Designing authentic ecologies to address the biodiversity crisis: Grasslands as designed landscapes.

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Abstract

Temperate grasslands are among the most threatened biomes globally, with urbanisation and land-use change driving widespread degradation. In Australia, temperate grasslands represent ecologically distinctive systems shaped by long histories of Indigenous land management, yet today they persist only as small and isolated remnants following extensive agricultural conversion and urban expansion.

A key challenge in restoring temperate grasslands in Australia is overcoming the legacies of past land use - particularly persistent weed seedbanks and anthropogenically elevated soil nutrient levels - that reinforce degraded ecological states. State-and-transition models describe how ecosystems can shift between alternative stable states, emphasising that restoration is not about returning to a previous condition but about guiding ecological processes toward biodiverse, self-sustaining systems. To achieve this, interventions are required that disrupt the feedbacks maintaining degradation and promote trajectories in which native species can establish, compete, and persist. In severely altered post-agricultural and urban sites, passive recovery is rarely sufficient, requiring forms of ecological reconstruction that actively design and manipulate conditions to enable grasslands to establish and endure.

Using the Glenthorne Grassland Project in metropolitan Adelaide as a focal case study, this thesis develops and evaluates a suite of interventions to improve grassland restoration outcomes. These interventions draw on ecological evidence, local knowledge, and practitioner insight to test how different pre-sowing treatments and establishment strategies influence the assembly of native grassland communities.

Designing and restoring urban grasslands requires integrating ecological processes with social, economic, and governance factors, as outcomes depend on both site processes and the systems that sustain them. To reflect this complexity, the thesis adopts a socio-

ecological systems perspective, recognising that effective restoration must align ecological feasibility with stakeholder expectations, resource constraints, and long-term management structures. Each chapter therefore engages with a different combination of these considerations, using experiments and case studies to examine how they shape the design, establishment, and performance of restored grasslands.

Following Chapter 1, the introduction, Chapter 2 investigates soil solarisation as a pre-seeding weed control strategy, demonstrating that sunlight intensity is a stronger driver of soil heating than air temperature. These findings refine recommendations for timing solarisation treatments in Mediterranean climates, allowing practitioners to optimise their soil heating interventions based on predictable solar radiation rather than variable air temperature.

Chapter 3 presents a randomised field experiment evaluating the effects of solarisation and topsoil removal as restoration techniques, showing that removing nutrient-enriched topsoil significantly enhances native forb and grass diversity by reducing weed pressure and altering soil conditions.

Chapter 4 focuses on earthworks and builds on these findings by introducing a decision-making framework that balances biodiversity benefits with economic constraints and soil conservation considerations, guiding practitioners (including restoration ecologists and landscape architects) on when and how to implement topsoil removal based on site conditions.

To enable long-term monitoring of restoration outcomes, Chapter 5 applies Sentinel-2 EVI2 time-series data to assess grassland vegetation dynamics, demonstrating that peak-of-season spectral signatures reliably distinguish native from non-native grasslands. This research highlights how remote sensing can serve as a cost-effective tool for tracking restoration success over time, particularly in fragmented urban landscapes. Finally, Chapter 6

examines the role of values in shaping ecological restoration and landscape design, proposing the "evolving circle of landscape design" as a conceptual framework to guide practitioners in integrating ecological principles with shifting cultural and policy priorities.

By combining empirical research with decision-making tools and practitioner perspectives, this thesis advances a hierarchical framework for pre-seeding interventions, categorising restoration strategies into (1) biomass removal, (2) seed bank manipulation, and (3) soil reset. This framework positions restoration as an adaptive process, where interventions must be tailored to site conditions, financial constraints, and long-term ecological resilience. These insights contribute to an evidence base for both restoration ecology and landscape architecture, providing a practical and conceptual foundation for fostering resilient, biodiverse temperate grasslands in urban and post-agricultural landscapes. This thesis will therefore be of use to all those seeking to restore and repair Australian native temperate grasslands in urban areas with its mix of practical insights and theoretical guidance for practitioners across the restoration ecology and landscape architecture spectrum.

Declaration

I certify that this work contains no material that has been accepted for the award of any other degree or diploma in my name, in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. In addition, I certify that no part of this work will, in the future, be used in a submission in my name for any other degree or diploma in any university or other tertiary institution without the prior approval of the University of Adelaide and, where applicable, any partner institution responsible for the joint award of this degree.

The author acknowledges that copyright of published works contained within the thesis resides with the copyright holder(s) of those works. I also give permission for the digital version of my thesis to be made available on the web, via the University's digital research repository, the Library Search, and through web search engines, unless permission has been granted by the University to restrict access for a period of time.

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11/03/2026

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Chapter 1

GENERAL INTRODUCTION

1.1 Introduction

My journey into environmental science didn't begin in a laboratory or a lecture hall - it started in the garden. I started small, in high school with a single corrugated iron raised bed where I grew vegetables and flowers. Then, when I moved to Adelaide for Uni, my Pop gave me a set of gardening tools -an unexpected catalyst that led me to transform my shared house front and backyard into a garden. What began as a simple interest in growing plants soon deepened into a fascination with soil science. That passion led me to major in Soil Science and Ecology and complete an honours project on plant-soil interactions. After university, work took me to Melbourne, where I found myself in lockdown with plenty of time on my hands. With limited access to outdoor spaces, I decided to scratch an itch and explore landscape architecture - completing a graduate certificate at Deakin University.

An opportunity to work as a field ecologist for the Australian Terrestrial Ecosystem Research Network brought me back to Adelaide, where I was fortunate enough to work in the wilder, remote, more intact parts of nature that I had come to love through hiking and camping. Conducting fieldwork across Australia, I developed an enduring appreciation for our native landscapes that remain relatively untouched by human influence. But a question developed in me: how could these dynamics of nature be meaningfully incorporated into the landscapes we build?

This question stayed with me as I worked across diverse ecosystems, from arid zones to coastal heathlands. Everywhere I went, I saw the same pattern: urban and constructed landscapes often lacked the richness and biodiversity of the natural ecosystems I surveyed. The contrast between wilderness and city was stark, and I wondered: could we design landscapes that felt alive, that functioned as real ecosystems rather than static, decorative

spaces? This ultimately led me to my PhD in landscape architecture and restoration ecology. When I came across the work of Paul Gibson-Roy and saw his photos of restored grasslands, something clicked. His approach - removing topsoil and directly sowing an ecosystem - seemed radical in its simplicity. But as effective as it appeared, I realised there was still room to test, refine, and experiment with the idea.

With these thoughts in mind, and encouraged by Scott, I applied for a Green Adeliade grassroots grant with the motto “go big or go home.” We proposed the reconstruction of a 5 ha native temperate grassland in an urban post-agricultural national park. Our grant application was successful, and this PhD is the result of that project. From the point that the project was funded, the restoration work and my doctoral research became deeply intertwined. This thesis is built around that project, the Glenthorne Community Grassland Project. It became both a testbed for restoration techniques and a lived example of how ecological science and design thinking can be applied in real, constrained, and publicly visible landscapes.

1.2 Socioecological framework: Aligning restoration with society’s needs

Ecological restoration does not happen on its own - it is driven by both nature and people. To succeed, restoration projects need ecological goals that align with what communities value and need (IPBES, 2018, 2019). In cities, this means weaving native biodiversity into everyday places where people live, work, and play. Landscape architects and restoration practitioners are the ones designing and shaping these environments that are both ecologically sustainable and socially sensitive.

In many cases, understanding this alignment necessitates a socio-ecological systems perspective. This is an academic way of seeing any site, say an urban grassland, as a living system made up of four parts (Ostrom, 2009; Poteete, Janssen and Ostrom, 2010; McGinnis and Ostrom, 2014):

1. Resource Systems:
   The ecosystem itself: the soil, water, plants and animals all interacting.

2. Resource Units:
   The species and their functions: the grasses, wildflowers and other plants we restore, and what they do (support pollinators, stabilise soil, provide habitat).

3. Governance Systems:
   The rules and plans: the local council policies, environmental regulations and guidelines that shape what we can do.

4. Actors:
   The people: designers, practitioners, volunteers and neighbours who manage, use and look after the grassland over time.

This living-system view shows that success comes when all four parts move in sync: A healthy ecosystem; The right species and their functions; Supportive rules and policies; and Engaged people. This means that while landscape architects take on many roles, they also require a solid grounding in ecology - how soil, water, plants and animals interact. Likewise, restoration practitioners need to understand the human side: community values, land-use needs, policy settings and the multifunctional roles these grasslands play. When ecology and social context overlap, restoration really works.

Grasslands exemplify this socioecological connection. Historically, these ecosystems have been vital to agriculture, local cultures, and global biodiversity (Whyte, Moir and Cooper, 1959; Linder *et al.*, 2018). Yet today, grasslands (especially temperate grasslands) are among the most imperilled biomes, with less than 0.5% remaining intact in some regions (Kirkpatrick, McDougall and Hyde, 1995; White, Murray and Rohweder, 2000). Their ongoing fragmentation and degradation including in urban areas underscore the need for reconstruction approaches that account not only for ecological resilience but also for human uses, aesthetic preferences, and institutional support. By applying the socio-ecological systems framework, Landscape architects and restoration practitioners can better anticipate how design decisions, governance arrangements, and community participation converge to influence the success and sustainability of urban grassland restoration.

1.3 The biodiversity crisis and societal concern

Since the Industrial Revolution, human activities such as land clearing, fossil fuel combustion and industrial agriculture have accelerated biodiversity loss to rates comparable with Earth's past mass extinction events. Scientists now describe this period as the sixth mass extinction, with over one million species at risk (Ceballos *et al.*, 2015; IPBES, 2019). The last mass extinction - caused by an asteroid - wiped out the dinosaurs. This time, the cause is us.

Australia is home to an estimated 566,000 species (Chapman 2009), many of which occur nowhere else on earth. Despite this extraordinary biodiversity, Australia has one of the highest mammal extinction rates globally (Woinarski, Burbidge and Harrison, 2015) and ranks ninth worst worldwide on several environmental performance indicators. Unlike many regions that have already lost the bulk of their native ecosystems, Australia still has much to lose, and much at stake. Biodiversity loss is a major global problem. When compared with other global issues, it is one of the most urgent (IPBES 2019). Scientists have warned that we are exceeding the "planetary boundary" for biodiversity loss (Rockström *et al.*, 2009, 2023; Richardson *et al.*, 2023), with serious consequences for the ecosystems and species that underpin food systems, climate regulation and human wellbeing.

This crisis is deeply interconnected with climate change. Both biodiversity loss and global warming are consequences of unsustainable land use, fossil fuel dependency and ecological degradation - and both are progressing in parallel. As illustrated in Figure 1, while global average temperatures have increased steadily since 1970, global biodiversity, as measured by the Living Planet Index, has sharply declined (Richardson *et al.*, 2023). These reinforcing trends highlight the need for integrated approaches to climate mitigation and ecosystem restoration, rather than siloed responses.

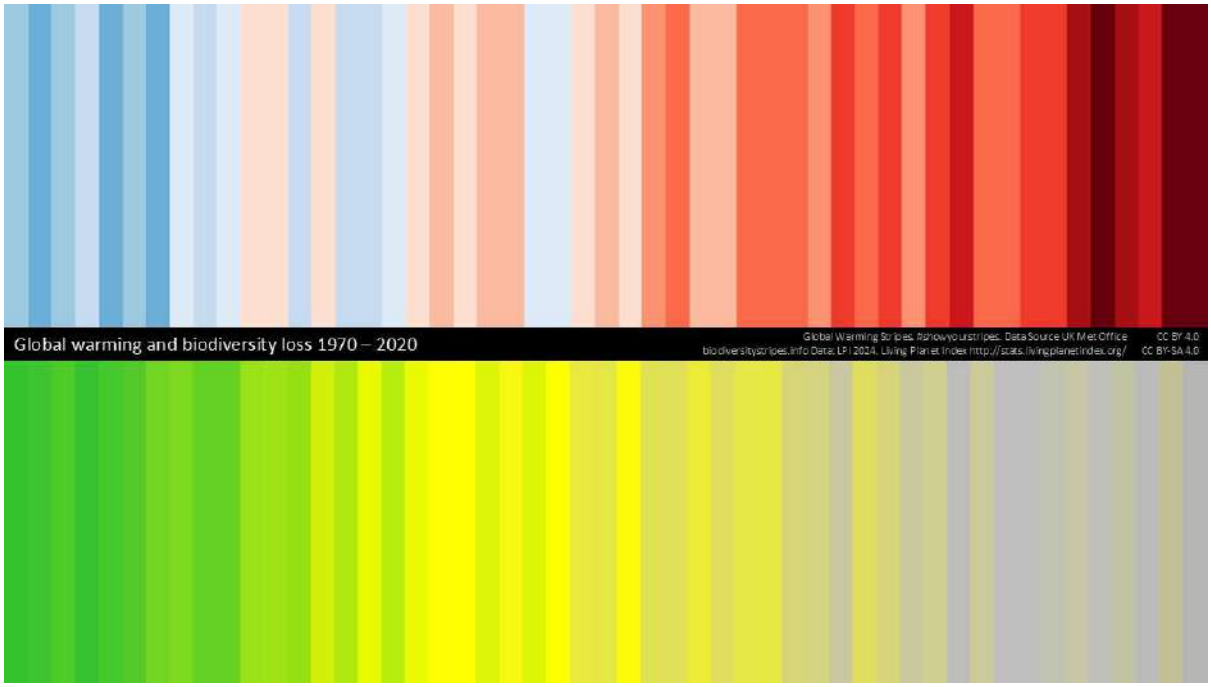

Figure 1. Parallel visualisation of global warming (top) and biodiversity loss (bottom), 1970–2020. Temperature data derived from the UK Met Office; biodiversity data based on the Living Planet Index, biodiversitystripes.info.

Enormous global initiatives have been proposed to combat ecological degradation, with ecosystem restoration and reconstruction emerging as an important tool to address the intertwined crises of biodiversity loss, climate change, and environmental degradation (Suding *et al.*, 2015), with a notable proposal being the Half Earth initiative – aimed at dedicating half of earth's land to conservation (Ellis and Mehrabi, 2019). International commitments, including the United Nations Decade on Ecosystem Restoration (2021-2030) (UNEP, 2021) underscore the urgency and seriousness of these efforts. In fact, an astonishing 750 million to 1 billion hectares worldwide are designated for restoration (Sewell, van der Esch and Löwenhardt, 2020), roughly equivalent to the combined land area of the United States of America and Mexico.

Despite the global nature of this crisis, the distribution of largely intact ecosystems - and the benefits they provide - remains uneven. Poor or Indigenous communities often rely directly on local ecosystems for subsistence and cultural practices (Alves and Rosa, 2007; Roe *et al.*, 2013; Fedele *et al.*, 2021). While policies to expand protected areas can bring tangible conservation gains, they may also impose high opportunity costs on groups whose

livelihoods depend on alternative land-use practices (Isbell *et al.*, 2017; Ellis and Mehrabi, 2019).

Although this thesis focuses on the restoration of temperate grasslands in an urban Australian context, it sits within a broader global conversation about equity, access, and the socio-ecological implications of large-scale restoration. In Australia, these dynamics include the ongoing exclusion of First Nations peoples from many conservation processes and the need to recognise and support Indigenous land management practices. This tension underscores the importance of integrating ecological objectives with social equity and economic realities in conservation strategies (Rockström *et al.*, 2023).

Furthermore, ecosystem services are often paramount in regions where populations are highly vulnerable, relying on local authority, knowledge, and leadership for effective stewardship (Obura *et al.*, 2021). However, broad-based community empowerment is also vital in more developed contexts. For instance, in South Australia, the Landscape Boards (including Green Adelaide) support local environmental stewardship through Grass Roots Grants, enabling communities to undertake restoration initiatives that balance ecological aims with social and economic considerations (Pascual *et al.*, 2021). This blend of policy support and on-ground engagement is crucial for truly comprehensive biodiversity strategies.

Public awareness has grown alongside these ecological realities, with most Australians (96%) believing more action is needed to protect nature and an overwhelming majority (95%) indicating they think more federal funding should go towards this (Biodiversity Council, 2025). Public awareness and support is leading to increased policy attention and funding for restoration (UNEP, 2021; WWF, 2022). No longer is it enough to preserve the few remaining intact ecosystems; many policy frameworks - including South Australia's new Biodiversity Act (Parliament of South Australia 2025) - now advocate the restoration and reconstruction of degraded landscapes. Landscape architects and restoration practitioners thus bear a heightened

responsibility to devise strategies that satisfy both ecological imperatives and societal expectations.

1.4 The role of landscape architects and restoration practitioners in biodiversity conservation

Landscape architects and restoration practitioners engage with biodiversity challenges in complementary ways. Over the course of the United Nations Decade on Ecosystem Restoration (2021–2030), emphasis on biodiversity and climate change has risen in professional discourse (Steiner *et al.*, 2019), prompting many landscape institutes to issue policy statements and guidelines linking these themes. A keyword analysis of Australian Landscape Architecture Magazine (Figure 2) illustrates this trend by showing increased references to “rewilding,” “restoration,” “ecology,” and, notably, “biodiversity” over the last decade.

This trend is not confined to Australia. For instance, the UK’s Environment Institute provides explicit guidance on biodiversity; Germany’s Bund Deutscher Landschaftsarchitekt:innen references the EU Biodiversity Strategy; and in Australia, the Institute of Landscape Architects recently published a “Biodiversity Positive Design” position statement (AILA 2023; European Commission 2020; Landscape Institute 2022). Meanwhile, professional bodies in the United States, the United Kingdom, Australia, Germany, and Singapore highlight the intertwined crisis of climate change as an urgent design challenge, reflecting the emerging perception of climate change as an emergency (UNDP, 2024).

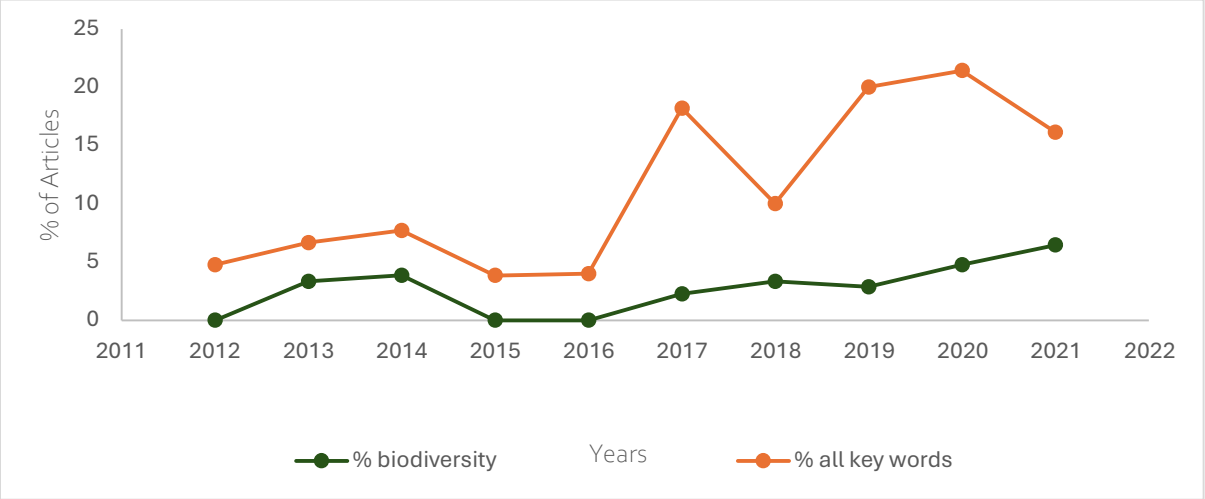

Figure 2: Keyword frequency analysis of *Australian Landscape Architecture Magazine*. Articles containing the terms 'rewilding', 'regeneration', 'restoration', 'conservation', and 'ecology' are graphed as a timeseries expressed as a percentage of total articles. 'Biodiversity' is tracked separately. The trend reflects a broader disciplinary shift toward ecological thinking within Australian landscape architecture practice. This analysis was developed during PhD candidature and benefited from discussions with Scott Hawken.

This thesis views design as a process of shared knowledge creation—where ecological function and cultural meaning are not separate goals but co-evolve through iterative experimentation. As Walsh and Allan (2023) argue, urban grasslands are not only ecological systems but 'settler commons', spaces that require ongoing engagement, cultural repair, and collaborative stewardship. Chapter 6 responds to this by exploring how adaptive design values and evolving frameworks can shape resilient grassland landscapes within living urban systems.

Landscape architects bring a design lens that explicitly integrates biodiversity outcomes with public use and amenity, translating ecological knowledge into built landscapes at multiple scales (ASLA 2025; IFLA Europe 2023; Ignatieva 2023) Restoration practitioners, working within the field of ecological restoration, focus on assisting the recovery of degraded ecosystems and the species and ecological processes that underpin them (Standards Reference Group SERA 2021) Yet both groups encounter critical, shared questions including:

- What type of biodiversity should be incorporated into designed landscapes?
- How do we create self-sustaining ecosystems that minimise ongoing intervention?

- How do we balance low maintenance needs with high biodiversity outcomes?
- How do we ensure long-term ecological stability and avoid domination by a few species?
- How can we create biodiverse plantings that do not require regular replanting?

Some of these questions can be considered “wicked problems” — complex, context-dependent challenges involving multiple stakeholders, conflicting values, and no definitive solutions. In landscape architecture, such problems are often approached through practice-based or practice-led inquiry, where knowledge is generated through direct engagement with real sites and their constraints. Activities such as earthworks, weed management, soil preparation, seeding, and ongoing maintenance are not merely applications of existing knowledge, but opportunities to produce new understanding when systematically documented and critically reflected upon.

Alongside this, landscape architecture has developed a substantial discourse on research-through-design (RtD), in which spatial design is used deliberately as an exploratory research strategy (Nijhuis & de Vries 2019). RtD requires explicit research questions, structured design investigations, and evaluative criteria that allow the design process itself to serve as a mode of inquiry. Reviews of RtD highlight, however, that such standards are met inconsistently, with “research” often used to describe routine design practice rather than rigorous, question-driven investigation (Lenzholzer et al. 2013).

This thesis is informed by that discourse but adopts a different methodological position. While the Glenthorne Community Grassland project required design work (e.g., visualising spatial layouts, selecting plant palettes, developing interpretation and communication content, planning pre-sowing interventions, and sequencing establishment), however, the design process was not the primary object of research. Instead, the site functioned as a real-world context in which ecological questions could be tested empirically. The core of the research centres on the ecological development of the grassland: how different

pre-sowing interventions influence species recruitment and weed suppression; how soil solarisation performs under field conditions; how species-rich seed mixes establish in nutrient-enriched soils; and how site-level decisions are shaped by practical, organisational, and community considerations.

In this sense, the thesis combines practice-based inquiry with empirical ecological methods. Design operations generated spatial and ecological propositions - for example regarding topsoil removal, solarisation, or vegetation arrangement - but these propositions were evaluated through replicated field experiments, quantitative vegetation surveys, and remote-sensing-based monitoring of vegetation dynamics (Lenzholzer et al. 2013; Nijhuis & de Vries 2019; Steffen Nijhuis 2011). The work therefore occupies a hybrid position: embedded in the practice traditions of landscape architecture, yet grounded in the systematic observation, hypothesis-testing, and evidentiary expectations of restoration ecology.

Wicked problems also characterise grassland restoration in urban contexts such as the Adelaide Plains, where native temperate grasslands once dominated much of the landscape (Hyde, 1995; Kraehenbuehl, 1996; Gammage, 2011; Shackley, Allanson and Kuys, 2015). Today, most remnants have been lost or severely degraded, largely due to agriculture and urban expansion (Bradshaw, 2018). Against this backdrop, practitioners must combine science-based techniques - like topsoil removal to reduce exotic seedbanks (Gibson-Roy and McDonald, 2014) - with context-sensitive design to create landscapes that function ecologically and resonate with local communities. For instance, Ignatieva and Hedblom (2018) advocate for replacing resource-intensive lawns with ecologically rich grassland typologies, including naturalistic meadows and spontaneous urban vegetation, reframing such spaces as legitimate and desirable urban landscapes. These ideas parallel local efforts to create biodiverse grasslands that resonate with community values (Marshall, 2013; Walsh and Allan, 2023).

Urban grassland design can benefit from frameworks like Biodiversity Sensitive Urban Design (BSUD), which embed ecological principles; such as habitat creation, dispersal potential, and long-term stewardship; into the design and planning process (Garrard *et al.*, 2017; Kirk *et al.*, 2021). By identifying species needs early and aligning spatial strategies with biodiversity goals, BSUD offers a structured pathway for ensuring ecological value in dense, fragmented urban landscapes. This aligns closely with Objective 5 of this thesis, which investigates how changing design values and frameworks can incorporate ecological rigour without sacrificing creativity.

Landscape architects are uniquely placed to act as 'boundary spanners'- professionals who operate across disciplinary, political, and cultural lines to connect ecological insight with spatial imagination (Moosavi et al., 2025). Their role is not only to shape space but to translate between knowledge systems, question normative assumptions in urban development, and foster the political and cultural conditions needed for climate-resilient and biodiverse cities. This remit is increasingly significant as urban grasslands emerge as a distinct planting typology globally and in Australia, where their establishment faces particular challenges: nutrient-enriched post-agricultural soils, persistent exotic seedbanks, water-limited growing conditions, and public expectations shaped by lawn-dominated landscapes. This research responds directly to these challenges by testing science-informed restoration methods that landscape architects can use to establish biodiverse, self-sustaining grassland systems in complex urban contexts.

1.5 Why science matters in landscape design

Restoration is not simply about reintroducing species; it involves rebuilding ecosystems that are functional, resilient, and capable of sustaining biodiversity. In urban grassland restoration, this is especially complex. These sites are often degraded, compacted, or nutrient-enriched due to agriculture, construction, or historic land use. Invasive species dominate, and original plant communities have been lost or fragmented. Natural processes like fire or grazing are

absent or incompatible with urban land uses. Climate change adds further uncertainty, making it harder to predict which species or strategies will persist over time.

Grasslands, once widespread, are now among the most threatened ecosystems, facing degradation from agriculture, urbanisation, and invasive species (White, Murray and Rohweder, 2000; Gossner *et al.*, 2016). This thesis responds to key knowledge gaps that limit urban grassland restoration in Australia, including:

- How to restore biodiverse vegetation on degraded, nutrient-rich soils;
- How to suppress invasive exotic species without long-term chemical inputs;
- How to rebuild native forb and grass diversity at scale;
- How to optimise pre-seeding interventions like soil solarisation and topsoil removal in urban settings;
- How to develop evidence-based tools for monitoring small urban restoration sites;
- How to integrate design values and ecological rigour in a way that supports long-term maintenance and community engagement.

One primary constraint in restoration is altered soil states, often enriched with nutrients from agricultural practices. This nutrient enrichment favours invasive species, which outcompete native plants (Ba and Facelli, 2022).

Techniques like topsoil removal are well-established for reducing nutrient levels and weed seedbanks (Gibson-Roy and McDonald, 2014), but their use in urban contexts is underexplored and key questions remain. These include how deep soil removal must be to achieve sufficient weed control, how subsoil chemistry and structure affect re-establishment, and what trade-offs exist between biodiversity outcomes and erosion risk (Török *et al.*, 2011).

Soil solarisation (a lower-impact method using plastic sheeting to heat and sterilise surface soils) has shown promise in agriculture, but its mechanisms and restoration potential

are not well understood, particularly under Australian conditions. Open questions include whether solar radiation or air temperature drives soil heating, how seasonal and site conditions influence outcomes, and how different seed mixes interact with treatment history to shape biodiversity and native cover.

The science of restoration also helps address site-specific conditions. State-and-Transition Models (STM) offer a framework for understanding ecosystem shifts and guiding interventions (Westoby, Walker and Noy-Meir, 1989; Briske, 2017; Sinclair *et al.*, 2019). These models show that restoration is not about returning to a previous state but about guiding ecological processes towards self-sustaining systems – which can be novel (Collier and Devitt, 2016; Higgs, 2017; Clement and Standish, 2018; Teixeira and Fernandes, 2020; Teixeira, Fernandes and Ahern, 2021); although this is also criticised (Murcia *et al.*, 2014).

In the context of temperate grassland restoration in south-eastern Australia, STMs provide a practical lens for interpreting why degraded post-agricultural sites resist passive recovery and what level of intervention is required to shift them toward native-dominated states. Where nutrient enrichment and exotic seedbank pressure have pushed a system into an alternative stable state, low-intensity interventions such as herbicide application alone are often insufficient to cross the transition threshold - the system simply reverts. Gibson-Roy and colleagues have applied this thinking directly to grassy community restoration in Victoria and the Cumberland Plain, demonstrating that interventions which address both the seedbank and soil nutrient legacy - particularly topsoil removal and direct seeding - are often necessary to achieve a lasting state change (Gibson-Roy et al. 2010; Gibson-Roy 2022; Gibson-Roy 2023). Their work built on foundational research by CSIRO which established the germination requirements and horticultural potential of many native grass and forb species that are now central to restoration practice, including work on dormancy breaking and establishment conditions for key grasses (Hagon, Groves and Chan, 1975; Hagon, 1976; Hagon, 1977) and assessments of forb species for urban landscape use (Hitchmough, Berkeley and Cross, 1989; Hitchmough, 1994; Hitchmough et al., 1996). This thesis tests those approaches in the

Adelaide Plains context, where similar degradation pressures operate on Kaurna Country soils, and examines whether the transitions documented in Victorian and New South Wales systems are replicable under South Australian conditions.

Grasslands worldwide face similar challenges, though the specifics vary by region. In Australia, soils that are relatively infertile by global standards - reflecting the continent's ancient, weathered geology and low phosphorus cycling (Lambers et al., 2011; McLaughlin, 1996) - once supported native plants adapted to drought and fire. In the temperate grassland zone, however, these systems often occurred on some of the most agriculturally productive soils on the continent, which is precisely why they were preferentially cleared for cropping and pasture (Kirkpatrick, McDougall and Hyde, 1995; Gott and Williams, 2015). In North America and Europe, invasive species and nutrient over-enrichment from agriculture also disrupt native grasslands (Bakker *et al.*, 2012). These changes complicate restoration efforts, as ecosystems are now dominated by flora that thrives in altered conditions. As global restoration efforts expand, science is crucial to developing solutions that are both effective and regionally tailored (UNEP, 2021).

There are additional challenges in urban settings. Small site size, high edge-to-area ratios, altered hydrology, short funding timelines, and high public scrutiny all shape outcomes. This thesis draws on scientific insights to develop a practical design framework (outlined in the emerging conclusions of the General Discussion), which introduces a continuum of pre-seeding interventions:

(1) **biomass removal** (which targets aboveground weed dominance),

(2) **seed-bank manipulation** (which reduces exotic seed reserves without disturbing soil structure), and

(3) **soil reset** (which removes or inverts degraded topsoil entirely).

These three categories are defined and tested throughout the thesis and form a conceptual scaffold for linking ecological science with site-specific design decisions. The framework enables landscape architects and planners to scale restoration ambition to context, aligning intervention intensity with ecological goals and budget constraints.

1.6 How science informs grassland restoration and management

1.6.1 Designing and maintaining self-sustaining urban grassland systems

Current urban planting often features native species without necessarily ensuring conditions for long-term self-organisation (Clements, 1916; Hobbs and Norton, 1996) and ecological resistance to climate change (Isbell *et al.*, 2015). For instance, native grasslands depend on disturbance regimes - historically grazing or fire - that frequently conflict with urban design norms or perceptions of 'tidiness' (Nassauer, 1988, 1995; Reseigh, Foster and Myers, 2009; Li and Nassauer, 2020). Meanwhile, the European 'New Perennialist' movement led by Piet Oudolf (Oudolf, 2019) borrows from the visual and spatial characteristics of meadow-like grasslands but relies on horticulturally intensive planting schemes that require ongoing maintenance to sustain their designed structure. Within landscape architecture, naturalistic perennial plantings are widely discussed as carefully composed vegetation that achieves particular aesthetic and seasonal effects through design and management, rather than as self-organising grassland ecosystems in the ecological sense (Dunnett & Hitchmough 2004). This distinction is particularly relevant in Australia, where nutrient-enriched soils, invasive grasses, and water constraints make ongoing horticultural maintenance difficult to justify at scale.

In contrast, restoration ecology seeks to initiate systems capable of adapting to stochastic events once intensive maintenance ceases, but it does not in itself address how such systems should be shaped aesthetically or experienced by people (Kennedy, pers. comm.; Gibson-Roy, pers. comm.). Practitioners note that conventional planting can degrade rapidly if management funds or labour are interrupted, whereas an 'ecosystem' that balances species diversity with functional resilience can maintain itself over time. However, policy and funding frameworks - often driven by narrow metrics or carbon-centric incentives - can marginalise more biodiverse approaches like grassland restoration.

Bridging these perspectives, research-based interventions - such as timed mowing (Van Dyke *et al.*, 2004; Piqueray *et al.*, 2019; Reinermann *et al.*, 2022), targeted soil

interventions (Gibson-Roy, Moore and Delpratt, 2010), as well as selective seed provenance (Gellie *et al.*, 2016) and soil-appropriate seed mixes (Barr, Jonas and Paschke, 2017; Freitag *et al.*, 2021; Hulting *et al.*, 2024) - have the potential to create landscapes that are both aesthetically appealing and ecologically robust, with lower long-term maintenance needs (Gibson-Roy and Carland, 2022).

This thesis develops ecological restoration techniques that can be applied and adapted by landscape architects, horticulturists, and restoration practitioners working in urban settings. From landscape architecture, it draws on principles of adaptive design, spatial composition, and sense of place, which frame how interventions such as soil reset or solarisation might support not only ecological integrity but culturally resonant forms. From ecology, it incorporates concepts of disturbance, successional dynamics, and long-term self-organisation that extend beyond static planting models. These priorities can diverge: as Kiers et al. (2022) argue in their analysis of urban lawn conversions, aesthetic expectations and recreational programming often conflict with ecological rigour. Yet they also note - echoing the approach taken here - that both disciplines contribute essential but distinct forms of knowledge to urban landscape transformation.

Rather than positioning the two fields as opposites, this thesis treats them as complementary: design provides the contextual intelligence and cultural framing needed for urban application, while ecology provides the process-based understanding required for systems to persist over time. By combining methods from both, the research develops a tested, site-responsive framework that neither discipline could produce alone: a continuum of interventions grounded in ecological science but configured for design application in complex urban settings.

1.6.2 Grasslands as a model for urban biodiversity

Grasslands provide an ideal model for urban restoration because they embody the intrinsic qualities of authentic ecologies while addressing practical urban challenges. They support high species diversity (encompassing a range of forbs and grasses adapted to low-nutrient soils and drought conditions (Eddy, 2002)) which underpins their capacity for self-regulation and resilience. In addition, grasslands offer aesthetic and recreational benefits that enhance urban liveability, creating open spaces that promote both environmental engagement and community well-being.

Critically, temperate grasslands are among the most endangered ecosystems (Kirkpatrick, McDougall and Hyde, 1995). In Australia, extensive conversion of native grasslands and grassy woodlands to agricultural landscapes has significantly reduced these ecosystems to the brink of existence (Paton *et al.*, 2000). Urban restoration projects that reintroduce native grasslands thus present a transformative opportunity; not only for biodiversity conservation and ecological function, but also for re-establishing these lost landscapes in ways that are visible, educational, and culturally resonant (Marshall, 2013; Walsh and Allan, 2023).

Another advantage of grasslands in urban settings is that their low biomass mitigates urban fire risks, a crucial consideration in densely built environments, while their inherent scalability - from mere square metres to large urban parks - ensures that restoration can be tailored to a variety of spatial contexts.

In this light, grasslands represent authentic ecologies in that they reflect the natural evolutionary processes and historical conditions of a region. Their restoration is not simply a technical intervention but also an act of cultural reclamation. Academic perspectives on sense of place emphasise that authentic, place-based ecosystems are vital for nurturing community attachment and reinforcing local identity. By restoring native grasslands within urban

contexts, we are not only reintroducing ecological complexity and resilience but also reviving a tangible connection to the landscape's historical and cultural narrative.

Thus, urban grassland restoration becomes a dual strategy: addressing the global biodiversity crisis through science-driven methods while simultaneously cultivating design-based solutions that enrich the urban sense of place. This approach ensures that the ecological and cultural dimensions of urban landscapes are integrally aligned, fostering both environmental sustainability and community well-being.

In response to these challenges, this thesis proposes a site-responsive framework that enables practitioners to balance visual ambition, biodiversity goals, and resource limitations across diverse urban contexts.

Australian grasslands are highly diverse, often containing a wide mix of native wildflowers and grasses. This diversity reflects the continent's ancient, nutrient-poor soils by global standards (Beadle, 1962; Groves, Whalley and Mallett, 2002; Morgan *et al.*, 2017), long dry seasons, and unique evolutionary history shaped by isolation (Lambers *et al.*, 2011). Many native plants are adapted to low phosphorus availability, seasonal drought, and intermittent disturbance from fire or grazing (Lunt *et al.*, 2006; Parkhurst, Standish and Prober, 2022). These adaptations align with what is commonly described as stress-tolerant strategies - traits that favour persistence in low-nutrient, drought-prone environments, but not rapid growth or competitive dominance (Grime, 1977). This leaves many native species well suited to stable, nutrient-poor environments, but poorly suited to the enriched, compacted, and disturbed soils common in cities and former farmland.

Unlike grasslands in Europe or North America, where native species often recover naturally after disturbance, Australian natives rarely regenerate on their own. In many northern hemisphere systems, early colonising plants thrive in disturbed sites because they are adapted to grow quickly and take advantage of change (Grime 1977). While passive regeneration is often fruitful in the northern hemisphere, grassland restoration in Australia

usually requires intensive intervention: reducing nutrients, suppressing weeds, removing topsoil, and reintroducing plants by direct seeding. These barriers are especially strong in urban areas, where surrounding landscapes offer little support for natural regeneration (Hobbs and Norton, 1996; Hahs *et al.*, 2009; Williams, Marshall and Morgan, 2015); as a result, in modified landscapes, Australian native vegetation rarely regenerates on its own. In contrast, in places like Germany, native meadows can return with minimal help (Zerbe *et al.*, 2003; Hedberg and Kotowski, 2010; Fischer *et al.*, 2013; Kowarik and von der Lippe, 2018; Freitag *et al.*, 2021).

Policy and governance frameworks also shape what's possible in restoration. In countries like Germany or the Netherlands, grassland restoration is often supported by clear biodiversity targets, financial incentives, and integrated planning at regional scales (Ignatieva *et al.*, 2020). By contrast, Australian grassland projects must often navigate fragmented governance, ad-hoc funding, and weak policy protections, especially outside formal reserves. This makes restoration more dependent on local champions, pilot projects, and short-term grants (Garrard *et al.*, 2017). Urban contexts add further complexity due to tenure constraints, development pressure, and limited public awareness of grassland value.

Much of the current practice in Australian grassland restoration draws on the work of Gibson-Roy and colleagues, who demonstrated that direct seeding and topsoil removal can create complex grassland communities (Gibson-Roy and McDonald, 2014; Cuneo *et al.*, 2018; Gibson-Roy and Carland, 2022). However, these methods were developed in agricultural landscapes. Urban sites present different challenges of higher weed pressure, more fragmented land, and different public expectations. This thesis builds on that work, adapting it for urban conditions and testing when and where more intensive interventions are worthwhile.

Native grasslands are beginning to gain traction in Australian landscape design, with growing recognition of their intrinsic, cultural, and aesthetic value (Marshall 2013; Walsh &

Allan 2023). They also offer low water use, strong seasonal variation, and support high biodiversity. These traits make native grasslands a powerful nature-based solution—addressing climate adaptation, biodiversity loss, and community liveability in tandem. More than just a planting option, they challenge conventional ideas of what urban nature should look like (Nassauer, 1988; Walliss, 2012; Tribot, Deter and Mouquet, 2018). As design elements, they bring ecological function together with new ways of thinking about care, authorship, and place-based identity (Marshall, 2013; Walsh and Allan, 2023; Moosavi *et al.*, 2025).

There are a small number of precedents in which landscape architects have delivered grassland projects. Wootten Road Reserve in Wyndham, Victoria, integrates restored grasslands into a new residential development, combining ecological value with public amenity and design. It has been critically acclaimed, winning the 2020 AILA Victoria Landscape Architecture Award for Parks and Open Space (Ricardo, 2018; GLAS, n.d.). Another project, Evans Street Grassland in Sunbury, Victoria, demonstrates how a remnant grassland of state significance can be successfully protected and presented within an urban context, supported by strong community involvement and thoughtful landscape infrastructure (Marshall, 2013). These examples show that native grasslands can play a meaningful role in urban design, but such projects remain the exception, and many questions remain. This thesis contributes to that gap, providing data, framing, and methods to better integrate grasslands into design practice.



Figure 3: Examples of urban grasslands in Australian practice. Top: Wootten Road Reserve, Wyndham, Victoria, a restored grassland integrated within a residential development. Bottom: Evans Street Grassland, Sunbury, Victoria, a remnant temperate grassland of state significance conserved within an urban setting (GLAS, n.d.; Marshall, 2013).

1.7 Conceptual framework: Systems thinking in restoration design

I argue that addressing ecological restoration in urban areas requires a systems-thinking approach that recognises the dynamic interplay between social values, design choices, and ecological processes. This thesis integrates restoration science, applied field research and design philosophy to build a framework in which each element reinforces the others. The following sections illustrate how the thesis chapters, on topics ranging from soil solarisation (a method that uses clear plastic to trap solar heat and sterilise the topsoil) experiments to the use of remote sensing for project monitoring, together form an evidence-based foundation for grassland restoration in complex urban environments.

1.7.1 The Glenthorne Grassland Project: A Real-World Case Study

This thesis is built around a real-world urban restoration project that I led alongside my PhD: the Glenthorne Community Grassland Project (https://www.glenthornegrassland.com/). Undertaken between November 2022 and February 2025, the project restored five hectares of cleared, post-agricultural land within Glenthorne National Park in Adelaide's southern suburbs. Funded by a Green Adelaide Grassroots Grant, with additional support from Marion Council, the project brought together multiple stakeholders, including: Seeding Natives, Nature Play SA, Friends of Glenthorne, Glenthorne Park Rangers, and researchers and students from the University of Adelaide and Flinders University.

This was not a hypothetical design or a detached monitoring study. It was a live, high-stakes, publicly visible restoration project that I initiated, designed, and managed, working across government, community, and academic sectors. The practical responsibilities: budgeting, stakeholder coordination, procurement, contracting, scheduling, documentation, and public engagement; were carried out alongside the conceptual and ecological work of restoration design. As such, this project was not only central to the thesis, it was also a professional undertaking in its own right.

The site reflects many of the ecological and institutional constraints that shape grassland restoration in urban contexts. Following prolonged agricultural use, vegetation was dominated by exotic pasture grasses and soils exhibited elevated nutrient concentrations, with native species largely absent at the commencement of the project. The site is located within Glenthorne National Park–Ityamaiitpinna Yarta, a recently established protected area that had only just opened to public access at the outset of the study. Situated on Adelaide's southern urban fringe, the park forms part of a broader regional initiative to restore, connect, and protect open space in a rapidly urbanising landscape.

Early colonial drawings and oral history (pers. comm. Richard Woods) indicate that the site historically supported Grey Box–derived grassy woodland and was maintained through cultural burning by the Kaurna people. In its current condition, the site occupies a contested urban setting where contemporary restoration efforts intersect with ongoing urban expansion, the recognition of Indigenous land management histories, and attempts to repair ecologically degraded systems through active intervention.

The restoration approach tested three separate pre-seeding methods: topsoil removal, solarisation, and chemical fallow; each applied to different areas across the site to evaluate their effectiveness in suppressing weeds and enabling native species establishment. These treatments were followed by the direct seeding of 68 native species, selected for their local provenance, functional diversity, and suitability to the site's altered conditions. The project also incorporated education and engagement components, with local school students and community volunteers involved in sowing, monitoring, and interpretation. Conducted within an active national park, the project required ongoing coordination with land managers, contractors, and policy frameworks.

The project also served as a direct application of the socio-ecological systems framework introduced earlier in this chapter. The Resource System (the degraded grassland ecosystem) was reworked through disturbance and reassembly. The Resource Units (the sown

species) were selected for ecological function and adaptability. The Governance Systems (grant funding, council policies, maintenance protocols) shaped what could be done, by whom, and when. And the Actors (including myself, contractors, community groups, land managers, and educators) were central to how the work was carried out and sustained. These components interacted constantly, often in tension, and it was within these frictions that the research questions of this thesis emerged.

While Glenthorne's scale and design support set it apart from many urban restorations, its constraints - nutrient-enriched soils, fragmented land tenure, and high public visibility - are typical of many peri-urban sites in Australia. Unlike in parts of Europe where restoration targets are often supported by national biodiversity frameworks (e.g. Germany's "Bundesprogramm Biologische Vielfalt"), Australian restoration operates within a fragmented governance landscape. State and local government responsibilities often diverge, funding is short-term, and biodiversity targets remain largely non-binding. In this context, Glenthorne offers a model not only for site design and implementation but also for navigating the institutional and social conditions that constrain urban grassland success.

Glenthorne can also be situated within the broader global trajectory of anthropogenic land use, as described by Ellis (2020) in his taxonomy of the human biosphere. This framework traces a progression from wildlands and lightly used ecosystems (foraging, fire regimes), through grazing and shifting cultivation, to continuous agriculture and ultimately to suburban and urban anthromes. Glenthorne occupies a late-stage position in this sequence: a post-agricultural, peri-urban landscape subject to soil degradation, seedbank loss, and long-term ecological fragmentation. Ellis also emphasises that human settlement patterns tend to follow ecological gradients, transforming the most fertile and accessible lands first. The Adelaide Plains reflect this pattern (Figure 4), having been rapidly converted from diverse native grasslands into farmland and subsequently urbanised. Restoration at Glenthorne thus represents an effort to re-engage with deeply altered ecosystems where original structure and species composition have largely been erased.

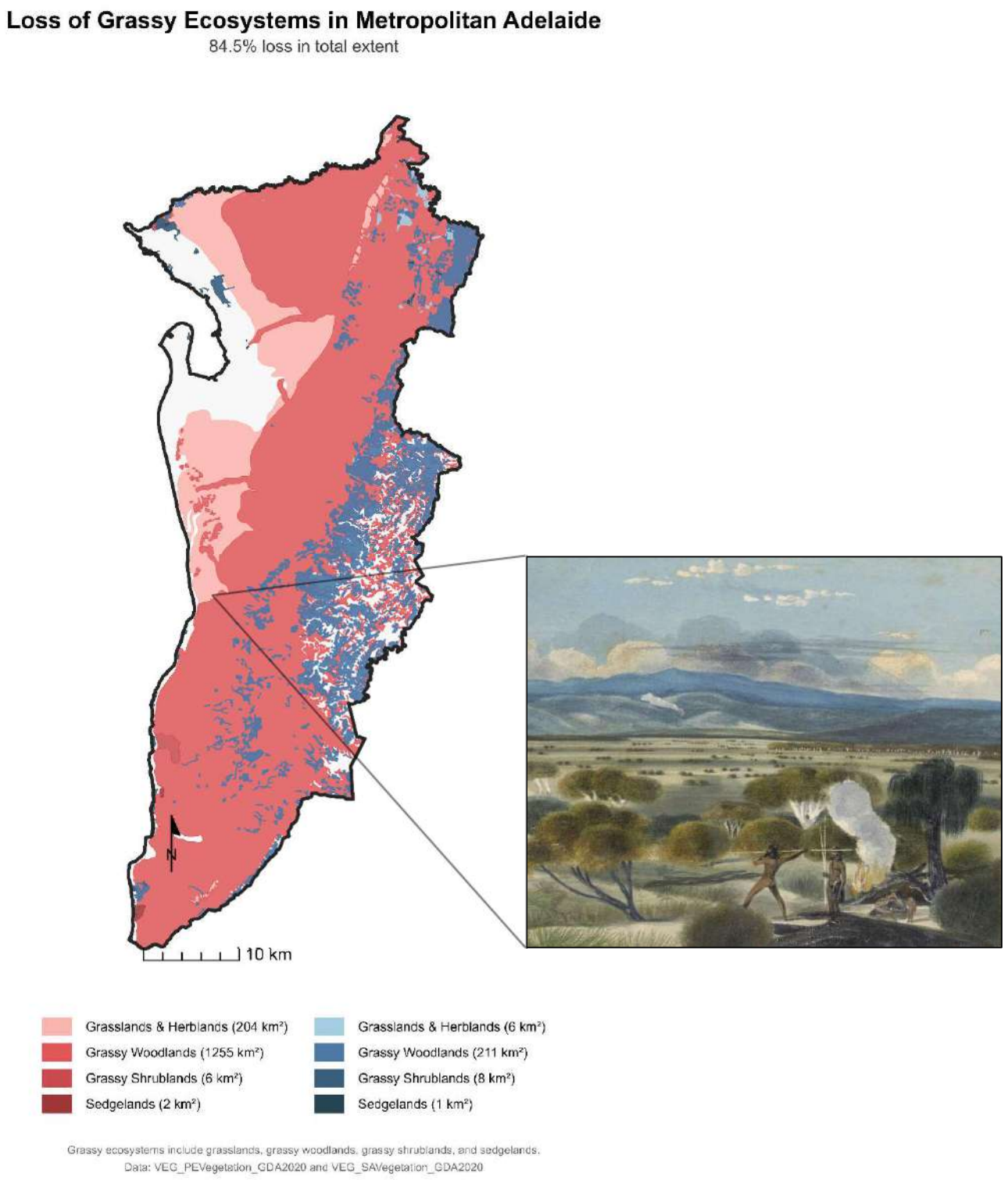

Figure 4: Pre-European Ecosystems of Adelaide. Map showing the extent of grassy ecosystems across the Adelaide region before European colonisation and their contemporary distribution. **Red** indicates the mapped extent of pre-European grassy ecosystems (including grasslands, grassy woodlands, grassy shrublands, and sedgelands). **Blue** indicates the remaining extant patches of these ecosystems in the present day. These layers highlight the substantial loss and fragmentation of grassy ecosystems since colonisation. Data sources: VEG_PEVegetation_GDA2020 and VEG_SAVegetation_GDA2020. View on the Glenelg plains, near the hills, South Australia, Skipper, John Michael. (1837). from http://nla.gov.au/nla.obj-135277269;

In sum, the Glenthorne Community Grassland Project (Figure 5) was the foundation of this research. It offered a rare opportunity to lead a large-scale urban grassland restoration from the ground up, with the dual perspective of practitioner and researcher. It exposed the logistical, ecological, social, and political realities of attempting restoration in public, altered, and constrained landscapes. And it allowed this thesis to remain rooted not only in theory, but in practice: in the messy, negotiated, hopeful work of putting native grasslands back into cities.



Figure 5: Aerial view of the Glenthorne Community Grassland Project site following initial topsoil removal and seeding, looking north-west toward the Gulf St Vincent. Drone photograph by M McQuillan, 2022.

1.8 Research aims, objectives, and questions

1.8.1 Overall research question

The core of this research examines the ecological development of a restored urban grassland: how different pre-sowing interventions influence species recruitment and weed suppression;

how soil solarisation performs under field conditions; how species-rich seed mixes establish in nutrient-enriched soils; and how site-level decisions are shaped by practical, organisational, and ecological considerations. These questions are addressed through a socio-ecological lens, recognising that effective restoration must align ecological feasibility with stakeholder expectations, resource constraints, and long-term management structures.

1.8.2 Research objectives

Urban grassland restoration sits at the intersection of design intent and ecological function. For landscape architects, it offers opportunities to reimagine neglected spaces as biodiverse, climate-resilient urban systems. For restoration ecologists, it presents a testbed for applied interventions under complex site constraints. This research contributes to both fields by addressing five objectives, each responding to a knowledge gap identified through the applied challenges of the Glenthorne Community Grassland Project:

1. Evaluate how soil solarisation can be optimised for weed and pathogen control in urban grassland restoration.
2. Investigate the effectiveness of topsoil removal in boosting native forb and grass diversity, and understand how this intervention interacts with soil conditions and weed pressure.
3. Develop a decision-making framework to balance biodiversity benefits with financial and environmental constraints in urban grassland restoration.
4. Assess the efficacy of remote sensing (Sentinel-2 EVI2 time series) for monitoring the success and trajectory of small-scale grassland restoration efforts.
5. Explore how evolving design philosophies intersect with ecological science to inform resilient, future-facing practices in landscape architecture.

The specific research questions associated with each objective are detailed in the context statement preceding each empirical chapter. Figure 5 and Table 1 summarise the thesis structure, mapping each chapter to its objective, research questions, knowledge gap, methods, and outcomes.

Table 1: Overview of Thesis Structure, Research Gaps, Objectives, Questions, Methods, and Outcomes.

Part	*Chapter / Title (Abbreviated)*	*Obj.**	*Quest.**	*Gap*	*Methods*	*Outcomes*
Part I Context & Rationale	**Ch. 1:** General Introduction	–	–	–	–	Frames the global biodiversity crisis, introduces the socio-ecological perspective, and outlines thesis aims, objectives, research questions, and structure.
Part II Empirical Investigations	**Ch. 2:** Sunlight vs Air Temperature in Soil Solarisation	OB1	Q1.1, Q1.2	G1 – Limited clarity on whether sunlight intensity or ambient temperature is the primary driver of topsoil heating for weed/pathogen control.	Field experiments Temperature & radiation measurements	Demonstrates that sunlight intensity can overshadow ambient temperature in soil heating, guiding the optimal timing for solarisation in restoration contexts.
	Ch. 3: From Bare to Bountiful – Topsoil Removal for Grassland Diversity	OB2	Q2.1, Q2.2, Q2.3	G2 – Insufficient empirical evidence on the role of topsoil removal in enhancing forb and grass diversity under urban restoration conditions.	Randomised field trials Soil parameter analysis	Shows that removing topsoil significantly increases native forb and grass alpha diversity.
	Ch. 4: Striking a Balance – A Framework for Topsoil Removal	OB3	Q3.1, Q3.2, Q3.3	G3 – Need for an integrated approach combining ecological benefits, economic costs, and soil conservation concerns when removing topsoil.	Economic modelling Soil assessment	Proposes a decision-making framework to balance biodiversity gains with financial and environmental constraints, offering guidelines on if, when, and how to remove topsoil.
	Ch. 5: Pixel Patchwork – Monitoring via Sentinel-2 EVI2	OB4	Q4.1, Q4.2, Q4.3	G4 – Lack of accessible remote-sensing methods for effectively detecting and monitoring vegetation changes in small or fragmented sites.	Sentinel-2 EVI2 Time-series analysis	Demonstrates that phenological metrics (e.g., peak-of-season greenness) differentiate native from non-native grasslands, providing a scalable tool for tracking restoration success and guiding management decisions.
Part III Synthesis & Design Perspectives	Ch. 6: The Evolving Circle of Landscape Design	OB5	Q5.1, Q5.2	G5 – Insufficient integration of evolving design values with ecological science in shaping landscape architecture practice	Reflective design & theory analysis	Advocates for a values-driven, science-informed approach for creating resilient urban grassland restorations that can adapt to future societal and environmental demands.

Legend
Obj.* = Objective(s) addressed by the chapter (e.g., OB1 = Objective 1).
Quest.* = Research Question(s) answered in each chapter (e.g., Q1.1, Q1.2).
Gaps = Specific knowledge or practice gaps that each chapter addresses (e.g., G1, G2).

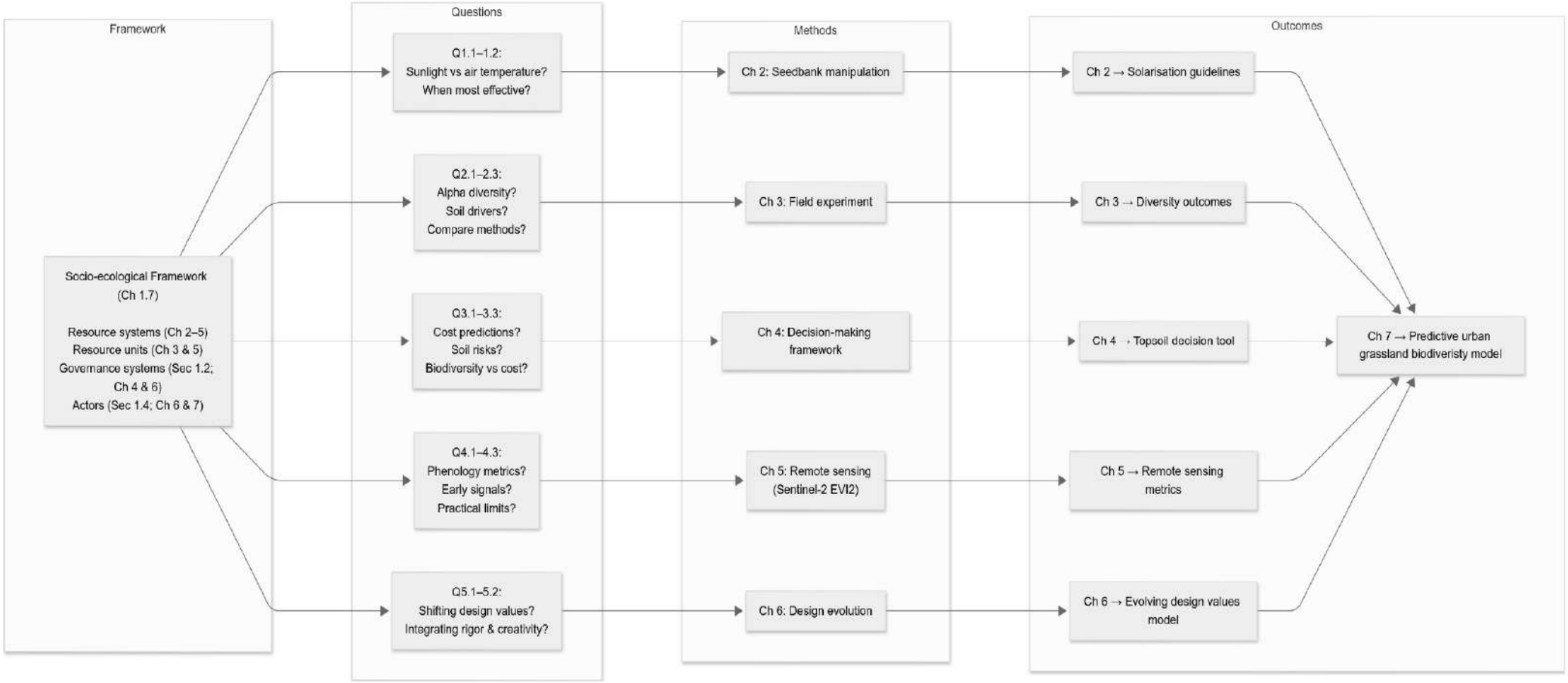

Figure 5. Thesis structure and logical flow. This flowchart illustrates how the research is structured around a socio-ecological framework, with each chapter addressing a specific knowledge gap through targeted research questions. Methods span empirical fieldwork, modelling, remote sensing, and reflective design analysis. Outcomes are synthesised to inform both ecological and design practice, culminating in a predictive model for urban grassland biodiversity. All investigations draw from the Glenthorne Community Grassland Project as a unifying case study.

1.9 Thesis structure

This thesis is organised into three main parts spanning five chapters, plus accompanying appendices. A brief description of each part follows, outlining how the thesis progresses from contextual background and empirical research to synthesis and future-oriented design insights. The research gaps are clearly and logically highlighted in each chapter's introduction, in line with the thesis by publication format.

1.9.1 Part I – Context and Rationale

Chapter 1 – General introduction This opening section establishes the global biodiversity crisis and situates urban grassland restoration within a socio-ecological framework. It highlights the aims, objectives, and research questions guiding the thesis, and concludes with an overview of the thesis structure.

1.9.2 Part II – Empirical Investigations

Chapter 2 – "Sunlight can have a stronger influence than air temperature on soil solarisation" Explores soil solarisation as a pre-planting weed and pathogen control method. This chapter presents experimental data demonstrating how sunlight intensity may outweigh ambient air temperature in heating topsoil, providing actionable timing strategies for practitioners.

Chapter 3 – "From bare to bountiful: Topsoil removal boosts forb and grass diversity in reconstructed Australian grassland" Shows how topsoil removal can significantly enhance native forb and grass diversity. It provides empirical evidence on the ecological outcomes of removing nutrient-enriched topsoil, demonstrating increased establishment success for sown native species under varying soil conditions.

Chapter 4 – "Striking a balance between biodiversity and banknotes: A framework to guide topsoil removal during grassland restoration" Builds on the findings in Chapters 2 and 3 to propose a decision-making framework. It addresses cost-effectiveness, soil conservation

concerns, and biodiversity outcomes, offering practitioners guidance on when and how to implement topsoil removal in large-scale restoration contexts.

Chapter 5 – "Pixel Patchwork: Monitoring Small-Scale Grassland Restoration Success Using Sentinel-2 EVI2 Time Series" Introduces remote sensing (Sentinel-2 EVI2) as a tool for monitoring the trajectory of grassland restoration over time. The chapter details how phenological metrics can distinguish native-dominated grasslands from non-native sites, supporting evidence-based management decisions.

1.9.3 Part III – Synthesis and Design Perspectives

Chapter 6 – "The Evolving Circle of Landscape Design" Examines how evolving values in landscape architecture - such as prioritising ecological function and resilience - can inform future landscape design. This chapter emphasises the interplay between scientific rigour and design innovation, proposing that a reflective, values-aware approach to urban grassland restoration can address ongoing and future environmental challenges.

Chapter 7 – General Discussion Integrates the empirical findings and design insights, showing that science-driven techniques - optimising soil solarisation, topsoil removal, and remote sensing - enhance urban grassland biodiversity and support evidence-based management. It also highlights how evolving landscape design values, centred on ecological function and resilience, can guide future restoration practices. This adaptive, systems thinking approach offers a blueprint applicable to other ecologies and biomes around the world.

Chapter 2

SUNLIGHT CAN HAVE A STRONGER INFLUENCE THAN AIR TEMPERATURE ON SOIL SOLARISATION: OBSERVATIONAL EVIDENCE FROM AUSTRALIA

Maximilian McQuillan, Ronald J Smernik, Ren Ryba

2.1 Context statement

Soil solarisation is an emerging pre-seeding technique in restoration because it provides a non-chemical alternative to soil fumigation, yet can still suppress dense, persistent weed seed banks within 4–6 weeks - addressing one of the major barriers to establishing native species in degraded sites. It involves covering moist soil with clear plastic to trap heat. Raising temperatures in the topsoil layer for several weeks reduces the number of viable non-native seeds and suppresses soil-borne pathogens (Katan 1981). This helps newly sown native plants establish. For restoration in public or urban areas, where large machines or repeated herbicide use are not ideal, solarisation is a lower-impact choice. It limits competition from unwanted plants, reduces the need for chemicals, and prevents the disturbance of soil removal, making it practical both ecologically and by design.

Solarisation sits within the continuum of pre-seeding interventions outlined in Chapter 1. It falls between lighter options such as chemical fallow (leaving the soil treated with herbicide to suppress weeds before planting) or mowing (periodically cutting aboveground vegetation), and more intensive resets such as topsoil removal (physically stripping off the upper soil layer). Its role is to manipulate the seed bank (the collection of all viable seeds present in the soil) and shift the odds of establishment in favour of native species, without erasing site history or requiring substantial soil movement. In that sense, it embodies the logic of restoration as design (Lenzholzer et al. 2013; Grose 2014; Higgs 2017). It is a deliberate, site-specific intervention intended to tilt vegetative trajectories at a crucial early stage.

As discussed later in this thesis in chapters 4 and 6, such interventions are never neutral. They directly shape design scope, project staging, and long-term maintenance requirements. For landscape architects, the choice of pre-seeding treatment is not simply technical preparation. It is a design decision with consequences for aesthetics (the visual and spatial character), legibility (how easily users can interpret the landscape), sustainability, and ongoing management.

Despite its significance, guidance on solarisation scheduling remains unclear. Many references simply suggest installing plastic 'in summer' or 'during the hottest part of the year,' which is not sufficiently specific for landscape professionals responsible for coordinating contractors, volunteers, or resources. The timing of the 'hottest part of the year' varies - December, January, or February - depending on weather. Restoration teams benefit from more detailed criteria, like whether to prioritise air temperature or sunlight intensity. Our findings indicate sunlight intensity is more useful, as it follows predictable seasonal changes, facilitating project alignment.

This chapter examines how sunlight and air temperature each heat soil in solarisation, and which is more important. With sunlight as the main driver, solarisation can follow seasonal radiation patterns, allowing earlier treatments while soil stays moist. If air temperature dominates, managers must use short-term forecasts, making scheduling harder. Either way, clarity lets solarisation become a set pre-construction step, not just a vague hope that plastic in summer will "do the job."

The research was conducted in the Adelaide Plains region of South Australia. This area has a Mediterranean-type climate, classified as Csa in the Köppen classification system indicating hot, dry summers and cool, wetter winters. Most published work on solarisation originates from the northern hemisphere, particularly the Middle East (Katan, 1981). Its relevance to southern hemisphere contexts remains uncertain. This chapter addresses this gap by analysing solarisation trials in South Australian grassland restoration. Here, differences in

day length (the number of daylight hours), sunlight intensity (the strength of sunlight reaching the ground), and seasonal rainfall create different constraints and opportunities. The experiment formed part of the Glenthorne Community Grassland Project, situated on Kaurna Country in Adelaide. Exotic-dominated post-agricultural pastures on fertile Vertosol soils (a soil type with high clay content and good fertility) provided a testbed for restoration. Replicated plots were established and clear polyethylene (a flexible, transparent, and weather-resistant plastic) sheeting was installed for six- and twelve-week windows, spanning from late summer into autumn. Microclimate loggers (devices that record environmental conditions) from Martin Breed's lab at the University of Adelaide recorded topsoil (the uppermost soil layer) and near-surface air temperatures. Complementary ultraviolet (UV) radiation data were drawn from the Australian Radiation Protection and Nuclear Safety Agency. This agency maintains ground-based UV monitoring in Adelaide and enables sunlight measurements to be used as a proxy (a substitute metric) for total incident solar radiation (total solar energy received at ground level) (ARPANSA 2023).

This project brought together park managers, researchers, volunteers, and members of the public who encountered the trial. Decision-making reflected both ecological and social factors. The work was embedded in a socio-ecological context (Ostrom 2009; McGinnis and Ostrom 2014; Bardgett et al. 2021); decision-making about treatments was influenced not only by ecological considerations but also by cultural, institutional, and practical ones. For example, both solarisation and topsoil removal were implemented in the newly established national park, yet their reception differed markedly. Solarisation's discreet visual profile and non-chemical character made it broadly acceptable (Image 1), whereas the more disruptive practice of topsoil removal attracted greater scrutiny and concern. Understanding this social dimension is important as methods in landscape architecture are evaluated for technical effectiveness and alignment with site identity, public expectations, and management goals.



Image 1. Soil solarisation at Glenthorne *National Park–Ityamaiitpinna Yarta (GNPIY): (left) installation of clear plastic sheeting, and (right) plots following installation.*

This chapter shows that both air temperature and sunlight heat soil, but sunlight has a stronger effect. Practitioners can schedule solarisation by sunlight, not unreliable temperature forecasts. Bright, cool days in late summer or early autumn can still warm soil if treatments last through sunny spells. Since sunlight patterns are predictable, this supports earlier procurement, scheduling, and clear communication, and lets treatments start while soil moisture remains. By reducing extra watering and expanding planning, solarisation becomes a more dependable, cost-effective tool.

