubestock became waterlogged as soil moisture drained into the more porous potting mix. After 2 years, planted native grasses had died with only a few trees and shrubs survived. After 5 years and surface applications of gypsum and mulch, replanted trees and shrubs have grown and now average 1 m tall, and the replanted grasses and forbs cover approximately 10% of the ground surface. This case highlights the importance of adhering to material specifications and incorporating soil amelioration measures during construction.



a) b) c)

Figure 5 Erosion and poor vegetation establishment from lack of amelioration of poor soil properties. One year after construction the site showed a) gully erosion b) dissolution holes and c) tunnel erosion with collapsed tunnels upslope and 'spew' holes downslope.

At a site located near Bacchus Marsh, 50 km west of Melbourne Australia, a final cover system received a surface application of compost, rather than the organic mulch of chipped timber incorporated into previous areas. The surface application was recommended to reduce erosion of the sodic soil. The area was hydroseeded with mixed wallaby grass (*Rytidosperma spp*). However, 12 months later and no germination of wallaby grass was observed only weeds (Figure 6a). The compost was mature and analytical results in accordance with AS 4454 did not indicate any potential issues. Additional testing conducted 1 year after application also indicated no potential chemical issues with the compost substrate. Despite this, the wallaby grass did not establish. The lack of germination is most likely caused by lack of contact with the soil surface (Figure 6b). Wallaby grass seed in small and requires soil contact to anchor and access moisture. The combination of mature compost and hydroseeding may have suspended the wallaby grass seed away from the mineral soil and thus prevented it from germinating and/or establishing. Over the next year, the compost will be incorporated into the soil to provide a more suitable seedbed for small grass seeds. This illustrates that even chemically suitable amendments can be ineffective if physical seedbed requirements are not met.

(a) (b)

Figure 6 Vegetation established on two rehabilitated areas. (a) Surface compost hydroseeded with wallaby grass showing only weeds; (b) Surface organic mulch incorporated into soil and hydroseeded with wallaby grass showing good establishment of wallaby grass and few weeds

4.2 Species selection

Indigenous ecosystems should be considered in all environments as these plants are typically better suited to the available soil type and climate, and more likely to provide vegetative cover across the range of seasons and years. Even in semi-arid and arid climates the small xerophytic plants, mosses, lichens and fungi which form the biological soil crust provide valuable cover and nutrient cycling assisting larger opportunistic plants to germinate and establish once the seasonal rains arrive (Belnap 2001).

Indigenous vegetation typically provides the best performance outcomes for the cover system, as the species are naturally suited to the site's climatic conditions. However, if the site conditions have significantly changed, such as changes in elevation the area may be more exposed, or where the available soil differ from the indigenous soil, then alternative species from nearby vegetation communities may need to be considered.

At a rehabilitation site on the Bellarine Peninsula in Victoria, the landform had been raised above the surrounding levels and the batter slopes were exposed to strong, gusty and salt-laden winds. The initial vegetation plantings establishment was poor and within two years had decreased from 1.1 plants/m^2 to 0.3 plants/m^2. Although over 1,000 wallaby grass and 1,300 Lomandra tubestock were planted in the trial area in 2019, only 65 wallaby grass and 152 Lomandra were present after 2 years and reduced further by 2025 (Table 3). In late 2021, BRM Environments were contracted to manage the vegetation and noted that *Poa* grasses had established on the site and provided micro-climates that supported the establishment of other plants. Boobialla (*Myoporum insulare*) is often found growing in coastal areas and hence was selected and planted to provide a protection barrier for the tubestock of indigenous vegetation including wattles and Eucalypts (Figure 7).

Through a planting program in 2022, 2023 and 2024, the diversity of species was increased from 15 different indigenous species to 24 indigenous species (Table 3). In 2025, the density of grasses reduced slightly, although there was an increase in coverage from the trees and shrubs. In 2024, trees and shrubs covered 30.1% of the site which increased to 42.3% in 2025. Conversely grasses decreased from 20.2% to 19.2% in the same period.

As the vegetation community becomes more mature, it is expected the groundcovers will reduce further while the trees and shrubs coverage will increase but by a lower density. Over the same period, the bare ground decreased from 33% in 2024 to 12.9% in 2025 while leaf litter increased from 13.3% to 17.8%. The increase in leaf litter will assist in recycling nutrients into the soil and protecting the surface from erosion. By using knowledgeable contractors, the site has gone from poor-coverage with a struggling vegetation to a complex, sustainable community able to withstand the harshness of the elevated environment.