Within the framework of state-and-transition models (Sinclair et al. 2019) (models describing how ecosystems can shift between different stable conditions), solarisation is a designed disturbance that reduces exotic propagule pressure (the introduction and presence of non-native seeds or plant material capable of establishing new populations). This increases the probability of transition from an exotic-dominated pasture state toward a native grassland state. For landscape architecture, this translates to more than just technical optimisation. It provides a way to integrate ecological thresholds (Westoby et al. 1989) (critical points at which small changes can trigger major shifts in ecosystem state) into project staging, provisional sums (budget allowances for uncertain costs), and risk contingencies (plans for managing uncertainty). An apparently "light touch" pre-seeding treatment can shape the trajectory of a project by influencing whether plantings establish successfully or collapse

under weed pressure. This has downstream consequences for maintenance, aesthetics, and stakeholder confidence.

This chapter was published as a Short Communication in Soil Research, a venue with a soil science readership. The decision to publish there reflected the study's narrow mechanistic focus on soil heating, as well as the urgency of providing practitioners with concise, evidence-based guidance. However, in the context of this thesis, the results are translated into interdisciplinary terms. They demonstrate how ecological insights about soil processes can be repurposed into design guidance for restoration planning, showing landscape architects that solarisation is not just an agronomic curiosity but a site preparation strategy with direct design implications.

The chapter serves two functions in the thesis: first, it establishes when solarisation is likely to succeed, addressing a knowledge gap in southern hemisphere applications; second, it links these findings to the restoration design framework from Chapter 1, where pre-seeding interventions are seen as thresholds on a continuum. Later chapters use this logic to compare methods like topsoil removal and develop frameworks weighing biodiversity gains against cost and soil risks. Together, these analyses advance the thesis's aim: integrating ecological science and landscape architecture to inform the design of biodiverse urban grasslands.

The specific research questions addressed in this chapter are:

- *Q1.1 To what extent does sunlight intensity versus ambient air temperature drive soil temperature during solarisation?*
- *Q1.2 Under which seasonal or environmental conditions is soil solarisation most effective as a pre-planting technique for ecological restoration?*

Statement of

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SOIL RESEARCH

Sunlight can have a stronger influence than air temperature on soil solarisation: observational evidence from Australia

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ABSTRACT

Soil solarisation is a method for pest and weed control pioneered in agriculture, and it is increasingly being adopted by restoration practitioners. Solarisation works by covering moist soil during hot periods with a sheet of clear plastic. The success of soil solarisation depends in large part on increasing the temperature of the topsoil. Topsoil temperature depends on several physical variables, including soil moisture content, ambient temperature, and sunlight intensity. In restoration scenarios, solarisation can be used to reduce weed and pathogen loads prior to planting target plants. It is rarely possible to have tight control over all the variables that are important for solarisation; however, practitioners can time interventions to maximise seasonal temperature and sunlight intensity. In this study, we investigated how these two key physical variables – temperature and sunlight – contributed to the success of soil solarisation. We found that while both ambient temperature and sunlight contributed to soil temperature, the data suggests that sunlight was the more influential driver of soil temperature. These results show that, when planning for soil solarisation during ecological restoration, land managers can benefit by considering sunlight as well as air temperature. The result that sunlight may be the more influential driver of soil temperature empowers land managers to better plan solarisation using sunlight projections, even when temperature is not optimal or is unpredictable.

Keywords: grassland restoration ecology, non-native plants, soil biology, soil seed bank, soil solarisation, soil temperature, southern hemisphere, tillage, weeds.

Introduction

Soil solarisation is a method of controlling soil pathogens and nuisance plants by covering moist soil during hot periods with a sheet of clear plastic. Soil solarisation was pioneered for agriculture in the Middle East in the mid-1970s (Katan 1981) and has reportedly been used in over 60 countries (Katan and Gamliel 2010). More recently, there has been renewed interest in solarisation because it uses less pesticide (D'Addabbo *et al.* 2010). Solarisation is a particularly promising option in restoration projects where it can reduce the abundance of exotic plant propagules from the soil seed bank over the course of only 4–6 weeks and alternative methods (e.g. topsoil removal with heavy machinery) are often unviable (Bainbridge 1990).

The sheet of clear polyethylene plastic used in soil solarisation generates a greenhouse effect in the upper layers of the soil. The soil temperature increases, which triggers a series of interrelated physical, biological, and chemical events that can destroy unwanted plant propagules (Horowitz *et al.* 1983; Bainbridge 1990).

It is well-accepted that in the absence of a complete, long-term weed survey, the success of soil solarisation can be estimated by measuring the short-term increase in soil temperature. Specifically, research over the past 40 years has indicated that the main driver of the success of solarisation is the ability to increase soil temperature, which kills target organisms (Horowitz *et al.* 1983; D'Addabbo *et al.* 2010). The greater the increase in soil temperature, the more effective solarisation becomes (Abed Gatea Al-Shammary *et al.* 2020). Previous studies have suggested that solarisation can successfully control plant seeds and microbial pathogens at depths of 0–10 cm, with some studies showing mortality of seeds and pathogens even beyond 20 cm depth (D'Addabbo *et al.* 2010; Gill *et al.* 2017).

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Many edaphic factors are known to influence the efficiency of heat transfer during solarisation. Such factors include soil texture, soil structure, and soil moisture content. Restoration practitioners have control over some factors known to influence solarisation, but not others. Specifically, practitioners do have control over the timing of solarisation, which determines the magnitude of both temperature and solar radiation – the drivers responsible for raising the temperature of solarised soil. In contrast, practitioners typically do not have the option to artificially water or till the soil due to infrastructure and equipment limitations (Cohen *et al.* 2019).

However, there is ambiguity around the optimal time to apply the solarisation treatment (Kanellou *et al.* 2023). Existing guides simply recommend that practitioners choose the hottest and sunniest part of the year (Bainbridge 1990) and to refer to meteorological data from previous years to determine when this is (Abed Gatea Al-Shammary *et al.* 2020). The weakness of this advice is that it does not help restoration practitioners understand the relative importance of different variables. How much should a land manager prioritise temperature, which can typically only be predicted days in advance, compared with sunlight intensity?

There are indications in the literature that solar radiation is an important contributing factor to the success of soil solarisation. One indication is the observation that translucent plastic works better than non-translucent plastic (Avissar *et al.* 1986; Al-Karaghouli *et al.* 1990), and another indication is mathematical modelling that uses solar radiation as one explanatory factor (Mahrer *et al.* 1987). However, for land managers, the relative contribution of solar radiation remains elusive. Furthermore, sunlight intensity varies globally, primarily with latitude, but also with altitude and cloudiness (Van Geffen *et al.* 2017). Most research on solarisation has been conducted in the northern hemisphere at high (e.g. southern Europe) to mid (e.g. the Middle East) latitudes, so the results of such research may not be applicable to restoration projects in the southern hemisphere (e.g. Australia). This study fills this gap by examining solarisation at a mid latitude site (approximately 35°) in the southern hemisphere.

Thus, the relative effects of solar radiation and temperature on soil solarisation are unknown. The aim of this study is to provide insights into the relative contribution of sunlight and air temperature to soil solarisation, enabling restoration practitioners – particularly in the southern hemisphere – to better plan and optimise this technique.

To address this gap, we conducted an analysis to statistically distinguish between the relative contribution to solarisation of air temperature and of sunlight. If sunlight is the primary driver of solarisation, then the statistical contribution of UV to soil temperature will be large while the statistical contribution of air temperature to soil temperature will be minimal. If air temperature is the primary driver of solarisation, then the reverse will be true. If sunlight and air temperature both make separate contributions to solarisation, then both UV and air temperature will display a moderate statistical contribution to soil temperature.

Materials and methods

Study site

The data were collected during a native temperate grassland restoration project on a Black Vertosol (Isbell 2021) soil (see Supplementary Table S2 for the full soil profile description) in an urban national park (35.06022S, 138.54983E) in Kaurna Country/Adelaide, South Australia. Eight temperature sensors (TMSs) (Wild *et al.* 2019) (TOMST, Czechia) were installed across eight 5 m × 5 m research plots (one per plot). These sensors recorded soil temperature 6 cm below the soil surface and air temperature 15 cm above the soil surface, at 15-min intervals.

Treatments

Each plot was randomly located within the grassland and randomly assigned to one of three treatments: (1) control (two replicates), where no plastic was placed over the soil; (2) 6-week solarisation treatment (two replicates), where clear 210-micron thick polyethylene (PE) plastic was placed over the soil and removed after 6 weeks; and (3) 12-week treatment, where the same plastic was placed over the soil and removed after 12 weeks (four replicates). This study reports on a subset of a larger randomised restoration experiment, which consists of five treatments (the two solarisation treatments described here, one herbicide treatment and two topsoil removal treatments) and one control, all with six replicates. Only the data pertaining to the solarisation treatments are reported on here.

The control serves one purpose in this study, which is to provide a baseline soil temperature measure to compare the solarisation treatments – this provided the necessary information to confirm that solarisation had successfully increased soil temperatures. However, the statistical analyses required capturing the seasonal variation in the environmental variables, and this necessitated using only the 12-week solarisation data. As such, more loggers were assigned to the 12-week solarisation treatment. It is also worth noting that the variation between replicates within each treatment was minimal.

Data collection

The experiment began on 26 January 2023, following two summer rainfall events[1], and corresponds to the middle of

[1]Total daily rainfall recorded at the closest weather station (Happy Valley Reservoir ~ 2 km east of the study site) was 2.4 mm on 18 January, and 4 mm on 24 January (BoM 2023).

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the local summer, with peak daily UV radiation (max. UV index 11). Details of how UV was quantified are given below. Data collection continued for 105 days, with milestones at 42 days (max. UV index 8.7) and 84 days (max. UV index 5). The experiment ended on 11 May 2023 (max. UV index 3.2). One logger (control plot) was destroyed by a wild animal during the experiment.

Sunlight data were obtained separately, from the Australian Government, which measures and publishes UV index data at 1-min intervals (ARPANSA 2023). This data is collected by sensors on the ground at a number of sites across Australia. The data for Adelaide City, located 15 km north-east of the study site, were used. UV radiation is strongly correlated with total solar radiation, enabling the use of UV radiation as a relative measure of solar radiation incident on the soil (Santos *et al.* 2011). The timing of the study, the weeks between the middle of summer and the middle of autumn, provides a natural gradient of different sunlight intensities. Furthermore, different days had different levels of cloud cover. The sunlight incident on the ground decreases when there is substantial cloud cover; attenuation by clouds is similar across different wavelengths of sunlight (Chen 1975). Therefore, the measured UV index is suitable as a relative proxy of total sunlight even on cloudy days. This is particularly useful for this study, as this means that the dataset contains both high-temperature, low-sunlight days and low-temperature, high-sunlight days.

Data analysis

The dataset was analysed as a set of 105 daily observations, with each day being assigned the soil and air temperature that was closest to the solar maximum on that day. This accounts for the fact that multiple temperature readings on each day would be very closely correlated with each other. Each day was then assigned a value for sunlight that was equal to the average UV intensity across the day from sunrise up until that day's solar maximum (i.e. the integral of the day's UV curve divided by time). This corrects for the confounding effect of day length and accounts for different durations of cloud cover each day. A daily timescale is also appropriate for the response variable; soil temperature which was measured 6 cm below the soil surface. At this depth, temperature changes relatively slowly with only small diurnal fluctuations. This resulted in a dataset containing daily observations of soil temperature at −6 cm, and air temperature at +15 cm, and UV radiation, across 105 days and three treatments. It is well-understood that daytime soil temperature is highest at the surface and decreases with depth (Reddy 2012). Practitioners are typically interested in the top few centimetres of soil. Therefore, 6 cm provides an appropriate, if conservative, measure of the effects of solarisation. Temperatures above 6 cm, including at the surface, would be slightly higher than the soil temperatures we report here.

First, to test the assumption that the clear plastic covers were successful in increasing soil temperature, the time series of soil temperature in each of the three treatments were visualised (Fig. 1*a*). The time series was visualised as the mean for each treatment; variation within treatments was minimal.

Second, using data from the 12-week treatment, statistical tests were conducted to measure the effect of air temperature and sunlight on soil temperature. A linear regression model was generated using the raw variables. Also, to provide results more readily interpretable by land managers, the two explanatory variables (air temperature and sunlight) were partitioned into categorical variables, with each data point being assigned to the factor 'high' (greater than the mean) or 'low' (smaller than the mean). The effects of these categorical variables on soil temperature were examined using an ANOVA and Tukey's *post hoc* test, and the results were visualised (Fig. 1*b*).

Third, to mitigate concerns about correlation between the two explanatory variables (air temperature and sunlight), a principal component analysis (PCA) was conducted, and the results visualised (Fig. 1*c*). The PCA axes were generated using the three key continuous variables: air temperature, sunlight, and soil temperature. The principal component plot was visualised, with points coloured by the predictor variables (sunlight and air temperature) for clarity (Fig. 1*c*).

All data analysis was conducted in R, using the packages myClim and factoextra for data analysis and ggplot2 and gridExtra for visualisation (Wickham 2016; Auguie 2017; Kassambara and Mundt 2020; Man *et al.* 2023; R Core Team 2023).

Results

On average, the soil temperature at solar maximum was 31.2°C (s.d., 5.0) in the solarised plots and 23.0°C (s.d., 4.6) in the control plots, over the 12-week period. This indicates that solarisation caused the soil temperature at solar maximum to increase by 8.2°C (s.d., 1.4). The mean air temperature was 24.7°C (s.d., 6.4°C), and the mean UV index (averaged between sunrise and daily maximum) was 2.5 (s.d., 1.1).

The visualised time series of the response variable (soil temperature; Fig. 1*a*), verifies that the clear plastic did indeed increase the soil temperature. This provides confidence that solarisation was performed successfully during the study. Time series of the input variables (UV index and air temperature) are visualised in the Supplementary material (Fig. S2), along with a time series of the difference in temperature between solarised and control treatments. Both the 6-week and 12-week treatments showed similarly higher soil temperature at daily maximum than the control plots over the first 42 days (Fig. 1*a*). When the solarisation plastic

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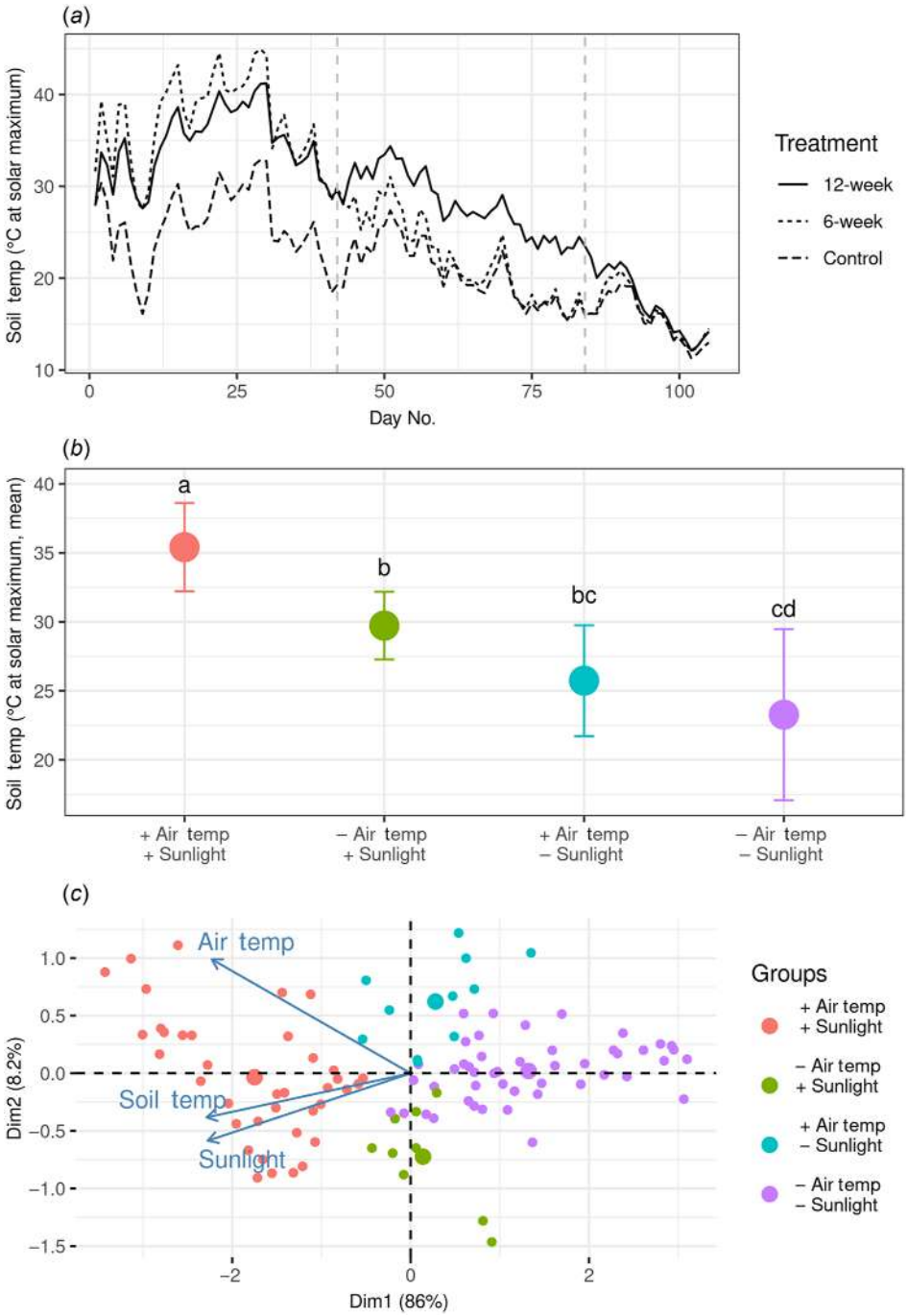

Fig. 1. The relative contribution of temperature and sunlight to soil solarisation. The main implication of this figure is that soil solarisation increases soil temperature (*a*), that air temperature and sunlight both contribute to the success of soil solarisation (*b*), and that the contribution of sunlight to solarisation may be more important than that of temperature (*c*). (*a*) Time series showing the daily topsoil (at −6 cm) temperature at the time of peak daily sunlight, averaged across the sensors within each treatment (vertical dotted lines delineate the 6 and 12 week treatment times); (*b*) Air temperature and sunlight partitioned into categorical variables ('+' indicates days with greater than the mean sunlight or temperature, and '−' indicates days with less than the mean sunlight or temperature). Letters indicate differences between groups as calculated using ANOVA and Tukey's *post hoc* test; (*c*) Principal component analysis (PCA), with data coloured based on the same grouping as in (*b*). Note that the vertical axes differ between plots.

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was removed in the plots assigned to the 6-week treatment, the soil temperature in those plots decayed to the same temperature as in the control plots over approximately 4 days. The difference in soil temperature between solarised and control treatments showed a gradual decrease over the 12-week study period, corresponding to the decrease in sunlight between summer and winter.

The linear regression (Table S1) revealed a strong effect of both sunlight (slope = 8.29; $P < 0.01$) and air temperature (slope = 0.988; $P < 0.01$), as well as their interaction (slope = −0.197; $P < 0.01$) on soil temperature. The ANOVA also supported this conclusion – the soil temperature was highest on days with both high sunlight and high air temperature (Fig. 1*b*). On days with high sunlight *or* high air temperature, the soil temperature was moderate, and still higher than days with low sunlight and low air temperature. Sunlight appears to have a slightly greater effect than air temperature, though this difference was not statistically significant (second and third columns in Fig. 1*b*).

The PCA supported this general conclusion. The biplot revealed that soil temperature was correlated with both sunlight and air temperature (Fig. 1*c*). Soil temperature appeared more strongly correlated (PCA vector more closely aligned) with sunlight than with air temperature.

Discussion

This study provides two main contributions to the field of existing knowledge on soil solarisation: (1) we use novel solar radiation data to differentiate the influence of air temperature from sunlight; and (2) the soil type in this study lies close to the edge of the texture spectrum (i.e. almost pure clay), providing useful information on the use of solarisation techniques on medium to heavy textured clay soils.

The relative effect of sunlight and ambient temperature on soil solarisation

Although solarisation is not used extensively in Australia, it has been noted that Australia should be a good candidate for solarisation due to its Mediterranean climate and long periods of intense solar radiation (Powles *et al.* 1988). However, to our knowledge, this claim has not been assessed using solar radiation data.

The results of the statistical analyses point to the conclusions that: (1) air temperature and sunlight are both correlated with high soil temperatures; and (2) solar radiation may influence soil temperature more than air temperature does. Fig. 1*b* reveals a higher mean soil temperature for days assigned low air temperature and high solar radiation than days assigned high air temperature and low solar radiation, though the difference between these two specific groups was not statistically significant. Likewise, Fig. 1*c* shows a higher correlation of soil temperature with sunlight than with air temperature. Therefore, both analyses point to the conclusion that soil temperature is more strongly correlated with solar radiation than with air temperature during solarisation. To account for correlation between the variables, these results should be verified with a perfectly crossed experiment – specifically, a laboratory experiment where air temperature and irradiation can be independently controlled.

If our interpretation is correct and is supported experimentally, then this has practical implications for practitioners who can use solar radiation rather than temperature to guide the timing and duration of solarisation. In global terms, temperate Australia (as represented by our study conducted in Adelaide) has extended periods of high solar radiation. To illustrate this point, we compared the local solar radiation regime to places where solarisation is more commonly used and not used. We use openly available global UV data from southern Australia, the Middle East, and western Europe. Fig. 2 shows the wide variance of solar radiation regimes across different parts of the world in terms of timing, duration, and magnitude.

The fact that southern Australia (and by extension, those parts of the continent characterised by a Mediterranean climate) receives high levels of solar radiation in global terms is significant as soil solarisation is still relatively uncommon in Australia. This contrasts with numerous countries where solarisation is employed more regularly (e.g. Argentina, Australia, Egypt, Cameroon, Greece, India, Israel, Italy, Libya, Morocco, Spain, Sudan, Tunisia, the USA, South Africa, and others) for a diverse range of applications (e.g. agriculture, restoration, archaeology) (D'Addabbo *et al.* 2010; Gill *et al.* 2017; Kanellou *et al.* 2023; Shinde *et al.* 2023).

Solarisation applied to high clay content soils

The study site's soil was classified as a Black Vertosol, which characteristically contain high amounts of clay, with medium to high clay textures recorded through the solum. These soils are a major feature of large parts of the Australian continent and are generally associated with major floodplains in central Queensland and New South Wales but are relatively uncommon in Southern Australia (Isbell 2021). The fact that they underlie the study site lends them as a good test case for the high end of the texture spectrum encountered in Australia's Mediterranean climate zone. The high clay content and strong structure of this soil results in a very high-water holding capacity. This may be advantageous in a restoration scenario where adding water prior to solarisation (as recommended as best practice (Abed Gatea Al-Shammary *et al.* 2020)) is often unfeasible.

In our study, solarisation caused soil temperature to be elevated by 8.2°C on average (s.d., 1.4°) compared to the control plots. This is comparable to other studies conducted on clay soils. For example, Linke (1994) reports a 6.9–8.5°C soil temperature increase for a 60% clay soil in Syria. Similarly, Jacobsohn *et al.* (1980) report a 8–12°C soil

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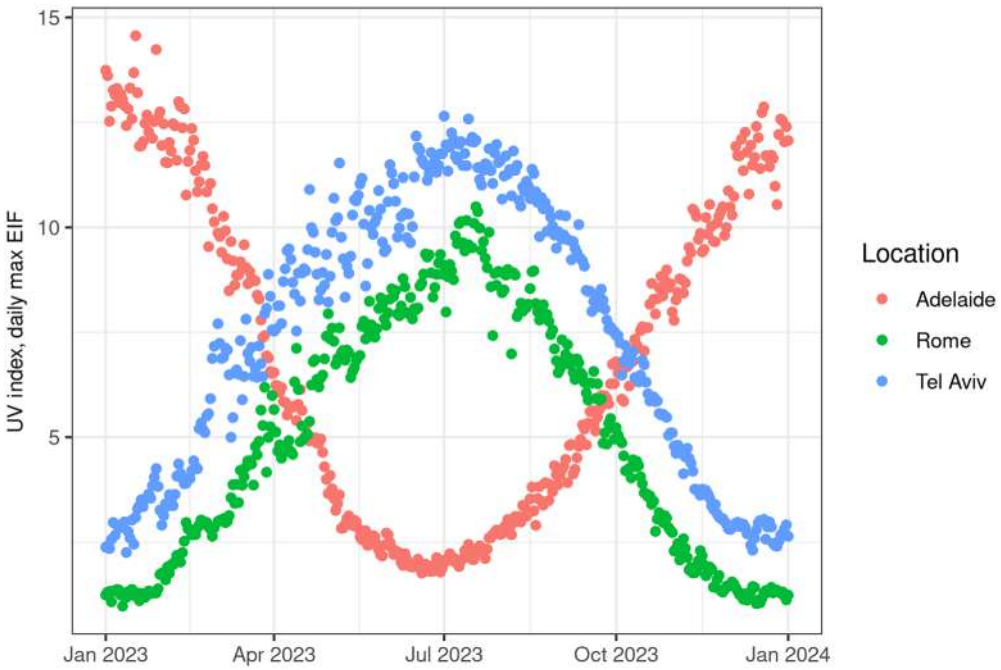

Fig. 2. Measured UV data from three locations, roughly representative of western Europe (Rome, Italy), the Middle East (Tel Aviv, Israel) and southern Australia (Adelaide) over 1 year. Data source: (Van Geffen *et al.* 2017).

temperature increase for a medium-fine textured soil in Israel, which therefore provides confidence that soil solarisation was performed successfully in this study. Furthermore, Sharma and Nene (1990) report a 10°C temperature difference at 5 cm in a clay soil in India. Interestingly, for the study most geographically proximate conducted just 13 km north-east of our study site, a higher difference in soil temperature between control treatments and solarisation treatments was reported (12°C higher than control at 5 cm; and 11°C higher at 15 cm, for a clay loam soil) (Powles *et al.* 1988).

It is not clear what variables impacted the higher temperature difference observed between our study site and that reported by Powles *et al.* (1988). The variation in seasons, stemming from the time disparity between our study and theirs, might have had an impact. Other factors may include the lighter soil texture and the fact that Powles *et al.* (1988) were able to irrigate their plots to field capacity prior to installing the plastic. Another factor that likely influenced our results is the dark soil colour (black), which would have caused the control soil to absorb more heat than a lighter coloured soil. Regardless, it is noteworthy that significant soil temperature increases are possible without prior irrigation to field capacity, at least on soils with significant clay content that impart a high water holding capacity.

Another observation, which is likely soil specific, was the time delay between the plastic being installed and the soil temperature rising. It took approximately 4 days for soil to reach this average temperature difference after the cover was installed and to return to baseline temperatures after the plastic was removed. This recharge and discharge time results from soil heat storage and highlights the need for consecutive days of high incident solar radiation and/or high ambient temperature for soil to reach high temperatures.

Study limitations

Our study encountered three main limitations. First, this was an observational study. The conclusions from this study could receive greater support from a fully crossed experimental study using different air temperatures and artificial light intensities. Secondly, the study was limited in space and in time. While we contribute to the literature by conducting a study in the under-represented southern hemisphere, it would be beneficial for this study to be replicated at a variety of locations across the southern hemisphere to ensure that the results are generalisable. Likewise, repeating the study at different parts of the year would provide further insight into the influence of timing on the effectiveness of solarisation. Thirdly, the outcome of solarisation depends on the soil type. We conducted this study at a site with a Self-Mulching Black Vertosol soil (Australian Soil Classification (Isbell 2021) code VE AE EI; see Table S2 for the full soil profile description). Repeating the study at additional sites would enable practitioners to be aware of how solarisation might be affected by soil type. Further research could also examine the relative effects of sunlight and temperature on mortality of microbial pathogens and plant propagules in soil. By complementing our results, such research would enrich practitioners'

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understanding of the effects of solarisation on both soil temperature and activity of different types of organisms.

Conclusions

The results of this study have practical implications for practitioners implementing soil solarisation in restoration contexts. Restoration practitioners often cannot control all the variables that influence the success of soil solarisation (e.g. temperature, moisture). Practitioners do, however, have relatively good control over the timing of the treatment, which influences the available sunlight. Practitioners can predict sunlight levels based on established models that primarily consider the position of the Earth relative to the sun (e.g. UV index). This has practical implications for practitioners, especially in light of the relative importance of solar radiation in solarisation as identified in this study.

Our findings lead to specific recommendations for conducting soil solarisation. For example, practitioners can focus on conducting solarisation over time periods that are likely to be bright, rather than warm. Cold, bright days are still suitable for solarisation, and practitioners can begin solarisation even during the spring, while soil moisture is still high and leave the plastic in place until the period of peak solar radiation.

Additionally, soil solarisation can be planned around sunlight projections (e.g. UV index) rather than temperature projections. Since sunlight projections are available much further in advance than temperature projections, this enables practitioners to plan soil solarisation much earlier while soil moisture is still relatively high. This reduces the need to artificially water the soil prior to laying the plastic. Consequently, understanding the important role that solar radiation plays during solarisation empowers practitioners to improve the timing of solarisation projects.

Supplementary material

Supplementary material is available online.

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Data availability. The data that support this study will be shared upon reasonable request to the corresponding author.

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Chapter 3

TOPSOIL REMOVAL INCREASES NATIVE FORB AND GRASS RECOVERY IN URBAN TEMPERATE GRASSLANDS

Maximilian McQuillan, Ren Ryba, Ronald J Smernik, Tarryn Davies, Heidi Rogers, Caleb Lilly, Scott Hawken, and Martin F. Breed

3.1 Context statement

People often misread restored grasslands as "messy" or neglected. In an early conversation with Caleb Lilly at Seeding Natives, we compared notes on how frequently native grasslands are mistaken for degraded paddocks. If public acceptance is shaped by what people see first, then composition and visual legibility matter. We wondered: could we design for a grassland that reads as intentionally cared for - by lifting forb and wildflower diversity - without compromising ecological goals? That conversation became the catalyst for the experiment reported in this chapter.

Here, I investigate how pre-seeding interventions - the site-preparation actions taken before broadcast seeding - shape who wins space in a restored grassland. For clarity, I use "pre-seeding interventions" to mean methods that adjust propagule pressure (the number and kinds of seeds present) and soil legacies (especially elevated nutrients from past agriculture). In practice, these fall into three tiers used by restoration practitioners and increasingly by landscape architects: (1) biomass removal (e.g., mow/slash/burn or knock-down herbicide) that clears canopy but leaves the seed bank intact; think of this as a 'light touch' approach. (2) seedbank reduction methods (e.g., soil solarisation) that thermally or chemically reduce viable propagules near the surface, functioning as a 'mid-range reset'. (3) soil reset methods (e.g., topsoil removal) that both remove most of the weed seed bank and lower nutrient loads, effectively providing a 'complete reset'. Each tier trades off cost, carbon, disturbance, and speed - issues central to design and delivery in landscape architecture.

This chapter builds directly on Chapter 2, which showed that sunlight can be a stronger proximate driver of soil heating during solarisation than ambient air temperature in South Australia's Mediterranean-type climate. There, I provided microclimate evidence to help with determining the optimal timing for solarization. This chapter shifts focus to vegetation outcomes, asking which pre-seeding interventions most effectively promote flower-rich grasslands with high native forb and grass representation?

To clarify the aesthetic and social implications of vegetation outcomes for landscape architecture, higher forb richness and greater spatial evenness are described here in terms of observable landscape attributes, including textural contrast, seasonal colour change, and fine-grained structural variation (Image 2). These characteristics are commonly associated with perceived care, legibility, and visual diversity in public landscapes. Such legibility can help counter the 'messy equals neglected' bias by making ecological intention visible (Nassauer, 1995). From a maintenance perspective, shifting communities away from non-native, nutrient-responsive pasture grasses can also moderate biomass pulses and reduce mowing inputs, which are important for cost-effective maintenance plans and carbon-positive site management. In short, species composition is not just an ecological outcome, it's a design lever with operational implications.



Image 2. Topsoil-removal plots in full bloom *(T_2, 18 months post-seeding; November 2024), Kaurna Country / Adelaide, South Australia (Csa). Photo: Scott Hawken. The forb-weighted seed mix produced a diverse wildflower layer (e.g., Chrysocephalum apiculatum, Wahlenbergia spp., Leptorhynchos squamatus), interwoven with less competitive native grasses (e.g., Austrostipa spp., Poa spp.). The resulting fine texture, seasonal colour, and vertical punctuation aim to increase visual legibility and perceived care - design qualities important for public acceptance of native grasslands in urban settings.*

Study aim and questions.

I test how three pre-seeding interventions: chemical fallow (biomass removal control), soil solarisation (6- and 12-week durations), and topsoil removal (7 cm and 14 cm); affect plant diversity and composition in a temperate, post-agricultural grassland in Kaurna Country / Adelaide, South Australia. The study asks: (1) Which intervention most increases native and sown alpha diversity at 6 and 18 months? (2) How does each intervention shift community composition toward native dominance? (3) Which options prioritize forbs over grasses for desired visual and ecological outcomes?

Design translation and seed mix strategy.

To explore a visually legible, wildflower-forward outcome, I partnered with Seeding Natives to formulate a forb-weighted seed mix (41 species) that departed from standard practice in Australian grassland restoration. Conventional seed mixes are typically grass-dominated, with forbs included at relatively low proportions and sown at modest densities to prioritise rapid ground cover and weed suppression (Barr et al., 2017). In contrast, the mix used here deliberately increased the proportional representation of native forbs and was sown at a higher-than-standard density (4.3 g m^{-2} compared with ~1.5 g m^{-2} commonly used in practice). This composition represents nearly triple the typical forb proportion, marking it as an explicitly experimental configuration intended to test whether greater floral richness and visual legibility could be achieved without compromising ecological establishment. By deliberately selecting less competitive native grasses, the mix created ecological "room" (Brunbjerg et al., 2017) for forb establishment, allowing the design decision to be evaluated through measurable vegetation outcomes.

3.1.1 Methods in brief

The field experiment employed a randomized block design in an urban national park (~180 m a.s.l.) on Black Vertosol/Black Dermosol clays, characterized by a warm-temperate, dry-summer climate (Csa). Five treatments were implemented prior to seeding: (i) biomass removal via glyphosate (control), (ii) solarisation for 6 weeks, (iii) solarisation for 12 weeks, (iv) topsoil removal 7 cm, and (v) topsoil removal 14 cm. All plots were broadcast-seeded on the same day and then managed with a light, time-limited mowing regime during the first growing season. Vegetation was monitored at 6 months (T1) and 18 months (T2) using 1 m^2 quadrats to quantify alpha diversity, community composition (NMDS/PERMANOVA), and abundance by plant functional group (generalized linear models, GLMs).

3.1.2 Theoretical positioning

I frame these interventions through two linked lenses that are used throughout the thesis. First, the State-and-Transition Model (STM): a practical framework that treats vegetation as occupying alternative, semi-stable states separated by thresholds; management aims to push systems across thresholds when they're "stuck" in degraded states dominated by non-native grasses (Sinclair et al., 2019). Second, a socio-ecological systems view that recognises biophysical processes and human decisions as coupled: restoration outcomes reflect climate and soil, but also governance, procurement, maintenance budgets, public preferences, and professional norms (McGinnis & Ostrom, 2014). In this study, "the social" is represented by who sets goals (agency clients, local government), who supplies and installs (contractors, nurseries), and who will live with and judge the outcome (communities). Explicitly naming these actors clarifies why a design-forward, forb-rich target has value beyond species counts.

3.1.3 Why These Methods Matter to Landscape Architecture.

- ***Topsoil removal*** *directly reduces both the dense non-native seedbank and elevated soil nutrients - primary obstacles to forb establishment in post-agricultural sites. This intervention most strongly favours the establishment of fine-textured, flower-forward swards, yielding plantings that appear intentional, support pollinators, and reduce the long-term need for weed control more effectively than the other interventions in this study.*

- ***Solarisation*** *is a lower-impact technique that uses plastic to suppress weeds where excavation is unsuitable. In this experiment, solarisation increased native grass establishment, but was less effective at increasing forb richness compared to topsoil removal. Therefore, solarisation is best when aiming for a grass-rich, rather than a forb-rich, sward. Chapter 2 provides timing guidance specific to South Australia's climate.*

- ***Chemical fallow*** *refers to a common business-as-usual practice in Australian grassland restoration and agricultural transition, in which repeated herbicide*

applications are used to suppress existing vegetation prior to seeding, without altering soil structure, nutrient levels, or the weed seedbank. This approach is widely used because it is relatively low-cost, familiar to practitioners, and logistically straightforward. In the present experiment, chemical fallow was included as a baseline comparator against which more intensive interventions could be assessed. Under conditions of high legacy soil nutrients and dense weed seedbanks, however, it was the least effective treatment for shifting plant communities toward native- or forb-rich states, with minimal impact relative to interventions that directly modified soil or seedbank conditions.

This chapter puts into practice the "intervention hierarchy" first introduced in Chapter 1: removing biomass, reducing the seedbank, and finally resetting the soil. It builds on Chapter 2 by showing how these different interventions affect vegetation outcomes. It demonstrates that in post-agricultural landscapes, stronger interventions (which address multiple barriers to restoration at once – soil nutrient legacies, exotic dominated seedbanks) are the most reliable way to shift the system towards being dominated by native plants, including a diverse mix of forbs. Finally, it sets up Chapter 4 which weighs the proven effectiveness of topsoil removal against its financial cost and long-term sustainability.

This is an ecological field experiment with explicit forward looking design intent, conducted in a single region/climate/soil type. In chapter 4 I discuss resource/trade-offs (soil as a valued resource and cost), while the ethics and values of active restoration are addressed in chapter 6, because these are precisely the constraints landscape architect's juggle. Publishing in an ecology-forward journal allows rigorous testing and peer scrutiny of the intervention effects; here, this discussion translates those results for an interdisciplinary audience and situates them within design practice and theory.

By comparing pre-seeding interventions under a forb-weighted seed design, I show that topsoil removal most strongly increases native forb and grass diversity and shifts community

composition away from non-native dominance, while solarisation can improve outcomes but tends to favour grasses; thus, intervention choice can be tuned to the desired experiential and maintenance profile of the finished landscape.

The specific research questions addressed in this chapter are:

- *Q2.1 Can forb rich grasslands be restored in urban areas in south Australia?*
- *Q2.2 To what extent does topsoil removal influence the alpha diversity of native forbs and grasses compared with other pre-sowing interventions?*
- *Q2.3 How do outcomes achieved through other preparatory measures (e.g., solarisation) differ from those achieved through topsoil removal?*

Statement of

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Contribution to the Paper	Conceptualisation, setting up experiment, methodology, fieldwork, data collection, data analysis, and writing.
Overall percentage (%)	80
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.
Signature	Date: 30/11/2025

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);
ii. permission is granted for the candidate in include the publication in the thesis; and
iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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3.2 Abstract

Introduction: Native temperate grasslands are globally threatened, biodiversity-rich ecosystems. In urban post-agricultural grasslands, restoration often achieves re-establishment of grass communities, yet associated forb recovery remains limited by weed seedbanks and nutrient-enriched soils.

Objectives: We tested whether pre-sowing interventions that target biomass, seedbanks, and soil nutrients can improve native plant recovery, especially forbs, in a degraded urban post-agricultural grassland.

Methods: We ran a randomised block experiment comparing biomass removal alone (chemical fallow), soil solarisation (for 6 and 12 weeks), and topsoil removal (to 7 and 14 cm). We sowed 41 plant species via a native seed mix, and vegetation was surveyed at 6 and 18 months to quantify alpha diversity, community composition, and plant functional group responses.

Results: Topsoil removal substantially increased native and sown alpha diversity 18 months after intervention, with pronounced gains for forbs and strongly reduced non-native grass dominance. Solarisation also promoted native alpha diversity but benefits were largely confined to grasses, limiting whole-community recovery. Chemical fallow offered little improvement.

Conclusions: Restoring forb-rich grassland communities in nutrient-enriched systems requires intensive soil reset via topsoil removal. Seedbank-focused (solarisation) and biomass-only (chemical fallow) interventions were insufficient to shift plant communities away from non-native dominance.

Implications for Practice: Intensive interventions that alter soil nutrients, seedbanks, and weed biomass can overcome persistent land-use legacies in post-agricultural urban grasslands.

Where feasible, topsoil removal is a powerful option for restoring diverse and resilient native grassland communities; solarisation may provide a lower-intensity alternative when only modest gains are required.

3.3 Introduction

Restoring ecosystems is a central response to biodiversity loss, climate change, and ecosystem degradation crises (Suding et al. 2015). Global commitments, including the United Nations Decade on Ecosystem Restoration (2021-2030) (UNEP 2021), highlight the urgency of action. Although restoration agendas have largely focused on forests, grasslands cover about one-third of Earth's land surface (Bond 2019) yet remain underrepresented in restoration science and policy (Dudley et al. 2020; Bardgett et al. 2021). Native grasslands face widespread conversion to agriculture and urbanisation (White et al. 2000; Gossner et al. 2016; Dudley et al. 2020). These pressures are compounded by invasive species (Lenz & Facelli 2005) and persistent soil nutrient legacies (Prober et al. 2005; Helsen et al. 2013; Ba & Facelli 2022), which hinder the recovery of native plant communities.

Invasive exotic plant species are a major biotic barrier to effective temperate grassland restoration (Prober & Thiele 2005; Lenz & Facelli 2005; Sinclair et al. 2019; Humphries et al. 2020). In degraded landscapes, weeds compete with native plants and often dominate soil seedbanks (Morgan 1998; Cohen et al. 2019; Morris 2022), preventing re-establishment of native plant communities (Lenz & Facelli 2005; Ba & Facelli 2022). Urban grasslands pose additional challenges due to novel ecosystem dynamics and anthropogenic pressures, such as infrastructure and pollution (Fischer et al. 2013, 2012; Kowarik & von der Lippe 2018). Persistent soil nutrient legacies from agriculture (Prober et al. 2005; Parkhurst et al. 2022) and atmospheric deposition (Clark & Tilman 2008; Power & Collins 2010; Řehounková et al. 2020) further amplify these barriers. Elevated nutrients constrain restoration in seminatural grasslands across Europe (Bakker et al. 2012, Freitag et al. 2021), and in the UK (Helsen et al. 2013), US tallgrass prairies (Clark & Tilman 2008; Bakker et al. 2012; McFarlane et al. 2023), and Australian native grasslands (Prober & Thiele 2005; Sinclair et al. 2019; Gibson-Roy 2023). However, interactions between invasive plant seedbanks and nutrient enrichment remain poorly understood. Resolving these dynamics is essential for effective restoration (Resch et al. 2022).

Unlike tree-based restoration that can often be achieved through planting, grassland restoration usually requires broad-area seeding (Hedberg & Kotowski 2010). Establishment depends on overcoming two major barriers: the presence of non-native propagules and elevated soil nutrient concentrations (Prober & Thiele 2005; Parkhurst et al. 2022; Morris 2022; Gibson-Roy 2023). Approaches to mitigate these barriers can be grouped into three categories: (1) biomass removal, which removes standing vegetation but retains the soil seedbank (e.g. by burning, slashing and/or herbicide application; Gibson-Roy et al. 2010); (2) seedbank removal, which removes both biomass and seed/bud banks (e.g., by residual herbicide application, soil fumigation, solarisation, pine oil application, or soil microwaving (Hanson and Shrestha 2006, Al-Shammary et al. 2020, McQuillan et al. 2024)); and (3) soil reset, which removes biomass and seedbank while also reducing nutrients (e.g., topsoil removal; sod cutting and topsoil inversion (Gibson-Roy et al. 2010; Bakker et al. 2012)). Together, these approaches form a gradient of management intensity, from light-touch biomass suppression to full soil resets that address multiple barriers simultaneously. Yet few trials directly compare these methods in grasslands, and their effects on plant diversity remain poorly quantified. This evidence gap is critical, as restoration outcomes are highly variable (Atkinson et al. 2023; Bertuol-Garcia et al. 2023).

While topsoil removal and glyphosate application are widely used, soil solarisation has only recently gained attention as a restoration technique (Cohen et al. 2019; McQuillan et al. 2024) due to its reduced reliance on pesticides (D'Addabbo et al. 2010). This method involves covering moist soil with clear plastic during hot periods, creating a greenhouse effect in the upper soil horizons that destroys plant propagules through physical, biological, and chemical processes (Horowitz et al. 1983; Bainbridge 1990).

Soil reset through topsoil removal is often considered the most effective intervention for medium- to long-term restoration, with studies reporting high native plant cover and enhanced biodiversity outcomes (Gibson-Roy & Carland 2022; Resch et al. 2022). However, other work shows incomplete recovery of soil functions despite topsoil removal and

subsequent restoration (Taboada-Castro et al. 2009), suggesting outcomes depend on soil type and local context. Soil reset also raises environmental concerns, as topsoil is slow to regenerate and is widely regarded as a scarce resource. Topsoil removal is also resource-intensive, but in urban settings costs can sometimes be offset through partnerships with authorities, NGOs, and community groups (Török et al. 2011). Funders and managers must weigh these trade-offs by considering both the effectiveness of the method and the durability of biodiversity outcomes.

Temperate Australian grasslands are among the continent's most endangered, now reduced to 1–5% of their original extent (Barlow & Ross 2001; Turner 2012), following widespread conversion to cropping, pasture, and urban land use (Kirkpatrick 1995). Indigenous fire once maintained biodiversity, but European settlement has increased the concentration of soil nutrients and exotic cover (Morgan 1999; Gammage 2011; Pascoe 2018; Bardsley et al. 2019). With parallels to prairies and steppes worldwide, these systems are a globally relevant test case. State and Transition Models (STMs) highlight how disturbance drives persistent alternative states and when strong interventions are needed to restore native dominance (Yates & Hobbs 1997; Hobbs 2008; Briske 2017).

Here we tested three pre-sowing treatments – chemical fallow, solarisation, and topsoil removal – in a post-agricultural urban grassland. Our aim was to evaluate how these interventions influence seeded vegetation under conditions of persistent soil legacies and invasive pressure. Specifically, we asked how treatments affected: (i) overall plant community composition, (ii) performance of sown native plant species, (iii) the balance between native and exotic plant species, and (iv) plant life-form responses (grasses, forbs, shrubs) at 6 and 18 months post-sowing. We hypothesised that (a) topsoil removal would most strongly enhance native richness, particularly forb recovery, by reducing nutrient legacies and weed seedbanks; (b) solarisation would improve establishment but primarily favour grasses, by reducing competition from shallow weed seedbanks while leaving nutrient legacies intact; and (c)

chemical fallow would provide limited benefits for native recovery, by suppressing aboveground weeds without reducing nutrient legacies or the soil seedbank.

3.4 Methods

3.4.1 Study site

The native temperate grassland restoration field experiment was set up in an urban national park (35.06022S, 138.54983E) in Kaurna Country/Adelaide, South Australia. The site sits at ~180 m above sea level and receives an average of 550 – 650 mm of rainfall annually. The soils underlying the site are Black Vertosols and Black Dermasols (Australian soil classification (Isbell 2021)), both with medium to heavy clay contents throughout the soil profile. The site is on former agricultural land, and the post-agricultural vegetation is dominated by non-native pasture species and invasive exotic plants. Historical records and sketches indicate that the area was originally covered by grassland or sparsely wooded grassy woodland and was maintained through cultural burning by the Kaurna people (pers. comm. Richard Woods). All fieldwork was conducted under a Scientific Research Permit (Q27266-1) issued by the Department for Environment and Water, South Australia.

3.4.2 Experimental design

The experimental design consists of a randomised controlled field experiment, with each experimental block randomised across the study area (Figure 2 and Table 1) and each treatment randomised within the block (Figure 2 and Table 1). Each treatment plot measured 5 × 5 m (25 m²), with three permanent 1 m² quadrats systematically established within each plot for vegetation monitoring. There were five treatments:

3.4.2.1 Biomass removal (control)

To evaluate grassland plant establishment without nutrient or seedbank manipulation, only the standing biomass was removed using glyphosate at the manufacturer-recommended rate of 0.54 g/m² for the control at six and two weeks prior to seeding.

3.4.2.2 Topsoil removal

To reduce soil fertility and remove a portion of the soil seedbank, topsoil removal treatments were implemented in mid-May 2023 by a licensed contractor using a 1.5-tonne excavator with a jig used to validate the depths at 7 cm and 14 cm. The excess soil was spread away from the experimental treatments.

3.4.2.3 Soil solarisation

To manipulate the soil seedbank, the soil was solarised using clear, 210-micron-thick polyethylene (PE) plastic as described by McQuillan et al. (2024). In brief:

- 6-Week Solarisation: PE plastic was placed over the soil in the middle of summer in January 2023 and removed after 6 weeks.
- 12-Week Solarisation: PE plastic was placed over the soil in the middle of summer in January 2023 and removed after 12 weeks.

The solarisation treatments also had glyphosate applied two weeks prior to seeding. No herbicide was applied to the topsoil removal treatments as no biomass was present.

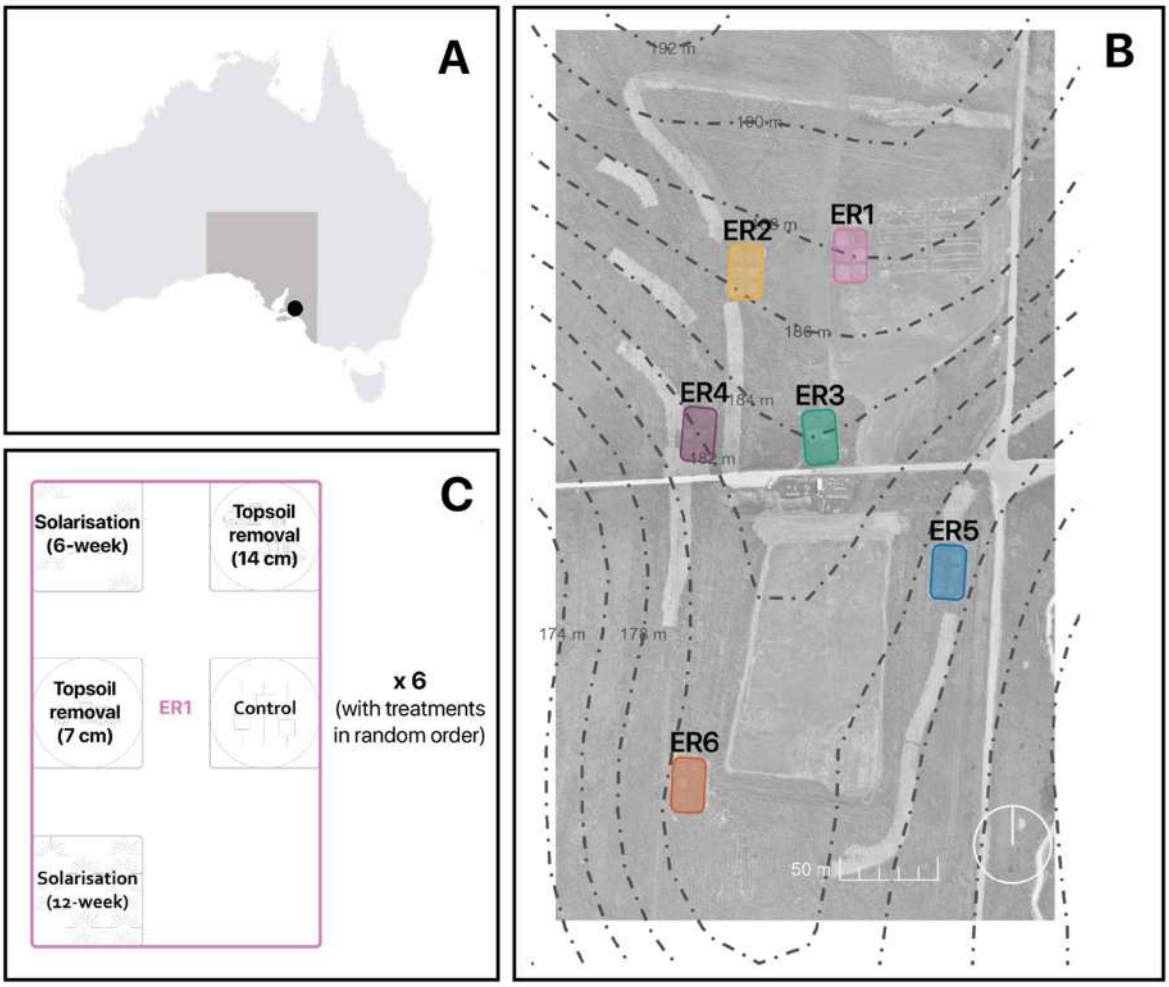

Figure 1. Site map and experimental treatment arrangement. A: Location in Australia B: Study area layout showing randomised controlled replicate blocks. C: Treatment allocation within blocks. Within each block, five treatments were randomly assigned to plots. The treatments included: (1) control (biomass removal with glyphosate only), (2) 6-week soil solarisation, (3) 12-week soil solarisation, (4) 7 cm deep topsoil removal, and (5) 14 cm deep topsoil removal.

Table 1. Summary of treatments applied in the grassland restoration experiment, including treatment descriptions, justifications, herbicide application details, and example photographs.

Treatment	Description	Justification	Example Photo
Control	Biomass removed with glyphosate applied 6 and 2 weeks prior to seeding; no additional treatments applied.	To evaluate grassland plant establishment without nutrient or seedbank manipulation.	
Solarisation (6 weeks)	Moist soil covered with clear polyethylene (PE) plastic for 6 weeks during summer (January–February); glyphosate applied 2 weeks before seeding.	To manipulate the soil seedbank; solarisation thermally reduces the seedbank and lowers levels of soil pathogens.	
Solarisation (12 weeks)	Moist soil covered with clear PE plastic for 12 weeks during summer (January–March); glyphosate applied 2 weeks before seeding.	To enhance soil seedbank manipulation; extended solarisation increases the thermal destruction of the seedbank and pathogens.	
Topsoil Removal (7 cm)	Excavation of topsoil to a depth of 7 cm.	To reduce soil fertility and remove a portion of the soil seedbank.	
Topsoil Removal (14 cm)	Excavation of topsoil to a depth of 14 cm.	To further reduce soil fertility and remove a larger portion of the soil seedbank.	

3.4.3 Seeding

Seed was supplied by Seeding Natives, who manage seed orchards for native grassland plants. A single experimental seed mix of 41 species was used across all plots (Table S1); one master batch was prepared, mechanically mixed thoroughly to ensure compositional consistency, and divided equally across the 30 treatment plots at an application rate of 4.3 g/m². Seeds were mixed with sawdust to prevent clustering and hand cast on 6 June 2023. Seed viability was not formally tested prior to sowing; species absent or poorly represented in the survey data should therefore be interpreted with this in mind, as low establishment may reflect seed quality rather than treatment effects alone. Post-seeding treatment consisted of monthly mowing with biomass removal (between September 2023 to February 2024) during the first growing season and selective dabbing for target species (*Phalaris aquatica*, *Silybum*

marianum and *Helminthotheca echioides*). In the second growing season, no biomass removal was employed.

3.4.4 Vegetation monitoring

To examine treatment effects on vegetation communities, vegetation was surveyed using 1 m² quadrats at T_1 (six months post seeding in November 2023) and at T_2 (18 months post seeding in November 2024), with three quadrats systematically installed in each treatment replicate, resulting in 90 quadrats. Each quadrat was marked with steel pegs for precise revisit monitoring. All plants within the quadrat were included in the survey. Each species was given a field name if unknown and later matched with botanical names with help from knowledgeable ecologists. Native status was assigned accordingly, and the dataset includes both native and non-native species and the count of individual plants of each species recorded within each quadrat.

3.4.5 Study site environmental conditions

Environmental variables, including soil moisture, soil temperature, ambient temperature, and rainfall, were monitored throughout the study to provide context for treatment outcomes (Figure S1). Soil moisture, soil temperature, and ambient temperature were recorded continuously using TMS sensors (TOMST, Czechia; Wild et al. 2019), installed in each of the six 7 cm topsoil removal plots; data were averaged across sensors for analysis. Rainfall data were obtained from the Bureau of Meteorology (BoM) Happy Valley Reservoir station. Soil moisture fluctuated seasonally, reflecting rainfall patterns, while soil temperature followed predictable seasonal trends, with peaks in summer months. Ambient temperature mirrored soil temperature patterns, and rainfall exhibited clear seasonality, with distinct wet and dry periods.

Between the first (T_1) and second (T_2) sampling dates, an unseasonally wet period occurred over summer, with two significant rainfall events (~40 mm each) recorded within a short timeframe. This resulted in a proportionate increase in soil moisture during a period

typically characterised by dry conditions. Such summer rainfall events are known to favour the germination of native C_4 grasses (Hattersley 1983; McDougall 1989) and summer ruderal weeds. These anomalous conditions may have influenced vegetation dynamics and treatment responses observed at T_2.

3.4.6 Statistical analysis

All analyses were performed using R (version 4.3.3) (R Core Team 2025), with the following packages: vegan (Oksanen et al. 2022) for NMDS, PERMANOVA, and PERMDISP analyses; pairwiseAdonis (Martinez Arbizu 2017) for pairwise PERMANOVA; rstatix (Kassambara 2023a) for Kruskal-Wallis and Wilcoxon tests; and multcompView (Graves et al. 2024) for compact letter displays.

3.4.6.1 Alpha diversity analysis

Alpha diversity was calculated using the inverse Simpson index to assess the richness and evenness of plant species within each treatment group at two time points (T_1 and T_2). The analysis was conducted separately for: (1) all plants, (2) native plants (sown + germinated from soil seedbank), (3) sown plants, and (4) non-native plants. To compare alpha diversity across the five treatments, a non-parametric Kruskal-Wallis test was used due to the lack of homoscedasticity, which is a key assumption for parametric tests like ANOVA. If the Kruskal-Wallis test indicated significant differences, pairwise comparisons were performed using Wilcoxon rank sum tests to determine which treatment groups differed from each other. This approach was applied to both T_1 and T_2 time points for each plant group.

3.4.6.2 Community composition analysis

Non-metric multidimensional scaling (NMDS) was used to assess the community composition of plants across treatments at both T_1 and T_2. NMDS is an ordination method that visualises the similarity or dissimilarity of species composition between treatments in a lower-dimensional space. The analysis was performed separately for: (1) all plants, (2) native plants, (3) sown plants, and (4) non-native plants. The stress value of each NMDS plot was

calculated to assess the quality of the ordination, with lower stress values (<0.2) indicating a better fit of the data to the two-dimensional space.

Differences in community composition among treatments were tested using permutational multivariate analysis of variance (PERMANOVA) based on Bray–Curtis dissimilarities with 999 permutations. Pairwise PERMANOVA tests were used where overall differences were significant, and p-values were adjusted with the Benjamini–Hochberg method. Homogeneity of dispersion was evaluated with a permutation test to distinguish differences in community location from differences in spread.

3.4.7 Plant type analysis

We assessed plant abundance at two time points (T_1 and T_2) using generalized linear models (GLMs) with a Poisson distribution. Four separate models were run for: (1) all plants, (2) native plants, (3) sown plants, and (4) non-native plants. In each model, plant type was included as a factor with three levels—grasses, forbs, and shrubs (with forbs serving as the reference level).

We tested the effects of each of the five treatments (topsoil removal at 7 cm, topsoil removal at 14 cm, solarisation for 6 weeks, solarisation for 12 weeks, and a control) on plant abundance. Treatment effects and their interactions with plant type were evaluated using z-values and p-values to determine statistical significance. This approach allowed us to assess overall treatment impacts as well as differences in response among plant types at both T_1 (6 months post-sowing) and T_2 (18 months post-sowing).

3.4.8 Data visualisation

Figures were produced using the tidyverse (Wickham et al. 2019), cowplot (Wilke 2024), ggpubr (Kassambara 2023b), gridExtra (Auguie 2017), and viridis (Garnier et al. 2024).

3.4.9 Significance testing

All statistical tests were conducted at an alpha level of 0.05. Where applicable, multiple comparisons were adjusted to control for Type I error. Tables S2-S4 report the detailed results of Kruskal-Wallis tests, Wilcoxon rank sum tests, GLM results, and post hoc tests for each comparison.

3.5 Results

3.5.1 Plant alpha diversity at six months (T_1) and 18 months (T_2) post-sowing

3.5.1.1 Native and sown plant alpha diversity

Six months after seeding (T_1), alpha diversity for native and sown plants differed amongst treatments (native: Kruskal-Wallis: $\chi^2 = 41.5$, df = 4, $p < 0.001$, Table S2A and Figure 2A; sown: Kruskal-Wallis: $\chi^2 = 47.3$, df = 4, $p < 0.001$, Table S2A and Figure 2C). For native species, alpha diversity for the topsoil removal treatments (7 cm and 14 cm) was higher than for the control (Wilcoxon rank sum tests: control vs topsoil removal 7 cm, $p < 0.001$; control vs topsoil removal 14 cm, $p < 0.001$, Table S3A and Figure 2A). For sown species, alpha diversity for the topsoil removal treatments (7 cm and 14 cm) was also higher than for the control (Wilcoxon rank sum tests: control vs topsoil Removal 7 cm, $p < 0.001$; control vs topsoil removal 14 cm, $p < 0.001$, Table S3B and Figure 2C). The 7 cm and 14 cm treatments had similar effects to each other (native: Wilcoxon rank sum test: $p = 0.338$; Table S3A and Figure 2A; sown: Wilcoxon rank sum test: $p = 0.352$; Table S3B and Figure 2C). In contrast, alpha diversity for the solarisation treatments (6-week and 12-week solarisation durations) did not differ from that of the control or from each other for either native or sown species (Wilcoxon rank sum test: $p > 0.05$ in all cases; Table S3A & S3B).

By 18 months post-seeding (T_2), native plant alpha diversity had either increased or remained stable across all treatments (Kruskal-Wallis: $\chi^2 = 45.6$, df = 4, $p < 0.001$, Table S2B). The topsoil removal treatments continued to maintain higher diversity than the 6-week solarisation and control treatments (Figure 2B; Table S3D). The 12-week solarisation increased in native alpha diversity compared to the 6-week solarisation treatment (Wilcoxon

rank sum tests: solarisation 12 weeks vs solarisation 6 weeks, $p = 0.046$), and the control (Wilcoxon rank sum tests: control vs solarisation 12, $p = 0.002$; Table S3D). However, solarisation remained less effective than topsoil removal treatments at increasing native plant alpha diversity, with differences between both solarisation durations and both topsoil removal depths remaining (Wilcoxon rank sum tests: $p < 0.05$ in both cases; Table S3D). The control treatment continued to show low diversity (Figure 2B).

For sown species, alpha diversity patterns closely mirrored those observed in native species. By T_2, the Kruskal–Wallis test indicated significant differences among treatments ($\chi^2 = 47.0$, $df = 4$, $p < 0.001$; Table S2B). Wilcoxon tests showed that topsoil removal treatments supported the highest sown species diversity, with no difference between the 7 cm and 14 cm removal depths ($p = 0.448$; Table S3E). Solarisation treatments exhibited moderate increases, with the 12-week treatment showing a higher sown species diversity compared to the control treatment (Wilcoxon rank sum tests: $p < 0.001$). However, sown species diversity in solarisation treatments remained lower than in both topsoil removal depths (Wilcoxon rank sum tests: $p < 0.001$). Control plots maintained the lowest sown species diversity.

3.5.1.2 All plant species alpha diversity

For all species combined (native + non-native plants), alpha diversity differed across treatments at T_1 (Kruskal-Wallis: $\chi^2 = 23.9$, $df = 4$, $p = <0.001$, Table S2A), with topsoil removal treatments exhibiting higher diversity compared to control and solarisation treatments ($p < 0.05$ in all cases, Table S3C). At T_2, however, no differences in alpha diversity were observed across treatments (Kruskal-Wallis: $\chi^2 = 8.1288$, $df = 4$, $p = 0.0870$, Table S2B), indicating that the initial treatment effects on overall diversity were not maintained over time.

3.5.1.3 Non-native plants alpha diversity

No differences in non-native species alpha diversity were detected across treatments at either T_1 (Kruskal-Wallis: $\chi^2 = 5.38$, $df = 4$, $p = 0.251$, Table S2A) or T_2 (Kruskal-Wallis: $\chi^2 = 2.24$,

df = 4, p = 0.692, Table S2B), indicating that the treatments did not differentially affect non-native plant alpha diversity.

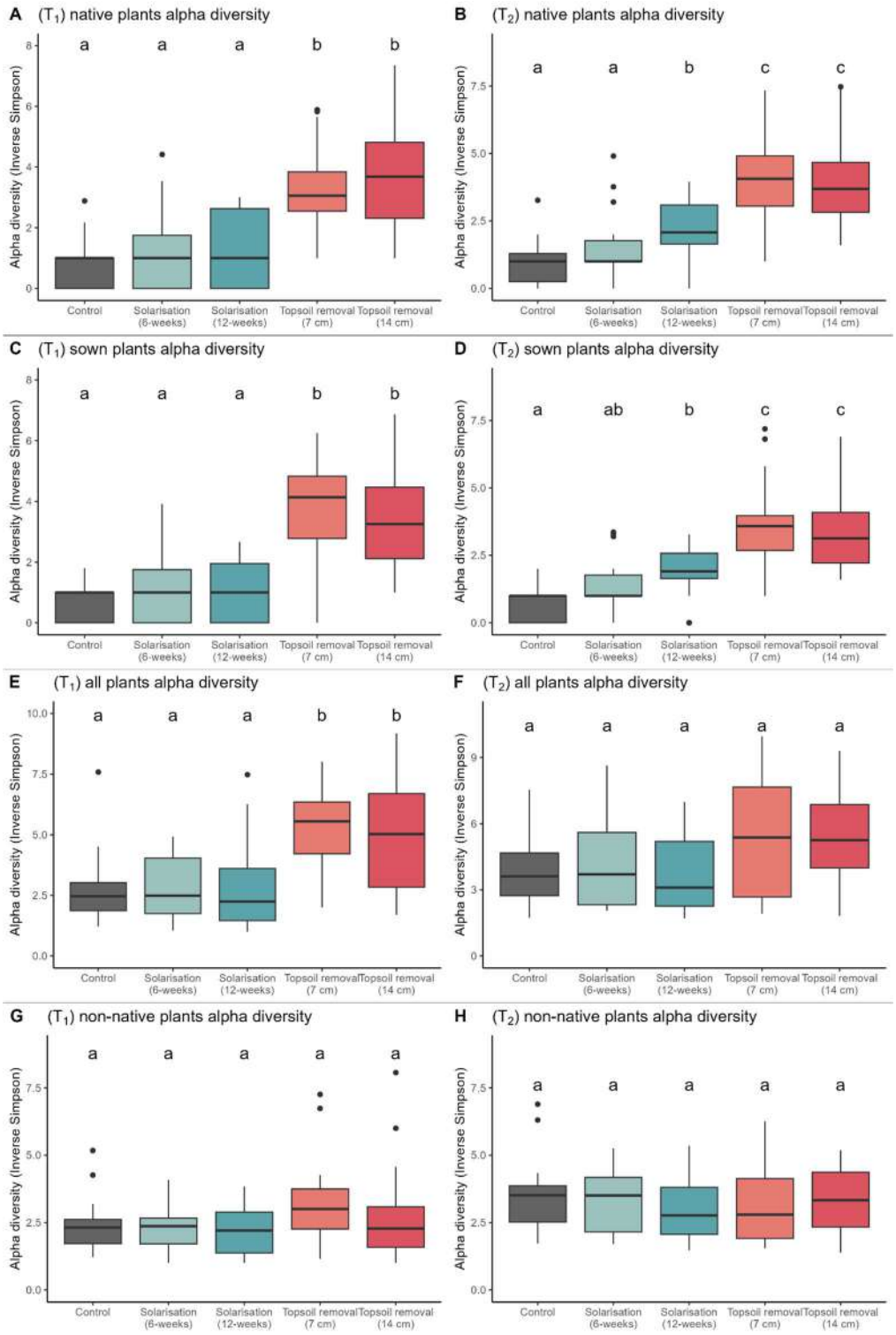

Figure 2: Plant alpha diversity across treatments at six months (T_1) and 18 months (T_2) post-sowing. (A) and (B) show alpha diversity of native plants at T_1 and T_2, respectively. (C) and (D) display alpha diversity of sown native plants at T_1 and T_2, respectively. (E) and (F) present alpha diversity of all plants (native + non-native) at T_1 and T_2, respectively. (G) and (H) depict alpha diversity of non-native plants at T_1 and T_2, respectively. Treatments include

control, solarisation (6-week and 12-week durations), and topsoil removal (7 cm and 14 cm depths).

3.5.2 Comparing plant community composition at six months (T_1) and 18 months post-sowing (T_2)

3.5.2.1 Native plants

At T_1, PERMANOVA indicated differences in native plant community composition (df = 4, F = 3.99, R^2 = 0.200, p = 0.001; Figure S2A; Table S4A), explaining 20% of the variation in community composition. Dispersion analysis showed differences in the spread of samples among treatments (df = 4, F = 3.01, p = 0.024; Table S5A), suggesting that variation within treatments influenced the results.

The NMDS ordination (Figure S1A, stress = 0.148) showed minimal differentiation between the control and solarisation treatments, with topsoil removal treatments (7 cm and 14 cm) more distinct. Pairwise PERMANOVA tests identified differences between topsoil removal treatments and the control (topsoil removal 7 cm vs control: p = 0.003; topsoil removal 14 cm vs control: p = 0.012; Table S6A).

At T_2, PERMANOVA confirmed differences in native plant community composition (df = 4, F = 4.73, R^2 = 0.202, p = 0.001; Table S4B), but the effect was less pronounced. Dispersion analysis at T_2 showed no differences in dispersion across treatments (Table S5B). Pairwise comparisons showed differences between topsoil removal treatments and the control (topsoil removal 7 cm vs control: p = 0.0025; topsoil removal 14 cm vs control: p = 0.0025; Table S6E).

3.5.2.2 Sown plants

At T_1, PERMANOVA showed differences in sown plant community composition (df = 4, F = 3.04, R^2 = 0.162, p = 0.001; Figure S2C; Table S4A), with treatment explaining 16.2% of the variation. Dispersion analysis indicated equal spread of sample variance (df = 4, F = 2.03, p = 0.101; Table S5A), suggesting that the observed differences were not influenced by dispersion. The NMDS ordination (Figure S1C, stress = 0.136) shows greater differences between the topsoil removal treatments and the control. Pairwise comparisons confirmed differences between topsoil removal treatments and the control (topsoil removal 7 cm vs

control: p = 0.003; topsoil removal 14 cm vs control: p = 0.005; Table S6B), but not each other (topsoil removal 7 cm vs topsoil removal 14 cm: p = 0.772; Table S6B); while solarisation treatments showed no differences compared to the control or each other (solarisation 6 weeks vs control: p = 0.385; solarisation 12 weeks vs control: p = 0.308; solarisation 6 weeks vs solarisation 12 weeks: p = 0.691; Table S6B).

At T_2, PERMANOVA indicated continued differences in sown plant community composition (df = 4, F = 4.38, R^2 = 0.193, p = 0.001; Table S4B). Dispersion analysis indicated equal dispersion (Table S5B). Pairwise comparisons identified differences between topsoil removal treatments and the control (topsoil removal 7 cm vs control: p = 0.003; topsoil removal 14 cm vs control: p = 0.003; Table S6F) and both solarisation treatments ($p < 0.05$ in all cases), but not each other (topsoil removal 7 cm vs topsoil removal 14 cm: p = 0.608), while solarisation treatments showed no differences compared to the control (solarisation 6 weeks vs control: p = 0.120; solarisation 12 weeks vs control: p = 0.120; Table S6F), or each other (solarisation 6 weeks vs solarisation 12 weeks: p = 0.394).

3.5.2.3 All plants (native + non-native)

At T_1, PERMANOVA revealed differences in community composition across treatments (df = 4, F = 4.86, R^2 = 0.186, p = 0.001; Figure S2E; Table S4A), with treatment explaining 18.6% of the variation. Dispersion analysis indicated an even spread of samples across treatments (df = 4, F = 1.03, p = 0.40; Table S5A). The NMDS ordination at T_1 (Figure S2E, stress = 0.183) shows minimal differentiation between control and solarisation treatments, with topsoil removal treatments (7 cm and 14 cm) more distinct. Pairwise comparisons confirmed differences between topsoil removal treatments and the control (topsoil removal 7 cm vs control: p = 0.002; topsoil removal 14 cm vs control: p = 0.002; Table S6C), as well as each other (topsoil removal 7 cm vs topsoil removal 14 cm: p = 0.030; Table S6C); while solarisation treatments showed no differences compared to the control or each other

(solarisation 6 weeks vs control: p = 0.333; solarisation 12 weeks vs control: p = 0.173; solarisation 6 weeks vs solarisation 12 weeks: p = 0.594; Table S6C).

At T_2, PERMANOVA continued to indicate differences in community composition (df = 4, F = 2.65, R^2 = 0.11088, p = 0.001; Table S4B). The NMDS ordination at T_2 (Figure S2F, stress = 0.202) showed increased overlap among all treatments. Dispersion analysis indicated an even spread of samples across treatments (df = 4, F = 0.318, p = 0.865; Table S5B). Pairwise comparisons confirmed differences between topsoil removal treatments and the control (topsoil removal 7 cm vs control: p = 0.012; topsoil removal 14 cm vs control: p = 0.010; Table S6G), but not each other (topsoil removal 7 cm vs topsoil removal 14 cm: p = 0.099; Table S6G), while solarisation treatments showed no differences compared to the control or each other (solarisation 6 weeks vs control: p = 0.697; solarisation 12 weeks vs control: p = 0.697; solarisation 6 weeks vs solarisation 12 weeks: p = 0.887; Table S6G).

3.5.2.4 Non-native plants

For non-native plants at T_1, PERMANOVA indicated differences in community composition (df = 4, F = 2.86, R^2 = 0.119, p = 0.001; Table S4A; Figure S2G), with treatment explaining 11.9% of the variation. Dispersion analysis indicated differences in the spread of samples (df = 4, F = 2.67, p = 0.0377; Table S5A), suggesting that variation in sample spread may have influenced the results. The NMDS ordination at T_1 (Figure S2G, stress = 0.169) showed minimal differentiation between control and solarisation, with topsoil removal treatments more distinct. Pairwise comparisons confirmed differences between solarisation and topsoil removal treatments (solarisation 6 weeks vs topsoil removal 7 cm: p = 0.012; solarisation 6 weeks vs topsoil removal 14 cm: p = 0.010; Table S6D).

At T_2, the NMDS ordination (Figure 4F) showed near-complete overlap among treatments, indicating reduced differentiation. However, PERMANOVA still showed differences (df = 4, F = 2.18, R^2 = 0.09293, p = 0.002; Table S4B). Dispersion analysis at T_2 indicated an even spread across treatments (Table S5B), indicating stabilisation in community

structure. Pairwise comparisons confirmed differences between topsoil removal treatments and the control (topsoil removal 7 cm vs control: $p = 0.045$; topsoil removal 14 cm vs control: $p = 0.020$; Table S6H), but not each other (topsoil removal 7 cm vs topsoil removal 14 cm: $p = 0.102$; Table S6H); while solarisation treatments showed no differences compared to the control or each other (solarisation 6 weeks vs control: $p = 0.783$; solarisation 12 weeks vs control: $p = 0.796$; solarisation 6 weeks vs solarisation 12 weeks: $p = 0.789$; Table S6H).

3.5.3 Treatment effects on plant type and abundance

3.5.3.1 Native Species Abundance

At T1 (6 months post-sowing):

For native plants, topsoil removal increased the abundance of forbs compared to the control. In the native-only model (GLM: Table S7A), topsoil removal of 7 cm showed a strong positive effect (Estimate = 3.05, $z = 12.6$, $p < 0.001$) and topsoil removal of 14 cm also increased abundance (Estimate = 1.68, $z = 6.56$, $p < 0.001$). Solarisation treatments did not have any effects (solarisation 6 weeks: $p = 1.00$; solarisation 12 weeks: $p = 0.056$), and there was no difference between forbs and grasses (grass main effect: $p = 0.732$).

At T2 (18 months post-sowing):

The positive effect of topsoil removal on native forbs persisted over time. In the native model at T_2 (GLM: Table S7E), topsoil removal of 7 cm (Estimate = 2.94, $z = 9.93$, $p < 0.001$) and of 14 cm (Estimate = 2.96, $z = 10.0$, $p < 0.001$) both increased abundance relative to the control. Solarisation treatments had no effect. Notably, however, interactions with the grass plant type show that native grasses responded differently than forbs: while the main effect of grass was not significant, the interaction terms for solarisation (6 and 12 weeks) with grass were positive ($p = 0.048$ and $p = 0.008$, respectively), and the interaction between topsoil removal (7 cm) and grass was negative (Estimate = –0.757, $p = 0.040$). This suggests that while topsoil removal greatly boosted native forb abundance, grasses showed a smaller increase relative to forbs.

3.5.3.2 Sown species abundance

At T1 (6 months post-sowing)

In the model for sown plants (GLM: Table S7B), topsoil removal also increased the abundance of sown forbs. Specifically, removal of 7 cm (Estimate = 2.80, z = 5.44, $p < 0.001$) and 14 cm (Estimate = 2.42, z = 4.64, $p < 0.001$) both had strong positive effects. Solarisation treatments did not affect sown forb abundance.

Grasses showed a higher overall abundance than forbs, which is reflected in the positive main effect (Estimate = 1.39, z = 2.48, $p = 0.013$). This means that, regardless of treatment, grasses tend to be more abundant than forbs. However, because the interactions between treatment and grass were not significant, the treatments did not change the pattern of response for grasses differently from forbs. In other words, while grasses start off more abundant, both grasses and forbs responded similarly to the treatments applied.

At T2 (18 months post-sowing):

For sown species (GLM: Table S7F), the strong positive effect of topsoil removal on the reference forb group persisted. Removal of 7 cm (Estimate = 3.36, z = 5.71, $p < 0.001$) and 14 cm (Estimate = 3.81, z = 6.52, $p < 0.001$) both increased forb abundance. Solarisation treatments again did not influence sown forb abundance compared to the control.

Grasses were generally more abundant than forbs (grass main effect: Estimate = 2.037, z = 3.318, $p < 0.001$). However, with 14 cm of topsoil removal, the increase in abundance was significantly less for grasses than for forbs (interaction: Estimate = –1.39, z = –2.22, $p = 0.026$). This means that while topsoil removal boosted forb numbers, grasses did not benefit as much.

3.5.3.3 Non-native species abundance

At T1 (6 months post-sowing):

For non-native plants (GLM: Table S7D), solarisation treatments produced contrasting effects at T_1. Solarisation for 6 weeks increased non-native forb abundance (Estimate = 0.402, z = 3.22, p = 0.001), whereas solarisation for 12 weeks reduced it (Estimate = –0.866, z = –4.88, p < 0.001). Topsoil removal of 7 cm led to a modest increase in forbs (Estimate = 0.297, z = 2.33, p = 0.020) while removal of 14 cm was not different from the control.

Our goal was to reduce the abundance of non-native plants. We found that non-native grasses were initially much more abundant than non-native forbs across all treatments (grass main effect: Estimate = 1.83, z = 17.5, p < 0.001; Table S7D). Importantly, however, the treatments reduced non-native grass abundance more than they did non-native forbs. For example, solarisation for 6 weeks had a negative effect on grasses (interaction: Estimate = –0.589, p < 0.001). Topsoil removal of 7 cm also decreased grass abundance (interaction: Estimate = –1.31, p < 0.001), as did topsoil removal of 14 cm (interaction: Estimate = –0.954, p < 0.001). In simple terms, although non-native grasses were more abundant to begin with, topsoil removal treatments were especially effective at lowering their numbers relative to non-native forbs.

At T2 (18 months post-sowing):

In the T_2 model for non-native species (GLM: Table S7H), solarisation for 6 weeks had a small but positive effect on forbs (Estimate = 0.116, z = 2.04, p = 0.041), while solarisation for 12 weeks no longer had an impact on forb abundance. Interestingly, both topsoil removal treatments increased non-native forb abundance (7 cm: Estimate = 0.389, z = 7.27, p < 0.001; 14 cm: Estimate = 0.354, z = 6.56, p < 0.001).

At T_2, non-native grasses were still more abundant than non-native forbs overall (grass main effect: Estimate = 1.16, z = 24.4, p < 0.001). However, topsoil removal treatments had a stronger negative impact on grasses. Specifically, the interaction between grass and 7 cm topsoil removal was significantly negative (Estimate = –0.836, p < 0.001), as was the interaction with 14 cm removal (Estimate = –0.944, p < 0.001). This means that although

topsoil removal increased non-native forb abundance, it was effective at reducing non-native grass abundance.

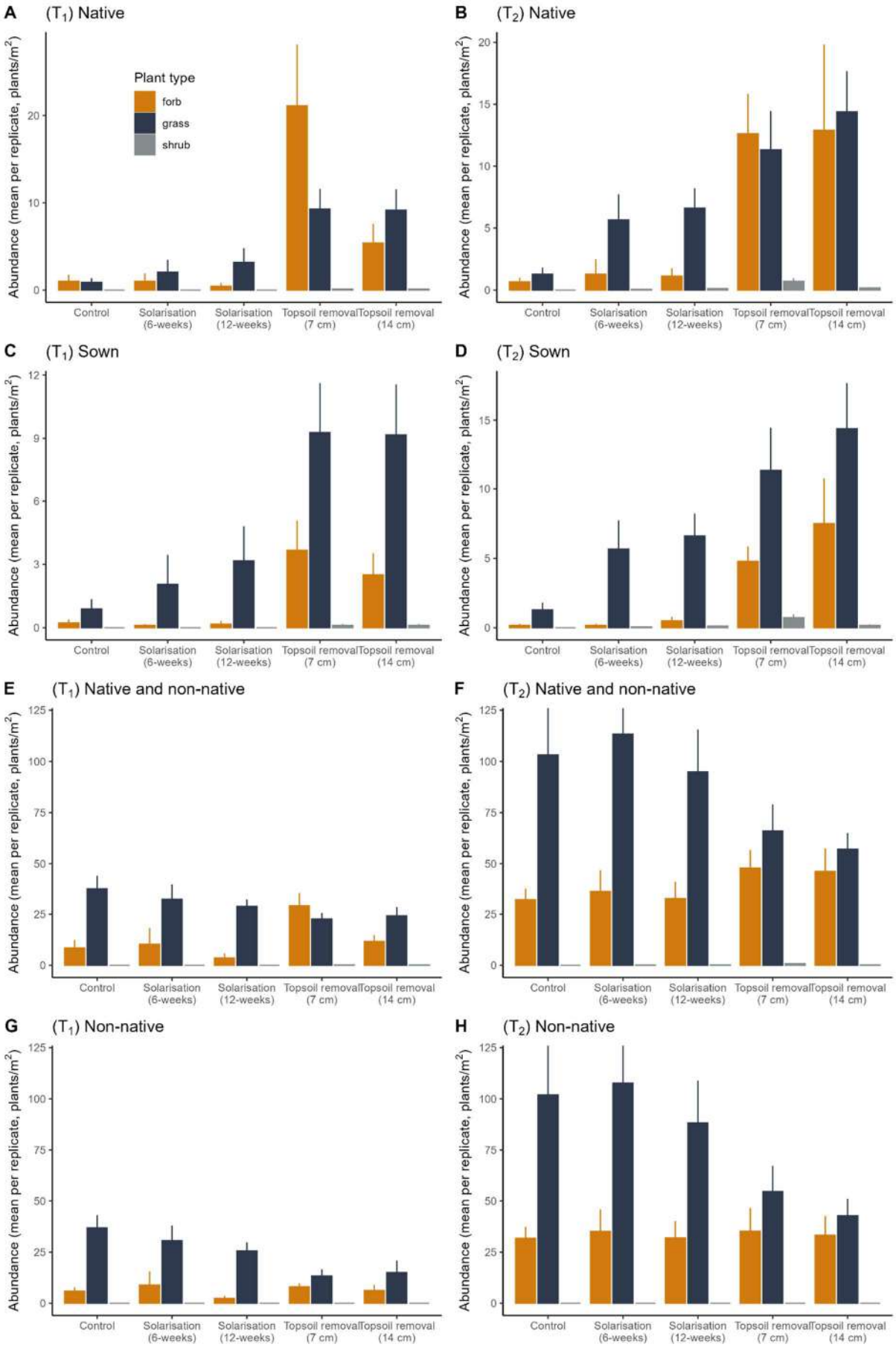

Figure 3: Mean abundance of plant functional groups (forbs, grasses, shrubs) (plants/m²) under different soil treatments at (T_1) 6 months and (T_2) 18 months post-sowing. (A, B) Abundance of native plants at T_1 and T_2 respectively. (C, D) Abundance of sown native plants at T_1 and T_2 respectively. (E, F) Abundance of all plants (native + non-native) plants at T_1 and T_2 respectively. (G, H) Abundance of non-native plant types at T_1 and T_2 respectively. Treatments include control, solarisation (6-week and 12-week durations), and topsoil removal (7 cm and 14 cm depths). Colours indicate plant functional groups: forbs (orange), grasses (blue), and shrubs (gray). Error bars represent the standard error (SE).

3.6 Discussion

Restoring highly modified grassland ecosystems - whether in urban parks, former agricultural land, or other human-dominated landscapes - often requires targeted interventions to overcome entrenched soil legacies, competition from persistent seedbanks, and nutrient enrichment. Yet it remains unclear how far these systems must be pushed to cross establishment thresholds for diverse native species. In this study, we experimentally compared a gradient of pre-sowing interventions ranging from biomass suppression to full topsoil removal. We found that topsoil removal generated the greatest gains in native plant diversity, especially for forbs, whereas solarisation offered moderate improvements largely limited to grasses. Although tested in an Australian temperate grassland context, these findings offer practical guidance for restoring degraded grasslands worldwide, particularly those burdened by long-lasting agricultural impacts.

3.6.1 Topsoil removal boosts native species abundance, especially forbs

Topsoil removal significantly increased native species abundance, with strong effects on sown forbs at both 7 cm and 14 cm depths. The largest gains occurred at 14 cm, though both depths were effective at 6 and 18 months. By lowering soil nutrients, topsoil removal reduces the competitive advantage of non-native species and facilitates native establishment (Prober & Thiele 2005; Lenz & Facelli 2005; Sinclair et al. 2019). These results support observational studies showing topsoil removal as one of the most effective interventions for promoting native diversity and abundance (Gibson-Roy & Carland 2022), particularly in post-agricultural and urban landscapes where non-native propagules persist (Fischer et al. 2013; Kowarik & von

der Lippe 2018). However, aboveground recovery often precedes belowground recovery, with soil biota and functions taking decades to approach reference conditions (Frouz et al. 2009; Resch et al. 2019, 2022). Long-term monitoring and, where appropriate, complementary soil interventions such as inoculation may therefore be required (Wubs et al. 2016, Peddle et al. 2025).

3.6.2 Topsoil removal reduces non-native grass dominance

Topsoil removal shifted plant community composition by reducing non-native grass abundance at both 7 cm and 14 cm depths. Reducing dominance of non-native species is critical for grassland restoration, as it allows native grasses and forbs to establish and persist. These results align with evidence that soil nutrient legacies facilitate non-native dominance (Prober and Thiele 2005) and with studies showing that topsoil removal effectively reduces exotic and promotes native plant recovery (Gibson-Roy et al. 2010; Sinclair et al. 2019; Gibson-Roy & Carland 2022). In contrast, solarisation produced weaker shifts: non-natives remained dominant, and while some native grasses established, forbs did not - underscoring the greater effectiveness of topsoil removal in reshaping community composition.

3.6.3 Topsoil removal enhances plant alpha diversity

We show that topsoil removal increased alpha diversity of both native and sown plant species in temperate grassland restoration. Native alpha diversity was higher following both 7 cm and 14 cm removal after 18 months, with no difference between depths. Enhancing alpha diversity, rather than only native cover, is critical for restoration success because it underpins ecosystem resistance and function (Isbell et al. 2015; Shackelford et al. 2021; Bertuol-Garcia et al. 2023). These results support earlier work showing that topsoil removal improves plant diversity by reducing nutrient availability, which favours native over exotic species (Gibson-Roy et al. 2010; Török et al. 2011; Řehounková et al. 2020). Our findings confirm topsoil removal as one of the most effective strategies for enhancing diversity in degraded temperate

grasslands, consistent with both experimental and observational studies (Gibson-Roy & Carland 2022).

3.6.4 Stable non-native diversity indicates other constraints on native establishment

We found no change in non-native species diversity, even as native alpha diversity increased. This suggests that native establishment is constrained more by soil nutrient levels and competitive dynamics than by the presence of non-natives alone. Our findings align with evidence that high native and non-native diversity can co-occur in novel urban ecosystems (Fischer et al. 2013) and that soil conditions strongly influence restoration trajectories (Prober & Thiele 2005; Parkhurst et al. 2022). These results add to growing recognition that plant–plant and plant–soil interactions are central to restoration outcomes (Lenz & Facelli 2005; Guerin et al. 2019; Ba & Facelli 2022; Resch et al. 2022).

3.6.5 Solarisation shows limited effectiveness in reducing non-native species

We show that solarisation had no effect on overall native diversity under the 6-week treatment, despite it being proposed as an effective way of suppressing non-native seedbanks. The 12-week treatment did increase native diversity, likely because it reduced non-native grasses and forbs at T_1 and allowed native perennial grasses to establish. However, solarisation remained less effective than topsoil removal in reducing non-native abundance and enhancing native diversity at T_2. These findings contrast with earlier studies suggesting solarisation should strongly suppress non-natives (Horowitz et al. 1983; Bainbridge 1990) and highlight the importance of environmental conditions such as soil moisture and solar intensity (McQuillan et al. 2024). Solarisation may therefore be most effective when optimised for local conditions or combined with other interventions (D'Addabbo et al. 2010; McQuillan et al. 2024).

Control plots consistently underperformed, underscoring the necessity of active restoration. More broadly, our results show that long-term success depends not only on suppressing non-natives but also on addressing soil conditions and plant interactions. This shifts the focus from weed control to fostering diverse, resilient plant communities – an urgent

need given the global decline of temperate grasslands and their importance for biodiversity and ecosystem services (Bardgett et al. 2021).

Our results show that pre-sowing treatments differ in both intensity and the barriers they address. Topsoil removal, the most intensive option, reduces nutrients, removes most of the seedbank, and eliminates aboveground biomass, producing the strongest plant community shifts. Solarisation targets seedbanks but leaves nutrient legacies intact, while glyphosate only suppresses standing biomass. This gradient explains the outcomes observed: topsoil removal delivered the greatest gains, solarisation moderate improvements, and fallow minimal effects. These findings support State and Transition Models, which stress that effective restoration depends on how strongly interventions push systems across thresholds that block native community establishment.

The policy message is clear. The light-touch restoration intervention failed to produce an effective outcomes in the nutrient-enriched post-agricultural system. Funding and management should prioritise interventions that address multiple barriers simultaneously, even if resource intensive, because they are more likely to yield long-term biodiversity recovery and resilience.

Ongoing monitoring is essential to track how above- and below-ground processes interact over time. Future work should integrate assessment of soils, microbes, and fauna to capture whole-ecosystem recovery. The dataset collected here would also support species-level indicator analysis - for example using the indicspecies package in R (De Cáceres & Legendre 2009) - to identify which species are driving community differences between treatments, a question of particular relevance to practitioners working within southern Australian grassland systems. Within plant communities, emphasis should be placed on life-form diversity, not just grass cover, to build functional, resilient grasslands. Integrating ecological science with landscape design will be key to embedding these outcomes in human-dominated landscapes worldwide.

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Chapter 4

STRIKING A BALANCE BETWEEN ECOLOGY AND ECONOMICS: A MORE HOLISTIC APPROACH TO OPTIMISING TOPSOIL REMOVAL IN GRASSLAND RESTORATION

Maximilian McQuillan*[1], Luke Finn[2], Carlos Bartesaghi Koc[3], and Scott Hawken[1]

4.1 Context Statement

The preceding chapters situated restoration interventions along a spectrum of management intensity, from light-touch approaches such as biomass reduction or chemical fallow, through moderate interventions that target biomass and seedbanks without addressing soil fertility (e.g., solarisation, fumigation), to full soil resets through topsoil removal (TR). This continuum is particularly relevant in post-agricultural landscapes, which characterise much of the peri-urban fringe and often represent a transitional stage before conversion to urban land use (Ellis, 2020). These sites carry strong land-use legacies, including elevated soil nutrient concentrations and dense exotic seed banks, that form a significant barrier to the establishment of biodiverse native grassland communities.

This ecological framing of interventions can also be viewed through the lens of contemporary landscape architecture, where ecology has evolved into both an analytical framework and a design metaphor. Over the past two decades, scholars such as Reed and Lister (2020) have argued that design disciplines must move beyond static, deterministic views of nature toward an embrace of systems characterised by resilience, adaptability, and change. Building on this, restoration is not only a technical challenge but also a cultural and political one: it asks how human interventions can engage with ecological processes in ways that are open-ended and future-facing.

In parallel with these theoretical developments, practice-oriented frameworks, such as Biodiversity Sensitive Urban Design (BSUD; (Garrard et al., 2017)), offer a practical perspective on integrating ecological knowledge into urban contexts. As the discussion shifts

from theory to practice, BSUD highlights the importance of embedding biodiversity considerations early in urban planning, rather than treating them as tokenistic add-ons. This approach resonates strongly with the challenges faced in post-agricultural landscapes: if a full reset intervention like TR is to be justified, it must be integrated into a design process that considers not only ecological feasibility but also social value, stakeholder expectations, and urban liveability.

Within this broader context, the Glenthorne Community Grassland can be understood as an instance of "research through designing" (Lenzholzer et al., 2013). While grounded in ecological science, the project also functioned as a design experiment - testing not only whether TR could enable diverse grassland communities, but also how such an intervention might be conceptualised, costed, and communicated within broader socio-ecological systems. This framing situates this chapter within the tradition of landscape architecture research, where design itself serves as a method of inquiry and a means of generating knowledge.

Having established the experimental and design context, Chapter 3 demonstrated that only TR, as a full soil system reset intervention, enabled the assembly of diverse vegetation communities. Recognising TR as ecologically effective raises a further question: can it be scaled from experimental plots to broadacre practice? Addressing this requires not only ecological data but a realistic appraisal of costs, logistical constraints, and soil conservation risks. Reflecting this need, this chapter turns to the economics of TR, using contractor quotes and case studies to develop tools for deciding when, where, and how to apply TR at scale.

During the Green Adelaide Grant application (2022), I consulted four local earthmoving contractors equipped with graders and land planes. Initial estimates suggested that TR might be achievable for around AUD $4,000–$5,000 per hectare, but these did not account for mobilisation/demobilisation (float) costs. Subsequent formal quotes revealed a much broader cost range. Topsoil removal rates (in units of ha^{-1} 10 cm^{-1} of soil removed) ranged from AUD ~$4,000 to ~$11,000, while float costs varied from ~$3,000 to ~$11,000 depending on travel

distance and machinery requirements. The total TR quotes for a single hectare ranged from approximately $12,700 to $22,200, representing a 74% increase at the upper end compared with the lowest quote. These discrepancies underscored both the complexity of accurately costing TR and the limited availability of reliable practitioner guidance at the time. Because I underestimated the true cost of TR, the budget allocation in the 2022 grant was insufficient once implementation began. This left very little contingency, and when early winter rains in 2023 restricted site access and delayed the earthworks, the project quickly faced financial pressure. As a result, completion of the earthmoving required an application for additional funds in 2024. The scale and complexity of the earthworks operation is illustrated in Image 3, which shows topsoil removal in progress across the restoration area in 2023.



***Image 3 Aerial view of topsoil removal operations at Glenthorne National Park**, showing earthmoving machinery actively stripping and redistributing topsoil across the restoration area. The stripped plateau (centre) and redistributed overburden (edges) are visible, along with the solarisation infrastructure (foreground). Drone photograph by the M McQuillan, 2023.*

Additionally, I faced considerable uncertainty in selecting a target depth for TR, since the literature offered little robust evidence on weed seedbank depth. Conversations with state

government stakeholders also revealed strong scepticism about the long-term sustainability of TR. Concerns focused on soil conservation (erosion, stability) and impacts on soil biodiversity (soil biota, burrowing invertebrates, microbiota). These challenges demonstrated the lack of a structured framework for evaluating TR risks and trade-offs. This gap motivated the framework presented in this chapter.

This chapter is presented in the format of a journal article to align with the thesis by publication approach. Although structured as a standalone publication, it is best understood within the broader context of the thesis. Preceding chapters provide foundational knowledge, and subsequent discussions explore broader implications and applications.

This chapter introduces a framework to guide decision-making in grassland restoration projects, with a focus on the economic and environmental sustainability of TR. The framework is intentionally designed to be adaptive and pragmatic, reflecting the realities faced by restoration practitioners, as demonstrated through its application at the Glenthorne Community Grassland. While it offers a novel approach to integrating economic and environmental considerations, it should be read in the context of its limitations, which are addressed here to provide transparency for future applications. The findings are derived from a single case study with soils (Vertosols/Dermosols) that cannot represent all post-agricultural landscapes. The impacts of TR on soil biodiversity (from microbiota to burrowing invertebrates) remain poorly quantified. Additionally, cost estimates are contingent on local contractor rates and mobilisation conditions. Additionally, the lack of long-term monitoring means that conclusions about sustainability are provisional. For these reasons, the framework is best understood as a practical tool for guiding future restoration work and prompting further testing, rather than as a prescriptive or definitive methodology.

Ultimately, this chapter aims to bridge disciplines and address a significant gap in restoration science by providing a structured approach to balancing economic feasibility with environmental sustainability. I hope that this work will serve as a starting point for more

comprehensive frameworks and inspire further exploration in temperate grassland restoration.

The specific research questions addressed in this chapter are:

- *Q3.1 What factors determine the cost of topsoil removal in urban restoration, and how can these costs be predicted reliably by practitioners?*
- *Q3.2 How can practitioners systematically weigh biodiversity outcomes against costs and environmental risks to determine whether topsoil removal is justified - and, if so, to what depth?*
- *Q3.3 What soil conservation concerns (e.g., erosion, sodicity, compaction) may arise when scraping soil, and how can these be assessed efficiently in the field?*

Statement of Authorship

Title of Paper	Striking a Balance Between Ecology and Economics: a More Holistic Approach to Optimising Topsoil Removal in Grassland Restoration
Publication Status	☐ Published ☐ Accepted for Publication ☐ Submitted for Publication ☒ Unpublished and Unsubmitted w ork w ritten in manuscript style
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Name of Principal Author (Candidate)	Maximilian McQuillan
Contribution to the Paper	Conceptualisation, methodology, data collection and analysis, and writing.
Overall percentage (%)	80
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.
Signature	Date

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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Statement of Authorship

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Signature		Date	1st December 2025

Please cut and paste additional co-author panels here as required.

4.2 Implications for Managers

- Use rapid soil-conservation screen (salinity, pH, erosion hazard, soil stability, buffer zones) to identify areas suitable for safe topsoil removal (TR), and determine when alternatives are necessary.
- Building on this, shallow removal (2–12 cm) across larger contiguous areas often provides the best cost-efficiency, as most weed propagules are concentrated in the upper horizons while float costs are spread over area.
- The cost–depth–area model helps practitioners choose depths that balance seed-bank reduction with budget and long-term management needs.
- Where erosion or soil-biodiversity risks are high, use alternative site-preparation methods (e.g., chemical fallow, mulching, tube stock planting) to avoid ecological damage.

4.3 Summary

Restoring native temperate grasslands in Australia increasingly relies on direct seeding to reestablish diverse and resilient ecosystems. However, restoration is often limited by nutrient enrichment (excess nitrogen or phosphorus in the soil) and dense non-native seed banks (seeds from exotic species) in the upper soil layers. Topsoil removal, which strips these layers, can overcome these barriers but is costly and presents ecological challenges, as these layers support soil stability and belowground biodiversity. Evaluating both costs and ecological trade-offs is crucial when planning topsoil removal.

This study presents a two-fold approach to assess the economic and environmental sustainability of topsoil removal. First, we develop a cost model to evaluate the efficiency of removal at different depths. Second, we propose a rapid framework to identify soil conservation issues, including erosion (topsoil loss due to water or wind), scalding (hard, bare soil surfaces resulting from lost structure), and sodicity (sodium accumulation impacting structure and infiltration), both at the surface and at the scraping depth. By integrating cost modeling with environmental assessment, our framework optimises cost-effectiveness and sustainability in temperate grassland restoration.

This study also proposes a spatial decision-making framework to inform ecological restoration design at the landscape scale. We validate this framework using a real-world test case at an urban national park in Adelaide, SA.

Keywords: environmental sustainability, grasslands restoration, topsoil removal, cost, scalping, weeds, soil conservation

4.4 Introduction

Grasslands cover about 40% of the world's land (Suttie et al., 2005); almost half of the world's grasslands (49%) are degraded (Gibbs & Salmon, 2015). These areas face ongoing threats from agriculture and urbanisation (Dudley et al., 2020; Gossner et al., 2016; White et al., 2000). Temperate grasslands are among the world's most threatened ecosystems (Richter & Osborne, 2014). Most native grasslands have been converted into farmland (Gott et al., 2015). In Australia, approximately 55% of the land is used for agriculture, which significantly degrades native ecosystems, particularly in areas where cropping and exotic pastures are prevalent (ABARES, 2025; Creswell et al., 2021). Native grasslands attracted European settlers during the colonial period as they were well suited to grazing introduced livestock. Consequently, native grasslands were replaced by farms, often followed by urban development - a process still underway in some areas (Ellis, 2020; Marshall, 2013). Ellis's concept of "anthropogenic biomes" (anthromes) reinforces this trajectory by recasting ecosystems through the lens of human activity, rather than solely through climate (Ellis, 2020). This reclassification highlights the degree to which landscapes like Glenthorne are not merely "degraded nature" but distinct socio-ecological systems shaped by agricultural legacies and urban expansion. Understanding such sites as anthromes underscores the need for restoration strategies that explicitly account for human histories, economies, and infrastructures alongside ecological processes.

The Natural Temperate Grasslands of Australia are among the continent's most threatened terrestrial ecosystems (Kirkpatrick, 1995; Williams & Morgan, 2015). Some, like those of the Gippsland Plain, are extinct (Lunt, 1994). Others, such as the Murray Valley Plains, survive only as fragments in less intensively used agricultural areas (Threatened Species Scientific Committee, 2010, 2012). Many now cover just 1-5% of their original range. High-quality remnants, like those in the Victorian Volcanic Plains (Barlow & Ross, 2001) and the Iron-grass Natural Grassland of South Australia (Turner, 2012), cover an even

smaller extent. With severe biodiversity loss and few intact areas left, restoration is critical for preserving biodiversity.

Australia's geological and evolutionary history has produced some of the most nutrient-poor soils on Earth. Native flora has evolved specialised strategies to thrive under these conditions (Beadle, 1962; Handreck, 1997; Lambers et al., 2011; McLaughlin, 1996). In this context, temperate grasslands naturally occur on comparatively naturally fertile soils in Australia. These soils are often young and formed from parent material in place (J. W. Morgan et al., 2017). Agricultural development has profoundly changed these systems. Past land use has enriched soils with nutrients - especially phosphorus, which is immobile and persistent (Parkhurst et al., 2022). Additionally, non-native species now dominate soil seed and bud banks (Morris, 2022). These species outcompete natives during establishment due to early germination and rapid growth (Ba & Facelli, 2022; Lenz & Facelli, 2005; Zeeman & Morgan, 2018). Many native plants show limited response to elevated nutrients, despite their adaptations (Hagon & Groves, 1977; Handreck, 1997; McQuillan et al., 2020). Together, nutrient enrichment and dense non-native seed banks are serious barriers to restoring biodiverse native grasslands on former agricultural land. Figure 1 shows how elevated nutrients and weed-heavy seed banks restrict native plant establishment. In such cases, a complete reset of the soil ecology may be necessary. This "ecological reconstruction" involves correcting abiotic damage and reintroducing native species with evidence-based methods (SERA 2021). When needed, it may include soil scraping or replacement. These approaches are more akin to mine rehabilitation than traditional restoration (Cooke & Johnson, 2002) and are considered ecological reconstruction rather than restoration (SERA, 2021).

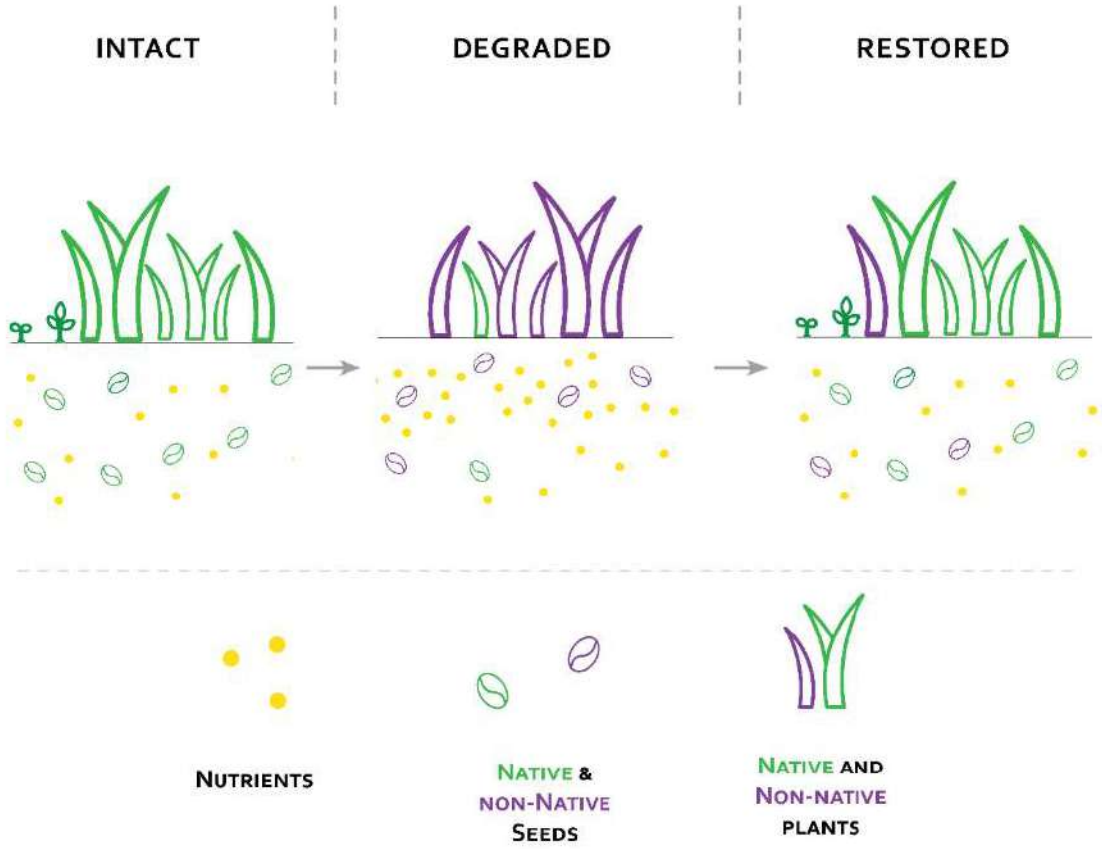

Figure 1: Conceptual model of intact, degraded, and restored systems. The intact (reference) system (either actual or theoretical) contains only native plants and propagules and relatively low concentrations of nutrients. The degraded system is typically dominated by non-native plants and propagules, as well as relatively high concentrations of nutrients. The restored system aims to be as close as practical to the intact system but may still contain a minority of non-native plants and propagules. Achieving and sustaining the restored system often requires the concentration of nutrients to be decreased to near that of the intact system.

Direct seeding is the primary method for reconstructing native temperate grasslands in Australia. It uses a diverse seed mix tailored to the target vegetation. However, the high soil nutrient levels and an abundance of non-native seeds and bud propagules hinder success (Gibson-Roy, 2023; Morris, 2022; Parkhurst et al., 2022). Without effective mitigation and ongoing management, vigorous non-native vegetation outcompetes sown natives.

Pre-seeding preparation methods are crucial in addressing these barriers. They fall into three main categories (illustrated in Fig. 2). Approaches to mitigate these barriers can be grouped into three categories: (1) biomass removal, which removes standing vegetation but retains the soil seedbank (e.g. burning, slashing, herbicide application, or long-term mowing with biomass removal — the latter can progressively draw down soil nutrients over time; Gibson-Roy et al. 2010); (2) seedbank removal, which removes both biomass and seed/bud banks (e.g., solarisation, soil fumigation, pine oil, or soil microwaving; Hanson & Shrestha 2006; Al-Shammary et al. 2020; McQuillan et al. 2024); and (3) soil reset, which removes biomass and seedbank while reducing surface nutrients (e.g., topsoil removal, sod cutting, soil capping; Gibson-Roy et al. 2010; Bakker et al. 2012; Gibson-Roy & Carland, 2022; Horsfall et al. 2025). Carbon addition and prescribed fire offer complementary in-situ nutrient manipulation with demonstrated benefits for native seeding outcomes (Prober et al. 2004, 2005).

The third category - soil reset - is considered the most effective for achieving restoration outcomes over the medium to long term (10+ years) on former agricultural lands where lasting effects of past land use remain. Topsoil removal (abbreviated as TR) reduces soil nutrient levels and removes many seeds and buds from invasive plants, as these are primarily located in the upper soil layer. Dr Paul Gibson-Roy brought attention to this technique in Australia (Gibson-Roy & McDonald, 2014). It is now widely used as a preparatory step for reconstructing temperate grasslands. Topsoil removal has been shown to work well in several case studies from farming and ex-farming landscapes (Gibson-Roy, 2023; Gibson-Roy & Carland, 2022; Řehounková et al., 2020; Resch et al., 2022). In certain situations, it lowers the presence of non-native plants to fewer than one plant per square meter (Gibson-Roy et al., 2010).



Figure 2: Pre-seeding preparation methods for temperate grassland restoration can be categorised into three main groups: (1) biomass removal: methods that remove standing vegetation but leave the exotic soil seed bank intact; (2) seed bank removal: methods that eliminate both standing vegetation and the exotic soil seed/bud bank; and (3) soil reset: methods that clear standing vegetation, reduce the soil seed/bud bank, and lower soil nutrient levels.

While techniques such as topsoil removal address the ecological legacies of past land use, their adoption and success also depend on broader social and economic contexts. Therefore, environmental restoration benefits from the cross- and trans-disciplinary perspective of socio-ecological system studies (Ostrom, 2009). A key insight is that successful restoration projects are founded on solid economic principles. Good ecology is necessary but not enough. Understanding economic and financial realities is essential for long-term

sustainability (Bodin et al., 2021; Iftekhar et al., 2016; Rohr et al., 2018). The limited resources for environmental restoration in many areas make it even more important to deliver the biggest 'bang for your buck'.

Despite its effectiveness, TR presents significant economic and environmental challenges. The costs of heavy machinery and logistics involved in soil removal make it an inherently expensive option, with previous estimates in Europe placing it at the upper end of restoration interventions (Török et al., 2011). Actual costs are highly variable and depend on factors such as soil type, the depth of removal, transport and disposal requirements, and broader economic conditions, including the availability of contractors and machinery. Site access also introduces uncertainty: acceptable soil moisture regimes for heavy equipment vary across different soil types, and season-breaking rainfall can render sites inaccessible, significantly delaying works if unexpected. Importantly, these costs are largely incurred upfront, in contrast to other interventions that may appear cheaper initially but often require higher ongoing management for years (Kimball et al., 2015). This concept is addressed through life cycle costing in other sectors (Heralova, 2017). This trade-off between high initial expenditure and reduced long-term maintenance is a critical consideration when evaluating the economic feasibility of TR.

A further challenge is the limited availability of comprehensive cost–benefit data, which makes it difficult for practitioners to assess the financial implications of different restoration methods or to balance costs with ecological outcomes (Kimball et al., 2015; Robbins & Daniels, 2011). Moreover, there is no clear relationship between the price of an intervention and its ecological effectiveness. What matters is not how much is spent, but the extent to which a treatment delivers the ecological “push” required to overcome barriers to recovery. For example, soil solarisation can require more time and resources per unit area than TR, yet TR often proves more effective because it resets both nutrient levels and seed banks. This suggests that cost alone is a poor proxy for ecological effectiveness. Proposed metrics, such as “native cover restored per dollar spent” (Kimball et al., 2015), gesture toward linking

investment more explicitly with outcomes, but direct measures, such as units of biodiversity per dollar, remain underdeveloped.

This does not mean that cost efficiencies are unattainable. Economies of scale can significantly reduce unit costs, particularly for inputs like seed, which remains one of the most substantial barriers to grassland restoration (Gibson-Roy, 2023; Gibson-Roy et al., 2021). Large projects can lower seed costs by establishing seed orchards in advance, thereby building reserves for future restoration (Cuneo et al., 2018). More broadly, scaling up has been shown to deliver benefits beyond cost savings, including accelerated biodiversity gains and improved outcomes at the landscape scale (Csonka et al., 2023; Lengyel et al., 2012). There is a pressing need for improved cost models that evaluate not only the upfront costs of TR but also the viability of soil seed banks at different depths, the logistical implications of earthmoving, and the potential for scaling to reduce costs. Without such tools, practitioners are left to make high-stakes decisions in the absence of reliable financial and ecological decision support tools.

Thus, one of the major gaps in current restoration practice is the lack of detailed information to guide decision-making about the costs and ecological consequences of TR. While depth of removal is recognised as a major driver of both cost and effectiveness, there is little empirical evidence to determine the point at which further scraping delivers diminishing ecological returns. This creates uncertainty for practitioners attempting to balance investment against long-term outcomes. Addressing this requires cost models that integrate seedbank viability with the costs of earthmoving across different depths, alongside frameworks that consider how scaling up restoration projects could deliver cost efficiencies.

In addition to the financial challenges, TR also carries significant environmental risks that must be carefully managed to prevent long-term harm. While the goal is to reset soil conditions and suppress non-native seed banks, the intervention itself is highly disruptive to the ecosystem. Scraping away the upper horizons can increase erosion risk, expose saline or sodic subsoils, and create disturbed surfaces that are vulnerable to secondary invasion (Frouz

et al., 2009; R. P. C. Morgan, 2005). In practice, the displaced topsoil is usually stockpiled or redistributed on site, but this does not eliminate the risk of spreading weeds or altering adjacent areas. Moreover, topsoil is a non-renewable resource on human timescales, and its removal compromises essential functions such as water retention, nutrient cycling, and soil biological activity (Frouz et al., 2009). TR also potentially reduces or removes the residual native seed bank. Although reintroduction is possible through direct seeding or planting, this presumes that all species can be sourced and overlooks the potential for hidden taxa to be lost, as well as the risk that seed orchard or nursery stock may offer lower or maladapted genetic diversity compared to *in situ* propagules (Gellie et al., 2016; Shackelford et al., 2021). The trade-off, therefore, lies in weighing the loss of topsoil functions and residual propagules against the opportunity to remove entrenched barriers and create conditions in which native grasslands can be re-established.

Therefore, without careful planning, TR has the potential to lead to net environmental harm (Frouz et al., 2009). The lack of a detailed framework to assess the suitability of target sites for TR represents a critical gap in current restoration practices. Without such a framework guiding pre-restoration assessments, practitioners may unintentionally cause more harm than good, undermining the long-term sustainability of the restoration effort. Key parameters, such as salinity and erosion potential, should be considered when determining whether a site is suitable for TR (Morgan, 2005). A robust environmental assessment framework would mitigate these risks and guide practitioners toward sustainable restoration practices.

This study argues that restoration planning must consider both ecological and economic realities simultaneously. For TR, this means not only asking whether it works for reducing weeds and nutrients, but also whether it can be afforded and sustained without causing new problems. To help with this, we propose two practical tools: (1) a cost model that links the price of soil removal to the depth and persistence of weed seed banks, and (2) an environmental checklist that highlights risks such as erosion, soil scalding, and impacts on

biodiversity. Used together, these tools provide practitioners with a clearer way to determine when TR is worth the investment and when it may do more harm than good.

4.5 Methods

The data were collected during a native temperate grassland restoration project in Glenthorne National Park-Ityamaiitpinna Yarta, an urban national park (35.06022S, 138.54983E) in Kaurna Country / Adelaide, South Australia. The site sits at ~180 m above sea level and receives an average annual rainfall of 550–650 mm. Soils consist of black Vertosol and black Dermosol (Isbell, 2021), both of which have medium to heavy clay contents.

The site has a long history of agricultural use, resulting in elevated soil nutrient levels and a vegetation cover dominated by non-native pasture grasses and invasive broadleaf weeds. Historical records and sketches suggest that the area was once covered by temperate grasslands or sparsely wooded grassy woodlands, with considerable overlap in understory composition. These ecosystems were historically maintained through cultural burning by the Kaurna people (pers. comm. Richard Woods). Restoration efforts, therefore, aim to reintroduce a diverse ground layer that reflects these vegetation types while addressing the persistent legacies of agriculture.

All fieldwork was conducted under a Scientific Research Permit (Q27266-1) issued by the Department for Environment and Water, South Australia.

4.5.1 Cost-efficiency model for topsoil removal

To optimise restoration efforts, we developed a cost model that integrates seed bank viability and earthmoving costs, allowing for informed decisions on the depth and scale of TR. A seed bank germination study informs viability, while the cost model helps balance restoration outcomes with cost efficiency. The detailed methodology is outlined in Appendices 1 and 2.

4.5.1.1 Soil seedbank germination study

A soil seed bank germination study was conducted to assess the depth of viable weed seeds and bud bank propagules across the project area. On 14 January 2023, soil samples were collected at 2 cm intervals to a depth of up to 20 cm from 11 locations, selected using a random location tool in QGIS (Table 1, Fig. 3). The design was not formally blocked; random sampling was considered appropriate for generating a site-level estimate of seedbank depth to parameterise the cost model.

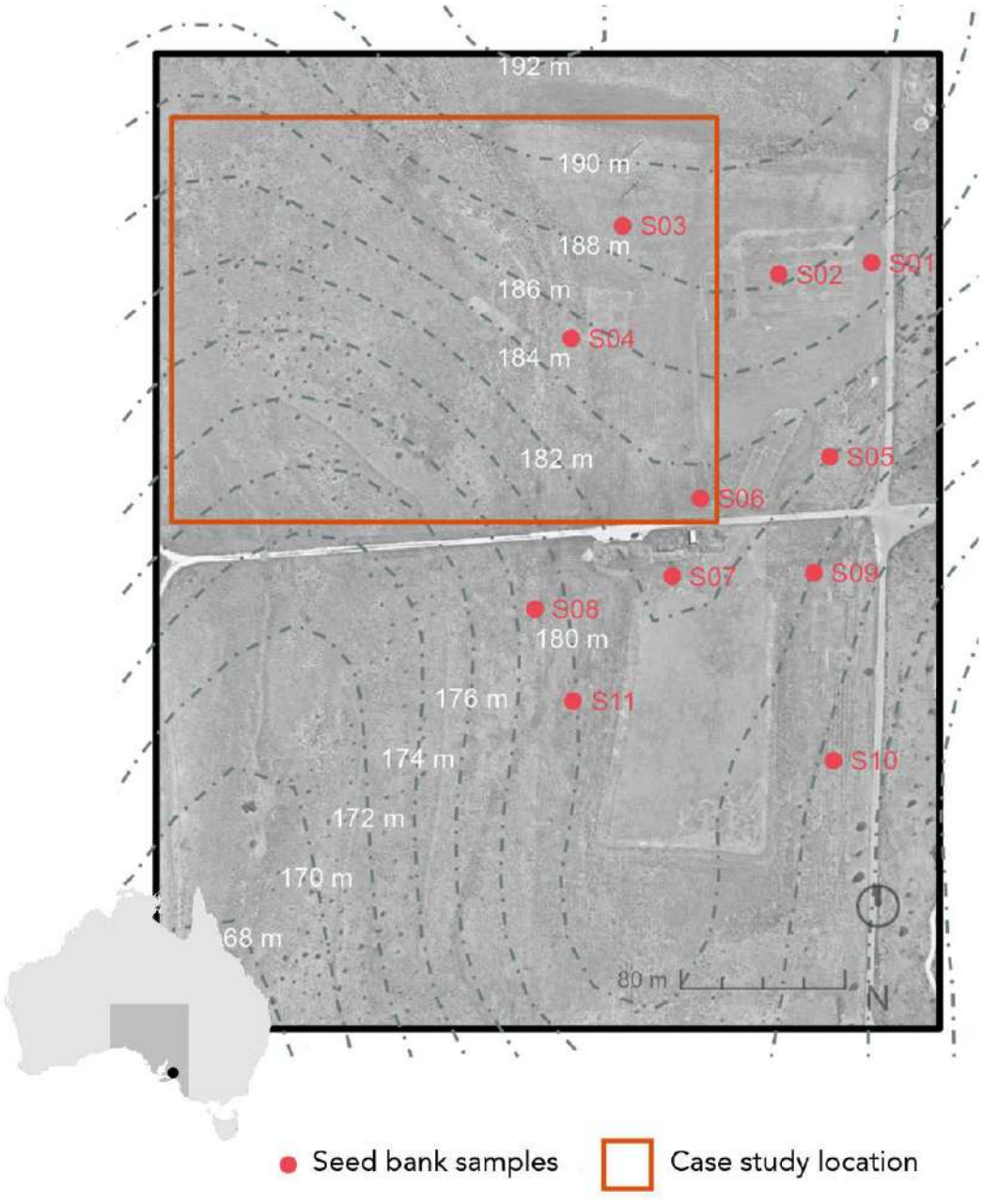

Figure 3: Study area. Red points mark 11 randomly selected soil sampling locations where samples were collected for the soil seed bank germination study. Contour lines with white text indicate topography and elevation. The orange box outlines the boundaries of the TR

suitability test case location. The linear, lighter-toned features are access roads and walking paths.

The samples were used in two germination trials: a summer trial from February 2, 2023, to February 21, 2023, and a winter trial from June 12, 2023, to July 13, 2023. The summer trial was conducted using sealed clear plastic containers to maintain moisture consistency across pots; condensation within the containers caused uneven water redistribution and likely contributed to mould development, resulting in early termination of the summer trial after 19 days. The winter trial used the same pot setup without enclosed containers and ran to completion. Germinants were counted at the conclusion of each trial. The detailed methodology, including sampling protocol, sample handling, germination trial setup, and seedling identification procedures, is provided in Appendix 1[1].

Table 1: Soil seed bank germination sampling information: sample ID, elevation, and horizons sampled.

Sample ID	Elevation (m AHD)	Horizon samples (cm BGL)
S01	188	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S02	187	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S03	189	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S04	186	0-2; 2-4; 4-6; 6-8; 8-10; 12-15; 15-20
S05	185	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S06	185	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S07	182	0-2; 2-4; 4-6; 6-8; 8-10; 12-15

[1] This material has been placed in the appendix to preserve the readability of the chapter at the decision-framework level (consistent with standard supplementary material practice in journal publications) while keeping the full protocol accessible to readers who require it.

S08	180	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S09	184	0-2; 2-4; 4-6; 6-8; 8-10; 12-15; 15-20
S10	180	0-2; 2-4; 4-6; 6-8; 8-10; 12-15
S11	180	0-2; 2-4; 4-6; 6-8; 8-10; 12-15

m AHD: metres Australian Height Datum
Projected Coordinate System: WGS 84 / UTM zone 54S (EPSG:32754)
cm BGL: cm below ground level

4.5.1.2 Topsoil removal cost data

A TR cost study was conducted by inviting 10 local heavy earthmoving contractors to provide quotes for an identical TR scope of work detailed in A2.1, with responses received from four contractors (A, B, C & D). All contractors provided a cost for mobilisation and demobilisation of earthmoving equipment (the float cost) and for topsoil removal (TR). The TR cost was standardised as the removal of 10 cm of topsoil across 1 ha, followed by stockpiling and levelling of the removed soil adjacent to the TR area. These costs were adjusted for inflation using the Producer Price Index (PPI) for road and bridge construction in South Australia between Q1 and Q3 of 2023 (see Appendix A2.2), applying an inflation rate of 0.72% to estimate real costs as of October 2023.

A simple linear cost model (Eq. 1) was applied to predict topsoil removal costs for the removal of 1 ha to 10 cm, and this was integrated with the seed bank germination data to create a cost model for restoration.

$$\boldsymbol{Eq\ 1}\textbf{: } \boldsymbol{cost\ of\ topsoil\ removal}\left(\frac{\boldsymbol{10\ cm}}{\boldsymbol{ha}}\right) = \boldsymbol{float} + \boldsymbol{area} \times \boldsymbol{TR}$$

4.5.2 Rapid soil conservation assessment framework (RSCAF)

To address sustainability risks, we developed a rapid soil conservation assessment framework (RSCAF) to identify potential issues such as erosion and biodiversity loss. The framework focuses on five soil conservation parameters: salinity, erosion hazard, soil stability, pH, and proximity to sensitive sites. The parameters were selected for their direct relevance to restoration outcomes, relative ease of measurement in standard soil surveys, and their role in determining whether topsoil removal would create irreversible risks.

Salinity:

Salinity levels were assessed using data from Hazelton & Murphy (2016) for a saturated paste analysis (ECe). The EC 1:5 water measurements were converted using soil texture multiplication factors to account for varying soil types. Soils with an electrical conductivity

(ECe) below 2 dS/m were deemed to have a **low environmental risk** for topsoil removal, while those with an ECe between 2 and 4 dS/m were classified as having a **medium environmental risk** and were deemed to require management interventions. Soils with ECe exceeding 4 dS/m were considered to have **high/ unacceptable environmental risk** due to the high risk of plant stress and reduced germination rates, consistent with thresholds reported for Australian vegetation (DEECA, 2020; DEW, 2012).

In practical terms, high-EC soils are unsuitable for topsoil removal because the stored topsoil would inhibit seedling emergence and require costly treatment (e.g., leaching or blending) to be usable. By contrast, low-EC soils can be stockpiled and reused with minimal treatment.

Erosion:

Soil erodibility was assessed using a simplified version of the Revised Universal Soil Loss Equation (RUSLE), which predicts annual soil loss based on rainfall erosivity, soil erodibility, slope length and gradient, erosion control practices, and ground cover (Morgan, 2005).

For this study, soils were classified into three erosion hazard categories: **low environmental risk** (<224 t ha^{-1} yr^{-1}): erosion is relatively modest even under bare soil conditions, and unlikely to prevent subsequent vegetation establishment; **medium environmental risk** (225–500 t ha^{-1} yr^{-1}): substantial soil loss is expected, but surface degradation remains potentially manageable with timely re-vegetation or erosion control; and **high or unacceptable environmental risk** (>500 t ha^{-1} yr^{-1}): soil loss at this magnitude represents massive degradation, rapidly stripping topsoil and compromising the ability of vegetation to establish or persist without intensive intervention.

These thresholds are adapted from Morgan (2005) as a comparative index under the artificial condition of complete ground-cover removal. They are necessarily high compared to natural background erosion rates (typically <1 t ha^{-1} yr^{-1} in undisturbed systems (Lu et al.,

2003)), but provide a relative scale of hazard when soils are fully exposed, where very high rates correspond to conditions in which vegetation recovery is severely constrained.

Soil stability (slaking and dispersion):

The Emerson test (Emerson, 1967) was used to assess soil stability by measuring the tendency of aggregates to slake or disperse when wetted. Following Emerson's classification system, soils that received a rating of 7 or 8 were classified as having a **low environmental risk**, reflecting stable aggregates that are unlikely to break down under field conditions. Ratings between 4 and 6 were considered **medium environmental risk**, representing partial stability where some breakdown or surface sealing may occur. Lower ratings (categories 1–3) were classified as **high or unacceptable environmental risk** because they indicate unstable soils with a high likelihood of structural collapse, dispersion, and associated erosion or infiltration problems. This categorisation reflects the progressive increase in susceptibility to soil structure breakdown across the Emerson scale, with low categories corresponding to the most severe functional risks for land management.

pH suitability for restoration:

Soil pH was assessed to determine its suitability for restoration, with optimal values ranging from 5.5 to 8.0. Soils within this range were classified as having **low environmental risk**. Soils with pH values between 4.5 and 5.4 or 8.1 and 9.0 were classified as having a **medium environmental risk**, while soils outside these ranges were considered to have a **high or unacceptable environmental risk** for restoration. These thresholds provide a general benchmark, recognising that tolerance varies among species and ecosystems (Hazelton & Murphy, 2016). For example, semi-arid to arid regions often have more alkaline soils, while wetter, more weathered regions (particularly on the east coast) are typically more acidic (Malone, 2022), with native plants adapted accordingly.

Buffer zones around sensitive sites:

Buffer zones were incorporated into the RSCAF to protect sensitive ecological and cultural sites. These include landscape features such as drainage lines and watercourses, riparian areas, lakes and ponds, known locations of threatened species, or Indigenous artefacts. Areas more than 100 m from such features were classified as having **low environmental risk**, as this distance provides a substantial and conservative buffer. Areas within 50–100 m were considered to have a **medium environmental risk**, and areas less than 50 m away were classified as having a **high or unacceptable environmental risk**, reflecting the greater likelihood of direct or indirect impacts. These thresholds are project-specific benchmarks, and managers may modify them as appropriate based on site conditions and available mitigation options.

4.5.2.1 Summary of RSCAF risk criteria

The RSCAF parameter ranges are summarised in Table 2. These criteria guided the identification of suitable areas for topsoil removal within landscape management units.

Table 2: Topsoil Removal Suitability: Rapid soil conservation assessment framework (RSCAF) for Temperate Grassland Restoration, outlining the criteria for salinity, erosion risk, soil stability, pH levels, and buffer zones surrounding sensitive sites.

Soil Conservation Issue	Low Risk	Medium Risk	High Risk
Salinity	<2 dS/m	2-4 dS/m	>4 dS/m
Erosion	0-224 t/ha/yr	225-500 t/ha/yr	>500 t/ha/yr
Emerson Test	Categories 7 & 8	Categories 4-6	Categories 1-3
pH	5.5-8.0	4.5-5.4 & 8.1-9.0	<4.5 & >9
Sensitive Site Buffer	>100 m	50-100 m	<50 m

Not all risks are permanent or equally correctable: pH and sodicity can often be corrected (e.g., lime, gypsum), while slope-driven erosion is more challenging to manage. The framework, therefore, serves as a precautionary screening tool rather than a definitive judgment on long-term site potential.

The decision pathway and its link to implementation are summarised in Fig. 4: the LHS presents the soil-conservation screening flowchart (buffers → erosion → pH/ECe → Emerson); and the RHS shows how conditional outcomes progress to safeguards and, where appropriate, the cost-depth-area optimisation prior to a final TR decision.

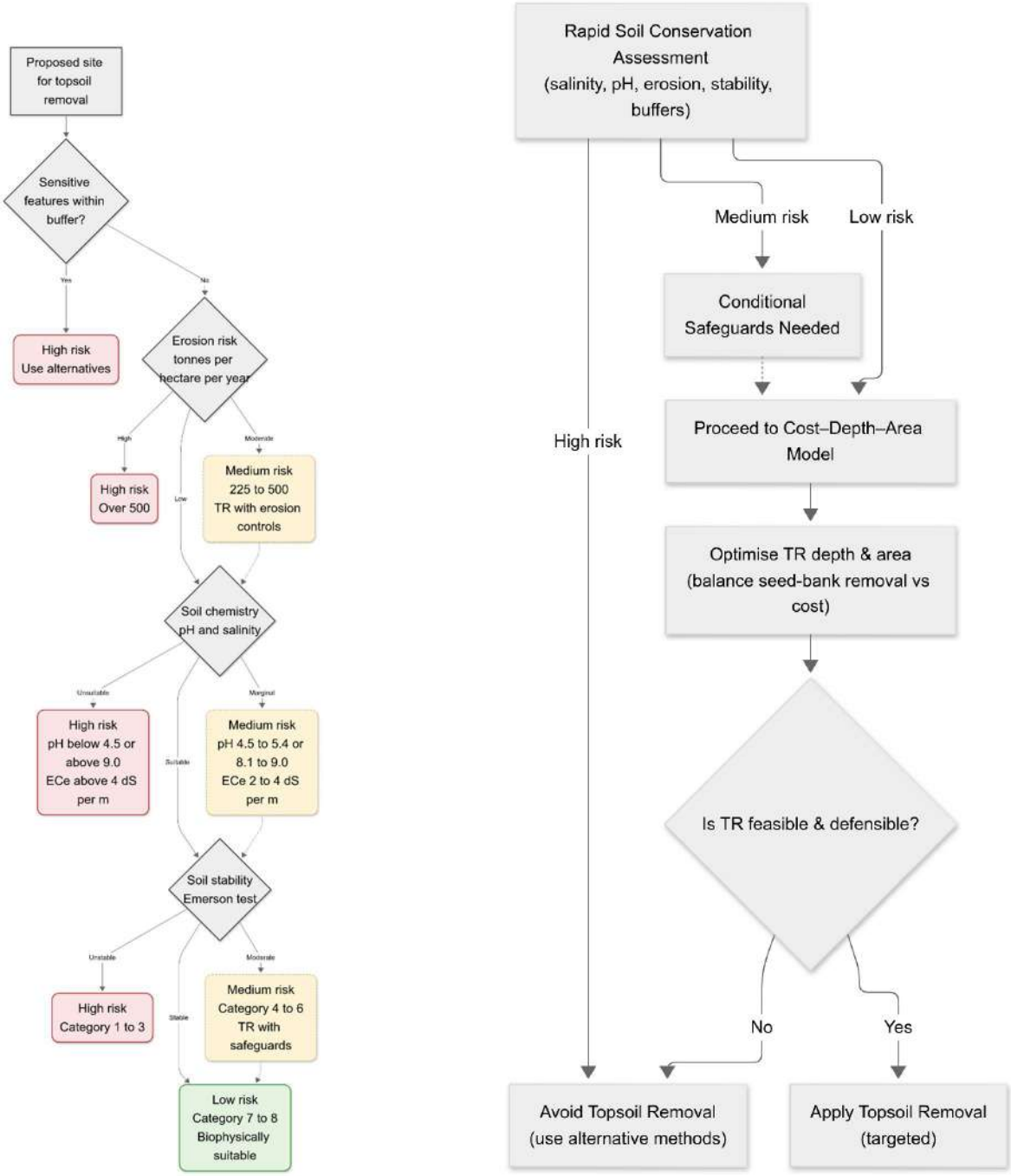

Figure 4: Soil-conservation decision pathway and strategic implementation workflow: (LHS) RSCAF Screening process for topsoil removal (TR). Sites are assessed for: (i) buffers to sensitive features; (ii) erosion risk (Low <225, Moderate 225–500, High >500 t ha^{-1} yr^{-1}); (iii) soil chemistry (pH 5.5–8.0 and ECe <2 dS m^{-1} = suitable; pH 4.5–5.4 or 8.1–9.0 and ECe 2–4 = marginal; outside these ranges = unsuitable); and (iv) soil stability (Emerson 7–8 stable, 4–6 moderate, 1–3 unstable). Outcomes are High Risk, Medium Risk (with safeguards), or Low Risk. Dashed connectors show that conditional cases may continue if addressed. (RHS) Strategic workflow from screening to implementation. High-risk sites avoid TR and use alternatives (e.g., chemical fallow, mulching, tube stock, etc.). Medium-risk sites proceed with safeguards, then cost–depth–area optimisation. Low-risk sites move directly to cost modelling. Final TR application depends on both biophysical suitability and economic defensibility.