Figure 7 (a) Poa and Boobialla providing sheltered micro-climates for other indigenous vegetation to establish; (b) Stunted she-oak from original planting showing aggressive environment prior to planting protection species

Table 3 Plant density of individual species increased over time through planting and natural succession

Species		**Plant Density (no/15 m^2)**		
Scientific name	**Common name**	**2021**	**2024**	**2025**
TREES & SHRUBS				
Allocasuarina verticillata	Drooping Sheoak	0.205	0.355	0.593
Eucalyptus camaldulensis	River Red Gum	0.423	1.098	1.159
Eucalyptus ovata	Swamp Gum		0.586	0.430
Myoporum insulare	Common Boobialla		1.077	0.914
Acacia acinacea	Gold-dust Wattle		0.130	0.170
Acacia implexa	Lightwood	0.075	0.082	0.170
Acacia mearnsii	Black Wattle		0.409	0.368
Acacia melanoxylon	Blackwood		0.164	0.136
Acacia pycnantha	Golden Wattle		0.477	0.586
Atriplex semibaccata	Creeping Saltbush	0.150	0.055	0.041
Bursaria spinosa subsp. spinosa	Sweet Bursaria / Blackthorn	0.102	0.041	0.020
Dodonaea viscosa	Sticky Hop-bush	0.225	0.266	0.245
Einadia nutans	Nodding Saltbush	1.445	0.273	0.266
Enchylaena tomentosa	Ruby Saltbush		0.048	0.082
Threlkeldia diffusa	Bonefruit	0.218	0.041	0.123
TOTAL		2.843	5.1	5.305
GRASS & FORBS				

Species		Plant Density (no/15 m^2)		
Scientific name	**Common name**	**2021**	**2024**	**2025**
Poa labillardierei	Common Tussock-grass	1.514	7.357	6.955
Dianella revoluta	Black-anther Flax-lily		0.116	0.061
Dianella brevicaulis	Small-flowered Flax-lily	0.280	0.075	0.055
Ficinia nodosa	Knobby Club-rush	0.007	0.068	0.007
Lachnagrostis filiformis	Blowngrass		0.020	0.000
Lomandra longifolia	Spiny-headed Mat-rush	1.036	0.089	0.041
Ryditosperma caespitosa	Common Wallaby-grass	0.443	0.041	0.041
Ryditosperma racemosum	Slender Wallaby-grass		0.136	0.027
Themeda triandra	Kangaroo Grass	0.245		0.007
Clematis microphylla	Small-leaved Clematis	0.041	0.014	0.007
TOTAL		3.566	7.916	7.200

4.3 Site constraints

Site constraints will often revolve around the proposed post mining land use which may limit the selected vegetation community. Site constraints also include the properties of the underlying waste mass, slope, aspect or exposure and profile depth; with these constraints varying from site to site. One aspect often overlooked is knowledge or understanding of vegetation management and this can be critical in supporting the success of the planted vegetation community regardless of other constraints. The authors have found that this has been the largest constraint and impact on the time required to establish a functioning and sustainable cover system.

Lack of knowledge typically contributes to the wrong species being planted. For example, at one site it was too late to plant native vegetation for a successful outcome. Instead, it was advised planting a sterile ryecorn crop to protect the site from erosion, limit weeds from establishing and allow weed control prior to the optimum planting time. Unfortunately, the site owner, purchased ryegrass and not ryecorn (Figure 8). This was only discovered during a site visit and the identified seed heads were full and sterile. As the seeds had already formed, the planting of the specified vegetation was delayed for a further year.

Lack of support typically results in failure to adequately control the weeds in the two years following planting. Australian native vegetation is slow to establish and requires pro-active weed control to prevent competition. The indigenous vegetation community for each site includes a variety of species and hence weed control can be difficult as even selective herbicides may detrimentally affect some of the selected species.

These cases highlight how small decision can have disproportionate impacts on closure schedules and risk.

Figure 8 Ryegrass growing well; however sterile ryecorn was meant to be planted which resulted in a one year delay to planting the indigenous species

5 Conclusion

This paper demonstrates that phytocaps represent a robust, sustainable and technically defensible alternative to conventional compacted clay barrier covers for mine waste landforms. Field-scale monitoring has shown that, when appropriately designed and vegetated, phytocaps can achieve equal or lower net percolation than compacted clay systems, while avoiding the well-documented long-term degradation associated with desiccation cracking, root penetration and changes in hydraulic conductivity. Importantly, the Adelaide A-ACAP trial highlights the performance of compacted clay barriers can deteriorate within only a few years of construction, whereas phytocap performance is largely dependent on the persistence and health of the vegetation community.

The case studies presented reinforce that successful phytocap performance is not driven by soil water balance principles alone, but by the deliberate selection, establishment and management of vegetation suited to site-specific soils, climate and post-closure land use. Poor outcomes were consistently linked to inadequate soil characterisation, inappropriate material placement, unsuitable seedbed preparation, incorrect species selection, or insufficient post-establishment vegetation management. Conversely, where soil properties were matched to plant requirements and vegetation communities were adapted in response to site exposure and constraints, resilient and self-sustaining ecosystems were established, delivering erosion protection, improved surface stability and enhanced long-term cover system performance.

These findings highlight the need for mine closure planning to move beyond short-term constructability and compliance-driven designs toward integrated, vegetation-led systems that recognise ecological processes as core functional components of engineered landforms. Phytocaps, when treated as living systems rather than static structures, offer substantial advantages in terms of cost, constructability, adaptability and long-term risk reduction. The outcomes presented in this paper support the adoption of phytocaps as a preferred closure strategy for suitable mine waste landforms and emphasise the importance of interdisciplinary collaboration between geotechnical, soil and ecological practitioners to achieve enduring mine closure outcomes.

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