4.5.3 Interpreting the framework

Effective application of the RSCAF demands not only adherence to quantitative thresholds, but also the exercise of professional judgement informed by contextual expertise. For instance, a soil classified as medium risk according to the Emerson test might still be considered suitable for topsoil removal by a practitioner who, based on prior restoration experience, recognises that targeted soil amendments or the implementation of erosion control measures can successfully mitigate the identified risks. Thus, practitioners must integrate their technical expertise with site-specific environmental conditions and a nuanced understanding of how multiple risk factors interact, thereby facilitating informed decision-making that is tailored to the unique characteristics and constraints of each restoration site.

In applying the framework, interpretation depends on professional judgement, recognition of compound risks, and site-specific knowledge. Quantitative thresholds may require adjustment when data are complex, and multiple soil conservation issues, such as erosion, stability, and salinity, may interact to amplify risk. Local conditions, including soil type, microclimate, land use history, and management practices, must also be considered to ensure that interventions are tailored and effective.

4.5.4 Application of the framework in the field

4.5.4.1 Delineating landscape management units

The RSCAF does more than classify risk; it sets intervention thresholds for planning. In practice, high-risk soils (e.g., steep slopes with high erosion potential or unstable aggregates) should be left undisturbed. Low-risk soils can be managed with greater flexibility, while medium-risk soils fall into a conditional category where intervention should proceed only with safeguards (e.g., erosion controls or soil amendments) and if projected biodiversity gains clearly outweigh the likelihood of degradation and the costs of remediation. This aligns ecological thresholds with defensible cost–benefit reasoning, recognising that financial outlays are best justified when tied to measurable biodiversity gains and a low risk of environmental damage.

We applied this logic at the Glenthorne National Park grassland restoration site. Landscape management units (LMUs) were delineated by integrating soil type, slope, and topography, initially drawing on the CSIRO soil map for Glenthorne National Park (Searle, 2021), and then refining boundaries with site-specific observations and a comprehensive on-site soil classification in December 2022 (Appendix 3). Three LMUs (~9 ha; Fig. 3, 7) were identified: **LU_1**: plateau tops with deep, black, cracking clays; **LU_2**: gently undulating to rolling rises with shallower, browner clays; and **LU_3**: lower slopes and drainage lines.

Sensitive areas, in our case all of LU_3 (i.e., drainage lines), were buffered and excluded from sampling. Within LU_1 (plateau) and LU_2 (slope), soils were sampled at one location per hectare to support mapping at a scale of 1:10,000 (Ringrose-Voase & Webster, 2007). Each location was tested at two depths (0–10 cm and 10–20 cm) and analysed for the parameters of the RSCAF.

4.5.5 Integrating assessment into restoration planning

The RSCAF results provided a precautionary zoning for the test site. High-risk soils were excluded from consideration for topsoil removal (TR), and medium-risk soils were considered only when supplementary measures could mitigate the risk. Critically, these zones were then integrated with the TR cost-efficiency model (Appendix 2), which combines contractor earthmoving costs with seed-bank viability by depth. Candidate interventions were retained only where: (i) the framework classified soils as biophysically suitable for TR (i.e., low risk); and (ii) projected costs provided economies of scale.

This integration yielded a replicable workflow: delineate LMUs → apply buffers and screen environmental risk → sample, analyse and rate soils against thresholds using the RSCAF → overlay the cost–depth–area model → select interventions that are both biophysically feasible and economically defensible. The resulting plan guided where TR was applied and where alternative preparations (e.g., chemical fallow, mulching, tubestock planting) were more appropriate.

4.6 Results

4.6.1 Soil seedbank germination study

The study revealed distinct seasonal patterns in seedling emergence, providing confidence that the study mimicked natural germination cues. During the cooler germination period, winter-active grasses germinated, while during the warmer germination period, only summer-active ruderal species germinated. While the number of germinants at each depth followed a similar pattern across both trials (Fig. 5A), the functional composition of germinants appeared to differ between seasons, consistent with the known seasonal germination patterns of exotic grasses and broadleaf ruderal species in temperate Australian grasslands — though systematic compositional data were not retained and this observation should be treated as indicative only. Figure 6A visualises the cumulative proportion of seed bank germination by depth for both trials, showing that most germination (~50%) occurred in the top 2 cm and then declined approximately linearly until all observed germination stopped beyond 15 cm.

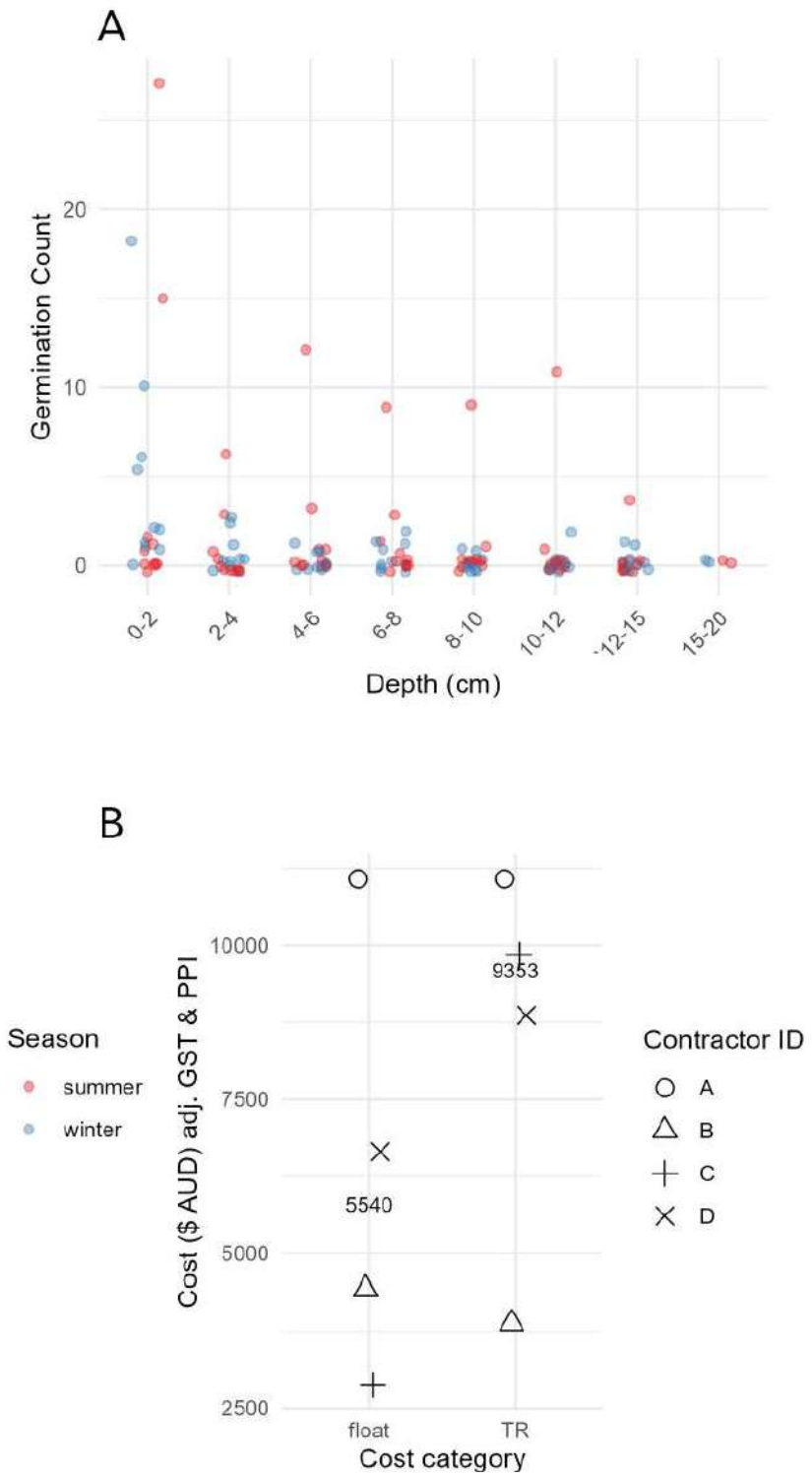

Figure 5: Soil seed bank germination and topsoil removal cost. (A) Summer and winter germinant counts: Scatterplot showing total germinants with slightly separated points for clarity. The x-axis shows different soil depths, and the y-axis shows the total number of seedlings. Dots are coloured by the season of the test. (B) Cost to move and remove topsoil: This figure breaks down the costs for bringing in and taking away earthmoving equipment, as well as removing a 10 cm layer of topsoil over one hectare. Costs are in Australian dollars, inclusive of taxes and inflation. Each job is represented by a different symbol, indicating the respective contractors.

4.6.2 Economic analysis of topsoil removal

Figure 5B shows quotes from four earthmoving companies, revealing that costs for moving equipment and removing topsoil vary widely. The typical price (adjusted for increases over the period) was about AUD $5,540 for moving the equipment and AUD $9,353 per hectare for removing the top 10 cm of soil. A simple formula (Eq. 1) was used to estimate the costs of topsoil removal for various areas and depths. Figure 6B shows that the cost rises steadily as area and depth increase. However, when we factored in the amount of seed bank removed at each depth, the cost per percent of seeds removed decreased for larger areas, with the best value in the top 2 cm of soil. For example, removing the top 2 cm over 1 ha costs approximately AUD $150 per 1% of the seed bank removed, but doing this over 5 hectares reduces the cost to about AUD $60 per 1% (see Fig. 6C).

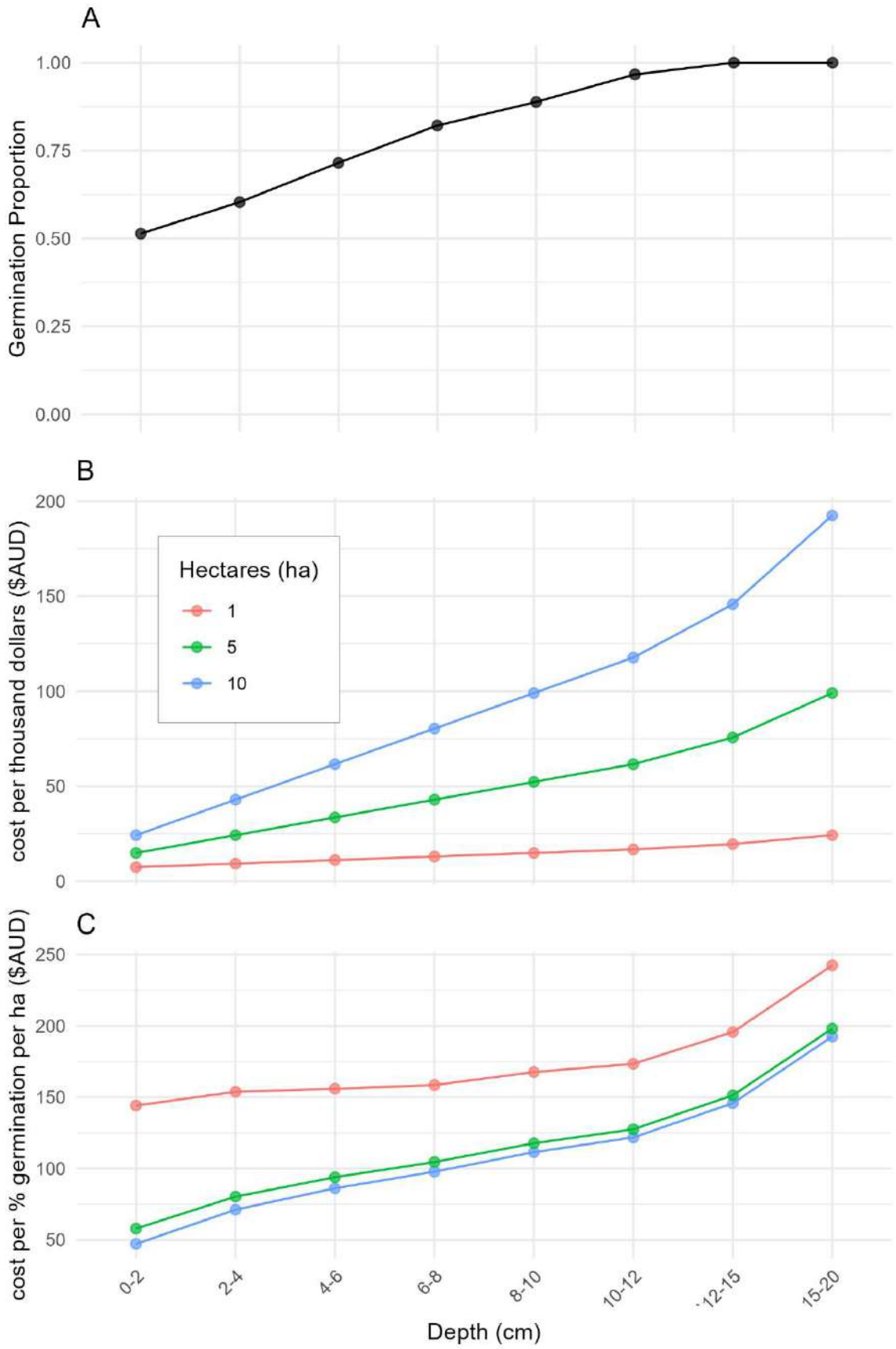

Figure 6: Determining a cost-effective TR target depth. (A) Cumulative proportion of soil seed bank germination aggregated across winter and summer trials, representing combined germination potential by depth; (B) Cost of topsoil removal earthworks to different depths across 1, 5, and 10 ha; (C) Cost per percentage of seed bank germination removed per hectare ($AUD) across topsoil removal to different depths for 1, 5, and 10 ha - lower values indicate greater cost-effectiveness.

In summary, these results provide a clear, data-driven basis for cost-effective, targeted restoration strategies by linking seed bank removal efficiencies with economic implications for large-scale projects.

4.6.3 Soil conservation factors for topsoil removal suitability

Table 3 summarises the test results for each landscape management unit, categorised by soil type and sampling depth. Salinity and pH risk levels across the assessed landscape units (LU_one and LU_two) were generally low, with most samples registering below the 2 dS/m thresholds and pH in the range 5.5-8, which we have classified as likely suitable for topsoil removal (Table 3). However, erosion risk increased significantly in LU_two, where erosion rates exceeded 150 t/ha/yr, classifying these areas as only marginally suitable for topsoil removal. Soil stability, as measured by the Emerson test, ranged from highly stable (ratings of 7 and 8) to unstable for sample LU_two_S_4_D_10-20, with lower ratings concentrated in deeper soil horizons, which indicates potential issues if the top 10 cm are removed in those locations, especially if compounded with additional issues, e.g., erosion potential.

Table 3: Soil features and suitability for topsoil removal. This table lists soil details for different landscape sections, including average pH, salt level (measured by electrical conductivity), erosion risk (the amount of soil that could be lost), and soil stability (how well soil clumps remain together). This information helps determine if it is safe and practical to remove topsoil for restoration, highlighting differences in salt content, erosion risk, and the strength of soil cohesion.

Sample ID	Soil Order	Sampling Depth (cm)	pH	ECe (dS/m)	Erosion Risk (t/ha/yr)	Emerson Stability Rating	Final land suitability class
LU_one_S_1_D_0-10	dermosol	0-10	7.4	0.6	10.5	8	Low risk
LU_one_S_1_D_10-20		10-20	6.6	1.8	10.5	4	Low-Medium risk
LU_one_S_2_D_0-10	vertosol	0-10	8.4	2	19.7	8	Low-Medium risk
LU_one_S_2_D_10-20		10-20	8.8	1.1	19.7	8	Low-Medium risk
LU_one_S_3_D_0-10	dermosol	0-10	7.5	1.9	15.8	8	Low risk
LU_one_S_3_D_10-20		10-20	6.6	1.9	15.8	3	Low- Medium risk
LU_two_S_4_D_0-10	dermosol	0-10	6.3	0.5	176.6	8	Medium risk
LU_two_S_4_D_10-20		10-20	7.1	2.8	176.6	2	High risk
LU_two_S_5_D_0-10	dermosol	0-10	6.1	1.9	235.1	8	Medium risk
LU_two_S_5_D_10-20		10-20	6.1	0.9	235.1	3	Medium risk
LU_two_S_6_D_0-10	dermosol	0-10	6	1.2	155.6	8	Medium risk
LU_two_S_6_D_10-20		10-20	6.4	0.8	155.6	8	Medium risk
LU_three	Excluded – drainage line buffered as sensitive ecological feature						

4.6.4 Application of RSCAF in restoration planning

Through assessing RSCAF results within landscape management units, the suitability assessment indicates which LMUs are at greater risk of problems from TR and which are at lower risk. Areas with high erosion risk or poor soil stability were excluded from topsoil removal plans or deemed marginally suitable. These marginal areas require additional management interventions. Figure 7 presents a spatial visualisation of the soil characteristics and management units. This visualisation clearly delineates areas of higher and lower restoration feasibility based on the RSCAF.

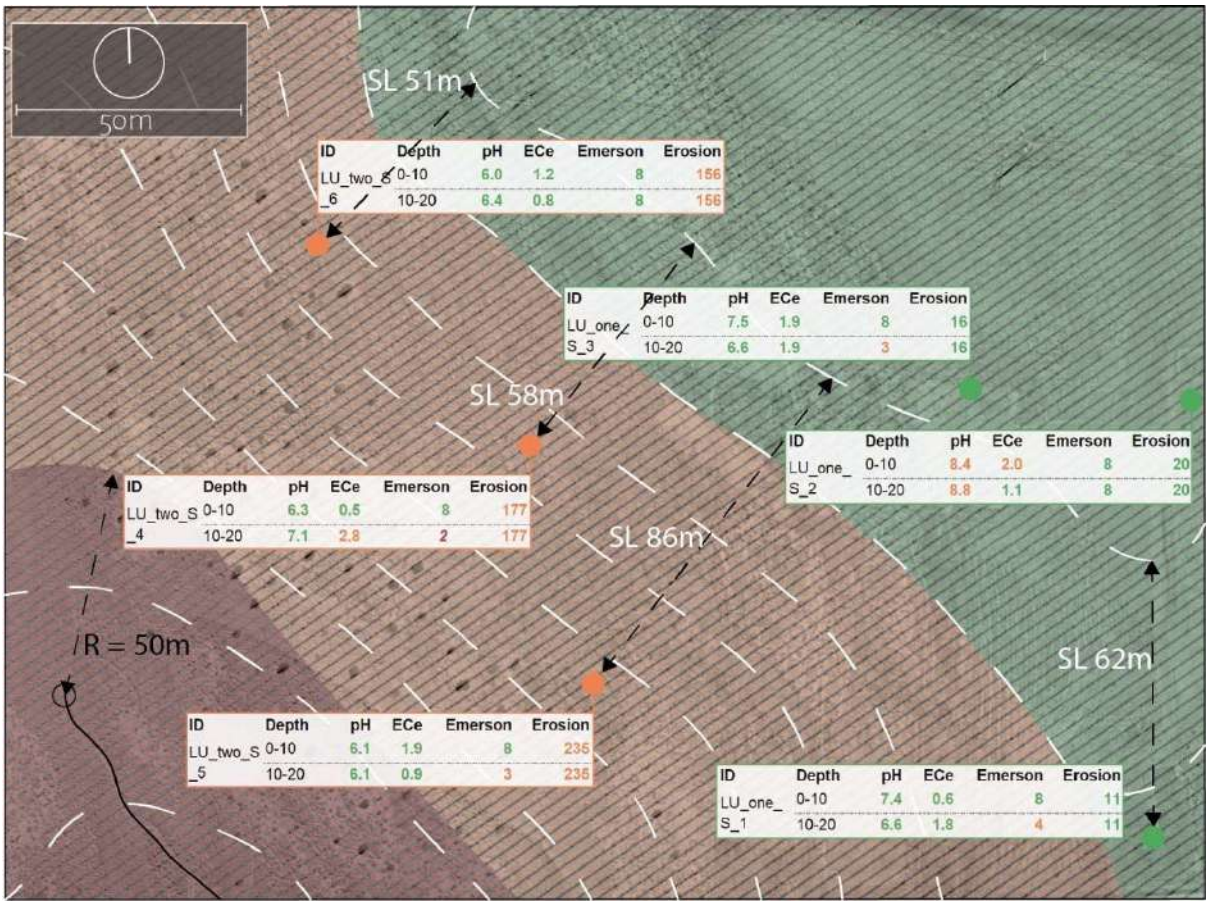

Figure 7: Soil characteristics and landscape management units. This figure presents a combined visualisation of soil characteristics and landscape management units. RSCAF results, including pH (average), electrical conductivity (EC in dS/m), erosion risk (in t/ha/yr), and Emerson stability ratings, are plotted for each sample within distinct landscape units. The figure illustrates how these factors vary across different soil types and sampling depths, informing the suitability of topsoil removal. Landscape management units are mapped based on soil type, topography, and proximity to sensitive ecological features, providing a spatial framework for evaluating the feasibility of restoration.

In summary, integrating the rapid soil conservation framework and the TR earthmoving cost model provided a comprehensive evaluation of both environmental risk and economic feasibility in the Glenthorne case study. The framework categorized LU1 (plateau soils) as low risk and suitable for TR, LU_TWO (slopes) as medium to high risk due to elevated erosion and occasional instability, and LU3 (drainage line) as unsuitable and excluded. The cost model indicated that shallow removal, especially the upper 2 cm, removed approximately half of the viable seed bank at the lowest cost per percentage removed, and exhibited economies of scale for larger areas. Overall, this study finds that shallow, targeted TR within stable landscape units is both the most cost-effective and the least risky approach.

4.7 Discussion

This study presents a practical decision-support tool, comprising the Rapid Soil Conservation Assessment Framework (RSCAF) illustrated as a flowchart (see Fig. 4), alongside a topsoil removal (TR) cost model. The flowchart's visual structure facilitates application by restoration practitioners, sequentially guiding users through designated thresholds - specifically, spatial buffers, erosion risk, soil chemistry, and stability - culminating in discrete categorisations of TR risk: low, medium, or high risk. This approach is consistent with ecological threshold frameworks in restoration (Hobbs & Norton, 1996; Suding et al., 2004; Suding, 2011) and aligned with the decision-making emphasis of the SERA Standards (2021). When integrated with the cost model and associated visual representations (Fig. 4, 7), this approach enables practitioners to systematically translate complex environmental and economic data into transparent, site-specific management decisions.

The RSCAF proved effective in the Glenthorne case study, where it assessed the suitability of TR by considering salinity, erosion risk, soil stability, and pH. This tool helps practitioners determine where TR is biophysically suitable and where alternative methods (e.g., soil solarisation (Al-Shammary et al., 2020) or mulching (Navarro-Pedreño et al., 2017)) may be more suitable. Similarly, the economic model highlights that economies of

scale can be achieved with TR, offering significant cost savings when larger areas are targeted.

The RSCAF and TR cost model expands site analysis from a static description into a way of thinking about landscapes as dynamic systems with different possible restoration trajectories (Hobbs, 2008). These systems have thresholds (Andersen et al., 2009; Holling, 1973; Suding & Hobbs, 2009) and trade-offs (Folke et al., 2004). Instead of only cataloguing soils and slopes, the framework invites practitioners to ask: *Where can we safely intervene, and where should we hold back? Where is it worth investing heavily, and where will a lighter touch achieve more?* This shift has consequences for how projects are scoped, staged, and communicated.

At a design level, zoning landscape management units into low, medium, or high risk reframes the drawing board. Grasslands on stable plateaus can be envisioned differently from those on slopes prone to erosion. Drainage lines demand minimal disturbance. These distinctions flow into spatial composition, circulation, planting palettes, and maintenance strategies (Ignatieva, 2010). They also sharpen the narrative of a project - why certain places become grassland, why others host woody planting, and why some are left alone.

Most importantly, the framework makes restoration a more legible design act. Maps and thresholds help clients, contractors, and communities see not just *what* is proposed, but *why*. They open up space for conversations about value - whether in terms of biodiversity gains, cultural resonance, or lived amenity. The framework provides common ground for engaging stakeholders including First Nations perspectives and ecological expertise alongside design intent. In this sense, it embodies socio-ecological thinking (McGinnis & Ostrom, 2014; Ostrom, 2009; Poteete et al., 2010) in practice. It renders the messy realities of soil, seed, and cost into designable choices that are open, transparent, and future-oriented.

4.7.1 Economies of Scale in Topsoil Removal

Analysis of germination data revealed that approximately half of all seedlings originated from the uppermost 2 cm of the soil profile. Germination frequency exhibited an almost linear decline with increasing depth, ceasing beyond 12 cm (Fig. 6A). A major finding of this study is the economies of scale that can be achieved with TR. The cost model demonstrated that while TR becomes more expensive in absolute terms with increased depth and area (Fig. 6B), when this is considered alongside the efficiency of removing non-native propagules, the increased depth and area boost cost-efficiency. To illustrate, Fig. 6C shows that the cost decreases significantly when comparing the 1 ha scenario with the 5 ha scenario. There are also diminishing returns - doubling the area again to 10 ha results in comparatively smaller savings. In our model, the savings are due to the high float cost being spread as the area is increased[2]. Capitalising on this cost-saving allows restoration projects to maximise their ecological impact while keeping costs manageable (Török et al., 2011). This information is relevant for decision-makers, who are often under pressure to minimise costs, but could use these results to balance cost-efficiency with ecological outcomes.

However, focusing exclusively on these upper layers risks insufficient reduction of the residual non-native seed and bud bank, potentially necessitating increased long-term management if invasive species continue to dominate post-restoration. These findings align with previous research (Glass, 1989; Weidlich et al., 2020), which emphasises the necessity of thorough seed bank depletion to ensure restoration success. Furthermore, Wakshum et al. (2018) and Weidlich et al. (2020) demonstrate that partial seed bank removal can lead to continued invasive species persistence and increase future management demands. Consequently, future research should systematically evaluate the optimal depth and extent of

[2] The high float cost may be a key negotiation point when working with environmentally conscious contractors, as businesses may be willing to absorb this expense as a co-contribution to support their Sustainable Development Goals (SDGs). For very large projects, this also raises the question of whether organisations could reduce reliance on contractors by investing in their own equipment, although such an approach would still require skilled operators and management capacity.

topsoil removal required to suppress weed re-establishment while balancing economic and ecological trade-offs. Longitudinal studies examining post-restoration outcomes across varying depths and management regimes would be particularly valuable in refining best practices for seed bank management and ensuring the long-term sustainability of temperate grassland restoration efforts.

Nevertheless, given that the greatest cost savings per unit seed bank removed are achieved at the shallowest depths, a fruitful area of investigation may be determining whether shallow topsoil removal could be combined with chemical fallow (Gibson-Roy et al., 2010) (i.e. repeated glyphosate knockdown spraying) over several years to achieve a similar reduction in the viability of the soil seed bank as topsoil removal to greater depths at a much lower cost.

A final consideration is the ongoing arrival of non-native seeds, which is particularly relevant in peri-urban grasslands. Even if the existing seed bank is completely removed, new seeds will continue to arrive via wind, water, animals, machinery, and foot traffic. This means that the benefits of deeper removal diminish once the remaining seed bank is similar to the amount of new seed arriving each year. The actual arrival rate is *very* hard to measure, so this point is necessarily theoretical, but it highlights an important principle: topsoil removal alone cannot keep a site weed-free in a landscape where new seeds are constantly introduced. Combining shallow removal with regular follow-up management, surveillance, or buffer treatments may therefore be a more practical and cost-effective long-term approach.

4.7.2 Application of the frameworks: case study at Glenthorne National Park-Ityamaiitpinna Yarta

The utility of this framework is illustrated in the Glenthorne case study, where it demonstrated positive potential as a practical and robust tool for assessing environmental risks associated with TR. Based on site-specific data (see Table 3 and Fig. 7), the RSCAF highlighted that the plateau (LU_one) is where TR will achieve the best economic and environmental outcomes. In contrast, soil erosion risk on the undulating slopes (LU_two) poses environmental

concerns, including the release of elevated nutrients and colloidal clays into the sensitive receiving environment (LU_three). Although TR is theoretically possible in other landscape units, altering the landscape restoration approach may lead to better environmental outcomes. The spatially translated knowledge from the conservation framework was used to inform a site-specific restoration plan, which is displayed in Fig. 8. In areas where TR is more likely to cause environmental issues, alternative methods such as chemical fallow and tube stock planting with mulching can be employed on slopes (LU_two) and drainage lines (LU_three) to manage erosion risks. Therefore, TR was only applied on the plateau (LU_one), where the framework indicated the lowest risk of soil degradation.

Topsoil removal produces displaced topsoil that must be managed. Two pathways exist: (i) keep on site, where it can be spread thinly and stabilised or stockpiled with erosion and weed controls; or (ii) remove off site, which is costly but may enable productive reuse, for example on roadsides in rural areas with adjacent broadacre fields where topsoil can be added to improve productivity. Off-site options may also require biosecurity checks for weeds and propagules, as well as soil contamination assessments and regulatory approvals. Detailed logistics and costing for managing the overburden are beyond the scope of this study. In the Glenthorne case study, overburden was redistributed to the plateau's edges, treated with chemical fallow, and seeded with a dense shrubby mix to suppress weed growth. In this scenario, the slope (LU_two) was assigned chemical fallow plus woody tube stock planting to mimic a grassy woodland structure, while the drainage line (LU3) was restored using bush care methods and selective tube stock planting to minimise disturbance to the riparian community.

This tailored strategy is just one example of how the RSCAF can be used to plan for landscape-scale grassy community restoration while minimising environmental risks and maximising restoration outcomes by applying the right techniques to the right areas. Once restoration practitioners decide to employ TR based on the RSCAF, they can fine-tune the TR area using the cost model. For example, practitioners may decide that, given the germination

rates, the target depth should be 12 cm. The cost model indicates that the greatest cost savings are achieved at a scale of approximately 5 ha at this depth. Therefore, land managers may opt to remove topsoil from the plateau to a depth of 12 cm over 5 ha. Fig. 6B indicates that based on the site-specific data from Glenthorne the cost should be around AUD 60k.

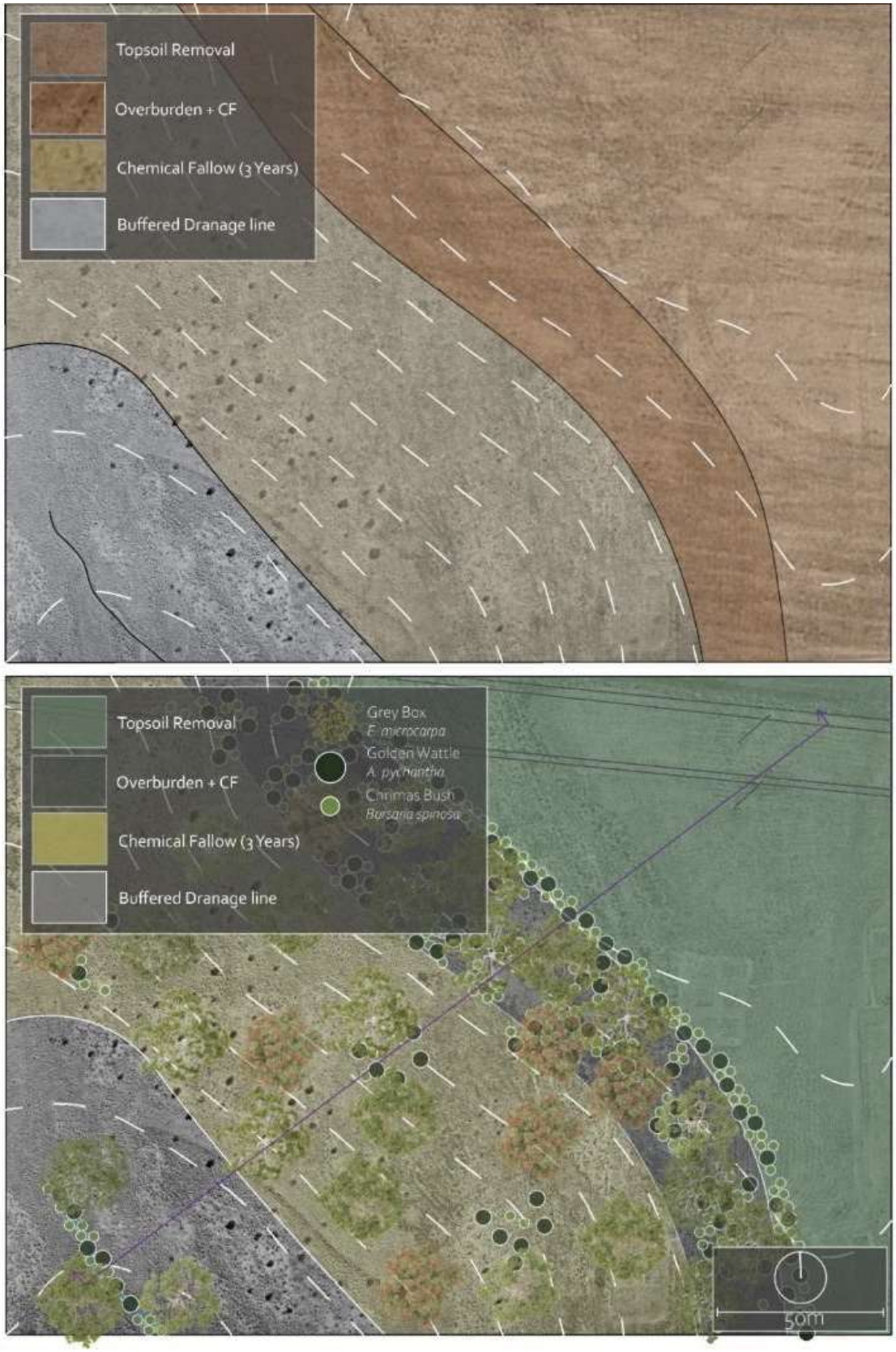

Figure 8: Integrating topsoil removal (TR) with landscape-specific restoration techniques at Glenthorne National Park. This figure demonstrates a restoration plan based on the TR cost model and RSCAF. TR is applied selectively on the ridge, where conditions are most favourable. The overburden is redistributed to the edges and treated with chemical fallow and eventually seeded with a dense shrubby mix to suppress weed growth. The slope is treated with chemical fallow and selective woody tube stock planting to mimic a grassy woodland, while the drainage line is restored using selective tube stock planting. The upper panel illustrates the restoration interventions across the landscape; the lower panel shows the planting plan.



Figure 9: Cross-sectional view of the Glenthorne restoration site, highlighting the plateau (LU_one) as the most suitable area for topsoil removal and direct seeding relative to the adjacent slope and drainage line units.

4.7.3 Challenges of restoring grasslands on different soil types

While applying site-specific data to the soil conservation framework is essential, we believe that the framework we developed can serve as a rapid assessment tool across all soil types and landscapes in Australia. This is because the soil conservation ranges it draws upon are intrinsic to soil function (Hazelton & Murphy, 2016) rather than place specific; for example, dispersive soils are unsuitable to leave bare regardless of climate, and soils with extreme pH present challenges under any conditions. At the same time, we acknowledge that each soil type, landscape element, and climate presents unique challenges for landscape restoration (Bertuol-Garcia et al., 2023). Therefore, thoroughly understanding and surveying your site's soil is crucial before commencing any earthworks. By feeding site-specific data into the framework, these unique challenges posed by different soil types should be effectively addressed.

The following examples highlight how these intrinsic soil characteristics manifest in different contexts. Sandy soils, for instance, likely show different patterns of non-native seed viability with depth (Glass, 1989; Wakshum et al., 2018). They can also present problems

such as non-wettability and low water-holding capacity, which require careful amendment and timing adjustments in restoration efforts. Likewise, TR in soils with elevated subsoil salinity, sodicity, or acidity may lead to scald formation once topsoil is removed (Hazelton & Murphy, 2016), exacerbating erosion risk and reducing native plant germination.

Such challenges are not just theoretical; the Glenthorne National Park case study demonstrates how soil properties can directly influence restoration outcomes. Scheduling earthworks when soils were moist from early autumn rainfall resulted in machinery losing traction, necessitating the suspension and subsequent postponement of the intervention until the following year. This field incident underscores the crucial importance of synchronising topsoil removal with optimal soil moisture conditions on Vertosols to ensure effective and uninterrupted operations. Moreover, deploying heavy machinery on clay-rich soils at or near field capacity substantially increases the risk of soil compaction, which can impair root penetration, reduce drainage efficiency, and exacerbate seasonal waterlogging. Thus, the incident highlights the importance of thorough site assessment and planning to prevent soil structural damage and avoid costly project delays in restoration activities involving Vertosols.

Vertosol soils are characterised by very high clay contents, high water retention, and high fertility (Isbell, 2021), and are relatively uncommon in the Adelaide region. These clays undergo significant seasonal movement, swelling when wet and shrinking when dry, leading to soil turnover. This process may result in deeper seed banks that are more effectively protected from physical and chemical disturbance. As such, removing 12 cm of topsoil may not be sufficient to eliminate non-native seed banks entirely in Vertosols. Additionally, their higher fertility can cause non-native disturbance specialist plants to regrow more vigorously after TR, complicating restoration efforts. Although Vertosols may complicate restoration, they also offer opportunities for long-term ecological resilience. Their shrink-swell properties and gilgai features may also help restore natural landscape heterogeneity, potentially benefiting native species through preferential germination of some species in the wetter depressions compared to the better-drained mounds. However, these soils must be managed

carefully to prevent erosion and maintain soil health, which necessitates ongoing monitoring and adaptive management.

4.8 Conclusion

This study offers actionable, evidence-based guidance for policymakers, practitioners, landscape architects, and planners to support the design and implementation of restoration projects that are both cost-effective and environmentally sustainable. Based on the presented findings, it is recommended that topsoil removal (TR) be preferentially considered for urban and peri-urban areas, where elevated land values, limited spatial extent, and immediate ancillary benefits, such as enhanced visual amenity, can justify the financial investment. For regional or large-scale restoration initiatives, stakeholders should conduct thorough site-specific cost analyses, as the higher absolute costs associated with TR may not consistently yield sufficient returns in ecological or economic terms. Furthermore, practitioners are advised to leverage economies of scale where feasible by consolidating restoration efforts and targeting areas identified as optimal using the RSCAF. The integration of these decision-support tools enables the strategic prioritisation of sites and tailoring of interventions to specific landscape and soil contexts. Collectively, these recommendations underscore the importance of aligning restoration strategies with local biophysical and economic conditions to enhance project viability and long-term sustainability.

The site-specific case study at Glenthorne National Park underscores the importance of tailoring restoration techniques to local conditions, particularly when working with challenging soils such as Vertosols. By combining topsoil removal with alternative methods, practitioners can reduce environmental risks while promoting long-term ecological resilience.

By integrating economic modelling with rapid soil assessments, it supports informed decision-making that can optimise resource allocation and enhance the effectiveness of large-scale restoration initiatives, aligning with broader environmental policy goals such as biodiversity conservation and the delivery of ecosystem services.

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Chapter 5

PIXEL PATCHWORK: MONITORING TEMPERATE GRASSLAND RESTORATION SUCCESS WITH SENTINEL-2 PHENOLOGY METRICS IN URBAN AUSTRALIA

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5.1 Context statement

I was fascinated by the potential of remote sensing in grassland restoration, and early in my PhD, I set out to explore how it could be applied to monitor grassland diversity. Initially, I focused on UAVs (drones), even obtaining my UAV licence and joining Bertram and Sami's research group to investigate drone-based monitoring. However, the results were underwhelming, and my enthusiasm began to fade. Around the same time, I attended Sami's remote sensing course, where I was introduced to Google Earth Engine - a tool that would change my entire approach.

After discovering Google Earth Engine, I shifted my focus from using drones to collect images toward extracting meaningful information from freely available satellite data for restoration monitoring. This chapter applies that approach to temperate grasslands in South Australia, using Sentinel-2 imagery to track both the initial establishment of restored grasslands and their subsequent performance compared with remnant patches. It then considers what these patterns mean for long-term monitoring and adaptive management. For a landscape architecture audience, the aim is to demonstrate a repeatable satellite-based workflow that distinguishes whether a grassland is functioning more like a native perennial or a non-native annual system. The chapter also helps guide the timing of interventions and communicates progress to stakeholders.

I focus on using Sentinel-2 imagery (satellite images from the European Space Agency's Sentinel-2 satellites, which provide high-resolution, multi-spectral images) and the

Enhanced Vegetation Index 2 (EVI2). EVI2 is a two-band "greenness" metric, meaning it uses only the red and near-infrared wavelengths of light to estimate the amount of green vegetation. In plain terms, EVI2 increases as leaf area and photosynthetic activity increase; unlike the classic NDVI (Normalized Difference Vegetation Index), it is less prone to "saturating" or reaching its maximum in denser canopies with abundant green. From a year-round EVI2 time series (a sequence of EVI2 values collected throughout the year) we summarise grassland behaviour using two intuitive phenology metrics (measures of seasonal biological cycles). Peak of season (POS) is the height of the growing season - how "green" the community gets at its seasonal maximum. The length of season (LOS) is the number of days between the start and end of the rapid growth period - how long the community remains actively growing. These two numbers provide designers, ecologists, and managers with a common language: POS captures how "leafy" a system becomes; LOS reflects how long it remains productive.

The study spans a gradient of sites from the coastal plain of Kaurna Country/Adelaide (Csa, grading locally toward warm-summer Csb) to inland, semi-arid country in the Riverland (transitioning toward BSk). Across this gradient, we paired restored/reconstructed patches with unrestored controls where possible and included high-quality remnants as references. All data processing was done with Google Earth Engine for accessibility and speed, then analysed in R (a statistical programming language); the full approach is reproducible with modest coding and, increasingly, can be scaffolded with AI-assisted scripting.

Two main findings guide practice. First, POS reliably distinguishes between native and non-native grasslands: unrestored, non-native sites reach higher seasonal peaks, while restored and remnant native sites peak at lower levels - reflecting field observations. Second, LOS, influenced by climate and management, is less consistent. While restored sites may have longer seasons and remnants vary, POS remains a dependable indicator. LOS still aids scheduling but must be climate-contextualized.

This matters to landscape architects because monitoring underpins design intent. If a project promises a resilient, biodiverse grassland, evidence is needed to show that the vegetation is trending towards that state. A POS-based dashboard enables you to demonstrate that a muted summer signal exhibits native-typical function, facilitates time sensitive management actions, and tracks deviations for early correction.

This chapter aligns with the thesis's social-ecological perspective. Remote sensing, from a socio-ecological systems perspective (McGinnis & Ostrom, 2014), broadens the evidence available to actors such as Traditional Owners, councils, and design teams. Over time, POS and LOS track if management helps shift grasslands towards stable, native dominance, or if further interventions are needed. As seen in previous chapters, this approach tracks the effectiveness of interventions and guides future management.

The method is deliberately straightforward. Sentinel-2 images, with their 10-metre resolution, are taken over the mapped management areas, filtered for clouds, processed to calculate EVI2, and then smoothed into time-series to generate POS and LOS (in simple terms, this means measuring grassland greenness and tracking how it changes across the season). The workflow is designed to be accessible for small teams and provides quick feedback to support in-season decisions. The real advantage is the ability to compare the trajectories of actively managed grasslands against either successfully restored or remnant grasslands, and to detect in near-real time if those trajectories begin to diverge. This allows managers to step in with targeted actions before problems become entrenched.

Monitoring only matters if it highlights the things that count. The metrics used here focus on what is most important: whether seasonal growth patterns align with management calendars and whether the results can be easily understood and valued by stakeholders. POS and LOS are not a substitute for field observations or cultural knowledge, but a complement to them; providing a shared, regularly updated picture that supports better dialogue and decision-making.

This research has been submitted to Applied Vegetation Science to share practical satellite monitoring for urban and post-agricultural restoration. That outlet's audience aligns with the technical subject matter of this chapter. The work's contribution is twofold: demonstrating a conceptually simple method to distinguish between native and non-native grasslands, and framing it for practical design and management, not just remote sensing.

Finally, this chapter is linked to Chapter 6, which explores the values underlying restoration and landscape architecture. The satellite metrics here make value judgments visible and discussable - what counts as success, whose vision is being served, and whose care and aesthetics shape public space. POS and LOS are tools for practitioners and communities - supporting cultural continuity, seasonal patterns, biodiversity, public acceptance, stewardship, and equity.

The specific research questions addressed in this chapter are:

- *Q4.1 To what extent can phenological metrics derived from Sentinel-2 EVI2 (e.g., length of season, peak greenness) differentiate restored grasslands from unrestored or non-native vegetation?*
- *Q4.2 How effectively does a Sentinel-2 EVI2 time-series approach detect early indications of restoration progress or setback in small, fragmented urban sites?*
- *Q4.3 What practical constraints - such as spatial resolution, cloud cover, data availability, or minimum site size - affect the usefulness of Sentinel-2 for monitoring urban grassland restoration?*

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Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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Applied Vegetation Science

RESEARCH ARTICLE

Pixel Patchwork: Monitoring Temperate Grassland Restoration Success With Sentinel-2 Phenology Metrics in Urban Australia

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ABSTRACT

Questions: More than 99% of Australia's native temperate grasslands have been lost since European colonisation, creating a need for restoration to retain their biodiversity. Remote sensing is a powerful tool for monitoring restoration, yet accessible approaches are needed to extract meaningful, restoration-relevant information for diverse practitioners. Here we ask if *Sentinel-2*–derived time series can monitor the success of grassland restoration by identifying significant phenological differences between non-native and native grassland vegetation in time and space.

Location: The study spanned six geographically distinct regions in southern Australia, strategically positioned along a landscape and management gradient extending from unrestored to restored to remnant intact sites, covering a spatial continuum from the coastal mediterranean climate of Adelaide to the semi-arid landlocked Riverland region, which supports remnant temperate grasslands.

Methods: We analysed temperate grasslands using *Sentinel-2* EVI2 (Enhanced Vegetation Index) time series within the web-based Google Earth Engine platform and using R software. The study included three management categories (remnant, restored and control) with vegetation cover assessed qualitatively by experienced practitioners to confirm these classifications. Phenology modelling was used to fit seasonal growth curves and extract two key metrics: the length of the growing season (LOS) and peak of season EVI2 values (POS). Modelled LOS and POS values were compared across management categories using ANOVA (analysis of variance).

Results: While we did not identify significant differences in the LOS, we did find significantly different POS values between grasslands dominated by non-native plants and by native plants. Control sites had POS values 44% higher than restored sites, likely due to the greater leaf area of non-native vegetation. In contrast, restored grasslands had POS values only 8% lower than remnant reference sites, indicating restored grasslands closely resembled remnant grasslands. This suggests that POS is a reliable metric for distinguishing between native and non-native grasslands.

Conclusions: The results demonstrate that phenological metrics derived from Sentinel-2 pixels effectively capture differences in vegetation composition, providing a practical tool for assessing restoration success. Readily available remote sensing tools like this can support diverse restoration practitioners in prioritising conservation efforts and improving restoration strategies, particularly in urban contexts where grasslands are small and fragmented and continue to face significant anthropogenic pressures.

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1 | Introduction

Native temperate grasslands are among the most threatened ecosystems in Australia, with less than 1% of the area they occupied before European colonisation remaining today (Hyde 1995; Kirkpatrick et al. 1995; Williams et al. 2015). Restoring grassland ecosystems is a conservation priority globally (Strömberg and Staver 2022) and locally (Hyde 1995). Yet assessing the success of restoration efforts remains challenging. Traditional monitoring methods are often labour-intensive, costly, and may lack the spatial and temporal resolution necessary for comprehensive evaluation.

Satellites have been employed in monitoring grasslands, although analyses have predominantly occurred at broad spatial scales, have depended heavily on high-quality calibration and validation data and have required expertise in remote sensing (RS) (Ali et al. 2016; Fauvel et al. 2020; Thornley et al. 2023; Guevara-Torres et al. 2024). This severely limits the applicability of existing grassland RS methods for land managers. A new generation of easy to access tools to monitor restoration success are needed to address this gap.

Freely available remote sensing satellite imagery offers substantial opportunities for cost-effective, large-scale vegetation monitoring (Kuenzer et al. 2014; Pettorelli et al. 2017). Advancements in satellite technology and the democratisation of high-quality data have made such resources more accessible over the past two decades (Shoko and Mutanga 2017; Wang et al. 2024). Billions of dollars in funding from the US, the EU and other countries have supported satellite imagery, making key Earth observation datasets increasingly accessible; with platforms including Landsat and Sentinel-2 now freely available. Landsat's adoption of an open-access policy in 2008 paved the way for this democratisation (Woodcock et al. 2008), which was further bolstered with the launch of Sentinel-2 in 2015 (European Space Agency 2024a), with its open-access policy from Day 1. Despite increasing accessibility, the main barrier for practitioners remains the level of expertise needed to manage and interpret satellite data effectively.

The technology for accessing and analysing satellite imagery has undergone significant advancements. While traditionally confined to the domain of specialist RS, the release of tools and data processing capabilities is rapidly dismantling barriers. One leading powerful tool for increasing the accessibility of RS is Google Earth Engine (GEE). It was launched in 2010 and is free for non-commercial use (Gorelick et al. 2017). GEE hosts one of the largest publicly available data catalogues and is powered by Google Cloud which provides powerful parallel processing capabilities, allowing users with only an internet connection to explore changes across the entire planet (Google 2024). The combination of GEE for initial data processing and the open-source R statistics software (R Core Team 2025) for subsequent spatial analysis enables a comprehensive approach to RS data analysis. Ecosystem restoration practitioners no longer need to be remote sensing specialists to obtain meaningful results, with advances in artificial intelligence being able to convert natural language prompts into code (Becker et al. 2023) and provide intuitive explanations of complex scientific concepts (Krenn et al. 2022).

With these advances making remote sensing more accessible, practitioners can now more readily investigate ecological patterns such as the distinct seasonal cycles that characterise native temperate grasslands. Despite being composed largely of perennial species, native temperate grasslands exhibit strong seasonal phenology. Phenology is the study of periodic plant and animal life cycle events (Sparks and Menzel 2002). Grassland phenology is largely influenced by Southern Australia's Mediterranean climate (Csa in the Köppen classification), which is characterised by cool wet winters and hot dry summers. In a typical year, native grassland plants begin growing following the summer-drought breaking rains in autumn. Grassland plants continue to grow actively through winter and into spring, before being dormant again by Summer (Groves et al. 2002). In cooler regions the season can extend into early summer. Flowering typically begins in early spring for many species as temperatures warm (Williams et al. 2015), and characteristically native grasses begin flowering later than broadleaf flowering herbs (forbs) (Morgan 1999). To cope with the summer drought, perennial species die back to dormant buds during summer when conditions for growth are poor due to low soil moisture and high temperatures (Williams et al. 2015).

Understanding the growth patterns of C3 and C4 grasses is important in native temperate grasslands, as these functional groups often co-occur: C3 grasses grow actively through the cool, wet winters, while C4 species remain largely dormant until triggered by much warmer temperatures for germination, growth, and flowering (Hagon and Groves 1977; McDougall 1989). In non-native grasslands, both C3 and C4 species are also present, but these communities are typically dominated by annual grasses with shorter growing seasons that die back earlier. In contrast native grasses are almost exclusively perennial (Gott and Williams 2015).

Remote sensing has been widely used to track phenological differences in grassland communities (B. A. Bradley 2013). Monitoring shifts in phenology enhances our understanding of ecosystem functioning and provides valuable insights into grassland restoration progress (Guevara-Torres et al. 2024). For instance, Shoko and Mutanga (2017) showed that the Sentinel-2 camera can pick up small seasonal growth differences between C3 and C4 plants, which is crucial for assessing changes in exotic plant communities, such as those brought about by climate change (Wilsey et al. 2017; Shoko et al. 2019). Additionally, research by Cheng et al. (2020) and Guevara-Torres et al. (2024) highlights the capacity of multiseasonal RS imagery to track short vegetation seasonal cycles and accurately classify grassland condition, further contributing to restoration management decisions. To demonstrate the potential of this technology in aiding land managers, this paper explores the effectiveness of Enhanced Vegetation Index 2 (EVI2) time series–derived from the Sentinel-2 satellites evaluating the success of grassland restoration efforts in time and space including at relatively small scales (e.g., <0.5 ha) in and around Adelaide, South Australia.

The utilisation of Sentinel-2 EVI2 time series enables us to model and quantify grassland seasonality (Bradley 2013; Wilsey et al. 2017; Cheng et al. 2020) and investigate how the timing of phenological attributes can enhance sensitivity to detect management changes. We define successful native grassland restoration interventions as a replacement of predominantly non-native, largely annual vegetation communities with

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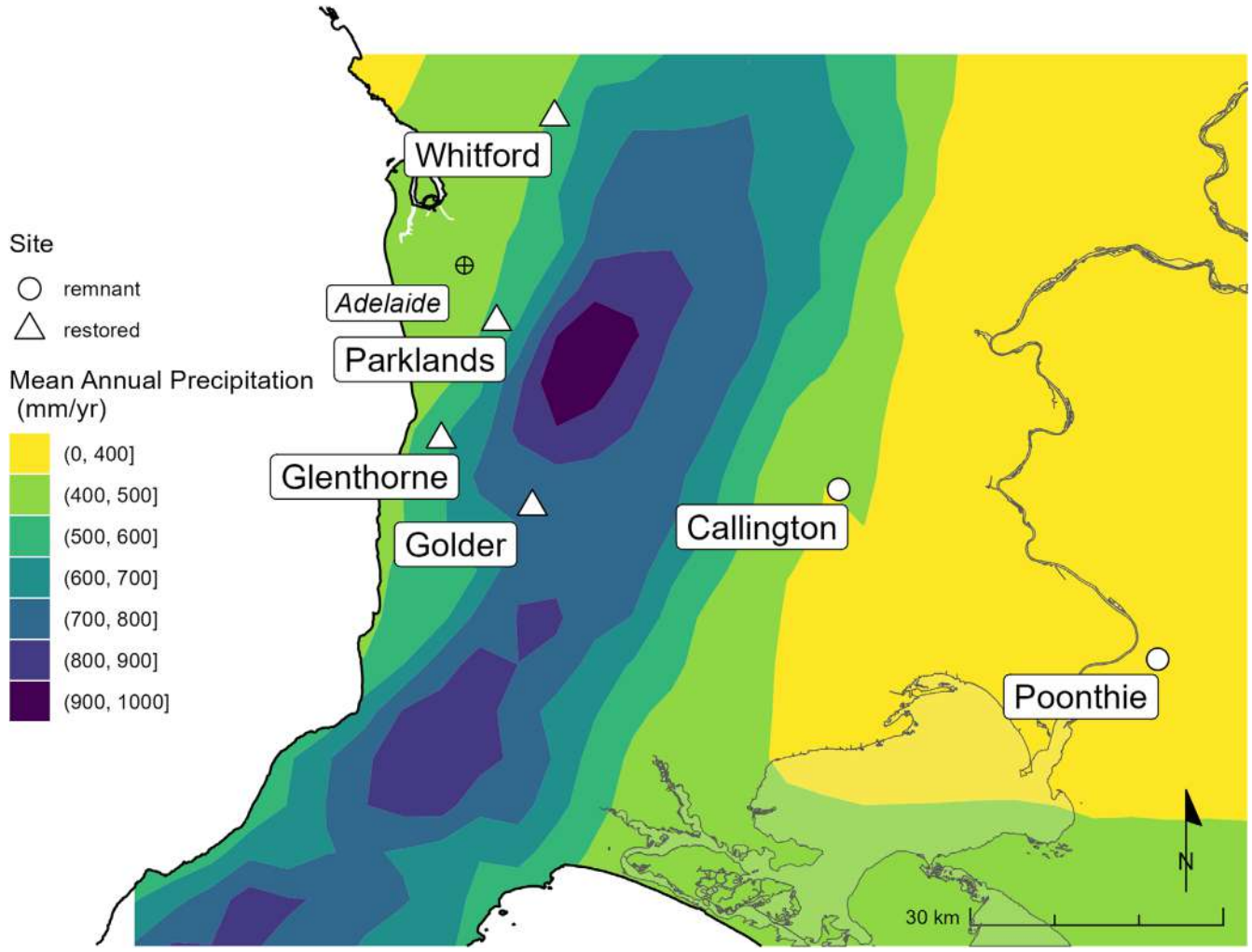

FIGURE 1 | Geographic distribution of study sites overlaying yearly rainfall. Symbols denote restored or remnant status. Adelaide city centre and the Murray River are marked for reference. Mean annual precipitation (mm/yr) calculated from 30-year monthly averages spanning 1981 to 2010, with a resolution of 2.5 km, data sourced from the Bureau of Meteorology (BoM 2024).

predominantly native, largely perennial vegetation communities. We ask the following: To what extent can RS tools like Sentinel-2 effectively monitor the success of grassland restoration projects by identifying key phenological differences between non-native and native vegetation communities? To test this, we investigated two hypotheses: (1) restored ecosystems should have a longer growing season compared to non-restored ones because they are dominated by perennial grasses, and (2) key phenological attributes in restored grasslands should resemble those in natural grasslands. To explore these hypotheses, we modelled the seasonal growth curve phenology (i.e., the annual green-up and brown-down of vegetation communities) across the management intensity gradient of control, restored, and remnant sites and compared two key phenological metrics: (1) the length of the season (LOS) and (2) the EVI2 value at the peak of the season (POS).

2 | Methods

We used Sentinel-2 satellite images to generate EVI2 time series across a gradient of management intensity, including control (dominated by non-native species), restored (areas where native biodiversity was enhanced) and remnant reference sites (intact native remnant grasslands). Data processing was conducted in GEE (Google 2024), while analysis and visualisation were performed using R (R Core Team 2025).

2.1 | Selection of Study Sites

The study spanned six geographically distinct regions positioned along a landscape and management gradient covering a spatial continuum from coastal mediterranean Adelaide to the arid landlocked Riverland. The native temperate grasslands included in this study are variously described as Iron-grass Natural Temperate Grassland of South Australia, Grey Box Derived Temperate Grassland of South Australia or more broadly as temperate grasslands. Scientific names for plant taxa follow the Australian Plant Census (Council of Heads of Australasian Herbaria 2024). The total distance covered by this study is approximately 90 km. Figure 1 shows the location of the study sites while Table 1 provides an overview of the key characteristics of the restoration gradient sites.

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TABLE 1 | Key characteristics of the restoration gradient sites in South Australia.

Site name	Restoration start date	Location	Restoration method	Vegetation community	Dominant species	Area (ha)
Glenthorne (Restored 2022)	2022	Adelaide	Topsoil removal, seeding	Restored Native C3 Grassland (not based on reference community)	Perennial native C3 grasses (*Rytidosperma* spp. And *Austrostipa* spp.)	0.8
Glenthorne (Restored 2023)	2023	Adelaide	Topsoil removal, seeding	Modelled on Grey Box Derived Grassland	Perennial native C3/C4 grasses, shrubs, forbs (*Rytidosperma* spp., *Enchylaena tomentosa*)	0.5
Glenthorne (Control)	—	Adelaide	Unrestored agricultural land	Exotic Agricultural Grassland (Unmanaged former pasture)	Non-native annual and perennial grasses/herbs	5.8
Parklands (Restored)	2013	Adelaide	Topsoil removal, seeding, supplementary planting	Modelled on Kangaroo Grass Grassland	Kangaroo Grass (*Themeda triandra, C4*), Wallaby Grass (*Rytidosperma* spp.), Native Daisies (*Vittadinia* spp.)	0.3
Parklands (Control)	—	Adelaide	Bush care (mowing, weed control)	Degraded Remnant Grassland (dominated by exotic species)	Non-native annual plants, native perennial grasses (*Rytidosperma* spp.), native forbs (*Oxalis australis*)	0.4
Whitford (Restored)	2000–2005[a]	Mt Lofty Ranges	Bushcare (mowing, weed control)	Modelled on Iron-grass Tussock Grassland	*Austrostipa* spp., *Aristida behriana*, *Enneapogon nigricans*	3.3
Whitford (Control)	—	Mt Lofty Ranges	Unmanaged, dominated by non-native plants	Exotic Agricultural Grassland (unmanaged former pasture)	Non-native plants (Wild Oats, Artichoke Thistle, Salvation Jane, Soursob)	1.5
Golder (Restored)	2015	Mt Lofty Ranges	Topsoil removal, seeding	Modelled on Iron-grass Tussock Grassland (offset restoration)	C3 Wallaby Grass (*Rytidosperma pilosum*), previously Native Wheat Grass (*Anthosachne scabra*)	3.5
Callington (Remnant)	—	Mt Lofty Ranges	Intact remnant	Iron-grass Tussock Grassland	Kangaroo Grass (*Themeda triandra*), Wallaby Grass (*Rytidosperma* spp.), Iron-grass (*Lomandra effusa*)	2.4
Poonthie Ruwe (Remnant)	—	Riverland	Intact remnant	Iron-grass Tussock Grassland (arid variant)	Iron-grass (*Lomandra effusa*), Wallaby Grass (C3), Kangaroo Grass (C4)	237

Note: The Site column lists each restored, control or remnant site included in the study; Restoration Start Date lists the year restoration began; Location indicates the region in which the site occurs; Restoration Method describes the primary techniques applied (e.g., topsoil removal, seeding, supplementary planting); Dominant Species identifies the main plant species present, distinguishing between native and non-native taxa; and Area (ha) denotes the size of each site in hectares.

[a]The exact year when grassland restoration began at Whitford is unclear from historic management plans. However, based on available information, it likely started between 2000 and 2005. Mowing and control of noxious weeds likely began around 1985, though these efforts did not explicitly consider native grasses at the time.

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We selected control sites using a paired-site design for accurate comparison with restored sites. Paired sites were chosen based on proximity to restoration sites (within 100 m) and similarity in vegetation structure, landform, slope and aspect to control for environmental variability. The selected control sites were dominated by non-native grassland vegetation and minimally managed, to act as a natural control.

2.2 | Detailed Descriptions of Restored Grassland Sites and Their Pairing With Remnant Grasslands

2.2.1 | Glenthorne Community Grassland Restoration Project (35.0602S, 138.5498E, Kaurna Country/Adelaide)

The Glenthorne Community Grassland restoration project is located in an urban national park on former agricultural land. Restoration began in May 2022 with a 1-ha native grassland, followed by the establishment of an additional 0.5-ha patch in May 2023. The first author managed the project during the study period, providing intimate site knowledge. Both areas were sown after topsoil removal, with the seed mix for the 2023 patch modelled on the 'Grey Box Derived Grassland' ecological reference community (DECCEEW 2010). The 2022 grassland is not modelled on a reference community and is dominated by perennial native C3 grasses (primarily Wallaby Grass (*Rytidosperma* spp.)). The 2023 site was still in early establishment and contained a mix of native perennial C3 grasses and chenopods (primarily Wallaby Grass (*Rytidosperma* spp.) and Ruby Saltbush (*Enchylaena tomentosa*) as well as non-native annual grasses primarily Wild Oats (*Avena barbata*) and Annual Ryegrass (*Lolium rigidum*)) and non-native herbs (primarily Bristly Ox Tongue (*Helminthotheca echioides*)). An adjacent section of unrestored, fallow, former agricultural land served as a control site for comparison. This site was dominated by the same non-native annual and perennial grasses and herbs as well as Variegated Thistle (*Silybum marianum*) and *Phalaris aquatica* (a perennial non-native grass).

2.2.2 | Adelaide Parklands Grassland (34.9345S, 138.6213E, Kaurna Country/Adelaide)

This longstanding grassland restoration project, initiated in 2013 (Haywood 2018), is located next to a degraded remnant native grassland, both sites are managed by the Adelaide city council. The restored grassland was sown following topsoil removal and there has been a significant amount of supplementary seedling planting and high intensity non-native plant control over the years (M. Starkey, pers. comm.) with the effect of the community being dominated by large native C4 grasses (mainly Kangaroo Grass (*Themeda triandra*)) which typically do not establish in large numbers following seeding in Adelaide's climate without above average summer rain events and/or fire. Wallaby Grass (*Rytidosperma* spp.) is co-dominant and the inter-tussock spaces are occupied by Native Daisies (*Vittadinia* spp.) and other forbs and low shrubs. Because the adjacent remnant grassland is heavily degraded and dominated by non-native annual plants it is included as a control site in this study.

2.2.3 | Golder Revegetation Project (35.1320S, 138.6685E, Peramangk Country/Adelaide Hills)

This project, managed by SA Water corporation since 2015, as a significant environmental benefit (SEB) offset, was sown and supplementary planted following topsoil removal. The restored grassland was dominated by the C3 Wallaby Grass, in particular *Rytidosperma pilosum*, but was initially dominated by Native Wheat Grass, *Anthosachne scabra* which is a relatively short lived C3 perennial known to colonise after disturbance. There is no paired control site for this project.

2.2.4 | Whitford Grassland (34.716939S, 138.697406E, Kaurna Country/Adelaide)

Located in the foot slopes of the southern Mt. Lofty Ranges (Durant and Berkinshaw 2005), this site was restored using bush care methods (Bradley 2002) namely mowing and weed control and there was no supplementary planting or sowing (J. Park, council employee, pers. comm., 2024). The site is dominated by *Austrostipa* spp. (Speargrass), *Aristida behriana* (Brush Wire Grass), *Enneapogon nigricans* (Black-head Grass), with non-native Wild Oats (*Avena barbata*) occurring between native tussocks. An unmanaged area dominated by non-native plants primarily Wild Oats (*Avena barbata*), Artichoke Thistle (*Cirsium vulgare*), Salvation Jane (*Echium plantagineum*) and Soursob (*Oxalis* sp.) to the south of the restored grassland was included as a control site.

2.3 | Intact Remnant Grassland Sites

2.3.1 | Callington Hill Grassland Flora Reserve (35.113521S, 139.067828E, Peramangk Country/Adelaide Hills)

This intact grassland reserve, the only publicly accessible protected grassland remnant in the Adelaide Hills region, was dominated by perennial native C3 and C4 grasses, with minimal non-native cover. Kangaroo grass and Wallaby Grass were the dominant species, with Iron-grass (*Lomandra effusa* and *Lomandra multiflora*) and Garland Lily's (*Calostemma purpureum*) being a more minor component, see Figure 2.

2.3.2 | Poonthie Ruwe Conservation Park (35.294S, 139.483E, Millewa-Mallee Country/Riverland)

A large remnant native temperate grassland in arid South Australia covering over 200 ha (Department for Environment and Heritage 2008). We only included the most intact southern half as mapped by (Guevara-Torres et al. 2024) to remove any potential confounding effect from the degraded vegetation (i.e., where vegetation cover is dominated by non-native plants). The study area was dominated by native vegetation, including the perennial herbaceous plant *Lomandra effusa* and *Lomandra multiflora* (Iron-grass) and the perennial grasses *Rhytidosperma* spp. (Wallaby Grasses, C3) and *Themeda triandra* (Kangaroo

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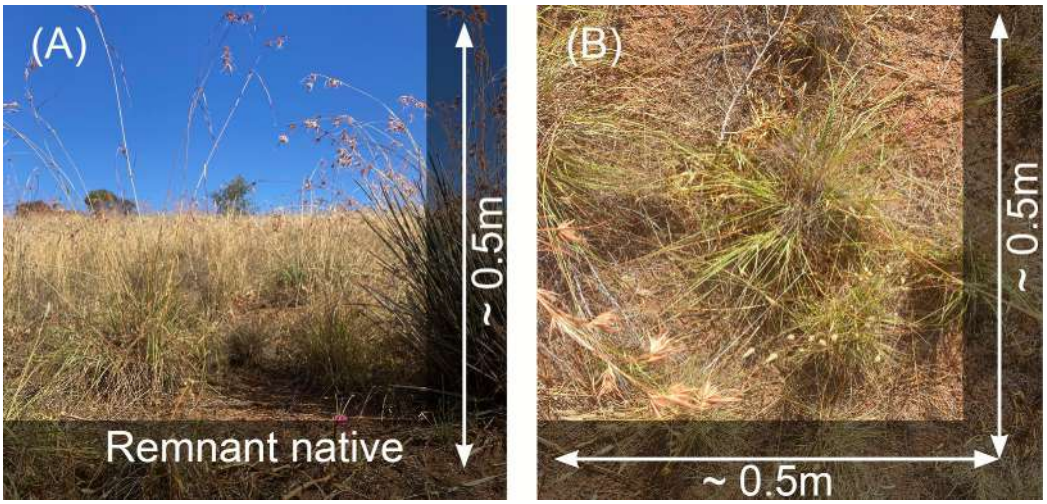

FIGURE 2 | Site photos from the Callington remnant site during the dry season in March 2024. Images are paired, with LHS (A) images taken in portrait at approx. 0.5 m, while RHS (B) images are taken from the top at the same spot and show approx 0.5 sqm.

Grass, C4) (Guevara-Torres et al. 2024). Non-native vegetation, primarily *Avena barbata* (Wild Oats), was also present in patches.

2.4 | Vegetation Cover Assessment and Verification of Management Categories

Vegetation cover was visually assessed by experienced practitioners to verify management categories (restored, control, and remnant). Management categories were defined based on practitioner knowledge of the site, such as areas where topsoil had been removed, areas where bushcare had been applied (Table 1), or areas that were unmanaged and treeless. The cut-offs for categorisation were as follows:

- Restored sites: > 50% of the vegetation cover is native
- Control sites: > 50% of the vegetation cover is non-native
- Remnant sites: > 50% of the vegetation cover is native in remnant ecosystems

This visual assessment was used to confirm the treatment areas, not for quantitative analysis, as the focus of this study is on comparing management categories.

2.5 | Satellite Imagery Processing

Sentinel-2 was selected for its high spatial resolution of 10 × 10 m pixels (compared to other freely available earth observation platforms), moderate temporal resolution (averaging around 5 days between acquisitions, compared to 16 days for Landsat 8) and availability of the required spectral bands to calculate EVI2 (red and near-infrared).

Surface reflectance imagery from Sentinel-2 (European Space Agency 2024b) was obtained from GEE, focusing on a defined region of interest (ROI), which were filtered for specific dates and cloud cover under 50%. Within GEE, the filtered image stack was cloud masked using the Sentinel-2 Cloud probability product (European Space Agency 2021). In R the imagery was clipped to match the boundaries of three management categories: restored grassland sites, control sites and remnant sites. To quantify the phenology of restored grasslands and control sites, we calculated the Enhanced Vegetation Index 2 (EVI2) (Jiang et al. 2008) for all cloud-free pixels. The analysis period reflects the timing of management interventions: For example, restoration at Glenthorne began in 2022, so earlier imagery was excluded. Remnant sites are long-established and stable, allowing analysis from 2019 onwards. EVI2 is calculated using the formula below:

$$\mathrm{EVI2} = \frac{2.5 \times (\mathrm{NIR} - \mathrm{Red})}{(\mathrm{NIR} + 2.4 \times \mathrm{Red} + 1)}$$

EVI2 is closely related to the Normalised Difference Vegetation Index (NDVI) (Xue and Su 2017) but offers several advantages that make it more suitable for this study. NDVI often saturates in areas with dense vegetation or high biomass, limiting its sensitivity in detecting variation under such conditions (Huete et al. 2002; Jiang et al. 2008). In contrast, EVI2 maintains improved sensitivity in high biomass environments and provides greater resistance to background noise from soil and atmospheric conditions (Miura et al. 2000; Gao 2000). These properties make it particularly useful in restoration contexts, where vegetation cover can be sparse, heterogeneous, or rapidly changing. Additionally, EVI2 omits the blue band used in the original EVI formulation, improving compatibility with sensors such as Sentinel-2, which offer consistent red and near-infrared bands but variable blue-band quality (Jiang et al. 2008).

After calculating EVI2 for each pixel, we generated time series for each management region using the median EVI2 value per acquisition date. Summary statistics, including minimum, maximum, mean, median, and standard deviation, were calculated to characterise pixel-level variation across management types and sites. The median was chosen as the primary summary statistic to reduce the influence of outliers, which are more likely in sites with fewer pixels or highly variable

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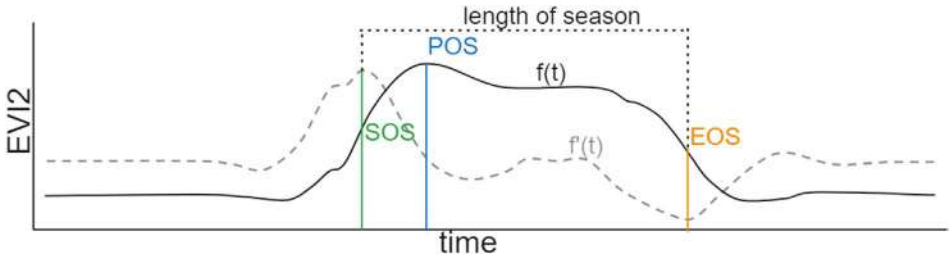

FIGURE 3 | Schematic of the vegetation phenology extraction methods employed in this study, including the EVI2 curve denoted as $f(t)$ and its derivative $f'(t)$, representing the rate of change. This figure is modified from a figure by Kong et al. (2022). Additionally, it illustrates key phenological parameters derived from the EVI2 time series: Start of Season (SOS), End of Season (EOS), Peak of Season (POS) and the length of the growing season.

vegetation structure. Although the exact monitoring years varied slightly between sites, all analyses used available data from within the study period (2019–2024). This was an intentional tradeoff as temperate grassland restoration is a relatively new field in the Adelaide region, with most projects initiated within the last decade. This approach allowed us to include a broader representation of existing management types, despite slight temporal variation across sites. These values are presented in Appendix S1. Figure 4 provides a time series for the Glenthorne ROI and shows raw EVI2 trajectories for three spatially paired management regions (a grassland restored in 2022, one restored in 2023, and an unmanaged, predominantly non-native control site).

2.6 | Spatial and Temporal Analysis

We exported EVI2 images for post processing in R. Images were spatially subset using polygons of defined management regions. To assess management-induced changes, we used statistical methods involving data smoothing, seasonal break detection and curve fitting to smooth the EVI2 time series, and then model each site's phenology. This approach enhances detection sensitivity by capturing phenological shifts and extracting key seasonal metrics—such as the start, peak and length of the growing season—across management areas. To do this we used *smooth_wWHIT* function in the *phenofit* R package (Kong et al. 2022). The function iterates the raw EVI2 time series data (Iteration 0) twice to find the best fit (Iteration 2). With this, we could quantify the length of the growing season (LOS), defined as the interval between the onset of rapid vegetation growth, that is, the start of the season (SOS) and the decrease in growth rate at the end of the season (EOS). These parameters are determined through analysis of the smoothed EVI2 curve (see Figure 3): SOS corresponds to the inflexion point where the first derivative of EVI2 reaches its maximum, indicating the start of rapid growth; EOS is identified as the point where the first derivative decreases to a minimum, signalling the end of rapid growth. Additionally, the peak of season (POS) is simply the maximum point of the EVI2 time series.

2.7 | Hypothesis Testing

LOS and POS values were grouped across years and by management categories. This process was repeated for all management categories for POS and LOS variables, resulting in a dataset with two outcome variables for each of the three management categories. Statistical analyses were then conducted in R on each outcome variable to evaluate the hypotheses. For Hypothesis 1, which posits that restored ecosystems should exhibit a longer growing season compared to non-restored ones, a one-way Analysis of Variance (ANOVA) was performed on LOS values across different management treatments. This was followed by Tukey's Honest Significant Difference (HSD) test for pairwise comparisons. Similarly, Hypothesis 2, asserting that key phenological attributes in restored grasslands should resemble those in natural grasslands, underwent similar analyses, this time focusing on POS values. LOS and POS values were visualised using box plots displaying the mean treatment outcome variable, allowing for easy comparison across different management treatments. The significance level for all statistical tests was set at $\alpha = 0.05$. All statistical analyses were conducted in R (R Core Team 2025).

3 | Results

Our analysis revealed distinct phenological patterns from EVI2 across varying management intensities.

3.1 | Grassland Phenology Modelling

The modelled Glenthorne EVI2 time series shown in Figure 4 depicts a gradient from native to non-native grasslands for the Glenthorne grassland restoration project. These management induced differences are clearly reflected by the modelled EVI2 time series, with the unmanaged control region characterised by a higher POS compared to both the grassland restored in 2022 and the early establishment phase of the 2023 restoration.

Prior to May 2023, the modelled phenology of the grassland established in 2023 was visually identical to the unmanaged control, which is expected because this area was also unmanaged before then. The LOS and POS were lower in the first growing season following restoration for both restored grasslands at Glenthorne—specifically, the grassland restored in May 2022 had a shorter LOS and a lower POS during the 2022 growing season, and the grassland restored in May 2023 exhibited the same pattern in the 2023 growing season (see Figure 5). This is because the pre-existing non-native standing vegetation, and its propagules were removed during topsoil removal. Because restored areas were seeded following topsoil

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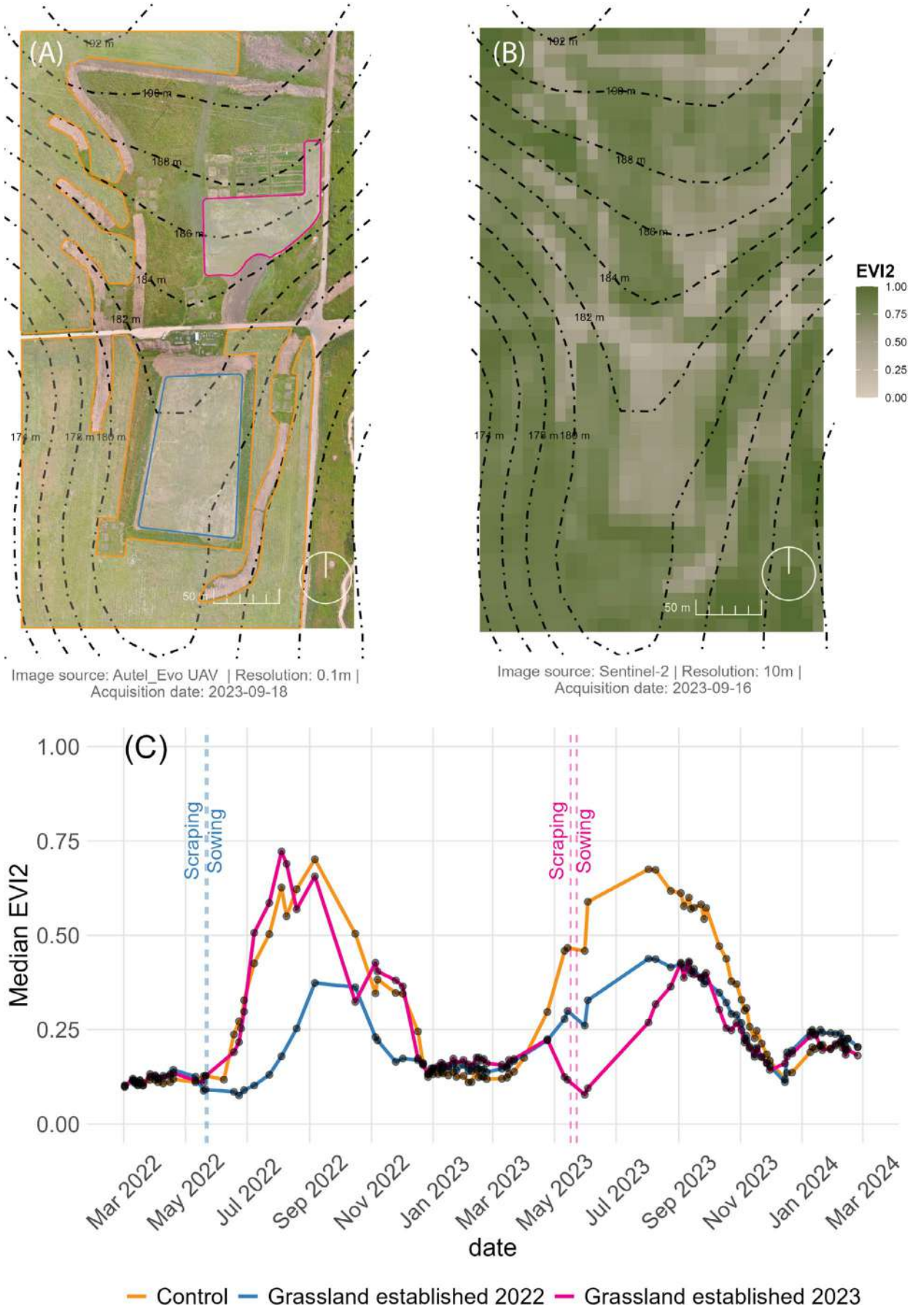

FIGURE 4 | Legend on next page.

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FIGURE 4 | Pixels to phenology: Panel A presents a high-resolution UAV aerial ortho mosaic with colour-coded management regions matching those in Panel C. Panel B shows the corresponding Sentinel-2 image captured within 2 days of the UAV mosaic in Panel A. EVI2 values from Panel B are used to calculate the median EVI2 for each management region, forming data points for the time series in Panel C across the three spatially paired management regions.

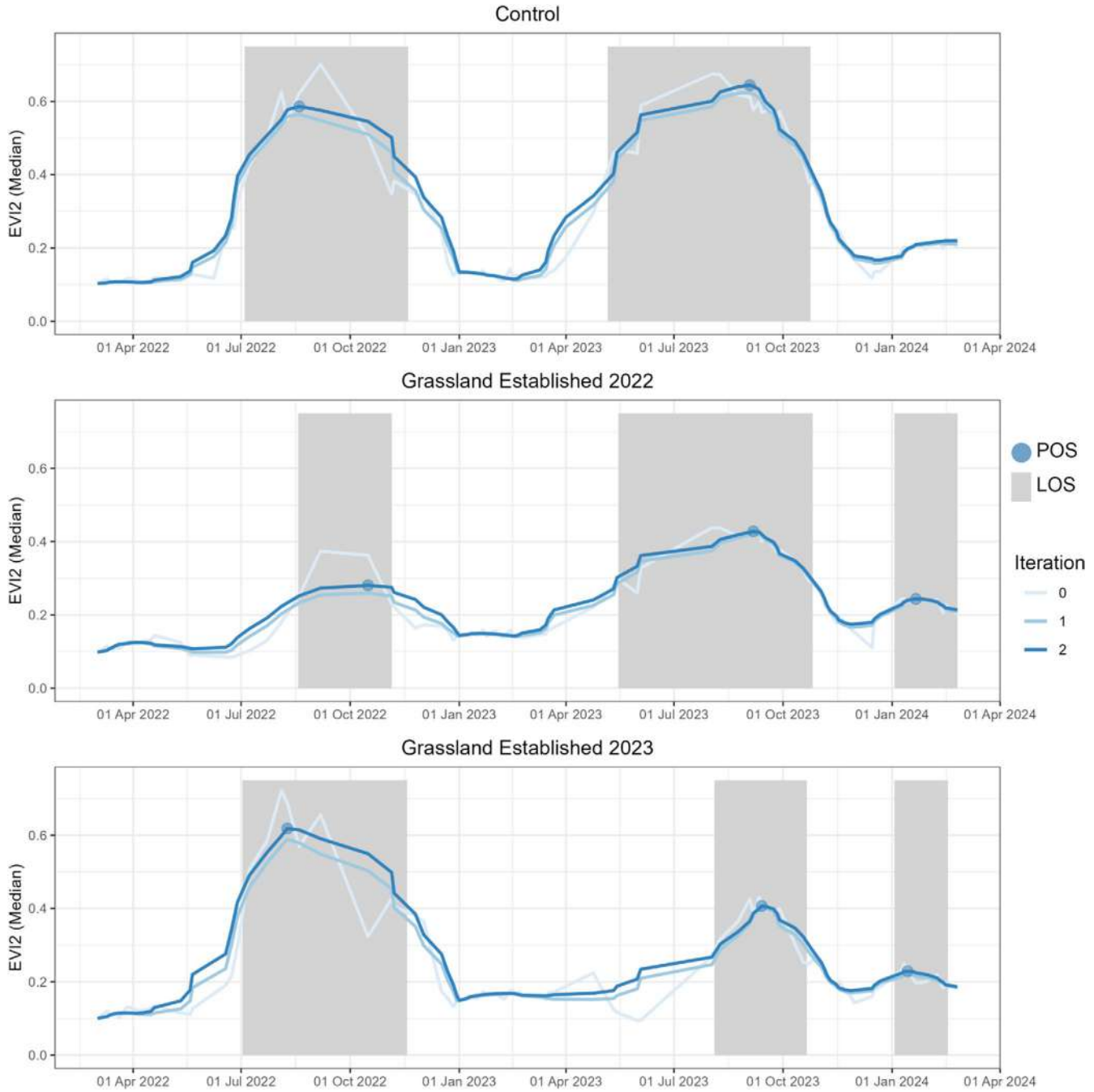

FIGURE 5 | EVI2 time series depict a gradient from native to non-native grasslands for the Glenthorne grassland restoration project. The unmanaged control site is dominated by non-native vegetation, while the grassland restored in May 2022 has greater native plant cover than the site restored in May 2023. The transparent blue point marks the peak of the season. High summer rainfall in December 2023, triggered a short second season for both restored sites. The dark blue line represents the second iteration of EVI2 smoothing; details on iterations are provided in the Methods section.

removal, vegetation had to establish from seed, a process slower than re-growth from dormancy or vegetative structures (e.g., tubers). Field observations confirmed that the plants in the grassland established in 2022 did not reach maturity until the second year (2023), which is reflected in both a longer LOS and a higher POS in 2023.

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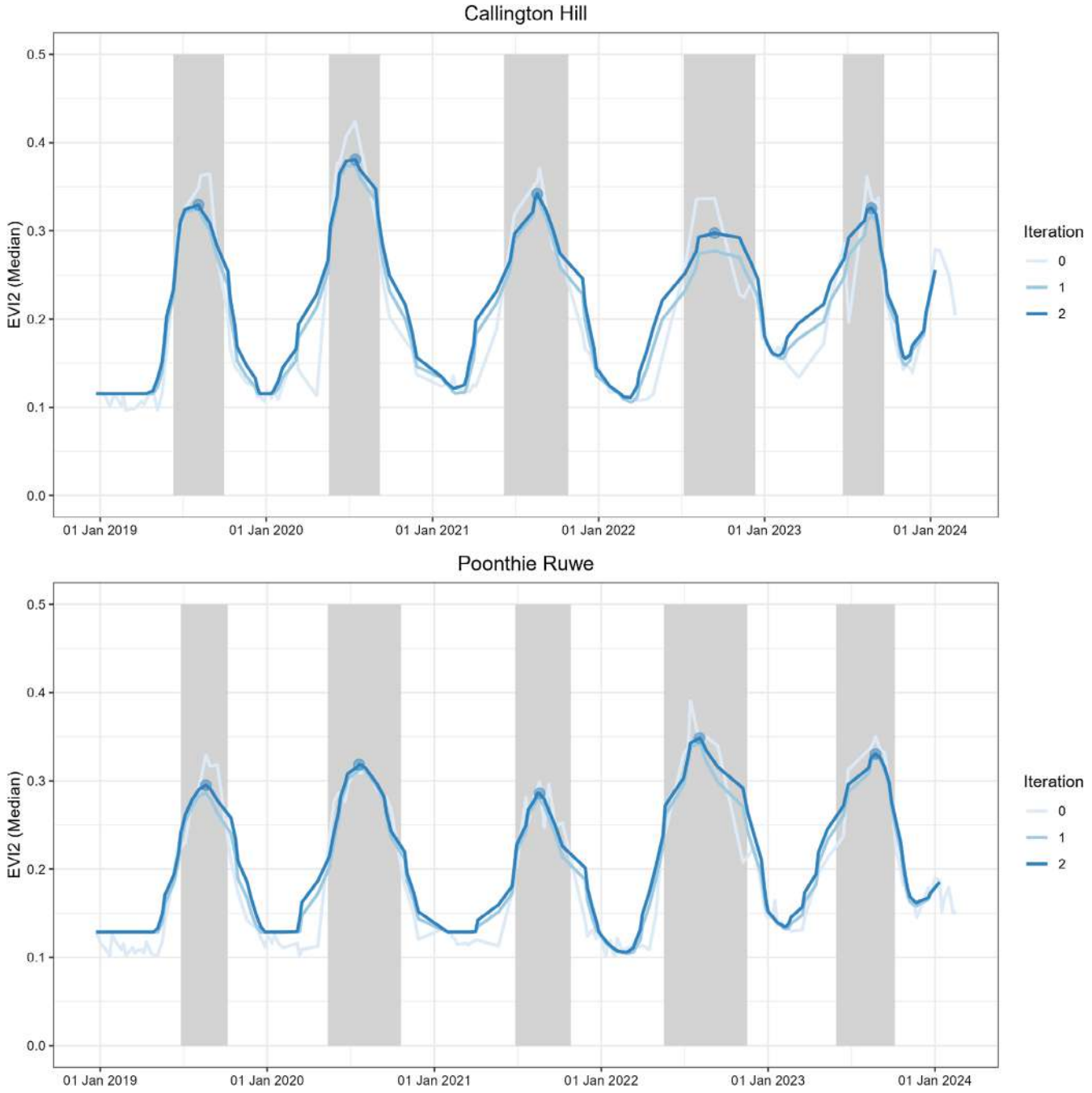

FIGURE 6 | The EVI2 time series illustrates the phenological dynamics of native remnant grassland ecosystems at the Poonthie and Callington sites. Phenology curves were modelled using phenofit in R. The grey box width represents the duration of the growing season (SOS to EOS). The transparent blue circle denotes the peak of the growing season. The dark blue line represents the second iteration of EVI2 smoothing; details on iterations are provided in the Methods section.

Phenology modelling picked up a second season at both restored sites during summer 2023/2024 (see Kong et al. (2022) for details), likely in response to unusually high December rainfall. Overall, modelled seasonality closely aligned with both management interventions and field observations.

To test whether these patterns held across other sites, the analysis was repeated for a further three grassland restoration sites. The results are presented in Appendix S1 and Figure 7, and overall, POS is consistently noticeably lower for restored versus control sites while the LOS appears slightly higher for restored sites versus control sites.

To test whether both POS and LOS resemble the characteristic seasonality of remnant grasslands, two remnant grasslands were analysed in the same way as the spatially paired restored/control sites. Figure 6 depicts the EVI2 time series and modelled phenology of native remnant intact grassland ecosystems at the Poonthie and Callington sites. Similar to the

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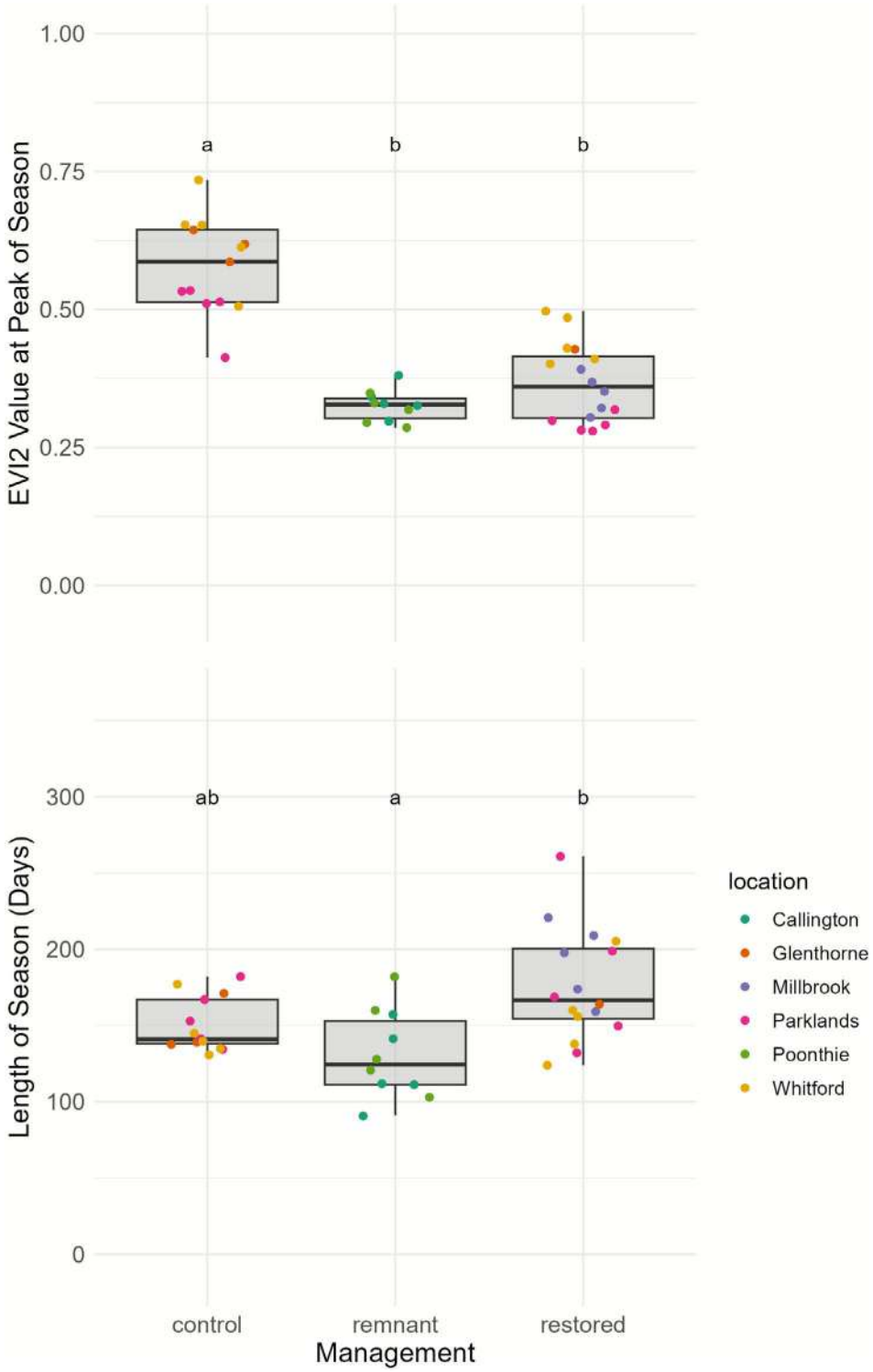

FIGURE 7 | Hypothesis testing: LOS (Length of Season) and POS (Peak of Season) values were combined across years and management categories for analysis. We tested hypotheses about LOS and POS differences between management categories using one-way ANOVA followed by Tukey's HSD test for comparisons. Significant differences are denoted using letters. Box plot graphs depicted mean LOS and POS values for each management treatment. Statistical significance was set at $\alpha = 0.05$.

restored sites, the remnant sites are also characterised by low POS values, however, the LOS is variable and consists of both long (e.g., both sites in 2022) and short seasons (e.g., Poonthie Ruwe in 2019).

3.2 | Hypothesis Testing

Contrary to our first hypothesis, we found no difference in the LOS between restored and control sites, and therefore accept

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the null hypothesis. The median LOS for remnant sites was 124 days, compared to control sites at 141 days. The median LOS of restored sites was the longest at 166 days, but also the most variable (Appendix S1 and Figure 7). LOS differed significantly between remnant and restored sites (Tukey's HSD post hoc test, $p < 0.05$; Figure 7), with no other pairwise comparisons showing significance.

Our second hypothesis was supported more clearly. We found that the median POS of restored systems was more similar between restored and remnant systems (8% lower for restored vs. remnant) than between restored and unrestored systems (44% lower for restored vs. unrestored), indicating that the non-native systems adjacent to restored grasslands have significantly higher EVI2 values at the peak of the season than restored and remnant systems (Appendix S1 and Figure 7). POS differed between control and restored sites as well as between control and remnant sites (Tukey's HSD post hoc test, $p < 0.05$ in both cases; Figure 7), but not between remnant and restored.

4 | Discussion

This study addressed the urgent need for monitoring of restored native temperate grassland ecosystems, which are threatened by habitat loss and fragmentation (Richter and Osborne 2014). In Australia, over 99% of native temperate grasslands have been mostly converted to urban or agricultural land uses (Williams et al. 2015; Williams et al. 2025), highlighting the critical importance of conserving and restoring remnants as well as reconstructing new patches—particularly within metropolitan, peri-urban and post-agricultural landscapes, where opportunities for ecological intervention exist.

We sought to determine the effectiveness of using Sentinel-2 in monitoring native Australian temperate grassland restoration success by detecting critical phenological differences between non-native and native grassland communities (Ali et al. 2016; Wilsey et al. 2017). We tested two key hypotheses: first, that restored grasslands would exhibit a longer growing season than non-native grasslands, and second, that restored grasslands would display phenological characteristics similar to those found in natural grasslands.

We show that Sentinel-2-derived data can be used to detect management-induced changes in grassland phenology. These results align with previous studies that have demonstrated the effectiveness of RS for monitoring vegetation responses to management (Ali et al. 2016; Shoko and Mutanga 2017; Wilsey et al. 2017; Shoko et al. 2019; Cheng et al. 2020; Wang et al. 2022). Our approach highlights the sensitivity of EVI2 time series in tracking management induced grassland responses. Notably, our method is simple, cost effective, and can be applied by practitioners without extensive RS expertise, making it a valuable tool for practical use in grassland restoration.

4.1 | Season Length Differences (LOS)

Our first main finding is that native grasslands do not seem to have a longer growing season compared to non-native grasslands, despite their perennial composition. The unexpected finding that the LOS was not significantly different in the direction we had anticipated challenges our assumptions about the distinctions along the management gradient. Instead, the finding suggests that control sites' biennial and perennial components, may effectively extend these grasslands' seasons—even though they are primarily dominated by annual non-native plants at the peak of the season. This prolonged seasonality may be driven by minor non-native components that persist later into the year and take fuller advantage of the period conducive to understory plant growth. Native perennials were largely absent in control sites during field visits, supporting the inference that the effect is likely due to non-native species. We note, however, that this interpretation is based on inference rather than direct vegetation survey data and should be treated with caution.

In contrast, the observation that restored sites had a longer season than remnant sites is more difficult to explain. It is widely recognised that restored sites generally do not have clear natural analogues and instead are novel ecosystems (Hobbs et al. 2009; Collier and Devitt 2016; Teixeira and Fernandes 2019). One key difference is the general lack of inter-tussock spaces in restored versus remnant sites (Williams et al. 2015), and another is the general dominance of C3 grasses in South Australian restorations. C4 grasses, which often contribute to later-season productivity, are typically absent in large numbers, although the Parklands site which showed the longest season (Figure 7) may be an exception due to substantial supplementary planting of C4 grasses (M. Starkey, former council employee, pers. comm., 2024).

Studies using RS have documented phenological differences between C3 and C4 species, with C3 grasses greening up earlier and senescing later in cooler, temperate regions like North America (Wilsey et al. 2017). However, in Mediterranean climates such as southern Australia, rainfall is more limiting than temperature. Similar studies in Kenya have shown that while C3 grasses green up earlier, both C3 and C4 species remain photosynthetically active in summer, responding to rainfall (Shoko and Mutanga 2017; Shoko et al. 2019; Cheng et al. 2020). Other studies comparing native and exotic phenology in grasslands show that exotic systems often green up earlier and senesce later due to (1) human selection for productive systems (Wilsey et al. 2014) and (2) filtering that selects for species growing outside of the normal growing period (Wolkovich and Cleland 2010).

The longer LOS observed at restored sites may also reflect local conditions: most are located in higher rainfall areas (Figure 1) and share land-use histories that likely increased soil nutrients compared to remnant sites. Additionally, restored sites often include patches of non-native vegetation, unlike the high-quality, low-exotic remnant sites included in this study. Collectively, these factors may explain both the extended LOS and higher variance observed in restored sites.

4.2 | Greenness–Intactness Relationship (POS)

Our second main finding is that there is an inverse relationship between 'greenness' (high EVI2) and 'intactness' (high % native cover). Looking at the time series for the establishing Glenthorne

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grassland sites, we observed that the restored areas exhibit lower POS values compared to the control site (Figure 4). The fact that the Glenthorne site was in its early establishment phase when the data were collected prompted us to question whether these lower values were a consequence of limited root development and reduced photosynthetic capacity of these early-stage communities. Data from longer-running restoration sites showed the same trend, indicating this is more likely driven by differences in adaptation between native and non-native species, especially in leaf area, which is known to influence EVI2 values (Mourad et al. 2020).

The inverse relationship between "greenness" and "intactness" was also evident at the restored Whitford site, which had a more even mix of native and non-native plants. As expected, its POS values were consistently intermediate; higher than those at more intact restored sites dominated by natives, but lower than would be expected in sites dominated by non-natives. This is not consistent with studies from other grassland ecosystems reporting increased productivity post-restoration using satellite-derived phenology metrics. For example, Liu et al. (2025) reported higher peak-of-season values in restored sub-tropical grasslands of the Nanling Mountains. By contrast our results suggest that temperate native grasslands may follow a different pattern.

At sites restored through TSR to address soil eutrophication and exotic species, a "productivity limit" may be imposed by the intervention. Native species adapted to low-nutrient soils and variable climates tend to prioritise resilience over rapid growth, relying on slow growth rates and efficient nutrient use in line with Grime's CSR theory (Grime 1977) and resource-use efficiency concepts (Reich 2014). In contrast, invasive exotic species thrive in eutrophic conditions due to rapid growth, reflecting competitive plant strategies. However, this altered phenology may come at a cost if exotics use up available limiting resources (especially soil moisture) by mid-summer, leading to a potential reduction in gross primary productivity (GPP) (Wilsey et al. 2014). While exotics tend to favour fast growth, natives are more 'stress-tolerant', focusing on long-term stability at the cost of higher productivity (Cornelissen et al. 2003). This may explain why our results appear to show lower productivity in restored grasslands.

4.3 | Synthesis of Findings

Together our two key findings challenge the narrative that native Australian grasslands, with their rich assemblage of grasses, forbs and other shrub and herbaceous species, are intrinsically better adapted to extract the most productivity from what the abiotic environment provides (Tilman et al. 1996). If native systems are indeed less productive, this is a significant distinction, because it suggests that the value of native grasslands is not necessarily their exceptional productivity (with possible implications for their ecosystem services such as carbon sequestration (Bardgett et al. 2021)), but rather that the meaningful distinction between native and non-native grasslands may be more subtle, centred on their role in maintaining finely tuned ecological relationships. For example, native grasslands may instead be defined by detailed phenological patterns of native species, such as the specific timing of growth, flowering and seed set (Groves et al. 2002), which align closely to the life cycles of native insects and other wildlife, enabling native systems to support a complex web of ecological interactions. This may contrast with introduced pasture crops, originally intended to provide forage for livestock precisely due to their exceptional productivity (Wilsey et al. 2014). These systems tend to simplify ecosystems by being too competitive and thereby disrupt subtle and intricate relationships established over evolutionary time scales. Their limited species diversity and potentially asynchronous phenological patterns can reduce ecological complexity (McKinney 2006; Gossner et al. 2016).

4.4 | Implications for Restoration and Monitoring

The first implication of our findings is that Sentinel-2-derived seasonal phenology is a useful metric for long-term monitoring, consistent with findings from other studies (e.g., Wilsey et al. 2017; Wainwright et al. 2011; Cheng et al. 2020), that demonstrates the potential of increasingly accessible open satellite data and freely available RS software to provide valuable insights into the condition and dynamics of these fragmented grasslands, including at relatively small scales (e.g., the restored Parklands site is only 0.3 ha). Additionally, utilising satellite-based monitoring technology, especially those with high temporal resolution and spatial accuracy like Sentinel-2, could allow for both retrospective and real-time analyses of restored grassland dynamics. This capability could support monitoring and interpretation of restored site trajectories over time and comparisons with successful past projects.

Beyond highlighting the potential of a free (for nonprofit organisations) or low cost (for commercial organisations) monitoring tool, our research also provides valuable insights for guiding grassland restoration efforts by informing adaptive management strategies. This approach empowers practitioners to prioritise conservation efforts, optimise restoration outcomes, and contribute to the long-term sustainability of urban grassland ecosystems (Bardgett et al. 2021). The insights gleaned from analysing grassland phenology at a site-specific or community scale hold significant promise for informing restoration management practices. Understanding the active growth and senescence periods of non-native plants can enhance the efficacy of management strategies, such as targeted herbicide applications. Another potential application of this method is in aiding site selection prior to large scale restoration projects. Presumably, out of a selection of potential restoration sites, those with a lower peak EVI2 value will have a lower/less vigorous weed load and therefore selecting the site with the lowest peak EVI2 may be the best candidate for restoration. The sensitivity of Sentinel-2 in detecting management induced changes at small spatial scales makes it a suitable tool for monitoring these urban grassland remnants. One manager can simultaneously track the phenological patterns across multiple sites and respond early to indicators of ecological shifts.

4.5 | Future Directions

We believe that the method we used for this case study is within the reach of restoration practitioners, particularly with the

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recent development of artificial intelligence (AI) tools which are able to convert natural language prompts into code (Becker et al. 2023) and explain complex scientific concepts intuitively (Krenn et al. 2022), significantly lowering the skill required to edit GEE and R scripts. Indeed, this analysis could be developed into a web app, removing the need for coding and improving accessibility for conservation practitioners further. Future research should evaluate the usability of such RS tools and the potential for AI to assist in conservation monitoring.

Practitioners could also integrate site-specific knowledge of native grassland phenology into the design of urban grassland systems. Our research is especially well suited to the study of urban and peri-urban grasslands, which are highly fragmented and patchy yet remain critical refuges for biodiversity and touchpoints for First Nations people to practise their culture. Native temperate grasslands have largely always been culturally managed landscapes (Gammage 2011; Gott et al. 2015). This is particularly poignant in public urban grasslands compared to private peri-urban and agricultural grasslands, where access and management opportunities for traditional owners may be more restricted.

Adelaide was established on culturally managed grassland landscapes (Hyde 1995; Kraehenbuehl 1996; Gammage 2011; Gott et al. 2015; Sandlos and Keeling 2016; Bardsley et al. 2019) that have evolved under the care of the Kaurna People for millennia (Bardsley et al. 2019; Gammage 2011). These systems have been largely erased and degraded in urban and agricultural areas (Williams et al. 2015; Morgan et al. 2017) making their care and monitoring critical for preserving both biological and cultural heritage. It was in an urban grassland remnant in Adelaide that the first cultural burn in an Australian capital city in almost two centuries took place (City of Adelaide 2021). Tracking phenological responses to these cultural management practices may support a shift in planning from a human-centred to a country-centred approach (Broffman 2022; Department of Planning and Environment 2023).

Urban governments, restoration NGOs and landscape architects involved in ecological design are increasingly conserving and reconstructing native grasslands, but the range of monitoring approaches available and the capacity for long-term monitoring are limited. Remote sensing (RS) approaches provide accessible and powerful techniques for assessing the impacts of urbanisation and management interventions on grasslands. Although biophilic design principles are increasingly applied to urban grassland conservation efforts (Marshall 2013; Marshall and Williams 2019; Walsh and Allan 2023), remote sensing evaluations of these projects are rare. This reflects a broader issue in urban ecology: integrating robust scientific methods into fast-paced urban development while monitoring outcomes in complex, ever-changing environments.

The interdisciplinary nature of this work highlights the need for continued collaboration between spatial scientists, ecologists and landscape architects in the design, implementation, management and monitoring of urban grassland systems (Opdam et al. 2001; Felson and Pickett 2005; Grose 2014; Musacchio 2018) By integrating RS data with ecological and cultural knowledge through landscape design principles, we can create resilient and functional urban green spaces that support biodiversity and enhance human well-being (McQuillan and Ryba 2023). Future research can focus on developing integrated approaches to inform the planning, implementation and evaluation of urban grassland restoration projects.

4.6 | Study Limitations

At the Glenthorne site, visually similar EVI2 trajectories masked very different on-ground conditions. The 2022 grassland, dominated by perennial wallaby grass (*Rytidosperma* spp.), had over a year to establish deep roots, with relatively low EVI2 likely reflecting life-history traits that limit leaf area under high solar radiation and seasonal aridity. By contrast, the 2023 grassland was still establishing, with smaller plants, bare ground, and broadleaf exotics. This illustrates a key limitation of the technique: similar EVI2 curves can arise from different ecological processes, underscoring the need to pair remote sensing with field observations. Future work can overcome this limitation by integrating quantitative ground truthing such as point intercept vegetation surveys which can be linked with RS phenological analyses (Wilsey et al. 2017).

In addition, variation in drainage, site age and local microclimatic conditions (e.g., shading, exposure, or soil moisture retention) may have also influenced the observed phenological patterns across sites, independent of the restoration treatments applied. These factors introduce further complexity to interpretation and underscore the importance of site-level contextual knowledge.

Another limitation of our study is the small number of sites, which prevented us from examining climate influences in depth. Future work could use a similar design across multiple sites along a longer aridity gradient to better assess how climate factors interact with restoration treatments and site conditions to shape grassland phenology.

5 | Conclusion

The ability to distil complex community-wide signals into meaningful phenological metrics is a powerful tool for ecosystem restoration. In this paper we have demonstrated that specific metrics—such as peak of season—can be tracked across time and space with enough sensitivity to distinguish between restored and unrestored sites.

Importantly, our analysis reveals that Australian native temperate grasslands exhibit lower 'greenness' compared to exotic communities, highlighting a key ecological distinction that is meaningful for management. Moreover, as Google Earth Engine is freely available for non-commercial use, it offers significant value to restoration ecologists and practitioners. It can help them better prioritise conservation investments and optimise restoration outcomes.

By utilising RS tools such as Google Earth Engine, we can advance understanding of urban grassland ecology and contribute to the development of effective strategies for restoring

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and conserving these threatened and culturally significant ecosystems.

Author Contributions

Conceptualization: M.M., S.W.R., methodology: M.M., S.W.R.; writing: M.M., C.L.; data collection and analysis: M.M., S.W.R.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study, including Sentinel-2 EVI2 composites, extracted phenological metrics and associated R analysis scripts, are openly available via Zenodo at https://doi.org/10.5281/zenodo.15273276.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Appendix S1:** Summary of modelled phenological metrics and summary statistics for peak of season and length of season across management areas.

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Chapter 6

THE EVOLVING CIRCLE OF LANDSCAPE DESIGN

Maximilian McQuillan*1, Ren Ryba2

6.1 Context statement

One of my most formative PhD conversations happened in a crowded café with my good friend Ren. The air was filled with the rich aroma of freshly ground coffee, and the ambient chatter created a lively hubbub that blended with the clinking of cups and saucers. As I tried and failed to articulate the link I felt between restoration ecology and landscape architecture, I sensed that both fields were embedded in ideas about land; however, I couldn't name the connecting thread. During our conversation, though, the missing link came into focus: values - what we choose to value in a landscape and why. Ren introduced the "moral circle," popularised by Peter Singer (1981), describing how societies widen (or reshape) which beings they consider deserving of care. This concept clarified what I saw in our fields. As biodiversity, climate resilience, and cultural legitimacy move to the centre of public concern, restoration and landscape design also converge. This chapter explores that convergence and proposes a simple lens: the evolving circle of landscape design.

Which futures should we choose for shared landscapes? This is the ethical question at the core of design and restoration, as they inherently involve value judgments; they are practices for choosing which values to express in the world. These choices evolve as society evolves, so practices also change accordingly. In the terms used throughout this thesis, institutions, professions, communities, and cultures (the 'actors and governance' side) shape ecological outcomes as much as the 'resource units' and 'resource systems' (McGinnis and Ostrom, 2014). Thus, restoration is not about reproducing a static past but about making clear, well-reasoned choices for diverse and functional futures. This work first appeared as a

short opinion piece in Restoration Ecology, precisely to initiate an idea-driven, invitation-based discussion between scientists and designers about their actual work in living places.

By "design" here, I mean integrating particular values into a functional whole. While this is clearly a human act in landscape architecture, it is not exclusive to it; keystone species and ecosystem engineers also "design" by shaping flows, habitats, and opportunities. Similarly, restoration ecology fits this definition. It isn't just technical weed control and seed addition; it's a social act involving normative choices - about which species matter, whose knowledge counts, which futures are legitimate, and who benefits. Because these choices happen in public space, designers and scientists have a responsibility: to name the values they express and notice when those values shift.

The "moral circle" is a useful (if contested) metaphor for seeing value shifts. This concept refers to the boundary we draw around those we consider deserving of moral consideration. Critics often dispute the scope of this circle and the criteria for inclusion, whether based on species, sentience, cultural values, or ecological contributions. Different cultures draw the circle differently; throughout history, the circle has expanded or contracted; centering humans is not the only approach (Singer, 1981). With this in mind, the key insight remains: as the circle moves, so too must our tools and standards for shaping land. For instance, if the circle now includes biodiversity as a public good and Country as a living presence, then the tools of design and restoration - guidelines, aesthetics, procurement, and monitoring - should reflect that.

You can see this evolving circle through time. Before "landscape architecture" was a profession, people made places expressing values: Neanderthal rings in Bruniquel Cave likely signified meaning; shifting from East to West Mound at Çatalhöyük suggests changing social ideas; Angkor married imperial planning with farmer-made water landscapes, mixing power and practicality; Persian gardens staged cosmology and luxury in ordered forms. More darkly, the Nazi rally grounds at Nuremberg show landscapes' use for violent ideologies, then

later as sites of warning and memory. Each example offers a lesson for today's practices: Neanderthal rings remind us that every landscape intervention carries cultural significance, like how modern designs should resonate with contemporary values. The shift at Çatalhöyük highlights the need for adaptable designs as social contexts change. Angkor's integration of power and practicality underscores the importance of balancing ecological function with user needs. Meanwhile, Persian gardens teach us about embedding narratives in design, similar to how green spaces today can tell stories of sustainability. Lastly, the Nazi rally grounds serve as a cautionary tale about the potential misuse of landscapes, reminding us that ethical considerations should steer our designs. The circle evolves, but not always for the good; it must be steered.

Today, steering is visible in institutions and projects. Landscape architecture has long integrated beauty, history, and function; over the last half-century, it has also adopted ecological thinking, universal design, and now explicit commitments to biodiversity and climate action (McHarg, 1969). Similarly, restoration ecology has expanded from site restoration to encompass governance, equity, and cultural practices. Yet tensions remain and are telling. For instance, the "new perennial" movement composes living tapestries that echo prairies and grasslands, but is direct about aesthetic authorship and maintenance, whereas restoration aims for self-organisation, ecological function, and species integrity. When aesthetics and ecological autonomy clash, which values should prevail? This pivotal question frames the debate between new perennial aesthetics and restoration goals, inviting a reflective weighing of the trade-offs inherent in interdisciplinary work. While both value change and seasonality, they weigh aesthetics, agency, and care differently. Beyond these, others push the circle outward: centering Indigenous knowledge, making wildlife crossings for animals as road users, or employing nature-based solutions at the city scale for water and heat. The pattern, then, is a growing sense of what, and who, counts.

The Glenthorne Community Grassland brought these frictions to life. Our aims were to restore biodiversity, engage with Kaurna people, create a welcoming public space, and

challenge expectations by making the area appear as native grassland, rather than a paddock. We met some of our aims, but missed others. Structural factors matter: restoration in Australia often defaults to a pre-colonial steady state, even as climate change and nutrient legacies make exact reconstruction unlikely. "Novel ecosystems" are now the norm in urban areas (Hobbs et al. 2009); genuine partnerships with Traditional Owners require time, resources, and trust, which are scarce in procurement cycles. The lesson was not to give up on multi-aim projects, but to state values and trade-offs explicitly, declare what the project prioritises, and keep widening the circle of decision-makers.

Looking ahead, the circle will continue to evolve, and practice must adapt. As we navigate this future, it is crucial to recognise areas where our understanding remains speculative, thereby inviting practitioners to engage in co-authoring potential solutions rather than viewing these scenarios as definitive predictions. Wilderness and rewilding are being re-examined in light of colonial histories and the demands of urban futures, presenting both uncertainty and opportunity. Calls to recognise animals, rivers, and other beings as stakeholders are increasingly driving conservation and design toward more-than-human ethics. New scientific insights, such as microbiome-inspired infrastructure, connect biodiversity design with human health, reshaping choices about plant selection and maintenance. Digital spaces (games, simulations, virtual reality) are teaching new generations to interpret and manipulate landscapes, embedding narratives about ownership, extraction, and belonging. The prospect of artificial agents gaining sentience or representing non-human interests in planning underscores the need for our professional circles to remain open and responsive. Embracing these uncertainties with intellectual humility allows us to place our rigorous and aesthetic practices within a broader moral context, extending our moral consideration while maintaining hope in our ability to adapt and create inclusive solutions.

Three practical consequences follow for designers, ecologists, and managers. First, list the values expressed in every project, named or not, and make time to identify those values

during scoping and engagement. Second, prototype for adaptation by designing with the understanding that values will change; combine durable infrastructure with adaptable stewardship, and track both cultural and ecological priorities. Third, co-draft authorship by involving Traditional Owners, community groups, and non-human perspectives early so they can influence project goals, not just implementation. In this thesis, the remote-sensing tools developed in Chapter 5 clarify these choices over time; they do not define what is good, but reveal the impacts of our decisions.

Ultimately, the evolving circle of landscape design invites honesty and hope. Honest that restoration and design are choices, not neutral acts; hopeful that reflection can align ecological integrity with cultural meaning and social acceptance. If we stay aware of how the circle shifts (and let it guide teaching, briefs, contracts, design, and care) we improve our chances of meeting challenges and leaving landscapes for future circles to inherit with gratitude.

The specific research questions addressed in this chapter are:

- *Q5.1 In what ways do values and priorities in landscape design — such as biodiversity or climate resilience — shift over time, and why are these shifts significant?*
- *Q5.2 How can an "evolving circle of landscape design" empower practitioners to integrate ecological rigour with creative design, ensuring urban grassland restorations remain adaptable to future challenges?*

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Signature	Date: 30/11/2025

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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RE-NEW OPINION ARTICLE

The evolving circle of landscape design

Maximilian McQuillan[1,2], Ren Ryba[3]

In landscape design and ecological restoration, humans shape the landscape. In doing so, practitioners express values that are not always stated explicitly. Here, we discuss how the values expressed by landscape designers and ecological scientists have constantly evolved through history and remain contested and re-negotiated today—a phenomenon we term the "evolving circle of landscape design." Values inherent in present-day landscape design and restoration (e.g. biodiversity, climate resilience) may be one transient point among many (e.g. power relations of societies in the past, digital landscapes of societies in the future) on this evolving circle. We believe that a critical, reflective understanding of the values expressed during landscape design and restoration will enable practitioners to choose more freely which values are expressed in landscapes, and that doing so will empower our disciplines to help society meet its present and future challenges.

Key words: culture, environmental management, ethics, landscape restoration, moral circle, normative, values

Implications for Practice

- Throughout history, people have designed and interacted with landscapes in ways that express specific values.
- The values expressed by landscape designers and restoration ecological scientists have constantly changed and will continue to change, and practice will change accordingly. We refer to this phenomenon as the "evolving circle of landscape design."
- We believe that if practitioners remain mindful and reflective about the values that underpin landscape design and restoration, our disciplines will be in a better position to help society meet its present and future challenges.

Introduction

In landscape design and restoration ecology, humans engage with and shape landscapes. There are numerous definitions of design, with no universal consensus. For example, Papanek (1985) defines design as "the conscious and intuitive effort to impose meaningful order" and Dieter Rams, emphasizing principles rather than a concrete definition, proposes 10 commandments of good design (Klemp & Ueki-Polet 2015). Design goes beyond humans—keystone species and ecosystem engineers may also be considered designers (Rosemond & Anderson 2003). In this article, we focus on the aspect of design that involves taking particular values and integrating them into a functional whole (Simon 1996). The practice of restoration ecology can be viewed along similar lines—rather than being purely technical, restoration ecology expresses many human values that are not always stated explicitly. In this sense, landscape designers and ecological scientists always choose which values to express in their practice. The ability to express values in the public realm means that designers and ecological scientists hold an important responsibility.

The key question that we explore is: "how have the values expressed by landscape designers and ecological scientists evolved through history"? We argue that these values have constantly evolved through history and remain contested and re-negotiated today—a phenomenon we term the "evolving circle of landscape design." If we take a critical view of how practitioners express values in the landscape, then we can more freely choose which values are expressed.

The Evolving Circle

If landscape design embodies human values, then how do human values change over time? To provide background context, there are ongoing debates about how human institutions—including politics, law, and so on—should include non-human entities. Such entities under debate include animals (Donaldson & Kymlicka 2011; Andrews et al. 2018; Vink 2020), ecosystem and landscape elements like rivers, and digitally sentient beings (Akova 2023).

One tool that can help us analyze human values is the "moral circle." The moral circle is a construct used by philosophers to distinguish the beings we care about—who are included in a person's moral circle—from those we do not. The moral circle was described by philosophers as early as the second century (Anthis & Paez 2021), though it has been made popular in modern times by Singer (1981).

Author contributions: MM, RR conceptualized and wrote the manuscript; MM was responsible for visualization; RR supervised the study.

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To be clear, the moral circle is a contested concept. First, it is hotly debated whether moral circles are expanding over time to include more beings, or contracting to include fewer (Freeling & Connell 2021). Some entities have found themselves freshly excluded from many cultures' moral circles: prisoners, fetuses, buildings, ancestors, deities, and many groups of farmed and wild animals (Branwen 2019). Second, moral circles are not uniform—different people and cultures have different ideas about which entities warrant moral consideration (Branwen 2019). Third, placing humans at the center of a concentric circle is inherently anthropocentric. Alternative perspectives that de-center humans, such as ecocentrism and even infocentrism, may also be valid (Torrance 2013). In any case, it is clear that the way cultures see the world changes over time.

If a culture's moral circle is evolving, then the way that culture interacts with its environment may need to adapt (Johansson-Stenman 2018; Freeling & Connell 2021). Many tools that are currently used in the West to analyze and engage with the world, such as goals and principles of restoration ecology and landscape design, were developed in the past. But if a culture changes how it views the world and its inhabitants, then tools developed yesterday may need to evolve to remain relevant tomorrow.

Landscapes in the Past

Let us begin by exploring the origins of how people engage with landscapes. As a recognized profession with its own body of knowledge, landscape architecture is young. Its origin is associated with the design of New York City's Central Park by Frederick Law Olmsted and Calvert Vaux in 1858. However, humans have been intentionally designing and shaping outdoor spaces for as long as our genus, *Homo*, has existed.

One ancient piece of landscape design is Bruniquel Cave in France (Fig. 1A). This cave contains elaborate ring structures that were constructed from deliberately broken stalagmites roughly 175,000 years ago, probably by Neanderthals (Jaubert et al. 2016). We do not yet know what values were expressed by these ring structures, but archeologists speculate that the structures may have served a ritual or symbolic purpose (Jaubert et al. 2016).

We can understand more about a newer example of urban landscape design—Çatalhöyük, an ancient city that existed around 7000 BCE in today's Turkey (Fig. 1B). The residents of Çatalhöyük may have only been a couple of generations removed from entirely nomadic lifestyles. Notable is the transition between two types of urban construction. In the East Mound, the older part of Çatalhöyük, domestic buildings tended to be similar in shape and size; in the newer West Mound, domestic buildings became much more varied (Orton et al. 2018; Newitz 2021). Perhaps this reflected a tension between views about how society should be organized—social cohesion and a flat hierarchical structure on one hand, and independence and individuality on the other (Newitz 2021; Hodder 2022). Of course, this interpretation is



Figure 1. Six examples of landscapes from human history, each a physical expression of a different set of human values. (A) Ring structures constructed by Neanderthals 177,000 years ago from broken stalagmites for an unknown, perhaps ritualistic purpose, Bruniquel Cave, France; (B) an artist's model of Çatalhöyük, a proto-city that flourished in 7000 BCE in modern-day Turkey; (C) Eram Garden, a Persian garden constructed in the 11th to 14th centuries CE in modern-day Iran; (D) Nazi party rally grounds in Nuremberg, 1938; (E) grassland restoration project in Glenthorne National Park-Ityamaiitpinna Yarta, 2023, Kaurna land (Australia); (F) a digital "landscape" in an artistic installation by WE ARE VIDEO in Germany. *Image credit*: (A) Luc-Henri Fage/SSAC. (B) Wolfgang Sauber. (C) Siamak.ess. (D) Public domain. (E) Render by Linfeng Huang, used with permission. (F) Sarah Kurig on Unsplash, used under the Unsplash license. (A–C) Published under the CC BY-SA 4.0 license and used under fair dealing.

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disputed (Orton et al. 2018), but it nevertheless suggests how people may have expressed dynamic cultural values in the landscape in this place and time.

Tension between value systems is also exhibited by the landscapes of greater Angkor. The capital of the Khmer Empire from around 800 to 1300 CE, Angkor consisted of magnificent temples, centrally planned landscapes, and urban complexes for the ruling elite. But Angkor also contained widespread decentralized design and landscapes that were constructed by farmers according to local traditions (Hawken 2013). The different patterns, orientations, and scales of the different designs are an expression of differences in power. Luxury and power also find expression in the gardens of Persia (Fig. 1C), which are frequently cited as one important, ancient tradition in the history of landscape architecture (Turner 2004).

Values embedded in landscapes can also be deeply harmful. In Nazi Germany, landscape design played an important role in the Third Reich (Fig. 1D). The regime's architects and landscape designers aimed to create a new aesthetic for Germany that reflected their nationalist and racist ideology (Hagen & Ostergren 2006). The Nuremberg Rally Grounds are one example. Crucially, Nuremberg has more recently become imbued with a new set of values through its intentional preservation as a memorial (Hagen & Ostergren 2006). Thus, the circle of landscape design certainly evolves, but we cannot claim that the circle is destined to evolve in a "progressive" or positive way.

Landscapes in the Present

How do landscape designers and ecological scientists express values in the present day?

Landscape designers and architects are trained in both environmental and social sciences, making the field of landscape design a place where diverse requirements are frequently weaved into a functioning whole (Dramstad et al. 1996). Over the last decade, these diverse requirements have been refined. Practice has traditionally been guided by beauty and aesthetics, historic preservation, and functionality. But the seminal publication *Design with Nature* (McHarg 1969), influenced by environmentalism, placed the values of environmental sustainability in landscape architecture practice. Social movements have also left their mark on practice, with a prime example being the emergence of "universal design" from the disability rights movement.

During the United Nations Decade on Ecosystem Restoration, there are indications that biodiversity conservation and restoration are newly included in the evolving circle of landscape design. Many landscape institutes consider biodiversity—the U.K.'s Environment Institute provides explicit resources, while Germany's *Bund Deutscher Landschaftsarchitect:innen* references the EU Biodiversity Strategy, and Australia's Institute of Landscape Architecture has recently published a Biodiversity Positive Design Position Statement. Likewise, every landscape institute in the United States, the United Kingdom, Australia, Germany, and Singapore has resources dedicated to addressing climate change, reflecting the emerging perception of climate change as an emergency (United Nations Development Programme 2021).

It is worth noting a tension here—landscape architects and restoration scientists both value biodiversity conservation and restoration, but they vary in how such values are perceived. Consider the European "new perennialist" movement, led by Piet Oudolf. This movement's landscapes share visual characteristics with grassland and prairie restoration. However, the perennialist movement explicitly seeks to abstract and enhance the aesthetic character, seasonality, and dynamism of perennial ecological communities; this is achieved through methodically planned and high-maintenance plant installations. In contrast, restoration ecology emphasizes species diversity, ecosystem functionality, as well as the self-organization and ecological stability of ecological communities. Exemplary researchers and practitioners who deal with the themes of this article in their work include: Julia Watson, a leading voice in communicating Indigenous values in landscape design through a collection of projects (Watson & Davis 2019); Anthony P. Clevenger, an early pioneer in the development of wildlife crossings to improve animal welfare outcomes around highways and is best known for his involvement with the animal crossing overpass in Banff National Park, Alberta (Clevenger & Huijser 2011); and Beijing-based Kongjian Yu, the leader of landscape architecture practice Turenscape, which explores innovative ways to mitigate environmental issues through nature-based solutions and green infrastructure.

The challenge in navigating complex value systems is highlighted by a current landscape restoration project led by the first author of this paper (MM). This project, the Glenthorne Community Grassland, is the reconstruction of a native temperate grassland in an urban national park in Kaurna Country/Adelaide, South Australia (Fig. 1E). The project has four main aims: to restore biodiversity; to engage the Indigenous community; to facilitate interactions between the public and the grassland; and to challenge mainstream aesthetic expectations by developing the park's aesthetic to closely resemble native grasslands.

The project has fallen short of achieving all four aims. In Australian restoration ecology, the implicit goal is almost always to return to the (imagined) static, Indigenous landscape that existed before European colonization (Trigger et al. 2010). This is justified by the desire to safeguard biodiversity. Given that climate change is causing many species' distributions to change, it is debated whether recreating past ecosystems is physically possible, let alone desirable (Harris et al. 2006; Hobbs et al. 2009). A related topic is the emergence of "novel ecosystems," where biotic, abiotic, and social drivers can cause entirely new configurations of plant and animal species to arise (Hobbs et al. 2009). Compounding this, is the challenge that can easily arise when grants are sought by white practitioners—an existing, trusting relationship with Indigenous communities is necessary to apply for grant money together with Indigenous communities, but existing grant money is necessary for building such a relationship. Where established practices and corporations can pay for Indigenous people's time and expertise, the power imbalance created by those situations can still be problematic.

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Thus, contemporary projects—including Glenthorne—often become technical challenges, where decisions are made by people highly trained in Western schools of thought and little space is left for meaningful engagement with Indigenous people. In modern, Western ecology programs, students rarely learn how science relates to other academic and non-academic forms of knowledge, despite restoration projects often depending on non-academic forms of knowledge (Higgs 2005). Balancing multiple aims will always remain a challenge, but as landscape managers, we can begin with honesty about the value systems that underpin our practice (Allison 2007).

Landscapes in the Future

As for the future, we can speculate about what changes may be on the horizon. Practice in landscape management is undergoing a process of contestation, perhaps as it always has (Martin 2022). What is considered appropriate practice is being re-negotiated. One well-known example is the concept of wilderness which, as is argued in the classic essay by Cronon (1996), is a "cultural invention" that depends on an unacknowledged history of colonialism. This suggests that the concept of "rewilding" may itself evolve, as suggested above in the context of novel ecosystems. Furthermore, as societies become geographically and culturally focused in urban areas, it grows increasingly important to find ways for people to interact with plants and animals in cities (Miller 2005).

Another debate is which rights we should grant to ("recognize in"?) non-human organisms. Wild animals, whose lives are profoundly affected by ecological restoration and landscape management, could be viewed as stakeholders in their own right (Capozzelli et al. 2020). This is mirrored by the emerging concept of more-than-human design, which seeks to provide tangible and intangible outcomes that are valuable to human and nonhuman stakeholders (Roudavski & Rutten 2020). Similarly, it remains vigorously debated whether compassion is valid in conservation (Coghlan & Cardilini 2022) and whether the concept of "nativeness," used to justify killing some organisms for the benefit of others, has any basis (Chew & Hamilton 2011).

Scientific discoveries can also cause new values to become incorporated into landscape design. For example, researchers have recently begun to understand how human health is affected by the microbiomes in environments. This has led to the new concept of "microbiome-inspired green infrastructure," which is just one way that landscapes and biodiversity are deliberately managed to meet the value of improving human health (Watkins et al. 2020).

More speculatively, even the concept of "landscape" may be transformed (Fig. 1F). Digital landscapes, shaped from scratch by high-budget companies, are inhabited by console gamers and users of virtual reality (Portman et al. 2015). Digital landscapes also express values. For example, the immensely popular *Minecraft* involves extensive interactions with a digital landscape, and this game has been criticized for reproducing narratives of settler colonialism and gendered, Eurocentric systems of ownership and property (López López et al. 2019; Brazelton 2020). As simulated landscapes and virtual ecosystems become more sophisticated and accessible, they may gradually come to replace physical landscapes in people's lives and experiences. This point is salient given the possibility of artificial sentience, in which digitally emulated minds may become not only highly intelligent but fully self-aware (Anthis & Paez 2021). Extending ethical consideration to digital beings and to non-biological forms of sentience may involve the promotion of more-than-human cultures (Brock & Roudavski 2020). Such adaptation to a post-anthropocene era would involve making creativity more inclusive of non-human beings (Canina et al. 2021).



Figure 2. A conceptual illustration of the "evolving circle of landscape design." Over time, the values of society change, and these values are expressed when humans shape the landscape—from early interactions natural landscapes to early cities (e.g. Çatalhöyük, Persian gardens), to modern structures (e.g. Nazi party rally grounds), to beyond (e.g. digital landscapes). *Image credit*: Maximilian McQuillan.

As landscape managers, our principles and practices express specific values. The shaping and restoration of landscapes always involves a choice of which values to express (Allison 2007). Society's values are evolving, and we can reflect on how to adapt our tools and practices to express society's emerging vision for the world. If we do so, we may be in a better position to help society engage with the challenges of today and tomorrow.

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Chapter 7

GENERAL DISCUSSION

7.1 Introduction

In Greek mythology, Sisyphus, a cunning and resourceful king of Ephyra, was notorious for his trickery and hubris. His exploits in cheating death and defying the gods provoked the wrath of Zeus. As punishment for his deceit, Zeus condemned him to an eternal task. He was forced to push a massive boulder uphill on a barren, unforgiving hillside beneath a vast, indifferent sky (Graves, 2017). This ceaseless struggle parallels the challenges faced by urban ecologists as they strive to restore degraded ecosystems. Just as Sisyphus strains to reach the summit only to watch the boulder roll back down, ecologists exert tremendous effort to make progress in restoration. Yet they often face setbacks as ecosystems tend to revert to their degraded states. This constant push against ecological degradation is the 'boulder' that modern scientists and restoration professionals must contend with (Hilderbrand et al., 2005).

The myth highlights two key concepts: ecosystem resistance and resilience. Resistance refers to properties that inhibit change, while resilience is the capacity of a system to recover its original state after disturbance (Holling, 1973; Folke et al., 2004). Restoration ecologists and land managers capture these dynamics through state-and-transition models (Westoby et al., 1989; Sadler et al., 2010; Sinclair et al., 2019). These models reveal that ecosystems often exist in multiple stable states, and that transitions between states are driven by internal feedbacks and external pressures.

To guide you through this chapter, you'll discover three major insights derived from Objectives 1–5. Firstly, I examine how pre-seeding choices influence establishment trajectories. This topic is discussed in Chapters 2 through 4. Secondly, the monitoring of these trajectories at scale is detailed in Chapter 5. Lastly, Chapter 6 situates these findings within the framework of landscape-architectural discourse and values. This chapter answers the

overarching research question posed in Chapter 1. That question is: “How can science-driven methods for restoring biodiverse, self-sustaining urban grasslands be leveraged to develop practice-based solutions that address the global biodiversity crisis and strengthen ecological resilience within metropolitan landscapes?” Where helpful, quotes from the published chapters are provided. I also link the synthesis back to the research objectives and guiding questions in Chapter 1(Section 1.8).

Within urban grassland restoration, the myth of Sisyphus - and other myths such as the “Field of Dreams” - reminds us that success requires more than repetitive interventions (Hilderbrand et al., 2005). It demands an adaptive strategy that works in harmony with natural processes (Steele & Harrow, 2014; SERA, 2021). Restoration must intelligently shift feedbacks so the system stabilises into a more desirable, resilient state (Andersen et al., 2009).

A central theme throughout this thesis is creating conditions that allow a reconstructed grassland to self-organise into a biodiverse and resilient ecosystem. Restoration should not be a rigid, predetermined spatial exercise. Instead, it should cultivate conditions that encourage self-sustaining and self-organising plant communities. Imagine a vibrant urban meadow, once a barren plot, now bursting with native wildflowers, their polinators navigating between diverse native grasses. This transformation, achieved by strategic design choices such as selective pre-seeding interventions and carefully matched seed mixes, showcases the subtle but powerful design agency in ecological restoration. Designers intentionally step back, allowing natural processes to take the lead while guiding the meadow’s development through initial interventions.

Effective ecological restoration must also align with society’s values and priorities. As outlined in the General Introduction (Chapter 1, Part I), a socio-ecological systems perspective is essential for addressing the complex challenges of urban grassland degradation. This approach considers resource systems, governance structures, and the roles of stakeholders. This thesis advances our understanding of temperate grassland reconstruction by

empirically evaluating technical restoration interventions. It categorises interventions as (1) biomass removal, (2) seed bank manipulation, and (3) soil reset, organised into a hierarchy of increasing cost but also likely increasing effectiveness. The thesis proposes a predictive framework for assessing how these measures influence the emergence of stable, desirable ecosystems. From a state-and-transition model perspective, it demonstrates how manipulating soil conditions and seed mixes drives feedback between ecosystem establishment and resilience. This guides decision-making and adaptive management strategies. Integrating scientific intervention data with social and economic perspectives is consistent with established socio-ecological models (Ostrom, 2007; McGinnis & Ostrom, 2014; Bardgett et al., 2021). These models highlight the interplay between ecological processes and human systems. This work forges a critical link between restoration ecology and landscape architecture. It blends scientific rigour with innovative, context-sensitive design.

In exceptionally challenging restoration scenarios, a more drastic approach is required. This method, known as “ecological reconstruction,” is a more suitable approach than traditional restoration. It is akin to mine rehabilitation techniques (Cooke & Johnson, 2002). According to the National Standards for the Practice of Ecological Restoration in Australia (2021), ecological reconstruction is used on severely degraded sites where natural regeneration is inadequate. It involves addressing root causes of degradation, repairing abiotic damage, and reintroducing key species using evidence-based practices. Preparatory work should be tailored to the site's condition. This strategy is implemented when the potential for natural recovery is extremely low, even with the use of regeneration techniques.

7.2 Thesis summary and conclusions

Part I – Context and rationale

The General Introduction connects native temperate grasslands to the global biodiversity crisis. It also places urban grassland restoration within a socio-ecological framework. This section stresses the importance of integrating scientific methods with design thinking to

address urban biodiversity degradation. By doing so, it prepares the reader for the empirical investigations in Part II and the synthesis of design perspectives in Part III.

Part II – Empirical investigations

Chapter 2 – Sunlight can have a stronger influence than air temperature on soil solarisation: Observational evidence from Australia

This chapter examines the optimal conditions for soil solarisation, used as a pre-seeding step to establish stable plant communities at urban and post-agricultural sites. The data show sunlight intensity controlled topsoil temperature more than air temperature, suggesting restoration practitioners may schedule solarisation based mainly on sunlight forecasts, even when air temperatures are low. While these findings are generalisable to a degree, they are shaped by the specific context of this experiment - namely a temperate Australian latitude, clay-rich soils, and dry summer starting conditions that influenced heat transfer and seed-bank sensitivity. Recognising these site and climate constraints helps clarify the conditions under which similar outcomes might be expected elsewhere. Focusing on improving pre-seeding conditions, this study provides a framework for optimising solarisation as a pre-seeding soil treatment, supporting more reliable establishment of a target native ecosystem in degraded areas.

Chapter 3 – Topsoil removal increases native forb and grass recovery in urban temperate grasslands

This chapter reports a controlled, randomised field experiment in a post-agricultural urban landscape that directly addressed Objective 2 (Q2.1–Q2.3) by comparing three pre-sowing interventions for native grassland reconstruction. The intervention outcomes were as follows: chemical fallow (biomass removal) resulted in no measurable biodiversity gain; solarisation (seed-bank manipulation) increased native alpha diversity over 18 months, predominantly benefitting grasses rather than the entire community; topsoil removal (soil reset) substantially increased native alpha diversity, with marked gains for both forbs and grasses, and improved floristic amenity. The underlying mechanisms reflect the intervention gradient: topsoil

removal reduced both the nutrient profile and the exotic seed bank, while clearing biomass; solarisation mainly targeted the seed bank; and glyphosate mostly suppressed biomass. Consistent with state-and-transition theory, the strongest intervention (topsoil removal) most effectively shifted the system past thresholds for native establishment. These outcomes provide a clear policy signal: light-touch approaches seldom succeed in nutrient-enriched post-agricultural areas; therefore, interventions should target multiple barriers simultaneously. Alongside the solarisation findings from Chapter 2, these results empirically support the hierarchical pre-seeding logic and STM transitions noted earlier, with topsoil removal firmly established as the most reliable pathway for achieving diverse and resilient urban grasslands at this site. However, the strength of these outcomes is shaped by the site's specific conditions - particularly its nutrient-enriched post-agricultural soils, exotic-dominated seed bank, and temperate climate - which means similar responses elsewhere will depend on local soil properties, seed-bank structure, and climatic context.

Chapter 4 – Striking a balance between ecology and economics: a more holistic approach to optimising topsoil removal in grassland restoration

This chapter builds on the results of Chapter 3 and establishes a framework to balance the biodiversity benefits of topsoil removal with its associated costs and risks. It combines a cost model with a rapid soil-conservation assessment tool to guide the depth and extent of soil removal. The framework assesses key risks - such as erosion, sodicity, and scalding - and evaluates cost-effectiveness. This helps practitioners know when and how to use topsoil removal. The approach joins cost analysis with environmental safeguards. It emphasises the importance of tailoring interventions to the specific soil type, seed bank, and costs at each site. The framework can be applied spatially to guide vegetation restoration plans at larger scales.

Objective 3 (Q3.1–Q3.3) highlights that long-term adoption depends on socio-economic factors: Good ecology on its own is not enough; long-term sustainability also depends on recognising economic and financial realities and ensuring that limited resources

achieve the greatest impact. The framework translates biophysical thresholds into practical guidance for managers, advising them to use rapid soil-conservation screens to identify where topsoil removal can be applied safely and cost-effectively. Shallow topsoil removal (2–12 cm) is often the most cost-effective approach because most weed seeds are concentrated in the upper soil layers and excavation costs rise with depth. However, leaving deeper layers of the seed bank intact can create long-term management challenges, as exotic species may continue to emerge and require ongoing control. The cost–depth–area model helps balance biodiversity gains with budgets and suggests alternatives - like chemical fallow, mulching, or tubestock planting - where soil degradation risks are high. Outcomes will also vary across sites depending on factors such as soil type, existing seed-bank composition, moisture regimes, and the degree of prior disturbance, all of which can influence both the ecological benefits and the financial viability of shallow topsoil removal compared with alternative treatments.

Chapter 4 connects ecological thresholds to procurement and early design decisions. This links ecological analysis with landscape-architectural work. The chapter extends the results of Chapter 3 to create a framework for managing ecological and financial risk. It also ensures that restoration goals fit with larger project plans (see also Chapter 6).

Chapter 5 – Pixel patchwork: monitoring temperate grassland restoration success with Sentinel-2 phenology metrics in urban Australia

This chapter introduces remote sensing as an approach to monitoring grassland restoration. It uses Sentinel-2 EVI2 time-series data for this purpose. By tracking seasonal metrics such as the length and peak of season (POS), the study shows restored grasslands “behave” like native remnants. POS differed clearly between native and non-native sites, demonstrating that remote sensing can distinguish between different types of vegetation. This method is scalable and cost-effective, which is critical in areas where urban grasslands are small, fragmented, and under development pressure.

Addressing Objective 4 (Q4.1–Q4.3), the chapter notes that simple phenological metrics give restoration managers powerful information. Metrics such as POS can reliably

distinguish restored from unrestored sites. This helps managers adapt their methods. The analysis also shows "Australian native temperate grasslands have lower greenness than exotic communities," a key management point. Because the Google Earth Engine workflow is free for non-commercial use, councils, non-profit practitioners, and researchers can access it.

Chapter 5 supports the plant establishment results from Chapters 3, and also extends them. It introduces a scalable method for monitoring restoration and links short-term experiments to long-term resilience. This aligns with STM-informed adaptive management.

Part III – Synthesis and Design Perspectives

Chapter 6 – The evolving circle of landscape design

This final chapter moves from data-driven work to a wider review of landscape design thinking. It proposes a design philosophy that applies ecological principles to every stage of planning and constructing out landscapes. It returns to the idea of designing to support self-sustaining ecosystems. The chapter states that landscape design is always evolving: the values and priorities in a design now may differ from those of the past or future needs. This perspective enables practitioners to decide which values - biodiversity, climate resilience, and/or cultural significance - should guide each project. The "evolving circle of landscape design" is a practical and conceptual tool that links restoration ecology to the broader fields of landscape architecture and urban planning. To strengthen this adaptive design approach, collaboration with civic partners, such as local community groups and municipal units, is essential. Naming these collaborators who co-define these values can transform this philosophy into a shared agenda and ensure that the designs truly reflect community needs and aspirations.

The chapter explains, “Our management expresses certain values… Restoring landscapes always means choosing which values to show… Society’s values change, so we must adapt our tools to fit society’s vision.” Paired with the results of Chapters 2–5, this view makes the STM hierarchy both an ecological model and a design tool. It structures site work,

planning, and long-term care. The chapter meets Objective 5 by demonstrating how science and changing design values can contribute to creating lasting, resilient projects.

7.3 Integrating empirical findings and design innovation

Across the empirical investigations, a consistent narrative emerges: effective urban grassland restoration requires a hybrid approach that combines robust scientific methods with innovative design thinking. The experiments demonstrate that pre-seeding interventions fundamentally shape the early assembly conditions of reconstructed grasslands, and that different interventions operate along a gradient of ecological leverage. Solarisation acts as a targeted seed-bank tilt influenced primarily by solar radiation, while topsoil removal delivers a deeper shift by reducing nutrient legacies and removing entrenched exotic seed banks. These interventions establish the initial conditions upon which plant communities self-organise.

The decision-making framework developed in Chapter 4 extends these findings by showing how ecological thresholds, soil constraints, and project budgets can be aligned to guide when and how such interventions should be applied. It reframes pre-seeding choices as design–procurement decisions, embedding ecological reasoning into early project planning rather than treating it as an isolated technical step.

Monitoring closes the loop. Sentinel-2 phenology provides a scalable way to verify whether intervention choices are producing the intended establishment trajectories, enabling managers to detect early deviations and adjust accordingly. This positions remote sensing as a practical tool for adaptive management, linking short-term intervention outcomes to long-term ecosystem resilience.

Finally, Chapter 6 situates these insights within a broader design discourse. It reframes restoration as an iterative, value-driven practice that integrates ecological function with landscape meaning and stewardship. Under this lens, interventions are not just technical

solutions but part of a governance system in which ecological, cultural, and financial values shape long-term outcomes.

Together, these strands show that hierarchically organised pre-seeding interventions can generate predictable establishment trajectories, which can then be monitored, adjusted, and embedded within a design practice that makes ecological and cultural values explicit.

In parallel with the empirical work presented in this thesis, an applied management plan was developed for the Glenthorne Community Grassland (Appendix D). While not examined as part of this thesis, the plan translates the hierarchical pre-seeding framework (Chapters 2–4), phenological monitoring insights (Chapter 5), and design principles (Chapter 6) into an operational document guiding on-ground restoration, adaptive management, and long-term stewardship. Its inclusion demonstrates how the research was directly used to inform real-world grassland management across zoning, intervention timing, monitoring, and governance.

7.4 Emerging conclusions: A hierarchical model of pre-seeding interventions

Proposed Theoretical Framework for Emergent Urban Grassland Restoration

Drawing upon the empirical evidence and design perspectives established in the preceding chapters, this thesis culminates in a conceptual model categorising pre-seeding interventions for post-agricultural and urban landscapes into three broad types:

1. Biomass Removal (e.g., burning, short-term chemical fallow)
2. Seed Bank Manipulation (e.g., solarisation, pine oil application, soil microwaving, long-term chemical fallow)
3. Soil Reset (e.g., topsoil removal or soil inversion)

While the dataset remains insufficient for definitive conclusions across all contexts, the findings of this thesis provide consistent initial evidence that these interventions form a hierarchy, wherein ecological benefits and predictability of outcomes generally increase from

(1) biomass removal to (2) seed bank manipulation to (3) soil reset. Essentially, the aims of a less comprehensive intervention (e.g., biomass removal) can be subsumed by a more comprehensive approach (e.g., seed bank manipulation or soil reset), but not necessarily vice versa. Figure 1 (below) visually depicts this hierarchy, illustrating how each approach leads to predictable optimal ecological outcomes; from (4) relatively low-diversity, (5) high-native cover to (6) fully biodiverse vegetation assemblages. The empirical steps outlined in Chapters 2–4 populate the rungs of this hierarchy, while Chapter 5 provides a monitoring grammar to evaluate movement between states.

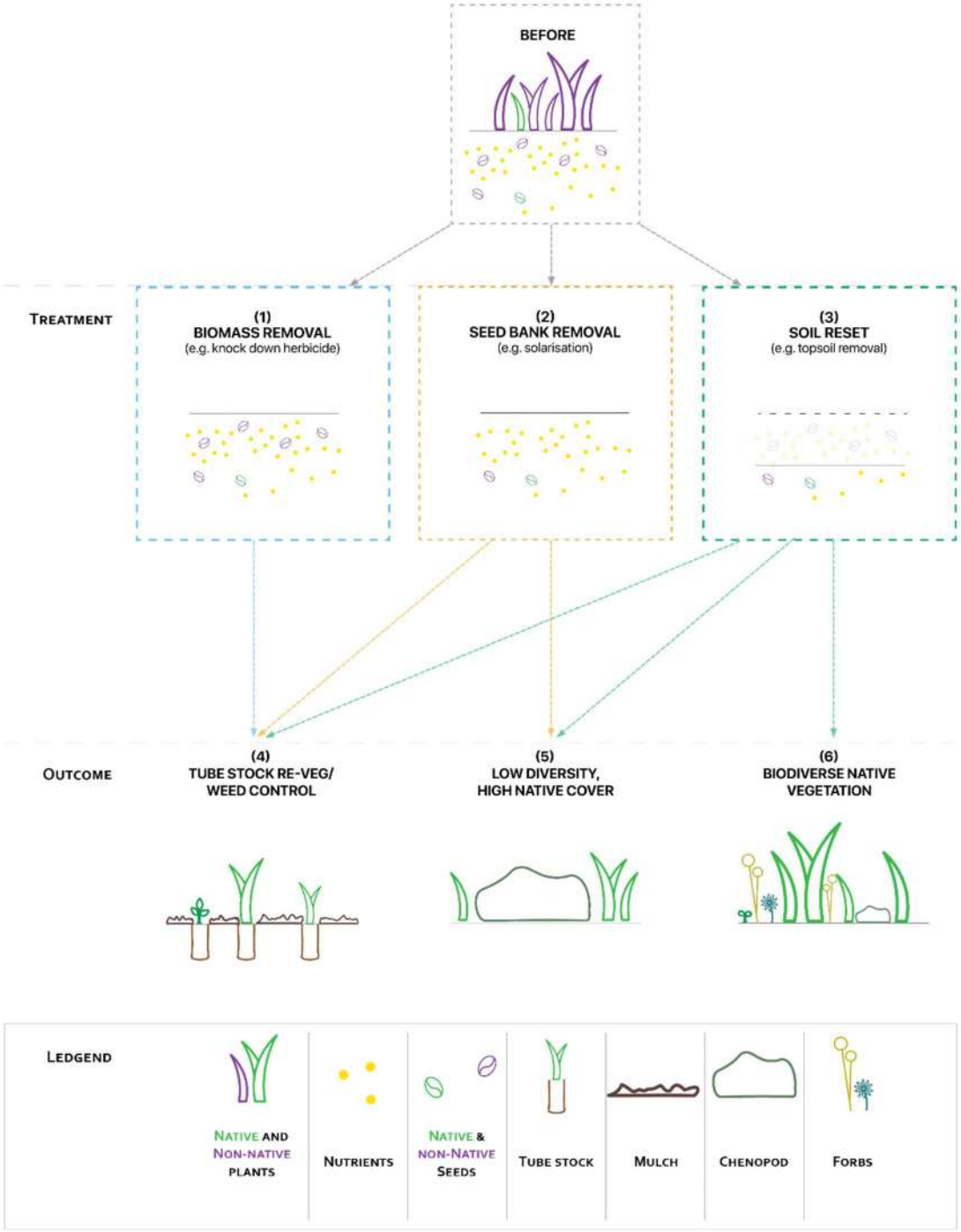

Figure 1: Conceptual framework linking pre-seeding interventions to restoration outcomes. The framework progresses from a degraded state dominated by exotic vegetation, elevated soil nutrients, and an exotic seed bank to three intervention pathways: (1) biomass removal, which reduces above-ground competition only; (2) soil seed-bank manipulation, which removes biomass and part of the exotic seed bank but leaves nutrient levels unchanged; and (3) soil reset, which reduces biomass, soil nutrients, and the exotic seed bank. These align with three outcomes: (4) mulching followed by tube-stock planting (low diversity and low native ground-storey cover); (5) direct seeding with fast-establishing competitive natives (e.g., perennial grasses and chenopods); and (6) high-diversity direct seeding of native grassland species (grasses and forbs), representing the highest restoration success.

7.4.1 State-and-Transition Models and Pre-Seeding Interventions

This model aligns with the state-and-transition framework - a widely used conceptual approach in rangeland and grassland restoration (Westoby et al., 1989; Hobbs & Norton, 1996). State-and-transition models (STMs) describe ecosystems as dynamic systems that can exist in multiple stable states, with transitions occurring due to disturbances, interventions, or environmental changes. Pre-seeding interventions can be understood as mechanisms that facilitate these transitions, shifting a degraded grassland towards a more biodiverse and resilient state (Yates & Hobbs, 1997; Sadler et al., 2010; Sinclair et al., 2019).

STMs acknowledge that certain degraded states are maintained by feedback loops, where the mere removal of a disturbance (e.g., cessation of grazing or mowing) is often insufficient to induce a return to a native-dominated grassland. Instead, restoration practitioners must actively disrupt existing feedback mechanisms to create conditions conducive to native species establishment. The hierarchical intervention model presented in this thesis mirrors this logic, wherein higher-intensity interventions (e.g., soil reset) have a greater capacity to push a degraded grassland beyond a threshold and into a new restoration trajectory. Conversely, lower-intensity interventions (e.g., biomass removal) may be insufficient to overcome persistent seed bank legacies or altered soil conditions, leading to partial or stalled transitions.

7.4.2 Restoration Trajectories and Thresholds

The alignment of pre-seeding interventions with STMs reinforces the importance of restoration thresholds - points at which a system requires a fundamental shift in structure or function to transition towards a desired state (Scheffer & Carpenter, 2003; Folke et al., 2004; Andersen et al., 2009; Rocha et al., 2015). While biomass removal may be effective at reducing immediate competition, it is unlikely to suffice if a site has a persistent exotic seed bank or highly altered soil chemistry. Seed bank manipulation provides a more targeted intervention, though its effectiveness may depend on site-specific factors such as residual high soil nutrient loads. In contrast, soil reset interventions are more likely to facilitate a transition

to a native-dominated state by directly addressing the underlying constraints to seedling recruitment and ecosystem reassembly (Yates & Hobbs, 1997; Sadler et al., 2010; Sinclair et al., 2019).

These findings highlight the need for context-dependent restoration strategies. In some cases, lower-intensity interventions may be appropriate where degradation is minimal or where native propagule pressure remains high. In more severely degraded contexts, stronger interventions (such as soil reset) may be the only viable option to shift the system over an ecological threshold (Andersen et al., 2009; Suding & Hobbs, 2009), allowing for the re-establishment of diverse native forb and grass assemblages.

7.4.3 Systems Thinking in Grassland Restoration

Integrating state-and-transition models with the hierarchical intervention framework aligns with a broader systems-thinking approach. This perspective recognises that ecological restoration does not occur in isolation but is embedded within a complex socio-ecological network of environmental, social, and governance factors (Ostrom, 2009; McGinnis & Ostrom, 2014; Bardgett et al., 2021). Urban grassland restoration must be approached as a dynamic, iterative process, where interventions interact with land-use history, urban planning policies, and stakeholder priorities (Bardgett et al., 2021).

By embedding pre-seeding interventions within a systems-thinking perspective, restoration practitioners can make more informed decisions that account for both ecological and social variables. For example, while soil reset may offer the most effective pathway to achieving a native-dominated state, its feasibility may be constrained by economic costs, land tenure agreements, or community concerns. In such cases, a phased or adaptive management approach, starting with seed bank manipulation and monitoring outcomes before escalating to more intensive interventions, may be a more practical and socially acceptable strategy.

Furthermore, the systems-thinking perspective reinforces the importance of monitoring and adaptive management. Interventions should not be applied in a rigid,

prescriptive manner (i.e., the myth of the cookbook which refers to the misconception that restoration techniques that work in one location or ecosystem will always work in others, ignoring the unique characteristics and complexities of each system (Hilderbrand et al., 2005)); instead, they must be continuously evaluated and refined based on ecological feedback. This adaptive approach is promoted by the Standards Reference Group SERA (2021) and recognises that grassland restoration is a long-term process influenced by uncertainty, external pressures, and emerging ecological insights, requiring ongoing adjustments to management strategies.

7.4.4 Testing the proposed predictive framework

To test the proposed hierarchical intervention model, a global meta-analysis could be conducted, synthesising data from peer-reviewed literature on direct seeding in grasslands. This analysis would focus on categorising the methods used in these studies into the three proposed intervention types: biomass removal, seed bank manipulation, and soil reset. By extracting key variables from these studies, such as measures of biodiversity (e.g., alpha diversity or plant species richness) and vegetation cover (e.g., % native cover or the number of individual plants), the analysis could provide insights into the efficacy and ecological outcomes associated with each intervention.

The meta-analysis would aim to identify correlations between biodiversity and vegetation cover, with the expectation that the results would generally follow the pattern outlined in the theoretical distribution presented in Figure 2. Specifically, biomass removal would be expected to result in low biodiversity and low native cover, seed bank manipulation would show moderate native cover but with limited biodiversity, and soil reset would be associated with both high biodiversity and high native cover. This approach would allow for the empirical validation of the hierarchical intervention framework, helping to refine the predictive model for grassland restoration outcomes; NutNet (Nutrient Network) studies and trait-based frameworks offer relevant methodological comparators for such a synthesis,

providing standardised nutrient manipulation data and functional trait metrics that could help contextualise the intervention hierarchy proposed here within a global grassland dataset (Borer et al., 2014).

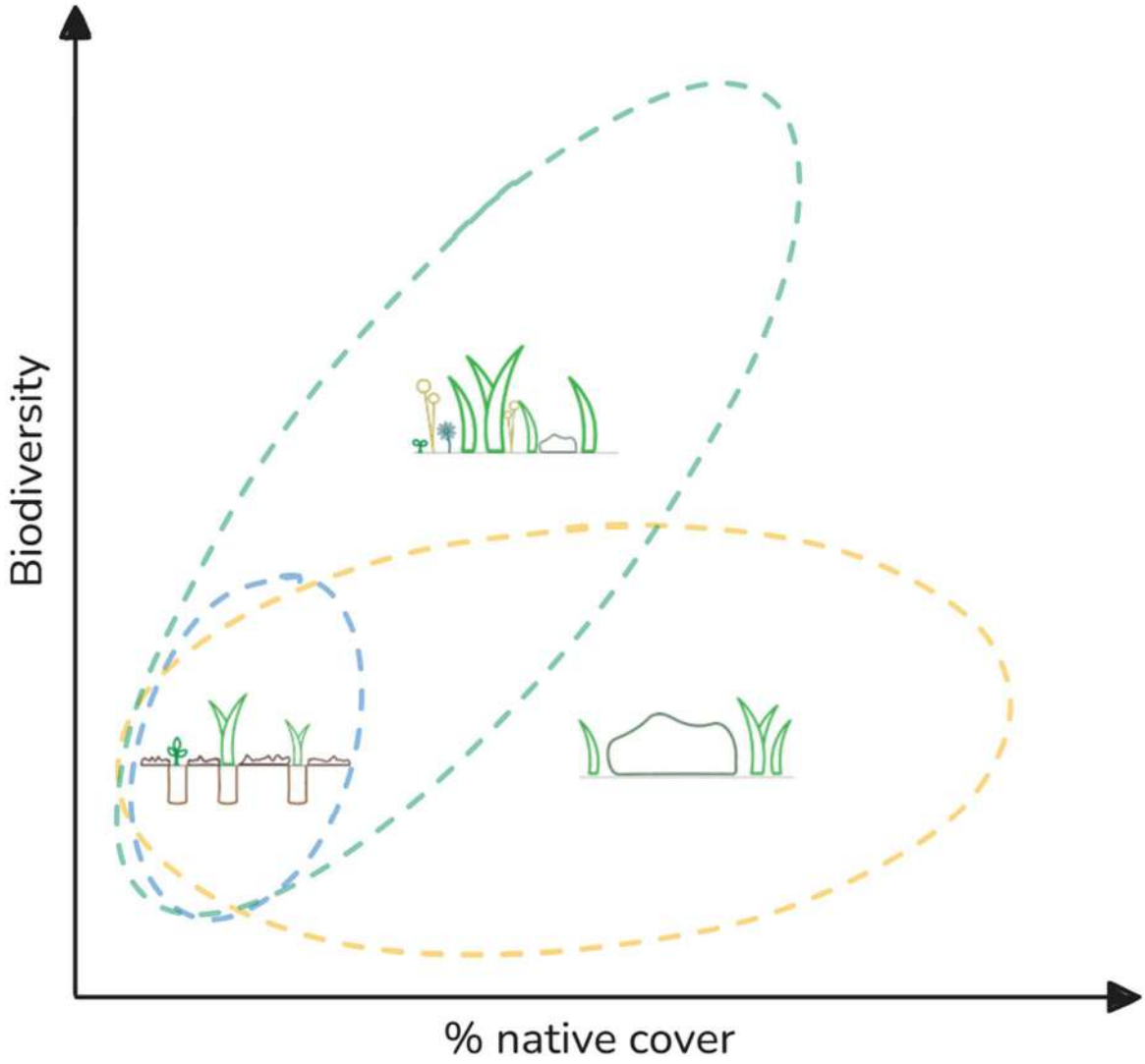

Figure 2: Theoretical relationship between biodiversity and % native cover in grassland restoration interventions. The y-axis represents biodiversity (e.g., alpha diversity or plant species richness), and the x-axis represents % native cover. Biomass removal (blue ellipse) is positioned near the origin, reflecting low biodiversity and low native cover. Seed bank manipulation (yellow/gold) is spread along the x-axis, showing higher native cover but lower biodiversity. Soil reset (green) is centred on the middle of both axes, representing high biodiversity and high native cover.

7.5 The importance of expanding urban grasslands in a changing world

7.5.1 Landscape-Scale Biodiverse Multifunctional Firebreaks

Recent extreme wildfire events, such as those in Los Angeles, California, in January 2025 and the catastrophic fires in Hawaii in 2023, have once again highlighted the vulnerability of urban and suburban areas to intense bush fires. These events demonstrate that even highly developed regions are not immune to the devastating impacts of wildfires. In Hawaii, for

example, scientists have long warned about the risk posed by invasive grasses, which cover a quarter of the Hawaiian islands and contributed significantly to the intensity and spread of fires (Kevany, 2023).

Similarly, in South Australia, the Black Sunday fires of 1955 in the Adelaide Hills demonstrated the region's susceptibility to severe fire events. These fires were exacerbated by altered vegetation structures (Lunt et al., 2006) and the challenges of managing fire risk in such areas only grows more pressing as both urban development and climate change intensify. As the climate dries and fire-prone conditions increase, regions like the Adelaide-Mt Lofty Ranges face heightened fire risks, with predicted increases in fire danger and area burned (Gill et al., 2014).

One major factor driving fire intensity and spread is the composition of natural fuels. Exotic pasture grasses - such as Phalaris aquatica - produce significantly higher biomass loads than native grasslands, leading to faster-moving, hotter, and more frequent fires (Rossiter et al., 2003; Walker & Morgan, 2022). Studies in temperate Australia have found that exotic-dominated grasslands can accumulate more than three times the fine fuel mass of native grasslands (Morgan & Lunt, 1999), with deeper, denser fuel beds that increase flame height and fire residence time (Cruz et al., 2018; Walker & Morgan, 2022). These grasses also dry out early in the season, creating a highly flammable fuel source at the peak of fire danger periods (Livingston & Varner, 2016).

In contrast, native grasslands - dominated by species such as *Themeda triandra* (kangaroo grass), *Austrostipa spp.*, and *Rytidosperma spp.*- tend to accumulate lower biomass, retain higher moisture content into summer, and burn at lower intensities, making them a preferable fuel type for fire management (Morgan & Lunt, 1999; Walker & Morgan, 2022). Additionally, research has demonstrated that native perennial grasses produce less bulk dry matter and maintain some live green material into summer, which reduces fire intensity and

slows the spread of flames compared to exotic-dominated landscapes (Livingston & Varner, 2016).

Australian cities are not well prepared for scenarios like the one in LA, but rather than accepting urban areas as vulnerable to wildfires, a landscape-scale approach is needed. Designing firebreaks that are both effective and sympathetic to biodiversity can help reduce risk while maintaining ecological value (Bowman & Sharples, 2023). This requires careful planning to balance fire safety with environmental concerns, ensuring boundaries between houses and bushland are managed without unnecessary destruction. The findings of this thesis support the case for restored native grasslands as landscape-scale firebreaks - topsoil removal demonstrated reliable suppression of exotic grass dominance (Chapter 3), and the significantly lower peak EVI2 values recorded at native-dominated sites relative to exotic-dominated controls (Chapter 5) are consistent with the lower fine fuel loads attributed to native grassland communities in the fire ecology literature. The use of these grasslands as firebreaks not only supports fire risk reduction but also contributes to biodiversity conservation in areas that are increasingly under pressure from both fire and land-use changes.

Restored native grasslands offer an attractive solution for creating landscape-scale firebreaks because they:

- **Support Biodiversity and Visual Amenity:** Native grasslands foster high biodiversity and can provide aesthetically appealing landscapes, helping to secure public support for fire management initiatives.
- **Enable Scalability and Multifunctionality:** These ecosystems can be established at a landscape scale to serve as effective firebreaks while also offering recreational and ecological functions.

- **Exhibit Lower Biomass:** Compared to forests or exotic grasslands, native grasslands accumulate less biomass, making them more manageable under controlled burning or mowing regimes.
- **Unlock Funding Opportunities:** Their restoration aligns with conservation priorities, potentially accessing environmental offset funding from programs such as SEB (Significant Environmental Benefit).

Implementing firebreaks using restored native grasslands represents an innovative, integrated approach to urban wildfire management - balancing ecological resilience with community safety, despite the upfront costs of soil reset and revegetation. Read through the lens of Chapters 3–4, these firebreaks are enabled by soil-reset interventions where warranted, and sustained through Chapter-5 style monitoring to verify fuel-load and compositional targets.

7.6 Co-implementing grassland restoration with road infrastructure upgrades

Another potentially effective strategy for grassland restoration at scale is to combine it with road infrastructure upgrades. Road projects often involve topsoil removal, creating opportunities to restore ecosystems during urban development. Gibson-Roy & Carland (2022) argue that this approach is cost-efficient. Using routine earthworks means restoration can occur with minimal additional cost, without the overheads associated with stand-alone projects (Gibson-Roy, 2022). This also brings clear visual and ecological benefits. Planting native grasslands instead of non-native species on verges improves transport corridors and boosts biodiversity. Roadsides restored in this way can become habitats for threatened plants which can be reintroduced to areas once dominated by weeds (Gibson-Roy, 2022).

On a large scale, transport corridors can act as continuous ecological links. They enhance connectivity across fragmented urban areas and facilitate the movement of wildlife, particularly pollinators and other insects. This supports broader conservation goals, such as

maintaining gene flow (Gibson-Roy & Carland, 2022). In fire-prone regions, roadside restorations can lower risk since native grasslands typically have less fine fuel than exotic grasses. (Gibson-Roy, 2022). Indeed, many of the highest-quality native grassland remnants in Western Victoria and comparable regions are found on roadsides that have historically functioned as firebreaks for rural communities (Marshall, 2013) - providing real-world precedent that the grassland-firebreak nexus already operates effectively at the urban-rural interface. However, long-term success depends on stewardship. Regular and well-timed burns, along with targeted weed control, is necessary to maintain stable community structure and prevent reinvasion. Road authorities should prioritise monitoring and maintenance as key components of project delivery (Gibson-Roy, 2022).

These restorations do many things. They act as ecological buffers, support environmental sustainability, and create more visually pleasing public landscapes. This brings infrastructure delivery closer to conservation goals. Urban projects can then help build ecological health. Integrating restoration reduces the costs of soil work while increasing biodiversity, enhancing connectivity, reducing fire risk, and fostering community engagement. For example, residents walking along a restored roadside may encounter native wildflowers and hear the buzz of native bees, creating a calm and inviting atmosphere. Landscape architects and restoration ecologists can play a crucial role in this process. Their skills in ecology and design help ensure interventions meet needs for fuel management and habitat provision. They can also help improve public acceptance and encourage stewardship. Including roadside restoration and large-scale firebreaks in planning helps bring lasting ecological and community benefits.

7.7 Final reflections and future directions

The STM-aligned hierarchy in this thesis extends the discussion on grassland restoration (Chapter 1) by demonstrating how pre-seeding interventions serve as transition mechanisms between states. It builds on existing STM logic (Westoby et al., 1989; Hobbs & Norton, 1996;

Sinclair et al., 2019) and challenges “cookbook” assumptions. By tying transitions to measurable seedbank and soil-nutrient limits (Chapters 2–4), it links theory to a practical monitoring framework (Chapter 5).

Methodologically, this thesis builds a hierarchical toolkit for each stage. In pre-seeding, it covers solarisation timing rules (Chapter 2). For establishment, it compares chemical fallow, TR and solarisation (Chapter 3). It helps guide mix selection and depth decisions through risk-aware costing (Chapter 4) and includes management and monitoring using satellite phenology (Chapter 5). From plot to satellite: methods evolve from plot-scale experiments to decision frameworks and large-scale sensing, giving a systems approach that others can adapt.

A priority for future work is a focused indicator species analysis examining species-level responses to pre-sowing interventions using the existing dataset from Chapter 3. Such an analysis would move beyond the community-level diversity metrics reported in Chapter 3 to identify which native forb and grass species are most reliably promoted by topsoil removal and solarisation - information with direct value for seed mix design and restoration benchmarking.

The experiments were necessarily limited by site and time. They include quick and mid-term observations that focus on establishment and early diversity. Some mechanisms, such as the responses of below-ground biota to TR or solarisation, remain unclear. Remote-sensing data show canopy changes, but should be combined with floristic surveys to watch forb richness and diversity. These limits do not weaken the hierarchy proposed. Instead, they highlight where conclusions are strongest.

Future research can build on these limits. Important questions include: How do below-ground biota respond differently to soil reset and seedbank manipulations across soil types and locations? Do long-term, broad-scale studies support the proposed hierarchy of pre-seeding interventions, and can they validate whether these intervention pathways produce

similar biodiversity outcomes across different soils and climates? How can remote sensing be improved to better monitor changes in functional diversity? By answering these questions, future studies can deepen our knowledge of urban grassland restoration.

7.8 Conclusion

This research has direct implications for restoring Australian temperate grasslands. The results show that light-touch methods are rarely effective in nutrient-enriched urban and peri-urban soils; successful reconstruction requires targeted and, at times, disruptive pre-seeding interventions that directly address nutrient legacies and entrenched exotic seed banks. For practitioners, this means prioritising interventions such as solarisation or topsoil removal where site conditions warrant them, and pairing these with adaptive monitoring to guide early corrections. For policymakers, the findings point to the need for evidence-aligned standards, including clearer guidance on pre-seeding treatments, criteria for topsoil removal, and the integration of scalable remote-sensing tools into monitoring frameworks. Strategically, the work supports the use of restored grasslands as multifunctional assets - providing firebreaks, habitat corridors, and biodiversity-rich green infrastructure within urban landscapes.

Restoration outcomes are often judged differently by ecologists, designers, and managers. This thesis proposes that success should be evaluated using integrated metrics. These include (i) ecological metrics of alpha diversity and phenology, (ii) design metrics of legibility, cultural fit, and aesthetic quality, and (iii) management metrics of cost-effectiveness, risk mitigation, and long-term viability. Such cross-disciplinary measures better capture the socio-ecological reality of urban grasslands (Klaus, 2013).

The research necessarily operated within specific sites, soils, and climates of southern Australia. This constrains the generality of some findings. The experimental work was limited to a few years of monitoring, leaving longer-term successional dynamics unresolved. Remote-sensing methods showed strong potential but may be limited in finer-scale or highly

heterogeneous urban mosaics. These boundaries underscore the need for cautious interpretation and future replication in other contexts.

Building on this thesis, three areas warrant priority. First, test STM-informed interventions across a wider array of urban contexts, including coastal, arid, and culturally significant landscapes. Second, to scale grasslands restoration, operationalise grassland firebreaks and roadside restoration as design experiments. These can deliver biodiversity, climate adaptation, and risk reduction simultaneously. Third, embed adaptive management through longitudinal monitoring. Utilise both remote sensing and ground-based methods to monitor how restored systems respond to shifting environmental and social pressures.

This thesis aimed to explore how urban grasslands can be restored in degraded landscapes by examining the impact of site preparation choices on biodiversity outcomes. In doing so, it contributes to ongoing efforts to better connect ecological science with landscape architecture. The research suggests that effective restoration is supported by science-driven methods combined with adaptive design philosophies that respond to societal values and priorities. The hierarchical model of pre-seeding interventions is presented as both an ecological and design tool for guiding biodiverse grassland restoration. Overall, this work suggests that we already possess much of the technical knowledge necessary to design cities with networks of biodiverse grassland corridors. It is now up to landscape architects and restoration ecologists to build coalitions to advance this vision.

This thesis has been prepared in accordance with the University of Adelaide's Specifications for Thesis and Academic Program Rules for the Doctor of Philosophy by Publication. All included publications were undertaken during the approved period of candidature, and each is prefaced by a statement of authorship detailing my contribution to the conceptualisation, execution, and documentation of the work.

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Appendices

Chapter 2: Sunlight can have a stronger influence than air temperature on soil solarisation: Observational evidence from Australia

Table S1 Linear regression model for effects of UV index and air temperature (°C) on soil (at -6 cm) temperature (°C) for the 12-week soil solarisation treatment. The coefficients for UV, air temperature, and the interaction between UV index and air temperature are statistically significant, indicating that both sunlight and air temperature contribute to soil solarisation.

	Dependent variable:
	Soil temperature (°C)
UV index	8.290^{***}
	(1.259)
Air temperature (°C)	0.988^{***}
	(0.165)
UV index: Air temperature interaction	-0.197^{***}
	(0.048)
Constant	-3.494
	(3.500)
Observations	105
R^2	0.771
Adjusted R^2	0.765
Residual Std. Error	3.546 (df = 101)
F Statistic	113.588^{***} (df = 3; 101)
Note:	*p**p***p<0.01

Table S2 Detailed classification information of soil at the study site: Self Mulching Black Vertosol, Australian Soil Classification Code: VE AE EI. All codes reference Australian Soil Classification system.

Horizon		**A1**	**B21**	**B22**	**B23k**
Boundary		clear	diffuse	diffuse	diffuse
Depth (cm)		0-10	10-50	50-90	90-100
Colour		black (10 YR 2/1)	black (10 YR 2/1)	black (10 YR 2/1)	black (10 YR 2/1)
Mottles		-	-	-	few, 2-6mm, orange
Texture		Light Medium Clay	Medium Heavy Clay	Medium Heavy Clay	Medium Heavy Clay
Structure	**Grade**	Moderate	Strong	Strong	Strong
	Shape	Angular Blocky	Prismatic	Lenticular	Lenticular
	Size	3	5	3	2
Coarse Fragments		-	-	-	-
Segregations		-	-	-	20-50%, calcareous, soft, medium

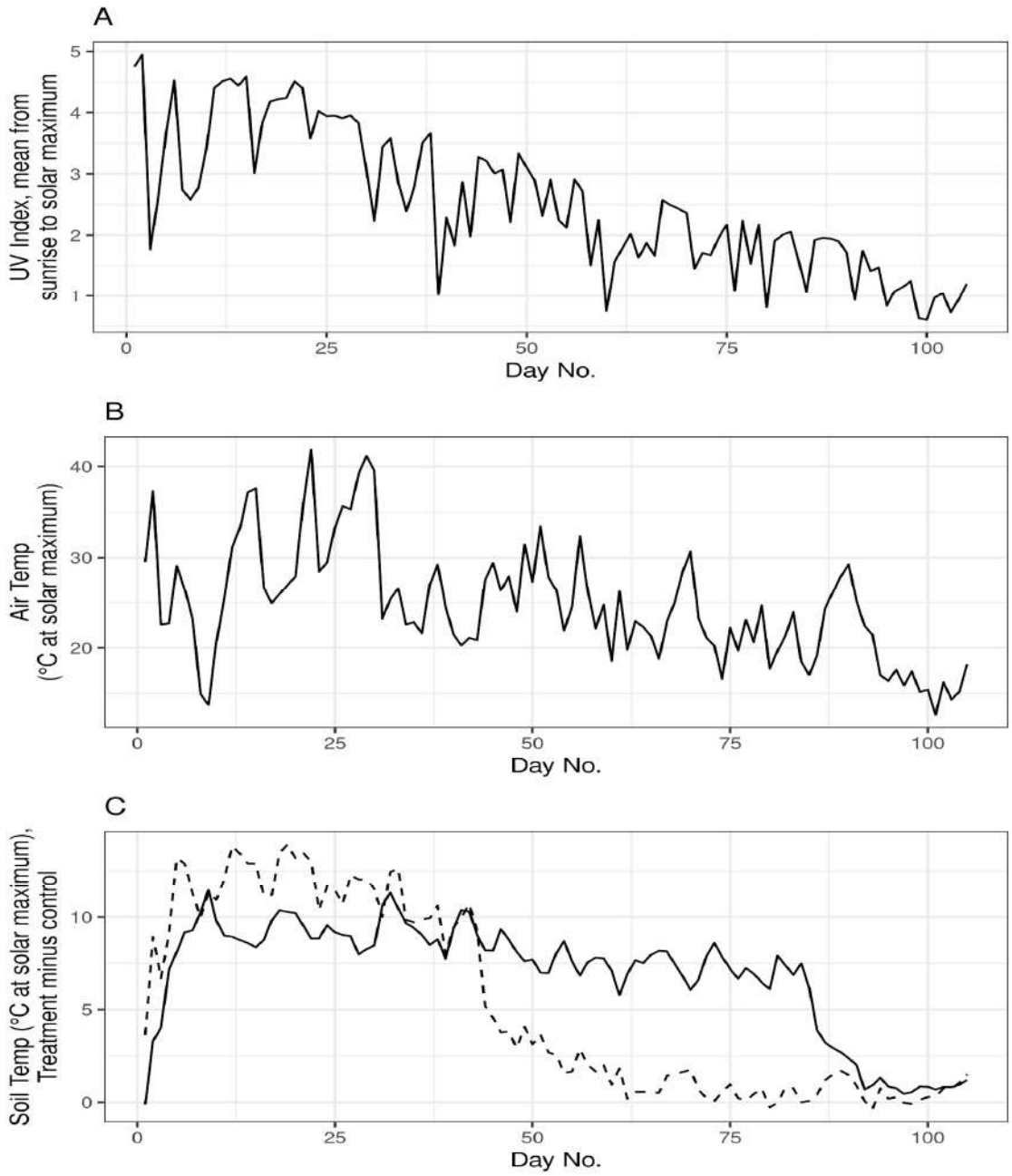

Fig. S1. Time series of ambient temperature and sunlight and soil temperature difference over study duration. (A) Time series showing the daily UV index mean calculated from sunrise to the time of peak daily sunlight, source: ARPANSA (2023); (B) Time series showing the daily air temperature ° C at the time of peak daily sunlight, averaged across the sensors in the 12 week treatment; (C) Time series showing the difference in daily soil topsoil (at -6 cm) temperature ° C for the 12 week and 6 week solarisation treatment at the time of peak daily sunlight, averaged across the sensors within each treatment (dashed lines indicate the 6 week solarisation treatment, while the solid line indicated the 12 week solarisation treatment).

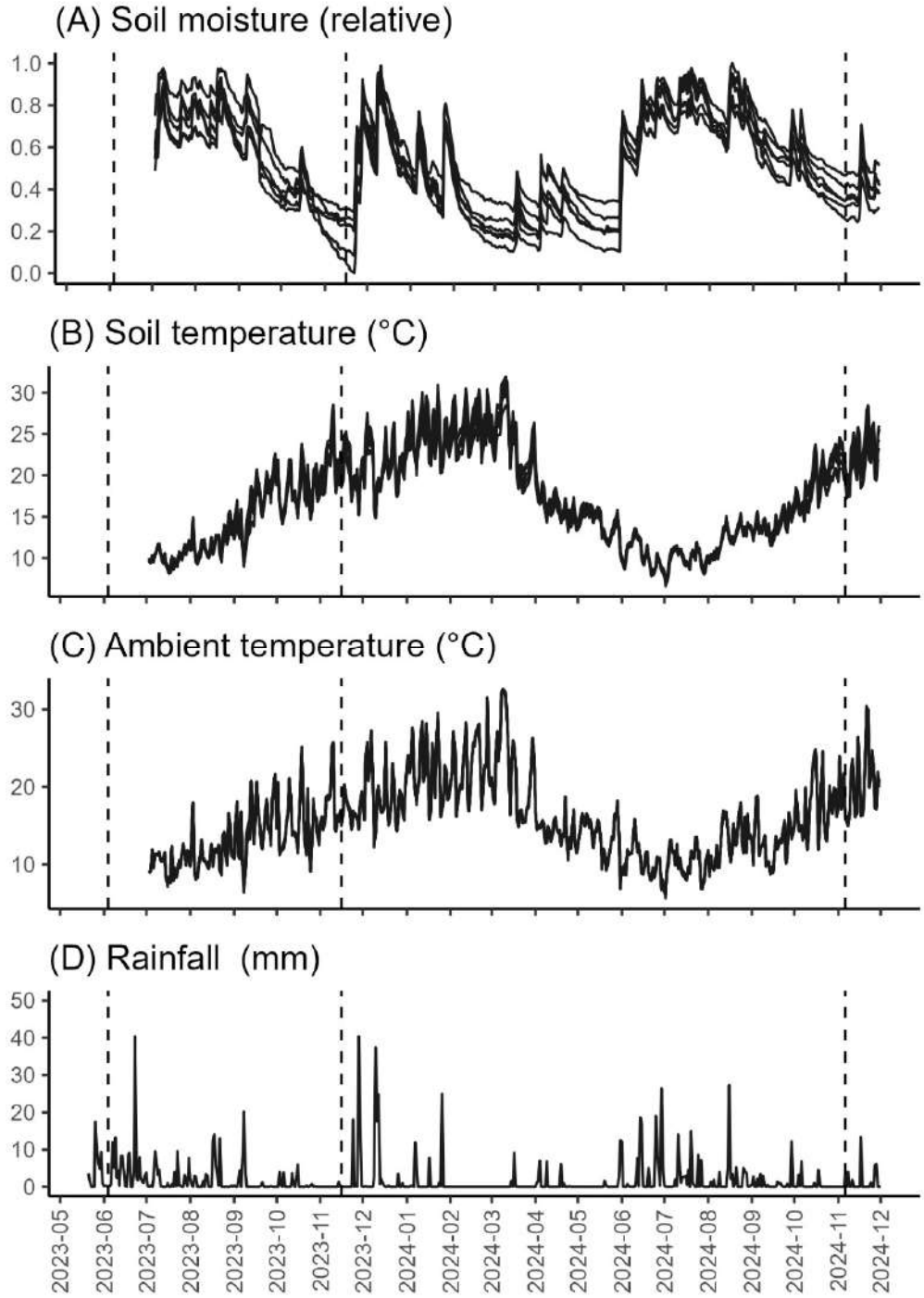

Figure S1. Temporal dynamics of environmental variables over two years. (A) Soil moisture (relative) indicates fluctuations in soil moisture content, showing seasonal patterns and response to ambient and subsoil temperature) and rainfall. (B) Soil temperature (°C) reflects variations in subsurface temperature measured at 10cm below the surface, peaking during warmer months. (C) Ambient temperature (°C) demonstrates daily and seasonal trends in air temperature. (D) Rainfall (mm) captures precipitation events, showing distinct seasonality throughout the observation period. Dashed vertical lines represent key time markers for the start and end of different observation periods (Soil sampling at T0; soil and vegetation sampling at T_1 – 6 months post seeding; and soil and vegetation sampling at T_2 - 18 months post seeding).

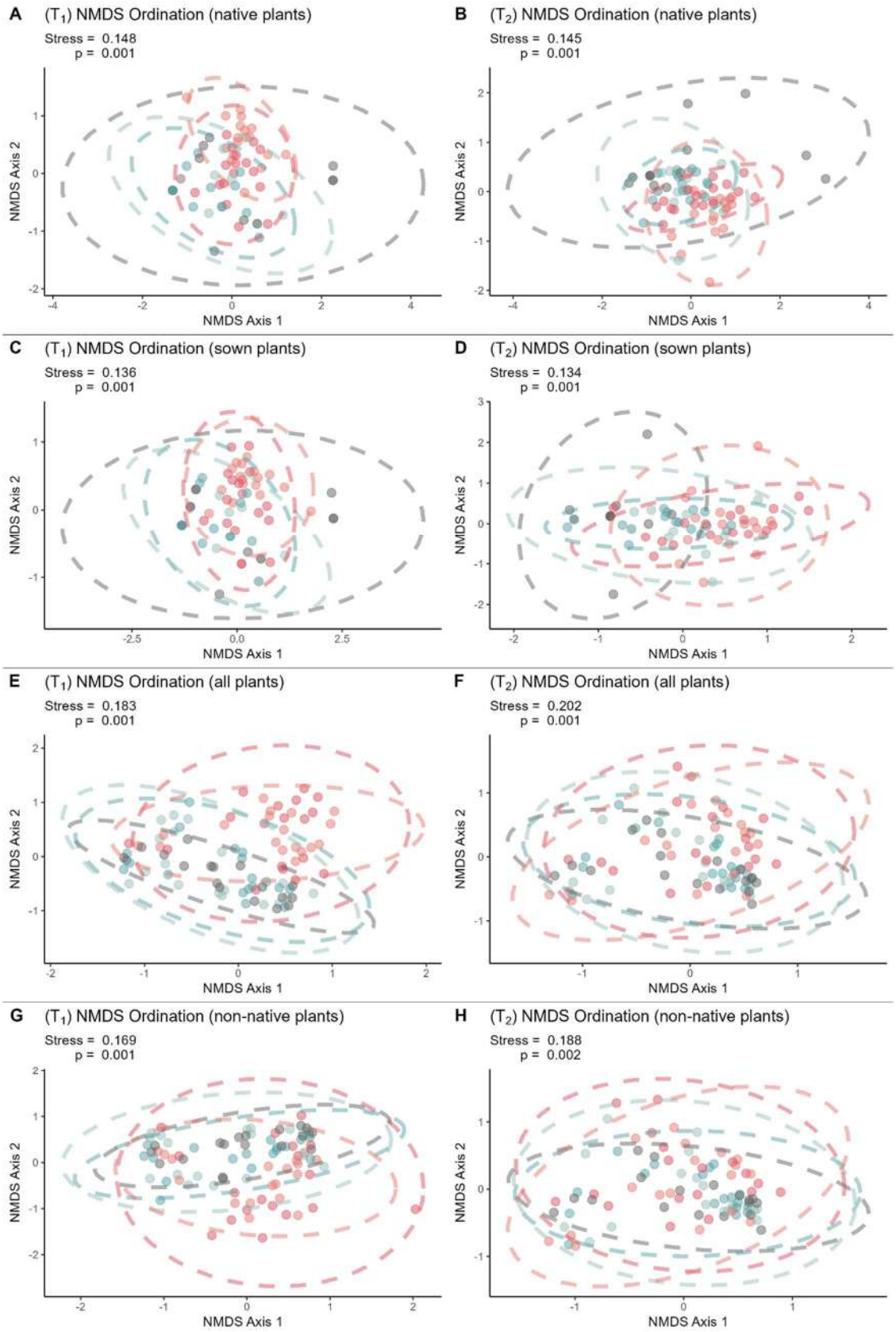

Figure S2: Plots showing community diversity and composition at 6 months (T_1) and 18 months (T_2) post-sowing. Panel (A) displays the NMDS ordination for native plants at T_1, and panel (B) shows the same for native plants at T_2. Panel (C) presents the NMDS ordination for sown native plants at T_1, with panel (D) showing sown native plants at T_2. Panel (E) represents the NMDS ordination for all plants (native and non-native) at T_1, and panel (F) shows the same at T_2. Finally, panel (G) presents the NMDS ordination for non-native plants at T_1, with panel (H) showing non-native plants at T_2. Treatments include control, solarisation (6-week and 12-week durations), and topsoil removal (7 cm and 14 cm depths).

Table S1. Photosynthetic pathway, plant type, and total seed weight (g/m^2) for 41 Australian native plant species. Photosynthetic pathway (C3 or C4) was assigned based on general knowledge and may not reflect all species-specific variations. Plant types are broadly categorized as Forb, Grass, Other Monocot, Shrub, or Vine. Treatment weight represents the total biomass allocated to each species per plot in a field experiment. *denotes species identified within replicates but not necessarily captures in quadrats for survey.

Species	Plant Type	Photosynthetic Pathway	Seed weight (mg/m^2)
*Anthosachne scabra**	Grass	C_3	224
*Austrostipa curticoma**			112
*Austrostipa elegentissima**			56
*Austrostipa scabra**			56
*Microlaena stipoides**			56
*Poa labillardieri**			24
*Rytidosperma caespitosum**			32
*Rytidosperma racemosum**			12
Aristida behriana		C_4	224
*Chloris truncata**			56
Cymbopogon ambiguous			56
*Dichanthium sericeum**			12
Enneapogon nigricans			24
*Themeda triandra**			112
Arthropodium strictum	Forb (other monocots)	C_3	224
Bulbine bulbosa			56
Burchardia umbellate			4
Callostemma purpureum			12
*Aceana echinata**	Forb (herbs)	C_3	24
*Eryngium ovinum**			56
*Geranium retrorsum**			76
*Goodenia pinnatifida**			168
*Leptorhynchos squamatus**			12
*Ptilotus spathulatus**			224
Ranunculus lappaceus			112
*Senecio quadridentatus**			24
*Vittadinia blackii**			112
*Vittadinia cuneata**			12
*Wahlenbergia stricta**			4
*Xerochrysum bracteatum**			76
*Calocephalus citreus**			776
*Chrysocephalum apiculatum**			668
Clematis microphylla	Forb (vines)	C3	168
*Convolvulus angustissmus**			76
*Convolvulus remotus**			112
*Hardenbergia violacea**			44
Kennedia prostrata			12
*Eutaxia microphylla**	Shrub	C_3	56
*Atriplex semibacatta**		C_4	56
Einadia nutens			56
*Enchylaena tomentosa**			56
Total			**4.3 g/m^2**

Table S2A: Kruskal-Wallis Test Results for Alpha Diversity Across Treatment Groups at T_1 (6 months post-seeding in November 2023).

Variable	Kruskal-Wallis Chi-Squared	Degrees of Freedom	p-value	Significance
Alpha all	23.898	4	8.45	***
Alpha native	41.538	4	2.1e-08	***
Alpha non-native	5.3763	4	0.25	ns
Alpha sown	47.312	4	1.313e-09	***

Significance Codes:
*** $p < 0.001$
** $p < 0.01$
* $p < 0.05$
ns $p \geq 0.05$

Table S2B: Kruskal-Wallis Test Results for Alpha Diversity Across Treatment Groups at T_2 (18 months post-seeding in November 2024).

Variable	Kruskal-Wallis Chi-Squared	Degrees of Freedom	p-value	Significance
Alpha all	8.1288	4	0.08697	ns
Alpha native	45.567	4	3.031e-09	***
Alpha non-native	2.239	4	0.6919	ns
Alpha sown	47.031	4	1.502e-09	***

Significance Codes:
*** $p < 0.001$
** $p < 0.01$
* $p < 0.05$
ns $p \geq 0.05$

Table S3A: Pairwise Comparisons of Native Alpha Diversity Across Treatment Groups at T_1 (6 months post seeding in November 2023) using Wilcoxon rank sum tests with Benjamini-Hochberg adjustment for multiple comparisons.

Comparison	Adjusted p-value	Significance
Control vs Solarisation 12 weeks	0.33785	ns
Control vs Solarisation 6 weeks	0.89335	ns
Control vs Topsoil Removal 14 cm	3.9e-05	***
Control vs Topsoil Removal 7 cm	3.9e-05	***
Solarisation 12 weeks vs Solarisation 6 weeks	0.55563	ns
Solarisation 12 weeks vs Topsoil Removal 14 cm	0.00020	***
Solarisation 12 weeks vs Topsoil Removal 7 cm	0.00017	***
Solarisation 6 weeks vs Topsoil Removal 14 cm	0.00055	***
Solarisation 6 weeks vs Topsoil Removal 7 cm	0.00055	***
Topsoil Removal 14 cm vs Topsoil Removal 7 cm	0.33785	ns

Table S3B: Pairwise Comparisons of Sown Alpha Diversity Across Treatment Groups at T_1 (6 months post seeding in November 2023) using Wilcoxon rank sum tests with Benjamini-Hochberg adjustment for multiple comparisons.

Comparison	Adjusted p-value	Significance
Control vs Solarisation 12 weeks	0.15468	ns
Control vs Solarisation 6 weeks	0.59600	ns
Control vs Topsoil Removal 14 cm	7.6e-06	***
Control vs Topsoil Removal 7 cm	1.2e-05	***
Solarisation 12 weeks vs Solarisation 6 weeks	0.55566	ns
Solarisation 12 weeks vs Topsoil Removal 14 cm	0.00018	***
Solarisation 12 weeks vs Topsoil Removal 7 cm	0.00015	***
Solarisation 6 weeks vs Topsoil Removal 14 cm	8.8e-05	***
Solarisation 6 weeks vs Topsoil Removal 7 cm	8.8e-05	***
Topsoil Removal 14 cm vs Topsoil Removal 7 cm	0.35222	ns

Table S3C: Pairwise Comparisons of All Plants Alpha Diversity Across Treatment Groups at T_1 (6 months post seeding in November 2023) using Wilcoxon rank sum tests with Benjamini-Hochberg adjustment for multiple comparisons.

Comparison	Adjusted p-value	Significance
Control vs Solarisation 12 weeks	0.91293	ns
Control vs Solarisation 6 weeks	0.91293	ns
Control vs Topsoil Removal 14 cm	0.00831	***
Control vs Topsoil Removal 7 cm	0.00066	***
Solarisation 12 weeks vs Solarisation 6 weeks	0.91293	ns
Solarisation 12 weeks vs Topsoil Removal 14 cm	0.00863	***
Solarisation 12 weeks vs Topsoil Removal 7 cm	0.00863	***
Solarisation 6 weeks vs Topsoil Removal 14 cm	0.00066	***
Solarisation 6 weeks vs Topsoil Removal 7 cm	0.00783	***
Topsoil Removal 14 cm vs Topsoil Removal 7 cm	0.91293	ns

Table S3D: Pairwise Comparisons of Native Alpha Diversity Across Treatment Groups at T_2 (18 months post seeding in November 2024) using Wilcoxon rank sum tests with Benjamini-Hochberg adjustment for multiple comparisons.

Comparison	Adjusted p-value	Significance
Control vs Solarisation 12 weeks	0.00243	**
Control vs Solarisation 6 weeks	0.35272	ns
Control vs Topsoil Removal 14 cm	1.5^{-05}	***
Control vs Topsoil Removal 7 cm	0.00020	***
Solarisation 12 weeks vs Solarisation 6 weeks	0.04546	*
Solarisation 12 weeks vs Topsoil Removal 14 cm	0.00176	**
Solarisation 12 weeks vs Topsoil Removal 7 cm	0.00015	***
Solarisation 6 weeks vs Topsoil Removal 14 cm	0.00019	***
Solarisation 6 weeks vs Topsoil Removal 7 cm	0.86184	ns
Topsoil Removal 14 cm vs Topsoil Removal 7 cm	0.86184	ns

Table S3E: Pairwise Comparisons of Sown Alpha Diversity Across Treatment Groups at T_2 (18 months post seeding in November 2024) using Wilcoxon rank sum tests with Benjamini-Hochberg adjustment for multiple comparisons.

Comparison	Adjusted p-value	Significance
Control vs Solarisation 12 weeks	0.00058	***
Control vs Solarisation 6 weeks	0.10733	ns
Control vs Topsoil Removal 14 cm	5.6^{-06}	***
Control vs Topsoil Removal 7 cm	1.4^{-05}	***
Solarisation 12 weeks vs Solarisation 6 weeks	0.06177	ns
Solarisation 12 weeks vs Topsoil Removal 14 cm	0.00511	**
Solarisation 12 weeks vs Topsoil Removal 7 cm	0.00021	***
Solarisation 6 weeks vs Topsoil Removal 14 cm	0.00021	***
Solarisation 6 weeks vs Topsoil Removal 7 cm	0.44760	ns
Topsoil Removal 14 cm vs Topsoil Removal 7 cm	0.44760	ns

Significance Codes:

*** $p < 0.001$

** $p < 0.01$

* $p < 0.05$

ns $p \geq 0.05$

Table S4A: PERMANOVA Results for T_1 (6 months post-seeding, November 2023). This table shows the results of the PERMANOVA analysis for the T_1 data, assessing treatment effects on vegetation composition across all plants, native species, non-native species, and sown species. Significant effects were found for all groups ($p < 0.001$), with native species showing the highest R^2 values.

Variable	Df	Sum of Sqs	R^2	F	Pr (>F)	Significance
All	4	4.677	0.18629	4.8648	0.001	***
Native	4	4.1733	0.19987	3.9967	0.001	***
Non-Native	4	2.8653	0.11873	2.8629	0.001	***
Sown	4	3.0097	0.16179	3.0401	0.001	***

Table S4B: PERMANOVA Results for T_2 (18 months post-seeding, November 2024)

The PERMANOVA results for T_2 indicate significant treatment effects on the overall community, native species ($p < 0.001$), and non-native species ($p = 0.002$), suggesting ongoing treatment impacts over time.

Variable	Df	Sum of Sqs	R^2	F	Pr (>F)	Significance
All	4	2.2177	0.11088	2.6501	0.001	***
Native	4	4.1257	0.20155	4.7331	0.001	***
Non-Native	4	1.7526	0.09293	2.177	0.002	**
Sown	4	3.4048	0.19347	4.3779	0.001	***

Significance Codes:

*** $p < 0.001$
** $p < 0.01$
* $p < 0.05$
ns $p \geq 0.05$

Table S5A: Dispersion ANOVA Results for T_1 (6 months post-seeding, November 2023). This table presents dispersion analysis results for T_1, revealing significant differences in dispersion for native and non-native species ($p < 0.05$), but no significant effects for the overall community or sown species.

Variable	Df	Sum Sq	Mean Sq	F value	Pr (>F)	Significance
All	4	0.075	0.0188	1.03	0.397	ns
Native	4	0.248	0.0619	3.01	0.0244	*
Non-Native	4	0.211	0.0527	2.67	0.0377	*
Sown	4	0.192	0.0480	2.03	0.101	ns

Table S5B: Dispersion ANOVA Results for T_2 (18 months post-seeding, November 2024). The dispersion analysis for T_2 shows no significant differences across any groups, indicating stabilisation of community structure over time.

Variable	Df	Sum Sq	Mean Sq	F value	Pr (>F)	Significance
All	4	0.0296	0.00739	0.318	0.865	ns
Native	4	0.155	0.0388	1.279	0.286	ns
Non-Native	4	0.0330	0.00825	0.338	0.852	ns
Sown	4	0.104	0.0259	0.755	0.558	ns

Significance Codes:

*** $p < 0.001$
** $p < 0.01$
* $p < 0.05$
ns $p \geq 0.05$

Table S6A: Pairwise Comparisons of NMDS Vegetation Community Composition for Native Plants Across Treatment Groups at T_1. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.157	0.599	0.0291	0.644	0.644	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	1.804	7.62	0.227	0.001	0.003	*
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.787	3.15	0.108	0.014	0.020	.
Solarisation 6 weeks vs Control	1	0.412	1.18	0.0586	0.310	0.344	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	2.114	9.55	0.254	0.001	0.003	*
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	0.764	3.28	0.105	0.009	0.015	.
Solarisation 12 weeks vs Control	1	0.470	1.48	0.0659	0.200	0.250	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.964	4.40	0.115	0.002	0.004	.
Topsoil Removal 7 cm vs Control	1	1.460	5.20	0.162	0.001	0.003	*
Topsoil Removal 14 cm vs Control	1	1.004	3.43	0.113	0.002	0.004	.

Note: In all models in this section, the reference category for plant type is "Forb," so the main effects for treatments indicate their impact on forb abundance.

Table S6B: Pairwise Comparisons of NMDS Vegetation Community Composition for Sown Native Plants Across Treatment Groups at T_1. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.140	0.581	0.0282	0.622	0.691	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	1.126	4.95	0.165	0.001	0.003	*
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.807	3.56	0.120	0.007	0.014	.
Solarisation 6 weeks vs Control	1	0.388	1.21	0.0597	0.302	0.385	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	1.076	4.93	0.154	0.001	0.003	*

Solarisation 12 weeks vs Topsoil Removal 14 cm	1	0.750	3.44	0.109	0.014	0.023	.
Solarisation 12 weeks vs Control	1	0.377	1.25	0.0563	0.308	0.385	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.122	0.577	0.0172	0.772	0.772	ns
Topsoil Removal 7 cm vs Control	1	1.325	4.80	0.156	0.001	0.003	*
Topsoil Removal 14 cm vs Control	1	1.266	4.62	0.146	0.002	0.005	.

Table S6C: Pairwise Comparisons of NMDS Vegetation Community Composition for All Plants Across Treatment Groups at T_1. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.171	0.688	0.0198	0.594	0.594	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	1.758	7.15	0.174	0.001	0.002	*
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	1.462	5.42	0.138	0.001	0.002	*
Solarisation 6 weeks vs Control	1	0.298	1.22	0.0347	0.300	0.333	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	1.809	8.12	0.193	0.001	0.002	*
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	1.398	5.67	0.143	0.001	0.002	*
Solarisation 12 weeks vs Control	1	0.394	1.78	0.0498	0.138	0.173	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.573	2.35	0.065	0.021	0.030	.
Topsoil Removal 7 cm vs Control	1	2.029	9.28	0.214	0.001	0.002	*
Topsoil Removal 14 cm vs Control	1	1.800	7.43	0.179	0.001	0.002	*

Table S6D: Pairwise Comparisons of NMDS Vegetation Community Composition for Non-Native Plants Across Treatment Groups at T_1. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.188	0.784	0.0225	0.517	0.517	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	0.830	3.21	0.0863	0.007	0.012	*

Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.985	3.34	0.0896	0.005	0.010	.
Solarisation 6 weeks vs Control	1	0.277	1.19	0.0339	0.296	0.370	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	0.885	3.82	0.101	0.005	0.010	.
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	1.155	4.31	0.113	0.002	0.007	.
Solarisation 12 weeks vs Control	1	0.350	1.70	0.0477	0.160	0.229	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.304	1.06	0.0302	0.389	0.432	ns
Topsoil Removal 7 cm vs Control	1	1.016	4.53	0.117	0.002	0.007	.
Topsoil Removal 14 cm vs Control	1	1.173	4.50	0.117	0.002	0.007	.

Table S6E: Pairwise Comparisons of NMDS Vegetation Community Composition for Native Plants Across Treatment Groups at T_2. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.236	1.26	0.042	0.261	0.261	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	1.461	6.66	0.177	0.001	0.0025	*
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.883	4.08	0.116	0.003	0.00429	.
Solarisation 6 weeks vs Control	1	0.527	1.98	0.071	0.065	0.0813	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	1.441	7.84	0.197	0.001	0.0025	*
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	0.739	4.08	0.113	0.003	0.00429	.
Solarisation 12 weeks vs Control	1	0.915	4.12	0.132	0.003	0.00429	.
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.345	1.64	0.046	0.131	0.146	ns
Topsoil Removal 7 cm vs Control	1	2.084	8.20	0.220	0.001	0.0025	*
Topsoil Removal 14 cm vs Control	1	1.708	6.79	0.190	0.001	0.0025	*

Table S6F: Pairwise Comparisons of NMDS Vegetation Community Composition for Sown Native Plants Across Treatment Groups at T_2. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.199	1.15	0.038	0.355	0.394	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	0.960	4.53	0.127	0.004	0.008	.
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.808	4.00	0.114	0.005	0.0083	.
Solarisation 6 weeks vs Control	1	0.431	1.97	0.076	0.096	0.120	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	0.958	5.41	0.145	0.001	0.0033	*
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	0.717	4.28	0.118	0.004	0.008	.
Solarisation 12 weeks vs Control	1	0.766	4.41	0.150	0.008	0.0114	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.156	0.773	0.0222	0.608	0.608	ns
Topsoil Removal 7 cm vs Control	1	1.926	8.86	0.247	0.001	0.0033	*
Topsoil Removal 14 cm vs Control	1	1.750	8.51	0.240	0.001	0.0033	*

Table S6G: Pairwise Comparisons of NMDS Vegetation Community Composition for All Plants Across Treatment Groups at T_2. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.0835	0.403	0.0117	0.887	0.887	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	0.581	2.65	0.0724	0.024	0.040	*
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.866	3.96	0.104	0.002	0.010	.
Solarisation 6 weeks vs Control	1	0.146	0.686	0.0198	0.627	0.697	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	0.674	3.32	0.089	0.006	0.012	*
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	0.742	3.66	0.0973	0.005	0.012	.
Solarisation 12 weeks vs Control	1	0.149	0.758	0.0218	0.562	0.697	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.408	1.90	0.053	0.069	0.099	.
Topsoil Removal 7 cm vs Control	1	0.886	4.24	0.111	0.004	0.012	.
Topsoil Removal 14 cm vs Control	1	1.008	4.84	0.125	0.001	0.010	*

Table S6H: Pairwise Comparisons of NMDS Vegetation Community Composition for Non-Native Plants Across Treatment Groups at T_2. This assesses treatment effects on community composition with adjustments for multiple comparisons using the Benjamini-Hochberg correction.

Comparison	Df	Sums OfSqs	F.Model	R^2	p-value	Adjusted p-value	Sig.
Solarisation 6 weeks vs Solarisation 12 weeks	1	0.097	0.489	0.0142	0.789	0.796	ns
Solarisation 6 weeks vs Topsoil Removal 7 cm	1	0.428	2.01	0.056	0.082	0.117	ns
Solarisation 6 weeks vs Topsoil Removal 14 cm	1	0.806	3.79	0.100	0.002	0.020	.
Solarisation 6 weeks vs Control	1	0.108	0.536	0.0155	0.783	0.796	ns
Solarisation 12 weeks vs Topsoil Removal 7 cm	1	0.546	2.78	0.076	0.034	0.068	ns
Solarisation 12 weeks vs Topsoil Removal 14 cm	1	0.669	3.41	0.0911	0.004	0.020	.
Solarisation 12 weeks vs Control	1	0.0811	0.439	0.0128	0.796	0.796	ns
Topsoil Removal 7 cm vs Topsoil Removal 14 cm	1	0.423	2.00	0.0555	0.061	0.102	ns
Topsoil Removal 7 cm vs Control	1	0.531	2.66	0.0724	0.018	0.045	*
Topsoil Removal 14 cm vs Control	1	0.694	3.47	0.093	0.006	0.020	.

Table S7A: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of native plants at T_1 (6 months post sowing, November 2023).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	5.445^{-15}	0.2357	0.000	1.0000	ns
Solarisation for 6 weeks (vs Control treatment)	-2.574^{-15}	0.3333	0.000	1.0000	ns
Solarisation for 12 weeks (vs Control treatment)	-0.8109	0.4249	-1.908	0.0563	ns
Topsoil Removal of 7 cm (vs Control treatment)	3.0500	0.2412	12.643	$< 2^{-16}$	***
Topsoil Removal of 14 cm (vs Control treatment)	1.6840	0.2566	6.563	5.27^{-11}	***
Grass plant type (vs Forb plant type)	-0.1178	0.3436	-0.343	0.7317	ns
Shrub plant type (vs Forb plant type)	-17.3050	817.4634	-0.021	0.9831	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	0.8383	0.4479	1.872	0.0613	ns
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	2.0810	0.5105	4.077	4.56^{-05}	***
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-0.7044	0.3559	-1.979	0.0478	*
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	0.6490	0.3666	1.770	0.0767	ns
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	-3.263^{-09}	1.156^{03}	0.000	1.0000	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	0.8109	1.156^{03}	0.001	0.9994	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	12.8743	495.7678	0.026	0.9793	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	13.7887	495.7678	0.028	0.9778	ns

Table S7B: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of sown plants at T_1 (6 months post sowing, November 2023).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	-1.50408	0.50000	-3.008	0.00263	**
Solarisation for 6 weeks (vs Control treatment)	-0.69315	0.86603	-0.800	0.42349	ns
Solarisation for 12 weeks (vs Control treatment)	-0.28768	0.76376	-0.377	0.70642	ns
Topsoil Removal of 7 cm (vs Control treatment)	2.80336	0.51493	5.444	5.2^{-08}	***
Topsoil Removal of 14 cm (vs Control treatment)	2.42037	0.52175	4.639	3.5^{-06}	***
Grass plant type (vs Forb plant type)	1.38629	0.55902	2.480	0.01314	*
Shrub plant type (vs Forb plant type)	-16.79851	1347.63534	-0.012	0.99005	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	1.53148	0.91626	1.671	0.09463	ns
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	1.55814	0.81448	1.913	0.05574	ns
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-0.45796	0.57762	-0.793	0.42787	ns
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-0.08701	0.58377	-0.149	0.88151	ns
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	13.30200	1347.63553	0.010	0.99212	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	13.68499	1347.63554	0.010	0.99190	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	1.262^{01}	4.958^{02}	0.025	0.9797	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	1.07^{01}	4.958^{02}	0.022	0.9828	ns

Table S7C: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of all plant types at T_1 (6 months post sowing, November 2023).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	2.1401	0.0809	26.471	$< 2^{-16}$	***
Solarisation for 6 weeks (vs Control treatment)	0.1845	0.1094	1.686	0.0917	ns

Variable	Estimate	Std. Error	z value	p-value	Significance
Solarisation for 12 weeks (vs Control treatment)	-0.8561	0.1481	-5.782	7.38^{-09}	***
Topsoil Removal of 7 cm (vs Control treatment)	1.2311	0.0919	13.397	$< 2^{-16}$	***
Topsoil Removal of 14 cm (vs Control treatment)	0.3167	0.1063	2.979	0.00289	**
Grass plant type (vs Forb plant type)	1.4917	0.0895	16.670	$< 2^{-16}$	***
Shrub plant type (vs Forb plant type)	-18.4427	495.7673	-0.037	0.9703	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	-0.3299	0.1231	-2.681	0.00735	**
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	0.5859	0.1591	3.682	0.000231	***
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-1.7443	0.1112	-15.682	$< 2^{-16}$	***
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-0.7634	0.1228	-6.219	4.99^{-10}	***
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	-0.1845	701.1209	0.000	0.9998	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	0.8561	701.1209	0.001	0.9990	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	12.8743	495.7678	0.026	0.9793	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	13.7887	495.7678	0.028	0.9778	ns

Table S7D: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of non-native plants at T_1 (6 months post sowing, November 2023).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	1.78246	0.09667	18.438	$< 2^{-16}$	***
Solarisation for 6 weeks (vs Control treatment)	0.40234	0.12488	3.222	0.00127	**
Solarisation for 12 weeks (vs Control treatment)	-0.86617	0.17767	-4.875	1.09^{-06}	***
Topsoil Removal of 7 cm (vs Control treatment)	0.29698	0.12763	2.327	0.01997	*
Topsoil Removal of 14 cm (vs Control treatment)	0.05456	0.13489	0.404	0.68587	ns
Grass plant type (vs Forb plant type)	1.82545	0.10417	17.523	$< 2^{-16}$	***
Shrub plant type (vs Forb plant type)	-19.08504	817.38210	-0.023	0.98137	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	-0.58889	0.13754	-4.282	1.85^{-05}	***
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	0.50345	0.18772	2.682	0.00732	**
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-1.31463	0.14820	-8.871	$< 2^{-16}$	***
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-0.95442	0.15299	-6.239	4.42^{-10}	***
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	-0.40234	1155.95285	0.000	0.99972	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	0.86617	1155.95286	0.001	0.99940	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	-0.29698	1155.95285	0.000	0.99980	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	-0.05456	1155.95285	0.000	0.99996	ns

Table S7E: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of native plants at T_2 (18 months post sowing, November 2024).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	-0.4055	0.2887	-1.405	0.16015	ns
Solarisation for 6 weeks (vs Control treatment)	0.6506	0.3561	1.827	0.06771	ns
Solarisation for 12 weeks (vs Control treatment)	0.5108	0.3651	1.399	0.16183	ns
Topsoil Removal of 7 cm (vs Control treatment)	2.9400	0.2962	9.926	$< 2^{-16}$	***
Topsoil Removal of 14 cm (vs Control treatment)	2.9618	0.2960	10.005	$< 2^{-16}$	***
Grass plant type (vs Forb plant type)	0.6506	0.3561	1.827	0.06771	Ns
Shrub plant type (vs Forb plant type)	-15.8971	495.7674	-0.032	0.97442	Ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	0.8389	0.4244	1.977	0.04807	*
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	1.1328	0.4304	2.632	0.00848	**
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-0.7574	0.3689	-2.053	0.04008	*
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-0.5405	0.3674	-1.471	0.14125	ns
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	12.7616	495.7684	0.026	0.97946	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	13.5945	495.7679	0.027	0.97812	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	13.0371	495.7675	0.026	0.97902	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	11.5490	495.7677	0.023	0.98141	ns

Table S7F: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of sown plants at T_2 (18 months post sowing, November 2024).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	-1.792	5.774^{-01}	-3.103	0.001913	**
Solarisation for 6 weeks (vs Control treatment)	-6.674^{-14}	8.165^{-01}	0.000	1.000000	ns
Solarisation for 12 weeks (vs Control treatment)	1.099	6.667^{-01}	1.648	0.099369	ns
Topsoil Removal of 7 cm (vs Control treatment)	3.356	5.873^{-01}	5.714	1.11^{-08}	***
Topsoil Removal of 14 cm (vs Control treatment)	3.807	5.837^{-01}	6.521	6.97^{-11}	***
Grass plant type (vs Forb plant type)	2.037	6.138^{-01}	3.318	0.000906	***
Shrub plant type (vs Forb plant type)	-1.451^{1}	4.958^{02}	-0.029	0.976650	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	1.489	8.485^{-01}	1.755	0.079186	ns
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	5.450^{-1}	7.045^{-01}	0.774	0.439157	ns
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-1.173	6.272^{-01}	-1.870	0.061417	ns
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-1.385	6.230^{-01}	-2.224	0.026163	*
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	1.341^{1}	4.958^{02}	0.027	0.978417	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	1.301^{1}	4.958^{02}	0.026	0.979069	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	1.262^{1}	4.958^{02}	0.025	0.979689	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	1.070^{1}	4.958^{02}	0.022	0.982774	ns

Table S7G: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of all plant types at T_2 (18 months post sowing, November 2024).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	3.47953	0.04138	84.087	$< 2^{-16}$	***
Solarisation for 6 weeks (vs Control treatment)	0.11626	0.05689	2.044	0.041	*
Solarisation for 12 weeks (vs Control treatment)	0.01698	0.05827	0.291	0.771	ns
Topsoil Removal of 7 cm (vs Control treatment)	0.38935	0.05359	7.265	3.74^{-13}	***
Topsoil Removal of 14 cm (vs Control treatment)	0.35393	0.05398	6.556	5.52^{-11}	***
Grass plant type (vs Forb plant type)	1.15682	0.04744	24.383	$< 2^{-16}$	***
Shrub plant type (vs Forb plant type)	-17.78211	182.38260	-0.097	0.922	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	-0.02178	0.06531	-0.334	0.739	ns
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	-0.10062	0.06723	-1.497	0.134	ns
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-0.83605	0.06521	-12.820	$< 2^{-16}$	***
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-0.94431	0.06652	-14.196	$< 2^{-16}$	***
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	11.29595	182.38535	0.062	0.951	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	12.08838	182.38398	0.066	0.947	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	13.58781	182.38282	0.075	0.941	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	12.15689	182.38352	0.067	0.947	ns

Table S7H: Summary of GLM (Poisson distribution) examining the effect of treatment and plant type on abundance of non-native plants at T_2 (18 months post sowing, November 2024).

Variable	Estimate	Std. Error	z value	p-value	Significance
Intercept (Reference: Control treatment, Forb plant type)	3.4795	0.0414	84.087	$< 2^{-16}$	***
Solarisation for 6 weeks (vs Control treatment)	0.1163	0.0569	2.044	0.041	*
Solarisation for 12 weeks (vs Control treatment)	0.0170	0.0583	0.291	0.771	ns
Topsoil Removal of 7 cm (vs Control treatment)	0.3894	0.0536	7.265	3.74^{-13}	***
Topsoil Removal of 14 cm (vs Control treatment)	0.3539	0.0540	6.556	5.52^{-11}	***
Grass plant type (vs Forb plant type)	1.1568	0.0474	24.383	$< 2^{-16}$	***
Shrub plant type (vs Forb plant type)	-17.7821	182.3826	-0.097	0.922	ns
Solarisation for 6 weeks x Grass plant type (vs Control treatment)	-0.0218	0.0653	-0.334	0.739	ns
Solarisation for 12 weeks x Grass plant type (vs Control treatment)	-0.1006	0.0672	-1.497	0.134	ns
Topsoil Removal of 7 cm x Grass plant type (vs Control treatment)	-0.8361	0.0652	-12.820	$< 2^{-16}$	***
Topsoil Removal of 14 cm x Grass plant type (vs Control treatment)	-0.9443	0.0665	-14.196	$< 2^{-16}$	***
Solarisation for 6 weeks x Shrub plant type (vs Control treatment)	11.2960	182.3854	0.062	0.951	ns
Solarisation for 12 weeks x Shrub plant type (vs Control treatment)	12.0884	182.3840	0.066	0.947	ns
Topsoil Removal of 7 cm x Shrub plant type (vs Control treatment)	13.5878	182.3828	0.075	0.941	ns
Topsoil Removal of 14 cm x Shrub plant type (vs Control treatment)	12.1569	182.3835	0.067	0.947	ns

Chapter 4: Striking a balance between ecology and economics: a more holistic approach to optimising topsoil removal in grassland restoration

Appendix 1: Soil seedbank germination study

A1.1 Sampling protocol:

Sampling date and location

Sampling to determine the depth of viable soil weed seeds and bud bank propagules was conducted on 14/01/23 at 11 randomly selected locations across the project area. Sampling was performed at 2 cm intervals down to a depth of 20 cm. The precise sampling information, including location coordinates and elevation, is provided in Table 1 (main text).

Sampling procedure

- The sampling locations were recorded using a GPS device.
- At each location, a hole approximately 20 cm^3 was dug using a mattock.
- The face of the soil profile was cleaned by chipping a vertical flat plane to expose a smooth surface for sampling.
- The soil was then sampled at specified depth intervals (as outlined in Table 1) using a measuring tape and a stainless-steel paint scraper. Reusable equipment was manually cleaned between sampling locations to avoid cross-contamination.
- Each sample was placed in a new plastic zip-lock bag and labelled with the sample ID and corresponding depth interval.

Sample handling

Soil samples were collected in plastic zip-lock bags and air-dried indoors for 18 days. However, some samples were exposed to rain during transport to the University and became

wet, remaining wet for two days before the greenhouse trial began. It is considered unlikely that this would have affected results.



Fig. A1.1: Soil horizon sampling for greenhouse germination study.

A1.2 Greenhouse germination trials

Summer trial (02/02/23 – 21/02/23):

- **Setup:** 100 g of each (unground and un-sieved) soil sample was weighed and placed into 65 x 85 x 55 mm pots. The pots were arranged inside larger clear plastic containers with lids to simulate a micro-greenhouse environment (see Fig. A2.2 in Appendix A).
- **Location:** The containers were placed inside Greenhouse No. 3 at the Plant Research Centre, University of Adelaide, Waite Campus. Initially, the containers were positioned on the western side of the greenhouse, which received full sunlight, but after one week they were moved to the shaded eastern side.
- **Watering:** Each pot was initially watered with 25 mL of water to approximate field capacity. However, due to uneven water redistribution caused by condensation inside the containers and water loss through evaporation, pots were manually watered on Mondays, Wednesdays, and Fridays to maintain moisture levels close to field capacity.
- **Trial duration:** The trial was scheduled to run for four weeks but was terminated early after 19 days due to detection of mould on the soil surface of some pots. The trial began on 02/02/23 (day length: 13.48 hours) and ended on 21/02/23 (day length: 13.08 hours).



Fig. A1.2.1: Soil seed bank germination greenhouse set up, first day.



Fig A1.2.2: Soil seed bank germination after 19 days of the Summer trial. Credit: M McQuillan.

Winter trial (12/06/23 – 13/07/23):

- **Setup:** The same experimental setup and soil samples were used for the winter trial, which was conducted at the first author's residence in Belair, South Australia. The pots were placed in full sun for the duration of the trial.
- **Trial duration:** The winter trial lasted four weeks, from 12/06/23 (day length: 9.50 hours) to 13/07/23 (day length: 9.57 hours).

Seedling identification and counting

- Germinants were counted and differentiated as monocots or dicots based on morphological characteristics.
- The results of both the summer and winter germination trials are displayed in Fig 3A.

Appendix 2: topsoil removal cost model

A2.1 Invitation to quote (ITQ) process

The scope of work was sent to 10 local heavy earthmoving contractors to request quotes for topsoil removal at Glenthorne National Park. Four contractors responded with quotes. The ITQ included the following project details and requested cost estimates for several scenarios, as some of the project parameters were still subject to change:

- **Location:** Glenthorne National Park
- **Project area:** See attached map (high-resolution aerial photograph and soil scraping plan provided)
- Soil Type:
    - North of the dirt road (running east-west): Heavy black cracking clay, with no to minimal coarse fragments, to depths greater than 1 meter.
    - South of the dirt road (running east-west): Brown clay, with shale fragments, to depths greater than 0.5 meters.
- **Access to site:** Access is provided via Majors Road onto Warhorse Track, which leads directly to the project area.
- **Soil handling:** Scraped soil is to be spread over the adjacent paddock.
- **Machinery storage:** Machinery can be stored on-site overnight. The project area is out of sight from major roads, and the gate at Majors Road is locked at 7 pm each night.
- **Safety supervision:** The work will be supervised by National Parks staff to ensure safety protocols are followed.

- **Topography:** Most of the project area is situated on a ridge or plateau, with generally flat topography.
- **Project timing:** The work must be completed by early May.

Cost scenarios requested:

Given uncertainties related to the depth of topsoil removal, which would be determined by ongoing soil seed bank germination trials, contractors were asked to provide quotes for the following scenarios:

- 2 hectares scraped to 10 cm
- 2 hectares scraped to 15 cm
- 3 hectares scraped to 10 cm
- 3 hectares scraped to 15 cm

Contractors were also informed that the most cost-effective method was desired. Similar jobs had successfully used a combination of a road scraper and an excavator, and contractors were encouraged to propose the most efficient and cost-effective approach.

Contractors were invited to schedule a site visit to view the area and clarify any project details, with assurances that additional information would be readily provided.

A2.2 Inflation adjustment:

To ensure accurate cost estimates, the quotes from January 2023 were adjusted for inflation using the Producer Price Index (PPI) for "3101 Road and Bridge Construction South Australia" published by the Australian Bureau of Statistics (2024). The formula used to calculate the inflation rate from Q1 2023 to Q3 2023 was:

$$Inflation\ rate\ = \frac{PPI\ in\ Q3\ 2023 - PPI\ in\ Q1\ 2023}{PPI\ in\ Q1\ 2023} \times 100$$

- PPI Q1 2023: 152.8
- PPI Q3 2023: 153.9
- Inflation rate: 0.72%

To adjust the original quotes from January 2023 to reflect their real value in October 2023, the following formula was applied:

$$Real\ Value\ in\ October\ 2023 = Original\ Quote\ from\ January\ 2023 \times \left(1 + \frac{Inflation\ rate}{100}\right)$$

This adjustment allowed for a comparison of quotes over different periods and ensured that cost estimations were current at the time of writing.

A2.3 Economic model development:

The economic model for the cost of topsoil removal and its effectiveness in seed bank removal was developed through a series of key calculations:

1. Germination data aggregation and cumulative calculation:

The total germination count was aggregated by depth intervals (e.g., 2 cm, 4 cm, 6 cm, etc.). For each depth, the cumulative sum of germinated seeds was calculated to understand the proportion of total seed bank germination at each depth. The percentage of total germination was then calculated relative to the maximum germination observed. This allowed for the determination of how much of the soil seed bank is removed at different depths, which is critical for assessing the efficacy of topsoil removal at specific depths.

2. Cost calculation for topsoil removal:

To develop the cost model, different scenarios of area (1 ha, 5 ha, and 10 ha) and topsoil depth (ranging from 2 cm to 20 cm) were considered. The upfront costs for each scenario were calculated based on the fixed mobilization and demobilization costs, as well as the variable

costs of scraping topsoil to different depths. These costs were adjusted for inflation and tailored to the project area and depth requirements.

3. Germination efficiency per dollar spent:

To evaluate the economic efficiency of seed bank removal, the percentage of germination removed was divided by the total cost for each scenario. This metric, "germination per cent per dollar spent," provided insight into how effectively each dollar contributed to removing the soil seed bank. This value was also inverted to calculate the cost per percentage of seed banks removed, offering a clear measure of cost-effectiveness across different depths and areas.

4. Modelling economic impact:

A linear regression model was developed to predict the economic efficiency of topsoil removal based on depth and area. This model helped to quantify how changes in these variables impact the cost-efficiency of seed bank removal. The model generated predictions that were used to assess different restoration scenarios and optimize the depth and area of topsoil removal for maximum seed bank reduction at the lowest cost.

5. Cost spent and cost per percentage germination:

To provide additional clarity on the costs, the economic model also calculated the cost spent on topsoil removal for each scenario. In parallel, the cost per percentage of seed bank germination removed per hectare was calculated. These metrics enabled the comparison of different scenarios (e.g., scraping 2 ha to 10 cm vs scraping 3 ha to 15 cm) to find the most cost-effective solution for the project’s goals.

6. Visual representation of economic models:

Visual graphs were created to illustrate the cost dynamics. One plot displayed the cost per $1000 for different area and depth scenarios, while another plot showed the cost per

percentage of germination removed per hectare. These visualizations helped to identify the scenarios with the highest cost-effectiveness and informed decision-making for the project.

A2.4 Software and data analysis:

All calculations and economic modelling were carried out using R (R Core Team 2023). The data was cleaned and aggregated, and the costs were calculated for each scenario. Regression models were applied to predict cost efficiency, and graphs were generated to represent the results.

Appendix 3: Soil classification and verification

A3.1 Soil pit excavation and classification

To verify and refine the soil types within the delineated landscape management units, a comprehensive soil classification was conducted on-site in December 2022. A soil pit measuring 2 x 5 meters (east-west by north-south) was excavated, and the following steps were taken:

1. Pit excavation:

   The soil pit was stepped down at 50 cm intervals to a total depth of 1 meter. Safety protocols were followed, including the construction of a ramp to prevent wildlife entrapment.

2. Soil classification protocols:

   The classification was conducted following the latest version (V3) of The Australian Soil Classification and the Australian Soil and Land Survey Field Handbook.

3. Soil pH and electrical conductivity (EC) measurements:

   Soil pH and electrical conductivity (EC) were measured using a 1:5 soil-to-deionised water solution. Soil samples were oven-dried at 40°C for 48 hours to prepare them for testing. All equipment was decontaminated between sampling locations to prevent cross-contamination.

4. Pit reinstatement:

   The soil pit was fully reinstated in January 2023, following the completion of soil sampling and classification.

Table A3.1: Soil classification data profile: Self Mulching Black Vertosol, Australian Soil Classification code: VE AE EI

Horizon		**A1**	**B21**	**B22**	**B23k**
Boundary		Clear	diffuse	diffuse	diffuse
Depth (cm)		0-10	10-50	50-90	90-100
Colour			black (10 YR 2/1)	black (10 YR 2/1)	
Mottles		-	-	-	few, 2-6mm, orange
Texture		Light Medium Clay	Medium Heavy Clay	Medium Heavy Clay	Medium Heavy Clay
Structure	*Grade*	Moderate	Strong	Strong	Strong
	Shape	Angular Blocky	Prismatic	Lenticular	Lenticular
	Size	3	5	3	2
Coarse Fragments		-	-	-	-
Segregations		-	-	-	20-50%, calcareous, soft, medium
Sample Intervals (cm)		0-10	10-20; 20-30; 30-40; 40-50	50-60; 60-70; 80-90	90-110

Table A3.2: Soil classification data profile: Bleached-Vertic Calcic Black Dermosol. Australian Soil Classification code: DE AE BD BB

Horizon		**A1**	**A2e/j**	**B21**	**B22**	**B23**	**B3k**	**C**
Boundary		-	-	-	-	-	-	-
Depth (cm)		0-10	10-25	25-40	40-50	50-65	65-75	75-100
Colour		10YR 3/2 M	10YR 3/1 M 10YR 7/1 D	10YR 2/1 M	2.5YR 3/2 M	2.5YR 3/1 M	7.5YR 7/4 M	Gley1 7/10X
Mottles		none	few, 2-6 mm, yellow	few, 6-20 mm, red	many, 20-60 mm, yellow	few, 2-6 mm, yellow	few, 6-20 mm, red	common, 2-6 mm, yellow
Texture		Medium Heavy Clay	Medium Clay	Medium Heavy Clay	Medium Heavy Clay	Medium Heavy Clay	Medium Clay	Medium Clay
Structure	*Grade*	-	-	Strong	Strong	Strong	Strong	-
	Shape	-	-	Lenticular	Lenticular	Lenticular	Polyhedral	-
	Size	-	-	20-60 mm	20-60 mm	20-60 mm	20-60 mm	-
Coarse Fragments		none	None	None	none	none	none	none

Horizon	**A1**	**A2e/j**	**B21**	**B22**	**B23**	**B3k**	**C**
Segregations	none	None	None	none	none	none	10-20 %, calcareous, soft, medium
sample intervals (cm)	0-10	10-25	20-30; 30-40;	40-50	50-60	65-75	80-90

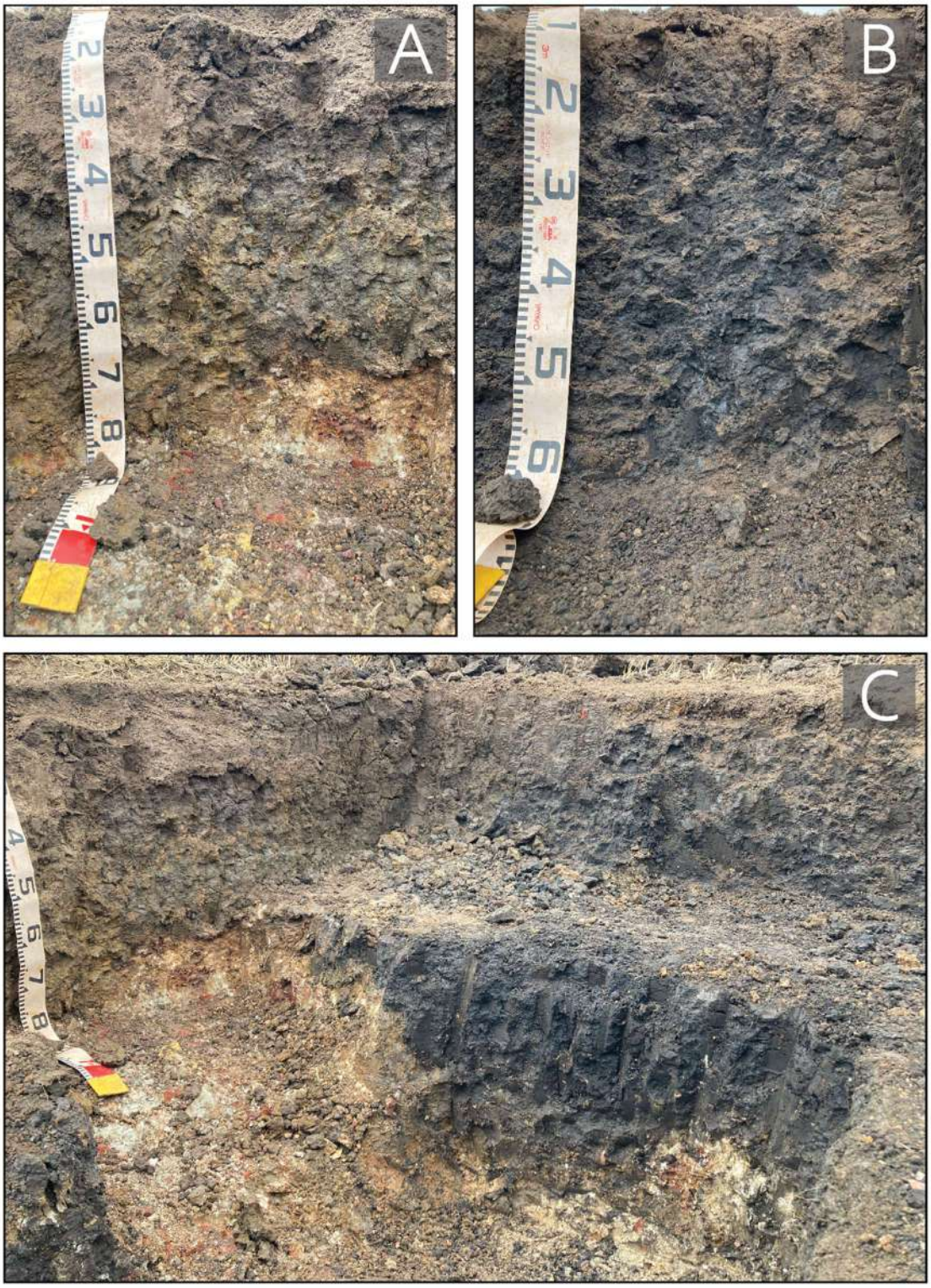

Fig A1.2: Soil Profile Characteristics: (A) Soil profile for a Bleached-Vertic Calcic Black Dermosol. This Dermosol soil order, as defined by the Australian Soil Classification (Isbell 2021), is characterised by a structured B2 horizon without a texture contrast. (B) Soil profile for a Self-Mulching Black Vertosol. The Vertosol soil order, according to the Australian Soil Classification (Isbell 2021), exhibits characteristics such as a moderate to heavy clay texture (>= 35%) throughout the profile, deep cracks wider than 5mm when dry, and the presence of slickensides and/or lenticular peds. (C) An image taken from further back to display both soil profiles side by side. Despite their proximity within one metre of each other, these profiles are classified as different soil orders.

Table S1. Summary of Modelled Phenological Metrics and Summary Statistics. This table presents the summary statistics for the Peak of Season and the Length of Season across management areas.

Phenological Metric	Summary Statistic	Control	Remnant	Restored
Peak of season (EVI2 value)	Min	0.41	0.29	0.28
	Median	0.59	0.33	0.36
	Mean	0.58	0.33	0.37
	SD	±0.086	±0.028	±0.071
	Max	0.73	0.38	0.50
Length of season (days)	Min	131	91	124
	Median	141	125	167
	Mean	150	131	176
	SD	±18	±29	±37
	Max	182	182	261

Chapter 7: Glenthorne Community Grassland Management Plan

This management plan was developed alongside doctoral research at the University of Adelaide and reflects applied interpretations of that research. It is intended as a guiding document for on-ground management rather than as a research output.



GLENTHORNE COMMUNITY GRASSLAND MANAGEMENT PLAN

National Parks and Wildlife Service South Australia

GREEN ADELAIDE

Seeding Natives Incorporated

THE UNIVERSITY of ADELAIDE

Updated: August 2025

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SECTION 1: SITE DETAILS

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PLAN APPROVAL

This plan remains in draft form to allow further input from key stakeholders and final approval by designated authorities.

Senior Ranger name	Signature	Date
Reece Boulden		
FoG President name	**Signature**	**Date**
Martin Schumacher		

SECTION 2: GLENTHORNE COMMUNITY GRASSLAND OVERVIEW

The Glenthorne Community Grassland, located within Glenthorne National Park | Ityamaiitpinna Yarta, is a reconstructed grassland that reflects the region's natural heritage, cultural importance, and historical significance. As part of the nationally threatened Grey Box (*Eucalyptus microcarpa*) Grassy Woodlands and Derived Native Grasslands ecological community, it represents an opportunity to preserve and restore one of Australia's most endangered ecosystems.

ECOLOGICAL AND HISTORICAL SIGNIFICANCE

Historical evidence indicates that, prior to European settlement, the Glenthorne Community Grassland, along with much of Adelaide, was managed as grassland ecosystems by the Kaurna people. The Native Temperate Grasslands of southern Australia are a nationally threatened ecological community, which today covers less than 1% of its pre-European range. Their decline is directly linked to the loss of plants and animals that specialise in this habitat. Most grasslands are cultural landscapes that rely on periodic disturbance (historically in the form of fire) to maintain a heterogeneous and biodiverse assemblage of plants and animals.

Remnant native grasslands can support exceptionally high levels of biodiversity, with over 100 species per hectare recorded in surveys. These ecosystems also hold significant cultural value for both Aboriginal people and European settlers. The Kaurna maintained grasslands across most of Adelaide to sustain their >65,000-year-long sustainable way of life. For European settlers, grasslands provided critical grazing resources in the early colonial period. The rapid shift in grassland management, coupled with a lack of understanding of these ecosystems, led to their near-total decline since 1836. Restoration initiatives like the Glenthorne Community Grassland are essential to preserving the biodiversity and cultural history embedded in these living landscapes.

CURRENT CONDITIONS AND CHALLENGES

Key challenges for restoration include:

- **Soil degradation and nutrient enrichment:** Favouring weed growth over native plant establishment.
- **Weed pressure:** From existing non-native plants and a persistent seed bank.
- **Land-use changes:** To agriculture and urban development; reducing connectivity and natural dispersal.
- **Limited remnant vegetation:** Resulting in reduced biodiversity and ecosystem functionality.

CONSERVATION PRIORITIES

The Glenthorne Community Grassland restoration efforts aim to:

- **Increase native vegetation cover** to outcompete weeds and establish a resilient seed bank.
- **Establish and maintain high biodiversity** areas of Grey Box Derived Grasslands.

GUIDING PRINCIPLES FOR RESTORATION

Restoration efforts adhere to sustainable Bushcare and Society for Ecological Restoration principles:

- **Adaptive Management:** Monitoring and refining restoration techniques.
- **Work Outwards:** Strengthen existing ecosystems by working from areas of healthy native vegetation.
- **Minimise Soil Disturbance:** To reduce the risk of weed resurgence.

RESTORATION AND EDUCATION INITIATIVES

Funded primarily by a Green Adelaide Grassroots Grant, with additional support from the City of Marion, the collaborative Glenthorne Community Grassland restoration project ran from late 2022 to early 2025. It involved key stakeholders, including the rangers at Glenthorne National Park, the Friends of Glenthorne, Seeding Natives SA, the University of Adelaide, Flinders University, and Nature Play SA. The project had three main aims:

1. **Ecological Restoration**: To recreate five hectares of high-biodiversity Grey Box Derived Grasslands using direct seeding methods.
2. **Grassland Restoration and Design Research**: Max McQuillan's PhD research focused on the restoration of the Glenthorne Community Grassland.
3. **Educational Outreach**: To provide a learning resource for STEM courses and creative content for broader educational opportunities.

OUR ROLE IN GRASSLAND CONSERVATION

Grasslands have always been cultural landscapes in Adelaide and the Mount Lofty Ranges; without human intervention, they transition to alternative ecosystems. The species that depend on grasslands require active vegetation management to thrive. The success of this restoration project depends on the collective efforts of the community. By supporting and participating in this initiative, we can help safeguard these remarkable ecosystems for future generations.

SECTION 3: GRASSLAND SEASONALITY PHENOLOGY

Phenology refers to the timing of natural events like plant growth, which in Southern Australia's Mediterranean climate is shaped by environmental factors. Grassland growth in this region depends on rainfall, temperature, and soil moisture, rather than fixed seasonal patterns. Figure 1 shows data collected from the Glenthorne Community Grassland (May 2023 to December 2024), illustrating how these factors influence grassland cycles. It also includes the Enhanced Vegetation Index (EVI2), which tracks plant greenness and leaf growth, providing a clear picture of seasonal phenology.

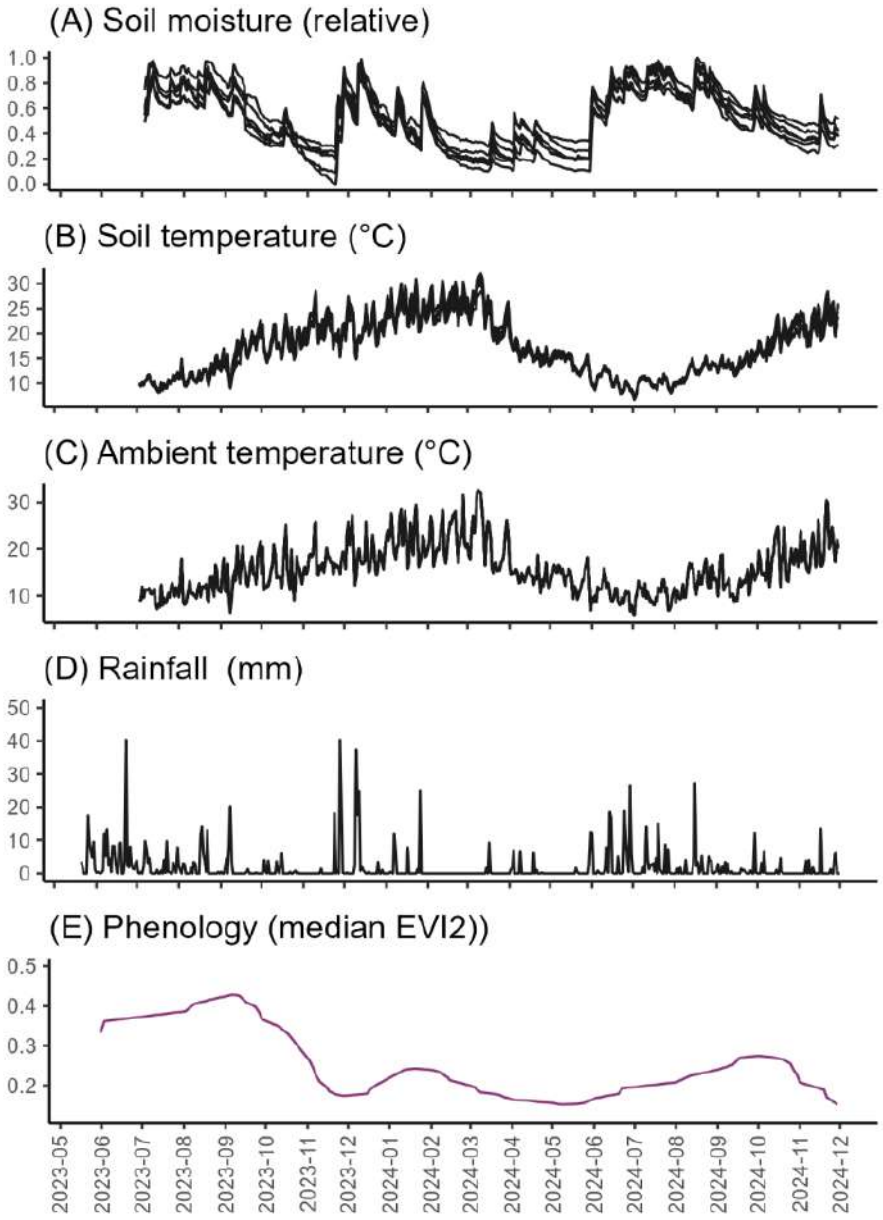


Figure 1: Environmental drivers of seasonal phenology in the grassland at Sam's Patch (Phase 1 of the restoration project – see Appendix A1). (A) Soil moisture increases sharply after rainfall events (D) but declines during dry periods, influencing plant activity. (B) Soil temperature and (C) ambient temperature exhibit clear seasonal trends, with warmer conditions during summer. High temperatures combined with low soil moisture restrict growth during summer, while cooler, wetter months drive plant activity, which peaks around September/October. (E) Phenology, represented by median Enhanced Vegetation Index 2 (EVI2– derived from Sentinel 2 Satellite imagery), reflects plant greenness and leaf growth, capturing the seasonal changes in grassland activity in response to environmental cues (higher numbers represent more leaf growth / greenness).

DYNAMIC SEASONAL PHENOLOGY OF GRASSLANDS

DRY DORMANCY (LATE SUMMER: JANUARY – MARCH)

During the hottest months, the grassland ecosystem enters a state of dormancy, with the landscape shifting from golden to rich golden-brown hues, except for C4 summer active grass species:

- **Dormancy:** Perennial species retreat to dormant buds, conserving resources to survive the intense heat and lack of soil moisture.
- **Seed dispersal:** Many species complete their reproductive cycles before this period, dispersing seeds to prepare for germination when conditions improve.

BREAK OF DROUGHT (AUTUMN: MARCH – MAY)

With the arrival of cooler temperatures and the first rains of the season, grasslands transition from dormancy to active growth, transforming the landscape from golden-brown to vibrant green:

- **Seed germination:** Rainfall triggers germination in many native species, including annual forbs and C3 grasses.
- **Initial growth:** Perennial species regenerate from buds, while soil moisture supports early root and shoot development.

COOL GROWTH PHASE (WINTER: JUNE – AUGUST)

Winter provides optimal growing conditions for many grassland species, particularly C3 grasses. The landscape remains lush and green during this period with many C4 grasses entering a version of dormancy:

- **Active growth:** Cool temperatures and ample soil moisture support vigorous growth in native species.
- **Nutrient storage:** Plants build reserves that will fuel flowering and seed production in the coming months.

SPRING FLOURISH (SPRING: SEPTEMBER – NOVEMBER)

As temperatures rise and day lengths increase, grasslands enter their most visually dynamic phase. The landscape is painted with dots of colours: pink, red, yellow, blue, and white:

- **Flowering:** Forbs bloom in early spring, providing critical resources for pollinators, followed by the flowering of native grasses later in the season.
- **Peak growth:** Native plants reach their maximum height and biomass, contributing to ecosystem services such as carbon sequestration and soil stabilisation.

SUMMER TRANSITION (EARLY SUMMER: NOVEMBER – JANUARY)

As soil moisture declines, grassland activity gradually slows, and the landscape takes on a golden appearance:

- **Seed maturation:** Grasses and forbs complete their reproductive cycles, producing seeds that are dispersed before the onset of the dry dormancy period.
- **Preparation for dormancy:** Perennial species shift resources to root systems and buds to survive the approaching dry months.

ECOLOGICAL GROWTH PATTERNS IN NATIVE GRASSLANDS

Native temperate grasslands in Southern Australia are dominated by perennial grasses and forbs, with a smaller subset of annual forbs. These ecosystems are adapted to the Mediterranean climate, with alternating periods of growth and dormancy that reflect environmental conditions rather than rigid seasonal boundaries.

- **C3 Species:** These plants dominate during the cooler months, with active growth and flowering occurring from autumn through spring. They thrive in lower temperatures and take advantage of winter's soil moisture.
- **C4 Species:** Adapted to warmer conditions, these species peak in activity during late spring and early summer when temperatures are higher. Their growth slows significantly during the cooler winter months.

Table 1 shows the interplay between C3 and C4 species which creates a year-round cycle of activity, ensuring that the ecosystem remains productive and resilient across varying conditions.

ECOLOGICAL PHENOLOGY CALENDAR

Table 1: This table outlines the seasonal phenology of the Glenthorne Community Grassland, indicating periods of dormancy, seed dispersal, germination, growth (C3 and C4 species), and flowering (forbs and grasses) across the months of the year. "X" marks the active periods for each stage.

Period	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Dormancy	X	X	X									X
Seed dispersal	X	X	X								X	X
Germination			X	X								
Growth (C3 species)			X	X	X	X	X	X	X			
Growth (C4 species)	X	X						X	X	X	X	X
Flowering (forbs)	X								X	X	X	X
Flowering (grasses)	X	X								X	X	X

Understanding these phenological patterns and the dynamic seasonality of grasslands has critical implications for management. Effective timing of interventions such as weed control, planting, and prescribed burns can align with natural cycles, enhancing the resilience and biodiversity of these vital ecosystems.

SECTION 4: WORK TO BE UNDERTAKEN

The principle of **adaptive management**[1] is critical for effectively managing the Glenthorne Community Grassland. This approach enables managers to respond to uncertainties and adapt strategies as new information becomes available.

The management framework outlined in this document provides a comprehensive overview of actions and specifies when and how to carry them out. In addition to targeted actions informed by the principle of **adaptive management**, annual tasks are essential to maintaining the grassland's health and preventing weed encroachment, particularly along its edges.

By following an adaptive management approach, the grassland can be maintained in good condition, addressing challenges like weed encroachment and seasonal variability. Regular monitoring and adjustments will ensure the effectiveness of these actions and improve the long-term health of the ecosystem.

RELEVANT POLICIES AND PROCEDURES

All activities undertaken in the Glenthorne Community Grassland must adhere to:

- **Volunteer Safety Framework (VSF):** Ensuring safe working practices.
- **Volunteer Code of Conduct:** Outlining ethical and behavioural expectations.
- **Planting Indigenous Species Policy:** Promoting the use of native flora.
- **Bushcare Hazard ID Checklist:** Identifying and mitigating hazards in the field.
- **Safe Work Practices (SWPs):** Detailed guidance on tasks such as chemical handling, weeding, and planting.

Refer to the **Glenthorne National Park – Ityamaiitpinna Yarta Volunteer Work Plan (GNP-IP Volunteer Work Plan)** for detailed forms, templates, and standard operating procedures.

[1] What is Adaptive Management?
Adaptive management is a flexible, step-by-step approach to managing natural environments. It's based on the idea that we don't always know exactly how ecosystems will respond to our actions, so we act, learn from the outcomes, and improve over time.
Here's how it works:

1. Plan: Create a plan using the best available information and set clear goals.
2. Act: Implement management actions, like mowing or spraying weeds.
3. Monitor: Observe the outcomes – What worked? What didn't?
4. Learn: Use this knowledge to understand the ecosystem better.
5. Adjust: Modify the plan to improve results in the future.

This cycle helps managers make decisions even when faced with uncertainties, ensuring continuous improvement in grassland management.

MANAGEMENT AREAS

The Glenthorne Community Grassland is divided into four management areas, each grouped according to their similar management strategies and structural characteristics. These management areas are visually represented in Appendix A2 – Management Areas.

AREA 1: WILDFLOWER MEADOW

- **Focus:** Supporting a high wildflower (forbs) to grasses ratio on red-brown clay soils[2] (see Appendix A4 - Soil Map).
- **Activities:** Precise weed control, eliminating target weeds, and monitoring biodiversity.

AREA 2: PERENNIAL GRASSLAND

- **Focus:** Black cracking clay soils seeded with a grass only mix of native C3 and C4 grasses.
- **Activities:** Broad acre weed control using selective herbicides and occasional weed control mowing.

AREA 3: INTERPRETATION AREA

- **Features:** Paths, seating, educational opportunities and interpretive signage.
- **Activities:** Maintenance of paths, planting of grasses, chenopods, and tube stocks to maintain the amenity.

AREA 4A SOLARISED PLOTS AND AREA 4B FIELD EXPERIMENT

- **Features:** Paths, plots of direct seeded grassy communities.
- **Activities:** Maintenance of paths, Precise weed control, eliminating target weeds, and monitoring biodiversity.

AREA 5: BUFFER ZONE

Biological Buffer (Area 5A)

- **Features:** Mixed planting of shrubs and small trees to create a buffer against windblown seeds.
- **Activities:** Substitute tube stock planting, weed control, remove any tall trees that may grow under powerlines, and monitoring buffer effectiveness.

Mown Buffer (Area 5B)

- **Features:** Mix of native grasses and non-native plants.
- **Activities:** Bi-annual slashing and annual broadleaf selective herbicide; optional preemergent herbicide once every 2-3 years to reduce annual weed competition.

Chemical Buffer (Area 5C)

- **Focus:** Zone of no plant growth maintained through annual herbicide application (pre-emergent AND/OR Glyphosate).
- **Activities:** Spraying pre-emergent and/or glyphosate before seed set.

[2] At Glenthorne, the red-brown clay soils foster richer plant diversity than the black cracking clay soils. This finding emerged from a randomised, controlled ecological experiment—led by Max—consisting of 36 individual 5x5 m research plots. Drawing on these site-specific insights, we refined our Stage 3 approach (see Appendix A1 for restoration phases): on red-brown soils, we applied a 50:1 forb-to-grass mix to capitalize on their higher biodiversity potential, while on black cracking clay soils, we planted only grasses. This targeted strategy simplifies weed management, as it allowed for the use of broadleaf-selective herbicides on the black clay. See A4 for a soil map of the grassland.

OVERVIEW OF ANNUAL ACTIONS:

BIANNUAL MOWING OF MOWN BUFFERS

Area 4B

- Mowing is crucial for reducing annual weed seed loads over the long term.
- The first mow should occur around the peak of the growing season in August–September.
- A follow-up mow should be conducted 4–6 weeks later to target regrowth.

BIANNUAL GLYPHOSATE SPRAYING OF CHEMICAL BUFFERS

Area 4C

- Persistent spraying of the chemical buffers is crucial for reducing annual weed seed loads over the long term – and should be kept up for at least 6 years (until 2030).
- The spray should occur before the peak of the growing season (around June – July) when weeds are small.
- a follow up spray should be conducted around the peak of the growing season around August–September.

HERBICIDE DABBING AND BRUSHING

Area 1,2,3 and 4A

- Herbicide use is focused on managing high-priority weeds (see the Priority Weeds section for details).
- Techniques include dabbing, weed brushing, and spot spraying to minimize harm to native vegetation.

PRIORITY WEEDS

The following weeds have been identified as being problematic during establishment; it is likely that the weeds that are problematic post establishment will differ slightly or substantially. The intent is not to remove every weed from the Grassland, rather to control the most problematic weeds and maintain good native cover to suppress windblown weeds. Table 2 outlines management actions for high and medium priority weeds at the Glenthorne community grassland. *Note – For a complete list of native and non-native plants see the Field Guides in Appendix D*.

Table 2: Priority Weed Species, Associated Control Methods, and Management Considerations. ***Photos taken by Max or from Wikipedia.***

Common Name	Species	Priority	Control Method	Photo
Variegated Thistle	*Silybum marianum*	High	Physical removal of seed heads/flowers followed by hand weeding for large individuals, or glyphosate dabbing/brushing for seedlings and smaller individuals	

Perennial Phalaris	*Phalaris aquatica*	High	Brush cutting and removal of seed heads/flowers followed by targeted glyphosate spraying once green blades reshoot	
Ox-tongue Thistle	*Helminthotheca echioides*	High	Physical removal of seed heads/flowers followed by hand weeding for large individuals, or glyphosate dabbing/brushing for seedlings and smaller individuals	
Stinkwort	*Dittrichia graveolens*	High	Hand weeding and/or targeted broadleaf-selective herbicide application	
Wild Oats	*Avena spp.*	Medium	Timed mowing and/or application of Raptor herbicide by contractors or rangers	
Mustard Weed	*Sisymbrium spp.*	Medium	Timed mowing and/or spot spraying (pre-emergent herbicide for severe cases)	

Annual Ryegrass	*Lolium rigidum*	Medium	Application of pre-emergent herbicide localized to outbreak areas	

ANNUAL WORK PLAN

This work plan outlines the key activities for the maintenance and restoration of the Glenthorne Community Grassland, based on the native grass maintenance schedule in Table 3. Tasks are timed to maximise ecological outcomes and align with grassland phenology, ensuring minimal disturbance to native species and the most efficient use of time, energy, and resources.

MAINTENANCE TASKS

BROADLEAF SELECTIVE HERBICIDE

- **Purpose**: Reduce competition from invasive broadleaf and bulbous weeds.
- **Details**: Spot-spray with metsulfuron-methyl (5–8g/100L water) and MCPA 750 (1-1.5L/100L water), adding a wetting agent and dye for accuracy (use higher rates for larger more established weeds). Ensure boom sprayers are calibrated for even application. Herbicide should be applied within the active growing season of the grassland. Triple rinse equipment after use to prevent residue contamination. Optimal period: May to October.

GLYPHOSATE HERBICIDE

- **Purpose**: Broad-spectrum control of persistent weeds, including grasses and broadleaf species.
- **Details**: Use a glyphosate-based herbicide (e.g., Roundup) at the recommended dilution for targeted weed control. Apply via spot-spraying, dabbing, or brushing to avoid damage to desirable plants. Glyphosate is most effective on actively growing weeds. Avoid application during drought, frost, or when weeds are stressed, as efficacy may be reduced. Ensure adequate drying time before rain to prevent runoff.

WEED DABBING AND BRUSHING

- **Purpose**: Target persistent weeds without disturbing the soil.
- **Details**: Use a high-concentration herbicide mix (20–50%; 1:5–1:2 ratio) for effective control. Apply directly to weed foliage using a brush or dabber to minimise herbicide drift. Collect ripe seed heads in a bag and dispose of them away from restoration areas. Optimal period: July to October.

TIMED MOWING OF ANNUAL WEEDS

- **Purpose**: Remove flower stems of annual weed grasses before they set seed (e.g., Wild Oat, Brome, Rye) while minimising damage to native grasses.
- **Details**: Adjust mowing height (>100mm) to protect native tussocks. Conduct multiple passes during the growing season. Timing is critical to intercept the main flush/elongation of weed flower stems, ideally before seed set. Remove cut material to reduce biomass. Optimal period: August to October.

MOWING FOR BIOMASS REDUCTION

- **Purpose**: Maintain grassland health by reducing accumulated biomass.
- **Details**: Conduct mowing every second year, timed after seed set of summer-active grasses such as *Dichanthium sericeum* (Silky Blue-grass) and *Themeda triandra* (Kangaroo Grass). Mow at a height of 100–150mm. This method can reinvigorate patches where burning is impractical. Remove mown material to avoid nutrient build-up. Optimal period: February to March (as needed).

Table 3: Seasonal activities and maintenance practices for managing the Glenthorne Community Grassland, including phenological stages, weed control, mowing, harvesting, and other interventions. Highlighted periods indicate optimal or acceptable timing for each activity. Refer to Figure 1 to see how seasonal triggers like temperature, rainfall and soil moisture interact with the seasonality of temperate grasslands.

Season	Jan		Feb		Mar		Apr		May		June		July		Aug		Sept		Oct		Nov		Dec	
Grassland Phenology																								
Broadleaf selective & glyphosate																								
Weed Dabbing / Brushing																								
Timed Mowing of Annual Weed																								
Mowing for biomass reduction																								
Harvest	C4	C4																			C3	C3	C3	C4
Burning									C3	C3									C4	C4				

Ideal time for maintenance

Acceptable time for maintenance

Not annually, but as required

Harvesting Native Grasses

- Purpose: Collect seeds for storage and propagation.
- Details: Harvest C3 (cool-season) species in November and C4 (warm-season) species in December to January. Use brush-cutters to detail isolated weeds and prevent seed contamination. Dry harvested material thoroughly in a well-ventilated area before storage. For mixed species, label seed lots clearly to maintain ecological integrity.

Burning

- **Purpose**: Stimulate germination of native seeds and remove accumulated thatch.
- **Details**: Consult with the NPWS Fire Management Team to develop a controlled burn plan. Burns should be conducted under appropriate weather conditions. Monitor post-burn recovery to assess germination success (>60 plants per m^2 is desirable) and manage follow-up weed control. Optimal period: April to May, in consultation with fire management authorities.

Preparation for New Sowing - Solarising or Topsoil Removal

- **Purpose**: Prepare areas for grassland expansion or enhancement.
- **Details**: Solarising involves using plastic sheeting to capture spring soil moisture and peak summer heat, effectively suppressing weeds. For shorter durations, wet the soil to field capacity (i.e., the maximum amount of moisture the soil can hold before drainage) and maintain coverage for approximately five weeks (late December to mid-February). For natural soil moisture, maintain plastic sheeting from September to February. Topsoil removal should target a depth of 12cm to remove the weed seed bank while preserving soil structure.

Over-Sowing and Sowing

- **Purpose**: Improve native cover and species.
- **Details**: Engage contractors to over-sow using the aerovator method to patch low-density areas. Focus on introducing a diverse range of native species to improve resilience against weeds. Optimal period: April-May, aligning with favourable germination conditions and preventing equipment from becoming bogged in black clay soils.

Pre-Emergent Herbicide

- **Purpose:** Control weeds before germination.
- **Details:** Apply pre-emergent herbicide (e.g., Indaziflam) between March 15th and May 15th. Ensure 10mm of rainfall occurs within six weeks of application to activate the herbicide. Avoid late applications, as root pruning during summer can stress native plants. Follow all product label instructions and environmental guidelines.

Grass-selective Herbicide

- **Purpose**: Target invasive annual grasses and reduce weed competition in restoration areas.
- **Details**: Apply herbicide containing the active chemical imazamox (e.g., Raptor) as a post-emergent treatment during the active growth stage of weeds. Use a calibrated sprayer (boom or knapsack) to ensure precise coverage, incorporating a non-ionic surfactant for increased efficacy. Apply between August and October to coincide with peak weed growth.

IMPLEMENTATION NOTES

- **Herbicide Application Timing**: Ensure applications align with active growth periods to maximise efficacy. Use weather forecasts to plan applications and avoid rain or wind.
- **Mowing Adjustments**: Timing and height must consider seasonal growth patterns and species phenology. Good timing significantly reduces workload and enhances effectiveness.

- **Burning Approvals**: Engage NPWS Fire Management Team for planning and approvals. Follow all safety protocols and monitor ecological outcomes.
- **Grassland Expansion**: Solarising and topsoil removal should be used strategically for grassland expansion, with minimal disturbance to existing vegetation. For small areas (e.g., 5 x 5m) of solarisation where wetting the soil to field capacity is feasible, a shorter solarisation period (~5 weeks between late December and mid-February) is possible. The longer timeframe from September to February is to capture natural soil moisture from early spring. Topsoil removal target depth should be between 7cm and 14cm. Small areas may be achieved with a tractor or bobcat, while large areas will require significant funding and project management. Solarised areas should be seeded with a high percentage of grasses and chenopods for best results.
- **Weed Disposal**: Collect and safely dispose of weeds removed during any activity to prevent re-seeding or reinfestation.
- **Glyphosate Herbicide use**: Triple rinse equipment after use to prevent cross-contamination. Always adhere to safety and environmental guidelines.

VEGETATION SURVEY

Vegetation surveys are optional but strongly encouraged. The Glenthorne Community Grassland website features video guidance on conducting rapid surveys in a systematic manner. Two primary methods are recommended:

1. **1m^2 Quadrats:**
   Place a 1m^2 quadrat at random locations within the grassland. Identify and count both native and non-native plant species to estimate species abundance and richness. See Appendix C1 for data structure.

2. **Bamboo Pole (Point Intercept Method):**
   Use a bamboo pole marked at 10cm intervals. Insert it randomly across the grassland and record which plant species touch each mark. This approach helps estimate overall species cover. See Appendix C2 for data structure.

These methods, used together, provide a comprehensive snapshot of vegetation composition and structure, informing management and restoration efforts.

Data can be sent to Max McQuillan for analysis and visualisation via email: maximilian.mcquillan@gmail.com.

This annual work plan provides a structured approach to maintaining the Glenthorne Community Grassland, ensuring long-term ecological health and sustainability. For more detailed procedural guidance, refer to the GNP-IP Volunteer Work Plan.

SECTION 5: STAKEHOLDER RESPONSIBILITIES

KEY STAKEHOLDERS AND THEIR RESPONSIBILITIES

The long-term maintenance of the grassland hinges on effective collaboration between the rangers at Glenthorne National Park and the Friends of Glenthorne National Park, working together in accordance with the Management Plan. The rangers are responsible for coordinating contractor activities and ensuring that all on-ground work aligns with the plan's objectives. Meanwhile, the Friends group supports these efforts by undertaking volunteer work and helping to secure funding from sources such as the Friends of Parks Grant and the City of Marion's conservation dividend.

Should financial or staffing challenges arise, these stakeholders (along with representatives from Seeding Natives) should convene to devise a sustainable strategy that upholds the ecological integrity of the grassland. **Appendix A3** visually illustrates stakeholder responsibilities across different zones of the grassland.

SHARED/COLABORATIVE - DEW RANGERS / FRIENDS

1. **Area 1 (High Biodiversity):**
   Focus on controlling priority weeds to safeguard native forbs and grasses growing in red-brown clay soils (refer to Appendix A4 – Soil Map). Volunteers should also carry out biodiversity monitoring to maintain ecological integrity.

2. **Area 4a (Solarised Plots) & Area 4b (Field Experiment):**
   Maintain pathways (via spraying with herbicide or mowing). This limits the weeds from spreading beyond the pathways and maintains the visual amenity.

FRIENDS OF GLENTHORNE VOLUNTEERS

1. **Area 3 (Interpretation):**
   Enhance the visitor experience around trails, seating areas, and interpretive signage by managing weeds and introducing appropriate native species to maintain the visual amenity.

2. **Area 5a (Biological Buffers):**
   Focus on controlling priority weeds to help the shrub and tree species that were sown establish. Supplementary tube stock planting with species used in the buffer seed mix (Appendix B2 – Direct Seeding Lists 2024) may help establish the buffer faster.

3. **All Areas:**
   Remove resistant high priority weeds (e.g., *Phalaris aquatica*) using Bushcare methods.

PARKS (DEW RANGERS) / CONTRACTORS

1. **Area 2 (Grass-Only):**
   Utilise contractors with specialised expertise for selective interventions such as broadleaf selective herbicide spraying. Funding for these services may be sourced through joint applications by the Friends of Glenthorne and Parks staff to grants such as the Friends of Parks Grant or the Conservation Dividend program – see Section 6: Grant Opportunities.

In addition, undertake harvesting from established areas and over sowing in bare patches or poorly established areas.

2. **Areas 5b (Mown Buffer) and 5c (Chemical Buffer):**
   Maintain these buffer zones through mowing (Area 5b) or, chemical weed control (Area 5c). Consider engaging contractors for these tasks to ensure effective edge management and limit the spread of windblown seeds. These buffers help define and protect core ecological areas.

3. **All areas / as required:**
   Coordinate and undertake vegetation surveys in collaboration with volunteer rangers, researchers, students, and restoration ecologists.

SECTION 6: GRANT OPORTUNITIES

Securing ongoing funding is essential to sustain the long-term management and restoration of the Glenthorne Community Grassland. Several grant programs are available to support both contractors and volunteer-led initiatives, particularly those that align with biodiversity conservation, community engagement, and ecological restoration.

Key opportunities include:

- **Friends of Parks Small Grants Program (DEW)**
  Provides annual funding to support on-ground conservation projects led by volunteer groups. Funding may be used for weed control, seed collection and/or biodiversity monitoring.

- **City of Marion – Conservation Dividend**
  A local government initiative that offers funding for projects improving biodiversity within the City of Marion. This can be directed towards contractor support for grassland management.

- **Green Adelaide Grassroots Grants**
  Competitive annual grants aimed at community groups and organisations restoring biodiversity and habitat in the Adelaide region. These grants can support large-scale interventions such as solarisation, topsoil removal, and sowing, as well as education and monitoring activities.

- **National Landcare Program & State NRM Support**
  From time to time, broader funding opportunities become available under national or state-level Landcare or NRM frameworks. These can support collaborative projects that demonstrate innovation in grassland restoration, resilience to climate change, and partnerships with schools or research groups.

Timing and alignment:
Most grants open once per year, typically between April and September. Applications should be tailored to the annual work plan (Section 4), highlighting how specific activities—such as pre-emergent herbicide application, sowing, or weed brushing—can be supported by external

funding. Joint applications by Friends of Glenthorne and Parks staff are encouraged to strengthen proposals and ensure alignment with the Management Plan.

To assist with planning, Table 4 identifies the seasonal activities most suitable for contractor delivery and potential grant funding, providing a practical guide for aligning funding applications with ecological management needs.

Table 4 outlines the key grassland activities most suited to contractor delivery and grant funding, providing a practical guide for aligning funding applications with seasonal management needs.

Season	Jan		Feb		Mar		Apr		May		June		July		Aug		Sept		Oct		Nov		Dec	
Solarising																								
Slashing for biomass reduction																								
Topsoil Removal																								
Over sow / sowing																								
Weed Dabbing / Brushing																								
Pre-emergent herbicide																								
Broadleaf selective & glyphosate																								
Annual grass herbicide																								
Timed Mowing of Annual Weed																								

SECTION 7: RISK MANAGEMENT

The safety of all participants in the Glenthorne Community Grassland restoration project is a top priority. Below are the key safety considerations for all activities. This document should be read and interpreted in conjunction with the **Glenthorne National Park – Ityamaiitpinna Yarta (GNP-IP) Volunteer Work Plan** ("the GNP-IP Volunteer Work Plan") for detailed procedures, forms, and comprehensive guidance.

KEY SAFETY CONSIDERATIONS

1. Adherence to Policies:
    - Follow the Volunteer Safety Framework (VSF), Volunteer Code of Conduct, and all relevant DEW policies, including planting and native plant material collection guidelines.
2. Personal Protective Equipment (PPE):
    - Volunteers must wear sturdy footwear, gloves, long-sleeved shirts and pants, hats, and sunscreen. Task-specific PPE such as eye and ear protection must be used as required.
3. Environmental Awareness:
    - Work must cease on days of "Extreme" or higher fire danger.
    - Plan for weather conditions, including heat, cold, and UV exposure, and adjust tasks accordingly.
4. Risk Assessments and Pre-Start Checks:
    - Conduct a site hazard assessment before starting any activity.
    - Inspect all tools and equipment to ensure they are safe and operational.

5. Wildlife and Hazard Awareness:
    - Be vigilant for snakes, spiders, and other wildlife. Follow safety practices for handling bites and stings.
6. First Aid Preparedness:
    - Ensure first aid kits are available on-site, and team members with allergies carry appropriate medications.
    - Key personnel must hold current first aid certifications.

REFER TO THE GNP-IP VOLUNTEER WORK PLAN

For detailed procedures, including:

- Safe Work Practices (SWPs) for specific tasks such as chemical handling, brush cutter use, and seed collection.
- Forms and templates for risk assessments and safety checklists.
- Operational guidance for specialised equipment and emergency responses.

Refer to the GNP-IP Volunteer Work Plan to ensure compliance and safety across all activities.

APPENDICES

APPENDIX A: MAPS AND SITE PLANS

Appendix A1 – Establishment Phases

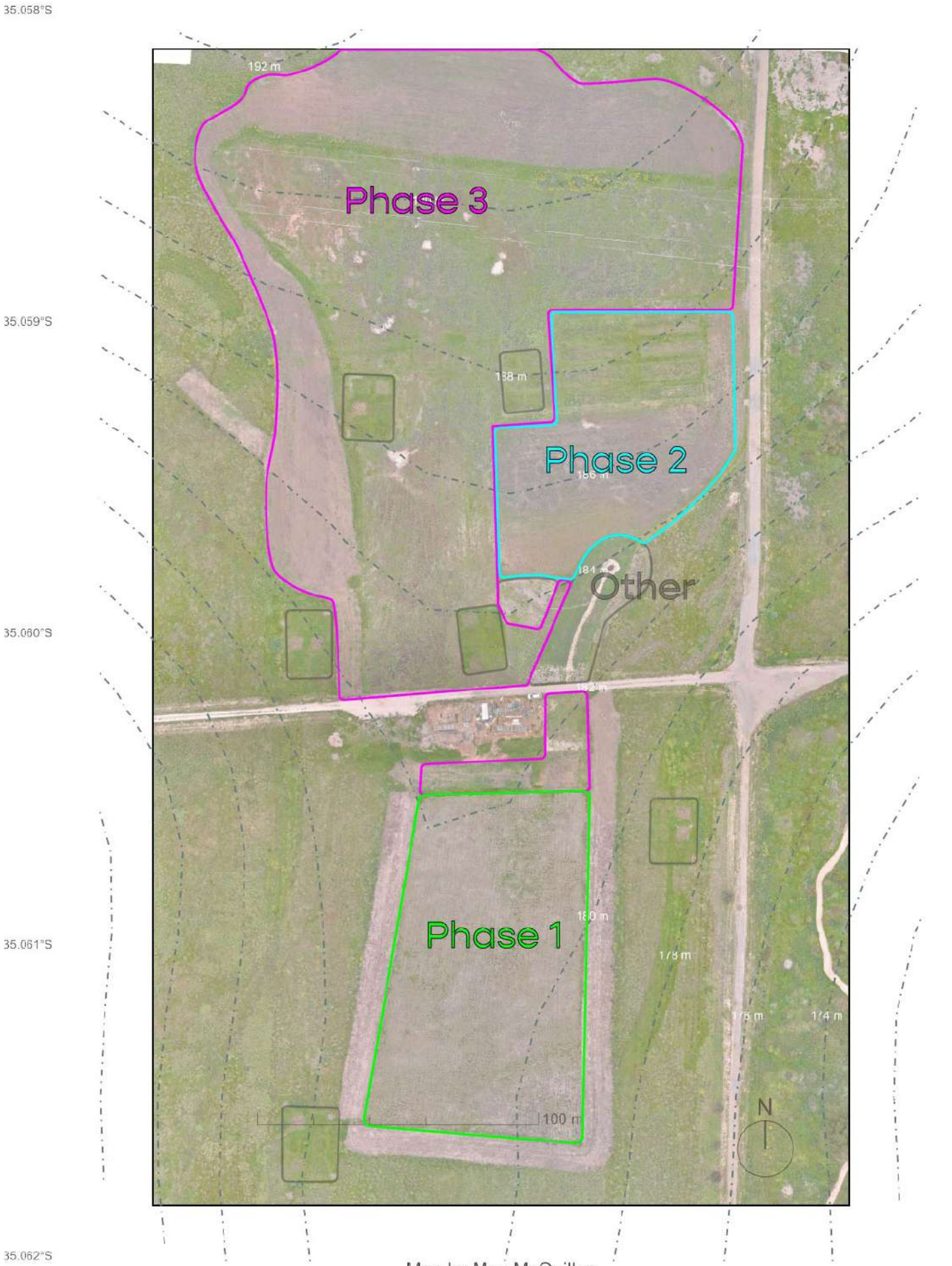
Appendix A2 – Management Areas

Appendix A3 – Stakeholder Map

Appendix A4 – Soil Map

Appendix A5 – Planning Map

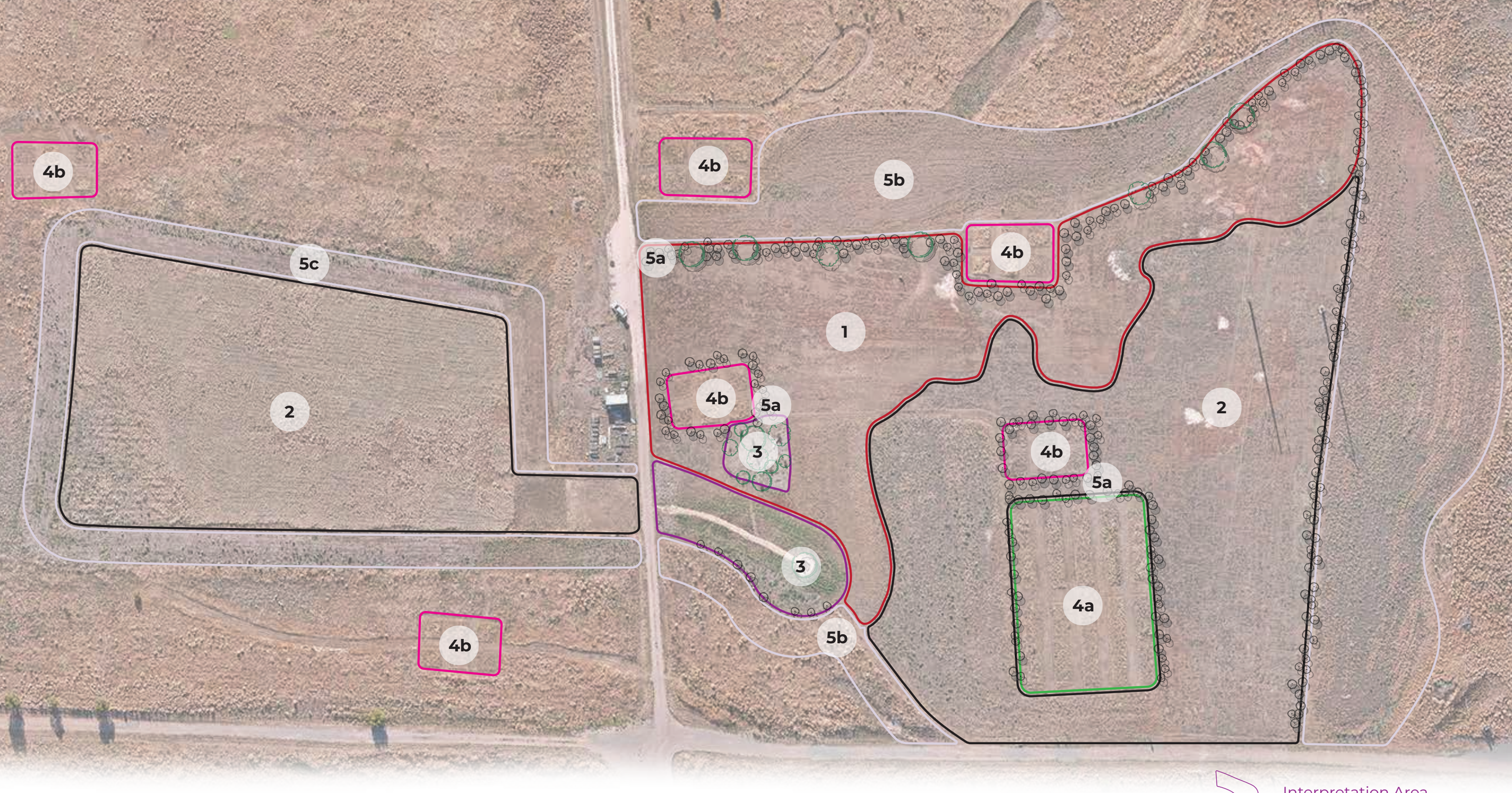
Appendix A6 – Concept Plan



Map by Max McQuillan

Image Date: 06/10/2024 | Mapping Date: 15/10/2024 | Resolution: 3.91 cm/pixel

138.5475°E 138.5480°E 138.5485°E 138.5490°E 138.5495°E 138.5500°E 138.5505°E



N 0 25 50 m

A2 - Management Areas

Interpretation Area

Buffer Zone

Wildflower Meadow

Perrennial Grassland

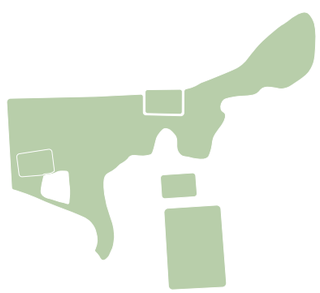
Field Experiment

Trees & Shrubs

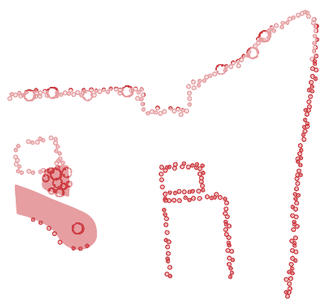
Solarised Plots

4b
5c
2
4b
4b
5b
5a
4b
1
4b
5a
3
2
4b
5a
3
4a
5b
4b

Shared



Friends



Parks



N 0 25 50 m

A3 - Stakeholder Map

Fill

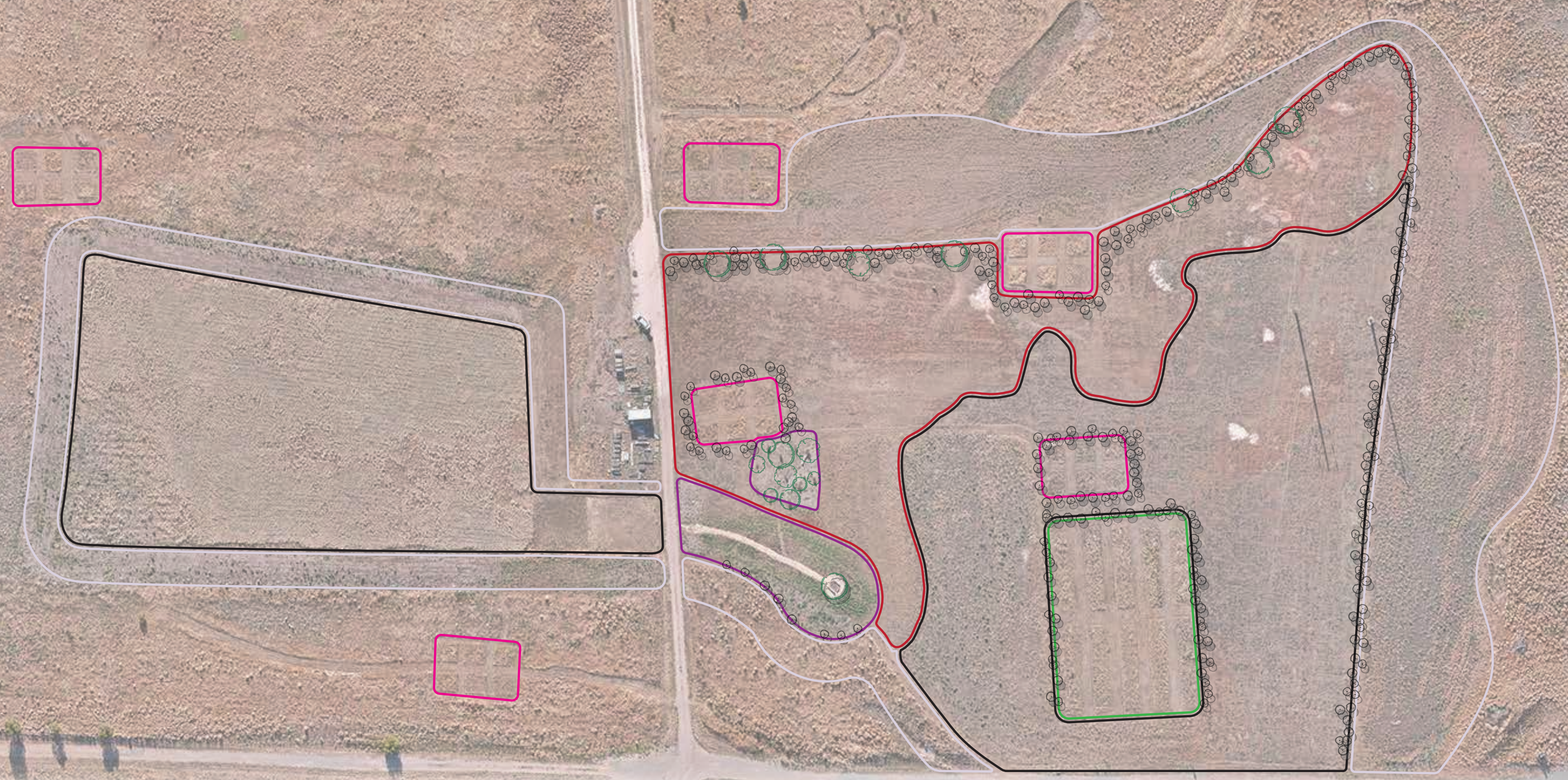
Red-brown Dermasol

Black Vertosol

N

0 25 50 m

A4 - Soil Map



N

0 25 50 m

A5 - Planning Map

Interpretation Area

Buffer Zone

Wildflower Meadow

Perrennial Grassland

Field Experiment

Solarised Plots

Trees & Shrubs



Map by Max McQuillan

Image Date: 03/12/2024 | Mapping Date: 12/12/2024 | Resolution: 15 cm/pixel

138.5475°E 138.5480°E 138.5485°E 138.5490°E 138.5495°E 138.5500°E

APPENDIX B: DIRECT SEEDING PLANT LISTS

Appendix B1: Direct Seeding Lists 2023

Appendix B2: Direct Seeding Lists 2024

2023 - Glenthorne Community Grassland
Grassland Seed Mix Lists

Experiment Mix		**Diverse Mix**		**Chenopod Mix**		**Grass only mix**	
Species	**Weight (g)**	**Species**	**Weight (g)**	**Species**	**Weight (g)**	**Species**	**Weight (g)**
Aceana echinata	20	*Aceana echinata*		*Anthosachne scabra*	500	*Anthosachne scabra*	500
Anthosachne scabra	200	*Aceana novae zelandiae*	100	*Aristida behriana*	200	*Aristida behriana*	200
Aristida behriana	200	*Anthosachne scabra*	50	*Atriplex semibacatta*	1000	*Austrostipa curticoma*	100
Arthropodium strictum	200	*Aristida behriana*	200	*Atriplex vesicaria*	200	*Austrostipa elegentissima*	100
Atriplex semibacatta	50	*Arthropodium strictum*	200	*Austrostipa scabra*	100	*Austrostipa nodosa*	200
Austrostipa curticoma	100	*Atriplex semibacatta*	50	*Chloris truncata*	300	*Austrostipa scabra*	500
Austrostipa elegentissima	50	*Austrostipa curticoma*	100	*Einadia nutans*	200	*Chloris truncata*	1000
Austrostipa scabra	50	*Austrostipa elegentissima*	100	*Enchylaena tomentosa*	1500	*Cymbopogon ambiguus*	500
Bulbine bulbosa	50	*Austrostipa scabra*	100	*Enneapogon nigricans*	100	*Dichanthium sericeum*	1000
Burchardia umbellata	5	*Bulbine bolbosa*	50	*Maireana brevifolia*	100	*Enneapogon nigricans*	500
Callostemma purpureum	10	*Calocephalus citreus*	50	*Rhagodia spinecens*	100	*Microlaena stipoides*	300
Calocephalus citreus	700	*Chloris truncata*	1000	*Rhagodia parabolica*	100	*Poa labillardieri*	100
Chloris truncata	50	*Chrysocephalum apiculatum*	200	*Rytidosperma caespitosum*	1000	*Rytidosperma caespitosum*	1000
Chrysocephalum apiculatum	600	*Convolvulus angustissmus*	500	*Rytidosperma setaceum*	1500	*Rytidosperma geniculatum*	200
Clematis microphyla	150	*Convolvulus remotus*	100	*Themeda triandra*	300	*Rytidosperma racemosum*	200
Convolvulus angustissmus	70	*Cymbopogon ambiguus*	100	**Total**	**7200**	*Rytidosperma pilosum*	500
Convolvulus remotus	100	*Dichanthium sericeum*	50			*Rytidosperma setaceum*	3000
Cymbopogon ambiguus	50	*Enchylaena tomentosa*	100			*Themeda triandra*	500
Dichanthium sericeum	10	*Enneapogon nigricans*	50			**Total**	**10400**
Einadia nutens	50	*Eryngium ovinum*	100				
Enchylaena tomentosa	50	*Microlaena stipoides*	50				
Enneapogon nigricans	20	*Rytidosperma caespitosum*	200				
Eryngium ovinum	50	*Rytidosperma setaceum*	2000				
Eutaxia microphyla	50	*Senecio quadridentatus*	2000				
Geranium retrorsum	70	*Themeda triandra*	50				
Goodenia pinnatifida	150	*Vittadinia blackii*	100				
Hardenbergia violacea	40	*Vittadinia cuneata*	100				
Kennedia prostrata	10	*Xerochrysum bracteatum*	500				
Leptorhynchos squamatus	10	**Total**	100				
Microlaena stipoides	50		**8300**				
Poa labillardieri	20						
Ptilotus spathulatus	200						
Ranunculus lappaceuus	100						
Rytidosperma caespitosum	30						
Rytidosperma racemosum	10						
Senecio quadridentatus	20						
Themeda triandra	100						
Vittadinia blackii	100						
Vittadinia cuneata	10						
Wahlenbergia stricta	5						
Xerochrysum bracteatum	70						
Total	**3880**						

2024- Glenthorne Community Grassland Seed Mix Lists

Shrubby Buffer Seed Mix

(For wind break)

- **Trees & Large Shrubs:**
    - *Acacia myrtifolia*
    - *Acacia pycnantha*
    - *Acacia acinacea*
    - *Acacia retinodes*
    - *Acacia verniciflua*
    - *Callistemon rugulosus*
    - *Eucalyptus microcarpa*
    - *Leptospermum continentale*
    - *Olearia ramulosa*
- **Medium & Small Shrubs:**
    - *Allocasuarina verticillata*
    - *Dodonaea viscosa*
    - *Rhagodia parabolica*
    - *Enchylaena tomentosa*
- **Grasses & Sedges:**
    - *Poa labillardierei*
    - *Microlaena stipoides*
    - *Austrostipa semibarbata*
    - *Dichanthium sericeum*
    - *Cyperus vaginatus*
- **Climbers & Ground Covers:**
    - *Clematis microphyla*
    - *Atriplex semibaccata*
- **Other:**
    - *Xanthorrhoea semiplana*

Lighter Clay Diverse Seed Mix

(For high floristic value)

- **Small Shrubs & Herbs:**
    - *Eutaxia microphylla*
    - *Callitris tetragona*
    - *Gonocarpus elatus*
    - *Pultenaea largiflorens*
    - *Goodenia pinnatifida*
    - *Linum marginale*
    - *Convolvulus remotus*
    - *Xerochrysum bracteatum*
    - *Senecio quadridentatus*
    - *Vittadinia blackii*
    - *Euchiton involucratus*
    - *Chrysocephalum apiculatum*
- **Grasses & Sedges:**
    - *Poa labillardierei*
    - *Themeda triandra*
    - *Rytidosperma caespitosum*
    - *Rytidosperma setaceum*
    - *Anthosachne scabra*
    - *Austrostipa nodosa*
- **Legumes & Climbers:**
    - *Hardenbergia spp.*
    - *Kennedia spp.*
- **Ground Covers:**
    - *Leptorhynchos squamatus*
    - *Acaena echinata*
    - *Acaena novae-zelandiae*
    - *Cynoglossum spp.*
    - *Wahlenbergia stricta*

- *Arthropodium strictum*
- *Ptilotus spathulatus*

Chenopod Mix Seed Mix

(For solarised areas)

- **Chenopod Shrubs:**
    - *Atriplex semibaccata*
    - *Atriplex vesicaria*
    - *Maireana brevifolia*
    - *Rhagodia spinescens*
    - *Rhagodia parabolica*
- **Grasses & Sedges:**
    - *Anthosachne scabra*
    - *Aristida behriana*
    - *Austrostipa scabra*
    - *Chloris truncata*
    - *Enneapogon nigricans*
    - *Rytidosperma caespitosum*
    - *Rytidosperma setaceum*
    - *Themeda triandra*
- **Ground Covers:**
    - *Einadia nutans*
    - *Enchylaena tomentosa*

Black Clay Grasses & Sedges Mix

(For seasonally wet black clay soils)

- **Grasses & Sedges:**
    - *Anthosachne scabra*
    - *Rytidosperma pilosum*
    - *Microlaena stipoides*
    - *Rytidosperma duttonianum*
    - *Rytidosperma geniculatum*
    - *Rytidosperma caespitosum*
    - *Rytidosperma setaceum*
    - *Chloris truncata*
    - *Setaria jubiflora*
- **Wetland & Riparian Plants:**
    - *Ficinia nodosa*
    - *Juncus amabilis*
    - *Juncus pallidus*
    - *Juncus subsecundus*
    - *Carex tereticaulis*
    - *Chorizandra enodis*
    - *Cyperus gymnocaulis*
    - *Cyperus vaginatus*
- **Special Collection:**
    - Glenthorne grass harvest 2023

APPENDIX C: GRASSLAND SURVEY

Appendix C1 – example Survey Solarised Plots 2023

Appendix C2 – Blank Survey

																		Management Area
3	2	1	3	2	1	3	2	1	3	2	1	3	2	1	3	2	1	Replicate
																		cover_native
																		cover_non_native
																		cover_bare
																		cover_litter
																		cover_rock

November 2023

Species Richness Survey Results Long solarised plots

Area	transect_position	seedmix	cover_native	cover_non_native	cover_bare	cover_litter	cover_rock	Microlaena stipoides	Avena barbata	Sonchus oleraceus	Helminthotheca echioides	Austrostipa curticoma	Lolium rigidum	Vulpia sp.	Rytidosperma racemosum	Rytidosperma caespitosum	Trifolium sp.	Plantago lanceolata	Atriplex semibaccata	Lythrum hyssopifolia	Atriplex semibacatta	Dittrichia graveolens	Malva parviflora	Phalaris aquatica	Anthosachne scabra	Medicago polymorpha	Bromus diandrus	Cucumis sp.	Polygonum aviculare	Conyza bonariensis	Vicia sp.	Kickxia elatine
solar	1	grassonly	25	5	70	1	0	1	1	1	1				1																	
solar	2	grassonly	50	5	45	1	0	1	1			1	1	1	1	1																
solar	3	grassonly	1	40	30	30	0	1	1				1		1	1																
solar	1	chenopods	30	20	30	20	0		1	1			1			1	1	1	1	1	1	1										
solar	2	chenopods	40	20	20	20	0		1	1			1	1		1			1	1			1	1								
solar	3	chenopods	20	10	50	20	1	1	1						1	1									1							
solar	1	diverse	40	20	30	10	1	1	1	1				1	1	1									1							
solar	2	diverse	20	10	50	20	1	1	1			1		1	1	1	1		1							1	1					
solar	3	diverse	30	30	20	20	0		1	1			1	1	1	1							1		1		1		1			
solar	1	chenopods	10	30	30	30	0		1				1			1													1			
solar	2	chenopods	25	30	30	1	1		1				1			1			1				1				1					
solar	3	chenopods	20	10	60	1	0		1			1				1			1				1			1				1	1	1
solar	1	grassonly	7.5	20	60	30	0	1	1				1		1	1								1						1		
solar	2	grassonly	10	5	70	15	0	1	1	1			1		1	1																
solar	3	grassonly	2.5	60	30	7.5	2.5		1	1			1	1	1	1								1								

1 = presence; 0 = absence

APPENDIX D – NATIVE AND NON-NATIVE PLANTS FIELD GUIDE

Appendix D1 – Native Plants Field Guide

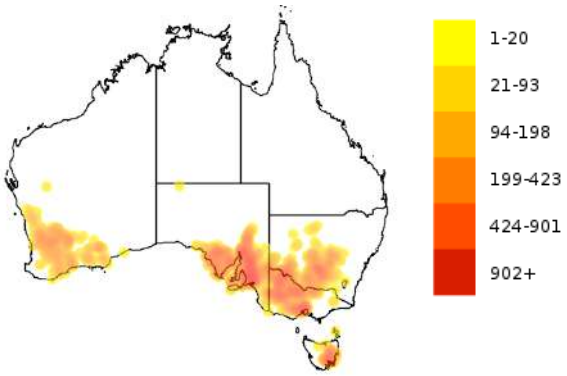
Appendix D2 – Non-native Plants Field Guide

This PDF was generated on 28 October 2024. View the original search query.

Family: **AMARANTHACEAE**

Scientific name: ***Ptilotus spathulatus***
Pussy-Tails

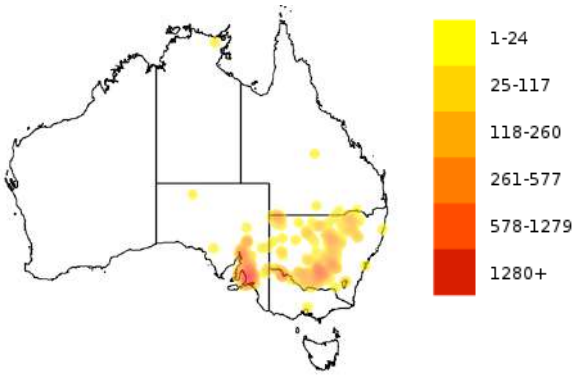




Family: **AMARYLLIDACEAE**

Scientific name: ***Calostemma purpureum***
Garland Lily

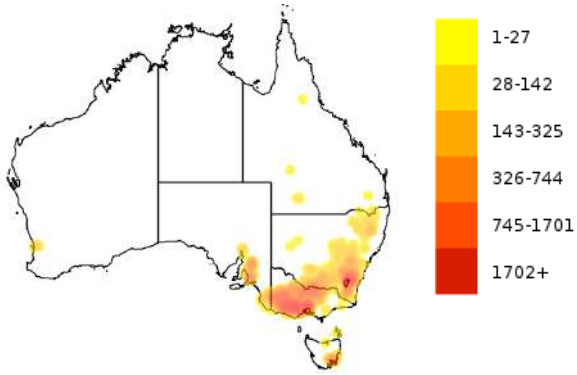




Family: **APIACEAE**

Scientific name: ***Eryngium ovinum***
Blue Devil

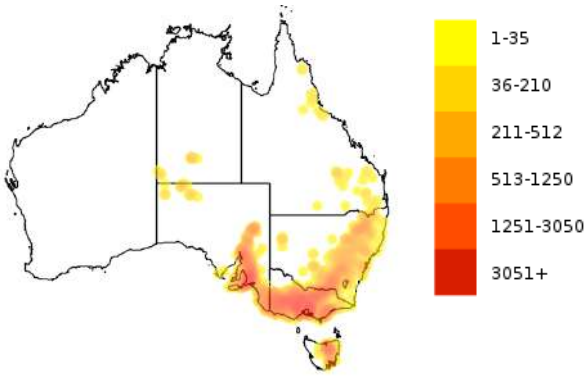




Family: **ASPARAGACEAE**

Scientific name: ***Arthropodium strictum***
Chocolate Lily

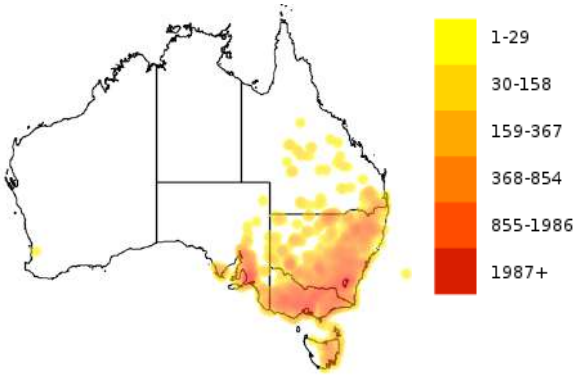




Family: **ASPHODELACEAE**

Scientific name: ***Bulbine bulbosa***
Bulbine Lily

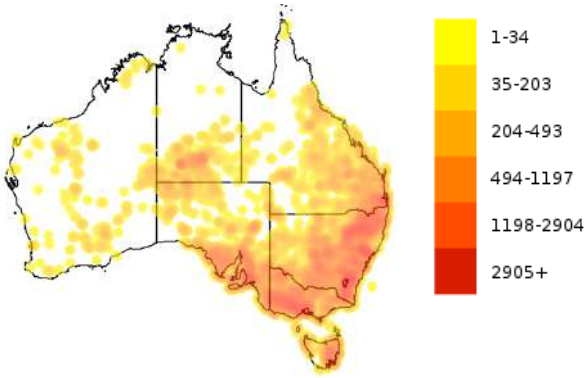




Family: **ASTERACEAE**

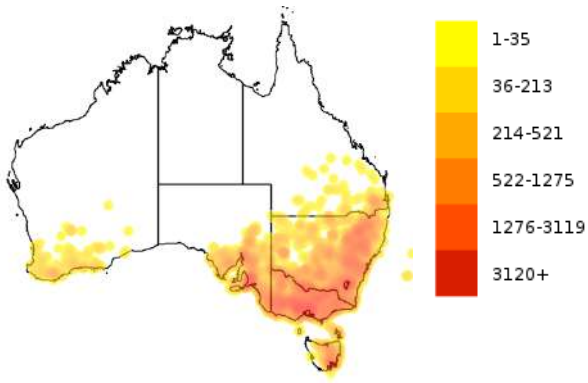
Scientific name: ***Chrysocephalum apiculatum***
Common Everlasting





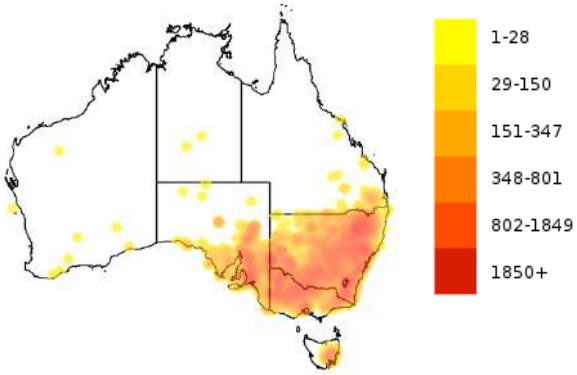
Scientific name: ***Senecio quadridentatus***
Cotton Groundsel





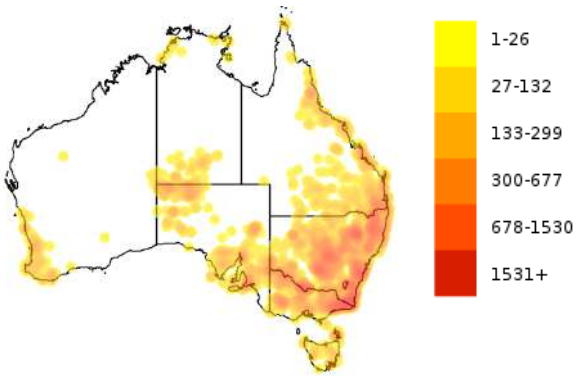
Scientific name: ***Vittadinia cuneata***
Fuzzy New Holland Daisy





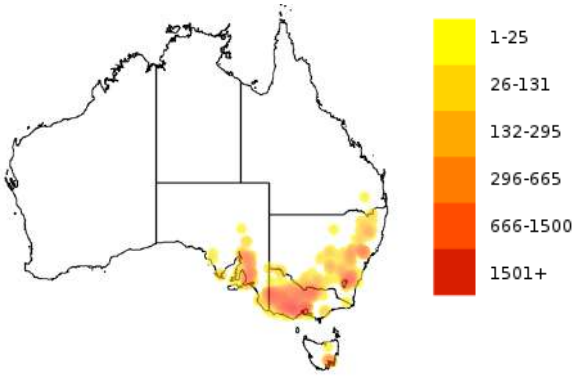
Scientific name: ***Xerochrysum bracteatum***
Golden Everlasting





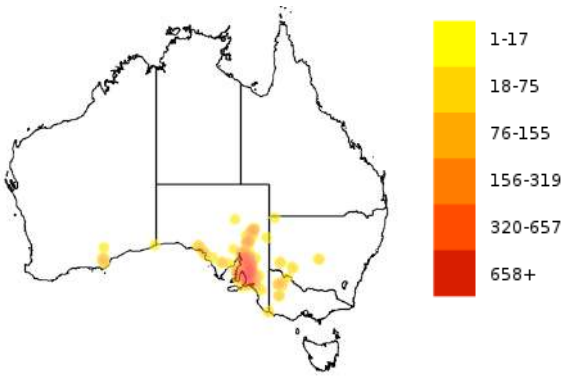
Scientific name: ***Calocephalus citreus***
Lemon Beauty-Heads





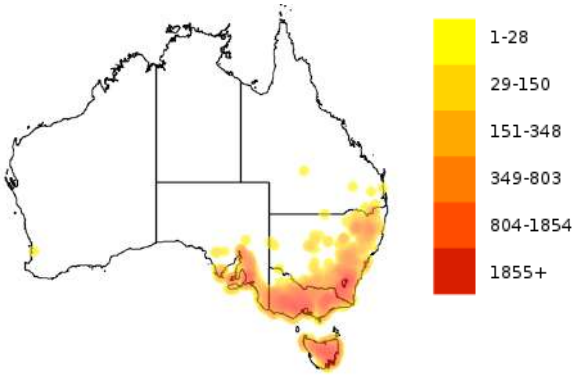
Scientific name: ***Vittadinia blackii***
Narrow-Leaf New Holland Daisy





Scientific name: ***Leptorhynchos squamatus***
Scaly Buttons

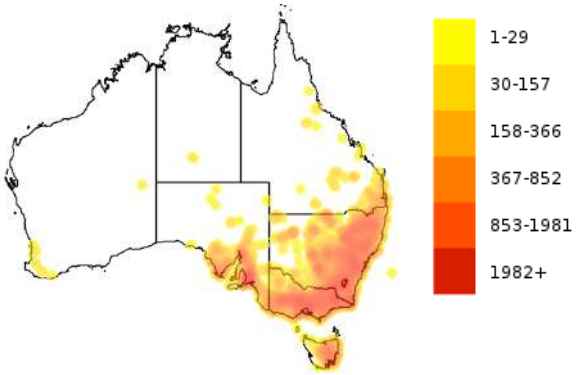




Family: **CAMPANULACEAE**

Scientific name: ***Wahlenbergia stricta***
Tall Bluebell

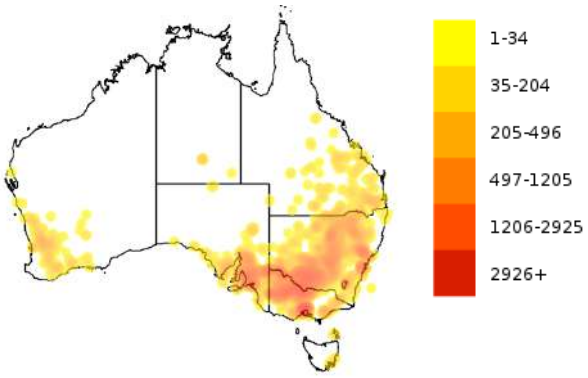




Family: **CHENOPODIACEAE**

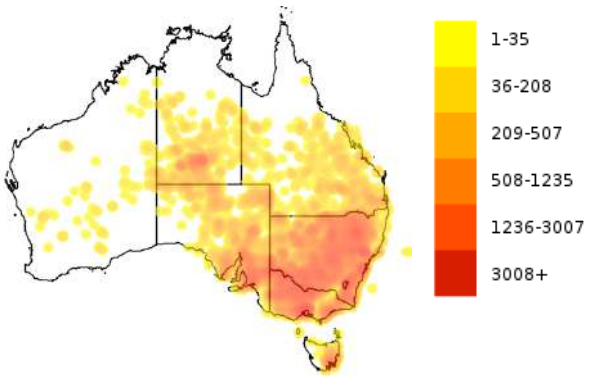
Scientific name: ***Atriplex semibaccata***
Berry Saltbush





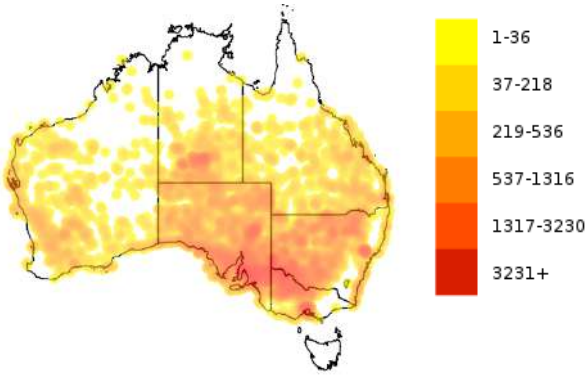
Scientific name: ***Einadia nutans***
Climbing Saltbush





Scientific name: ***Enchylaena tomentosa***
Ruby Saltbush

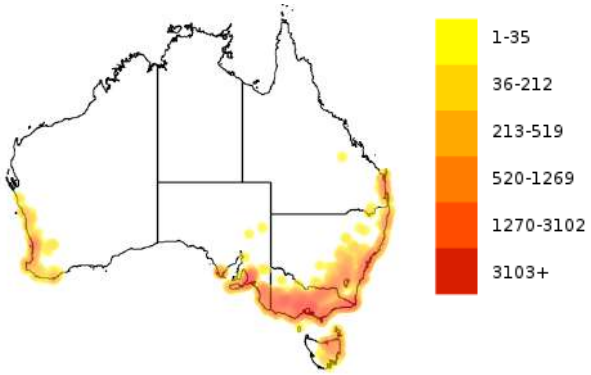




Family: **COLCHICACEAE**

Scientific name: ***Burchardia umbellata***
Milkmaids

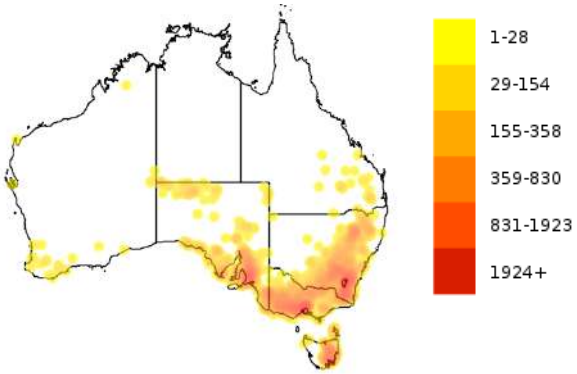




Family: **CONVOLVULACEAE**

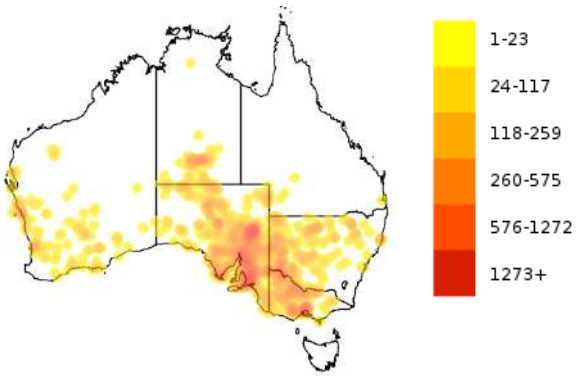
Scientific name: ***Convolvulus angustissimus***
Blushing Bindweed





Scientific name: ***Convolvulus remotus***
Grassy Bindweed

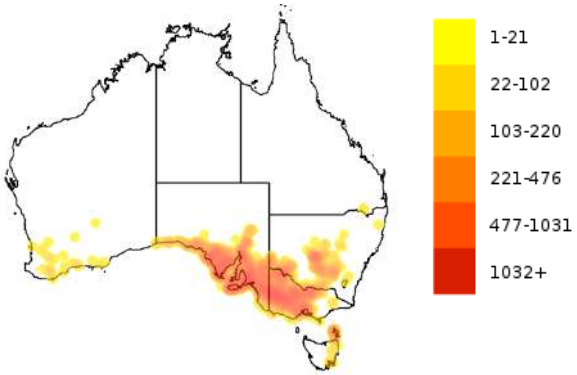




Family: **FABACEAE**

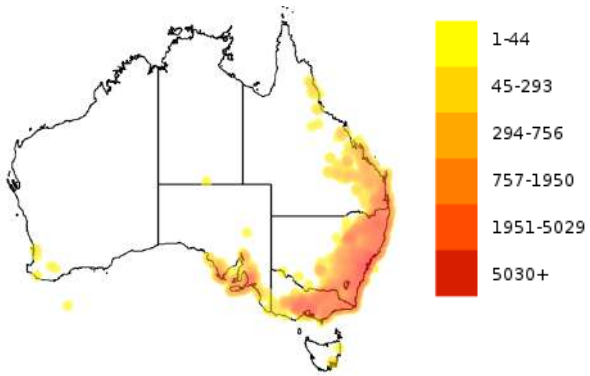
Scientific name: ***Eutaxia microphylla***
Common Eutaxia





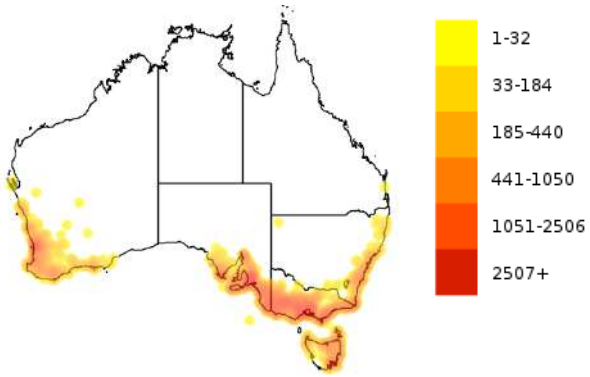
Scientific name: ***Hardenbergia violacea***
Native Lilac





Scientific name: ***Kennedia prostrata***
Scarlet Runner

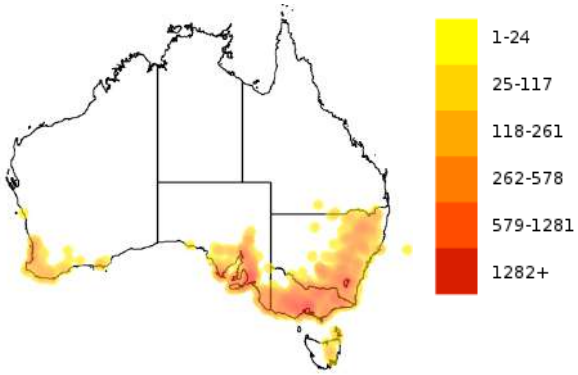




Family: **GERANIACEAE**

Scientific name: ***Geranium retrorsum***
Grassland Crane's-Bill

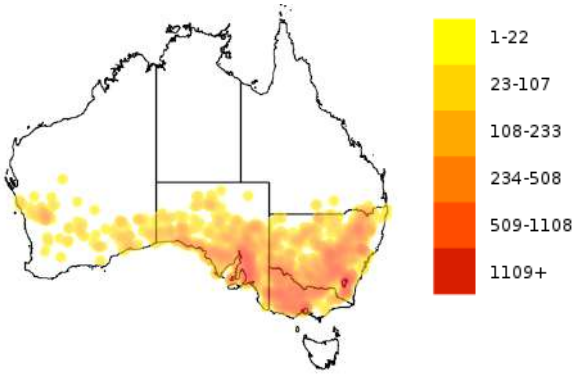




Family: **GOODENIACEAE**

Scientific name: ***Goodenia pinnatifida***
Cut-Leaf Goodenia

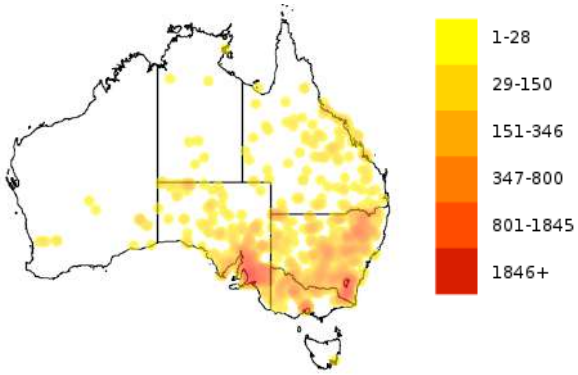




Family: **POACEAE**

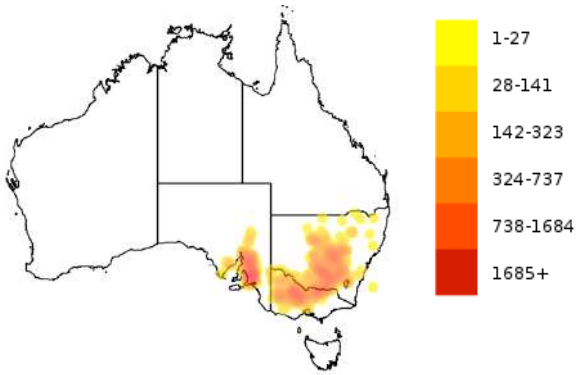
Scientific name: ***Enneapogon nigricans***
Black-Head Grass





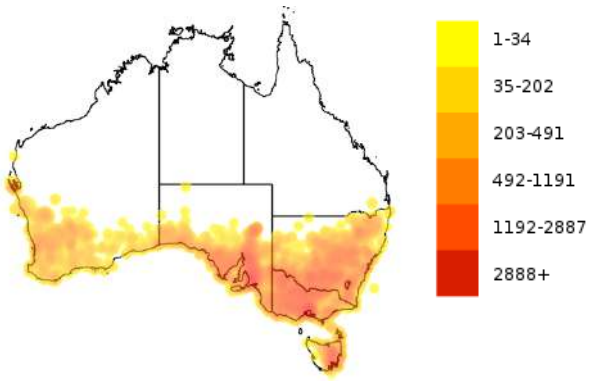
Scientific name: ***Aristida behriana***
Brush Wire-Grass





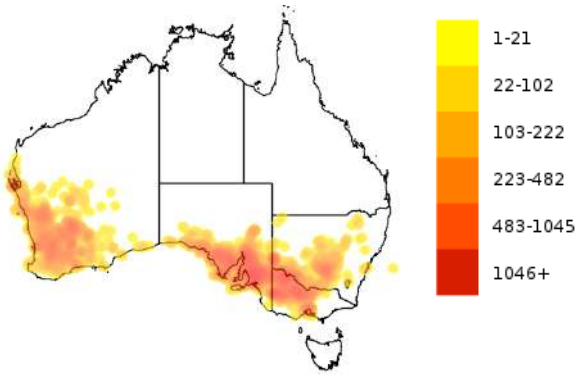
Scientific name: ***Rytidosperma caespitosum***
Common Wallaby-Grass





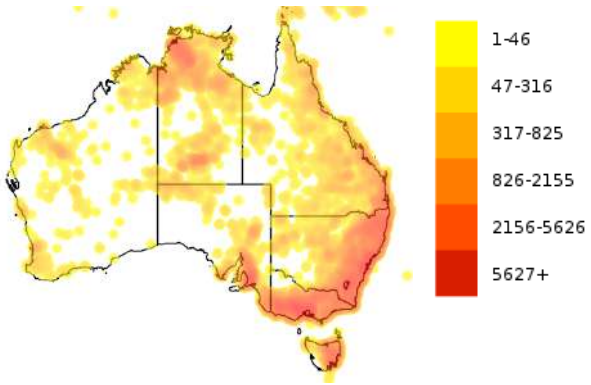
Scientific name: ***Austrostipa elegantissima***
Feather Spear-Grass





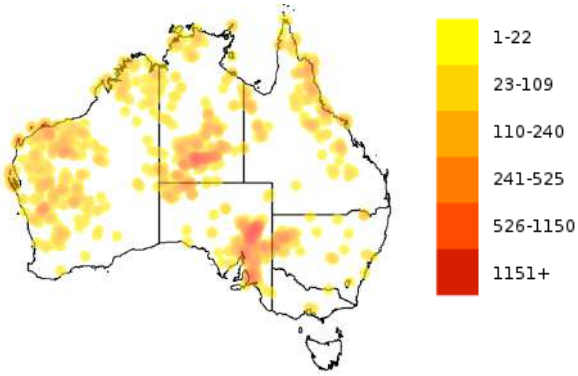
Scientific name: ***Themeda triandra***
Kangaroo Grass





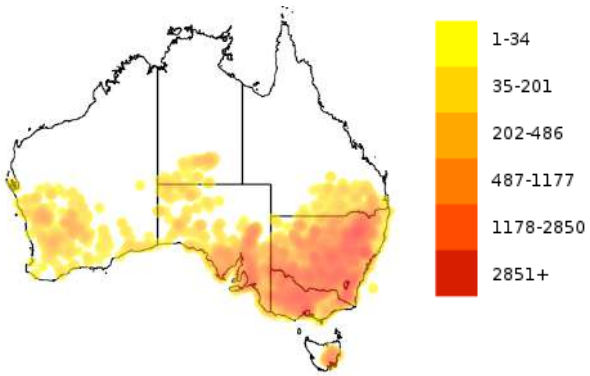
Scientific name: ***Cymbopogon ambiguus***
Lemon Grass





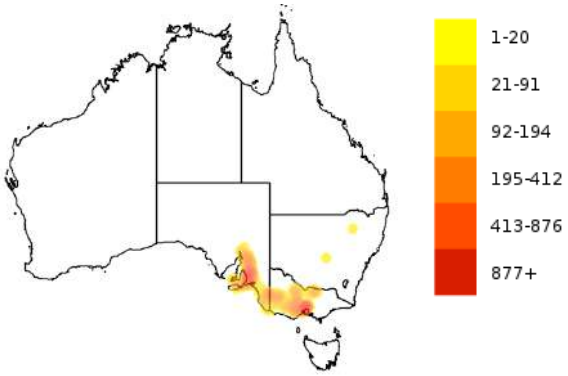
Scientific name: ***Austrostipa scabra***
Rough Spear-Grass





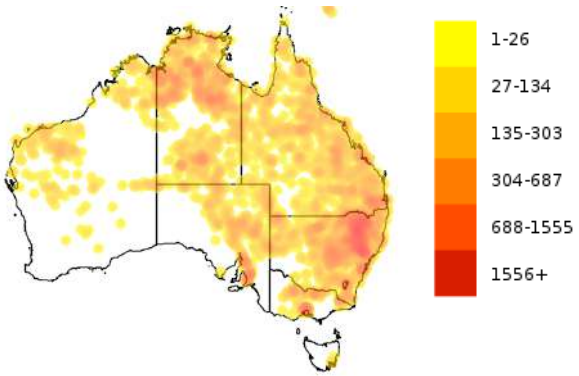
Scientific name: ***Austrostipa curticoma***
Short-Crown Spear-Grass





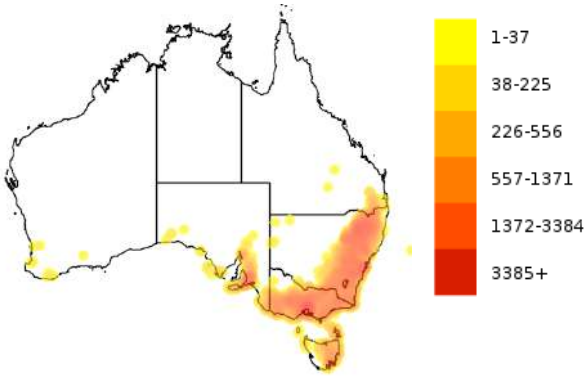
Scientific name: ***Dichanthium sericeum***
Silky Blue-Grass





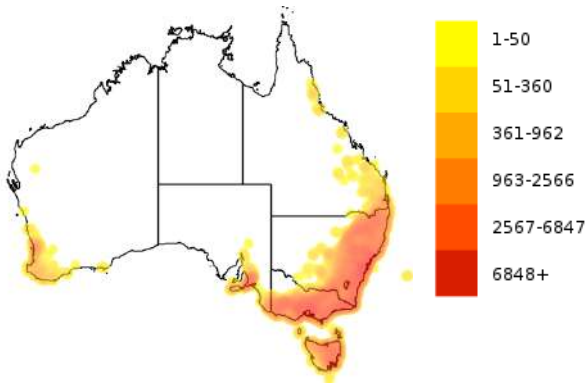
Scientific name: ***Rytidosperma racemosum***
Wallaby Grass





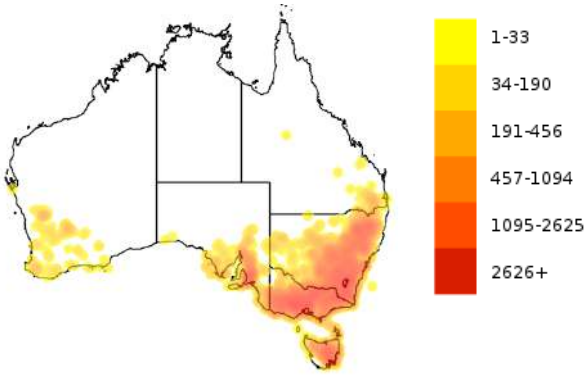
Scientific name: ***Microlaena stipoides***
Weeping Grass





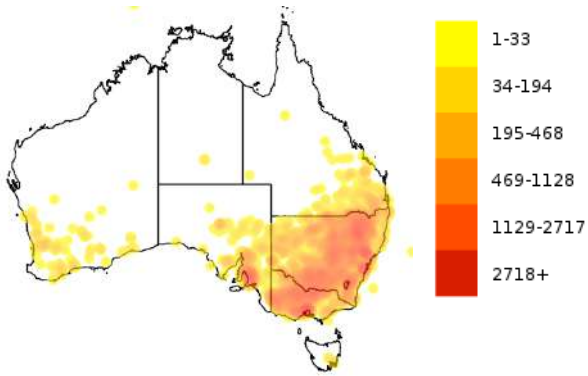
Scientific name: ***Anthosachne scabra***
Wheatgrass





Scientific name: ***Chloris truncata***
Windmill Grass

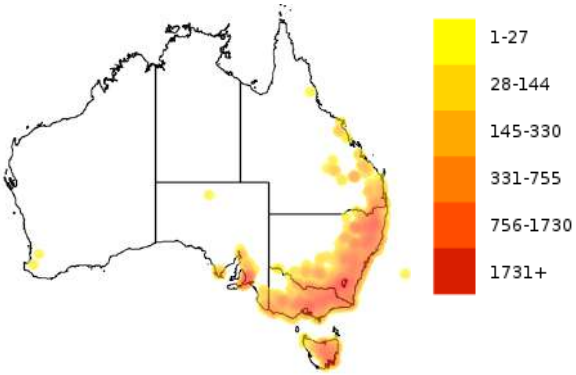




Family: **RANUNCULACEAE**

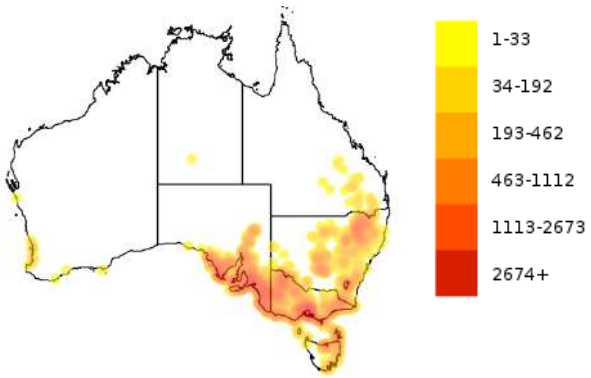
Scientific name: ***Ranunculus lappaceus***
Native Buttercup





Scientific name: ***Clematis microphylla***
Small-Leaved Clematis

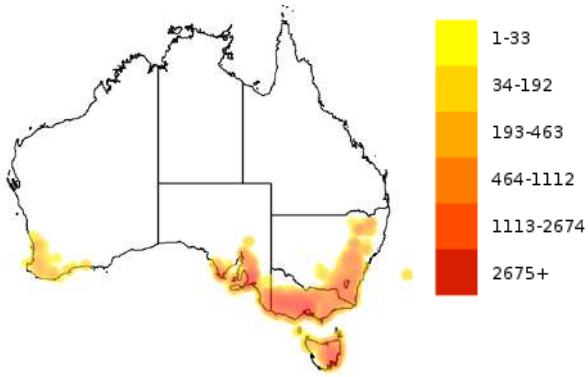




Family: **ROSACEAE**

Scientific name: ***Acaena echinata***
Sheep's Burr





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Acaena echinata

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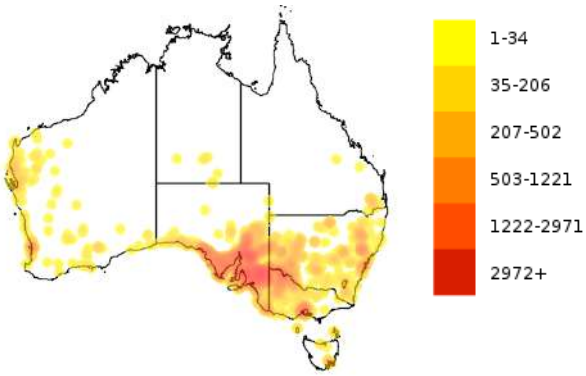
Family: **ASPHODELACEAE**

Scientific name: ***Asphodelus fistulosus***
Onion Weed

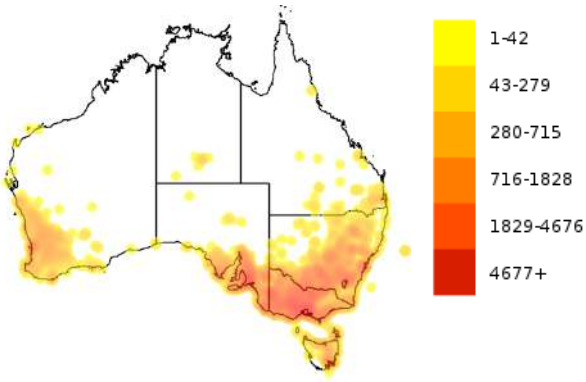




Family: **ASTERACEAE**

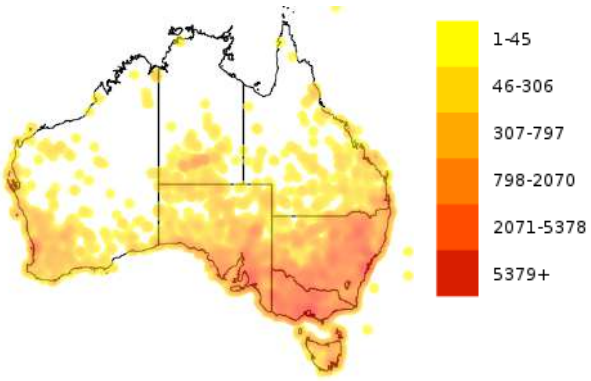
Scientific name: ***Arctotheca calendula***
Cape Weed





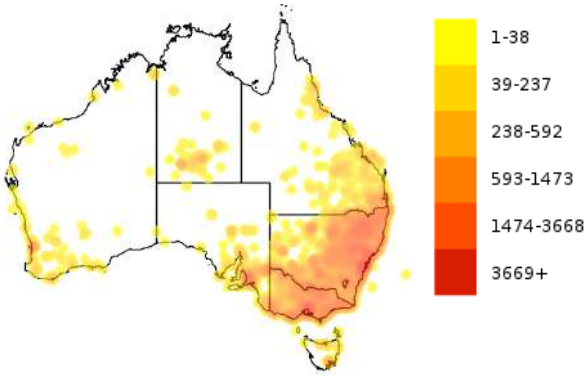
Scientific name: ***Sonchus oleraceus***
Common Sow-Thistle





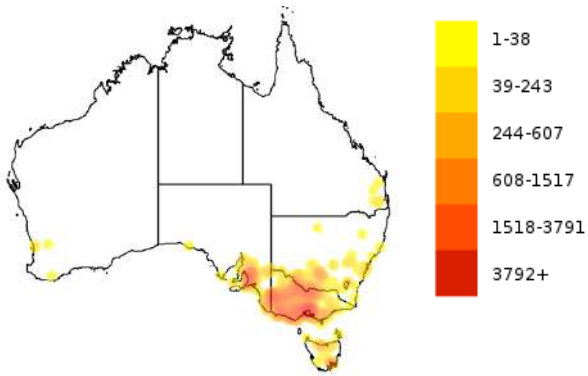
Scientific name: ***Erigeron bonariensis***
Flaxleaf Fleabane





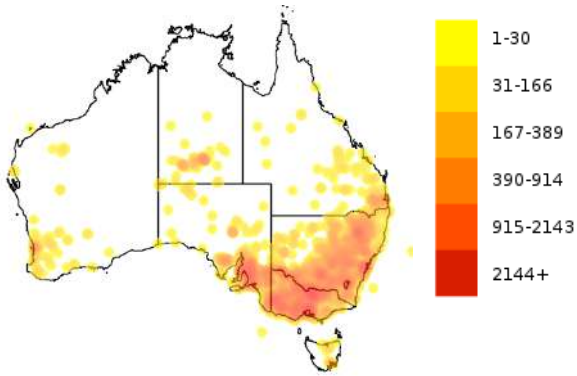
Scientific name: ***Helminthotheca echioides***
Ox-Tongue





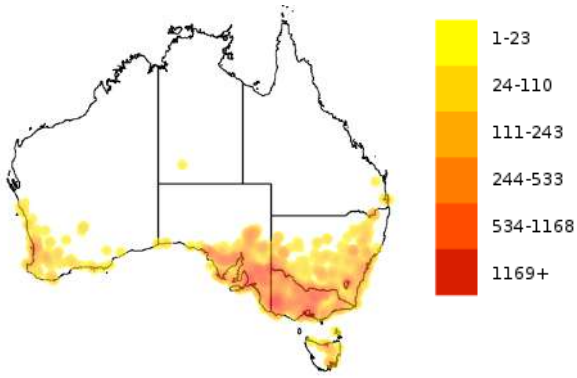
Scientific name: ***Lactuca serriola***
Prickly Lettuce





Scientific name: ***Dittrichia graveolens***
Stinkwort

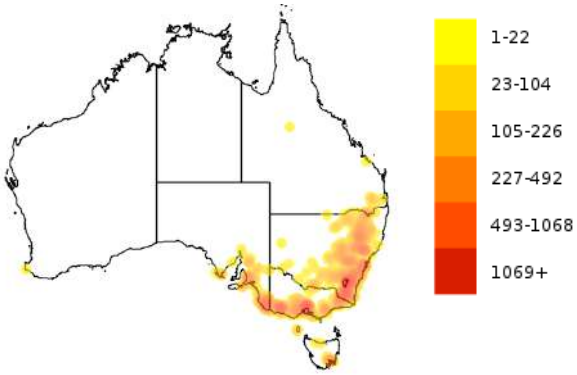




Family: **BRASSICACEAE**

Scientific name: ***Hirschfeldia incana***
Hoary Mustard

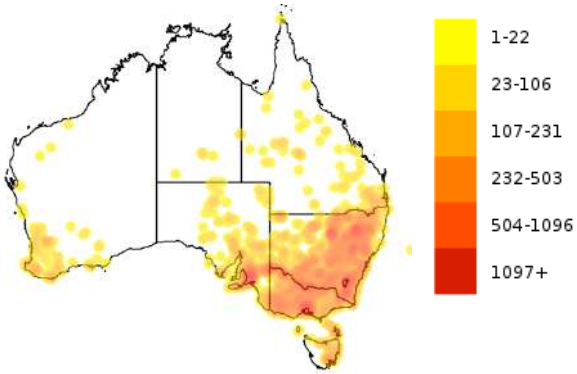




Family: **CHENOPODIACEAE**

Scientific name: ***Dysphania pumilio***
Small Crumbweed

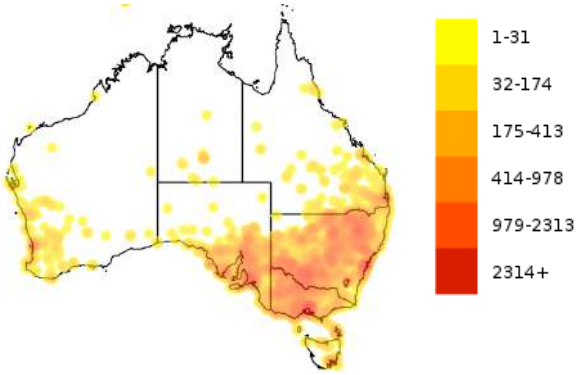




Family: **FABACEAE**

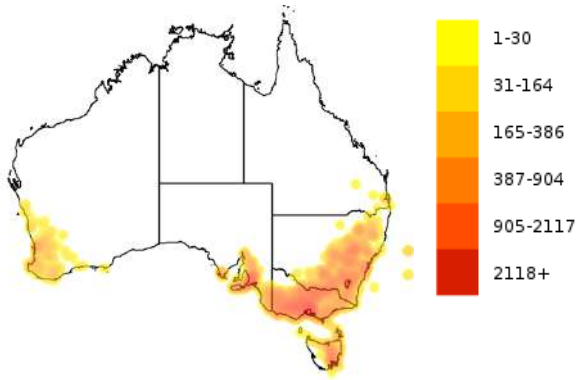
Scientific name: ***Medicago polymorpha***
Spineless Burr Medic





Scientific name: ***Trifolium subterraneum***
Subterranean Clover

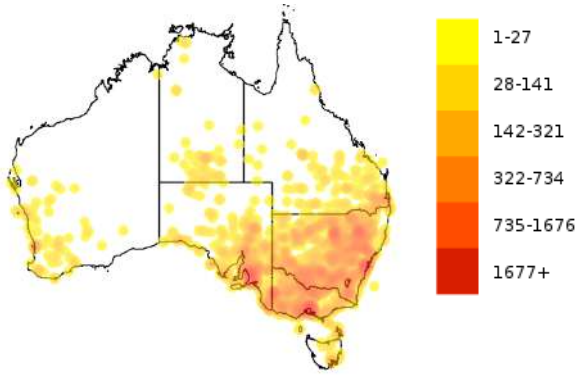




Family: **MALVACEAE**

Scientific name: ***Malva parviflora***
Small-Flower Mallow

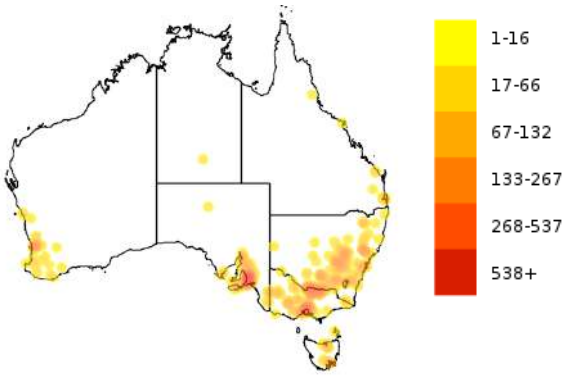




Family: **PLANTAGINACEAE**

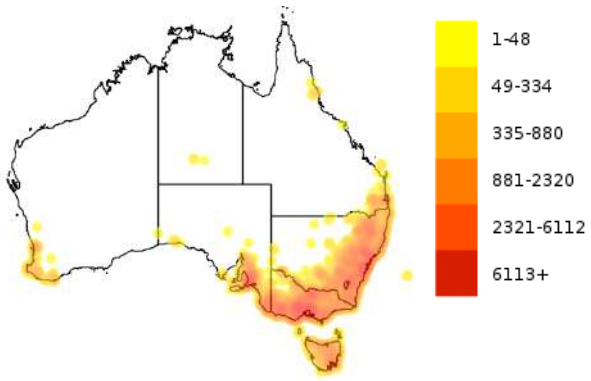
Scientific name: ***Kickxia elatine***
Pointed Toadflax





Scientific name: ***Plantago lanceolata***
Ribwort

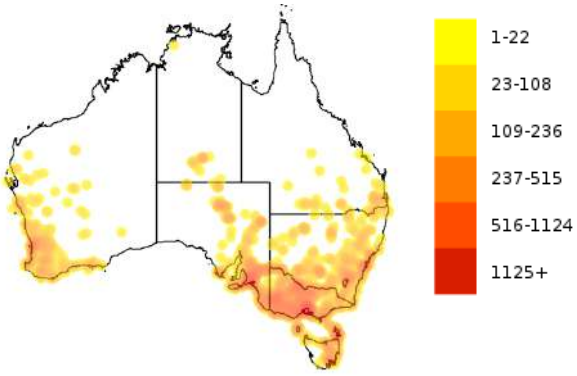




Family: **POACEAE**

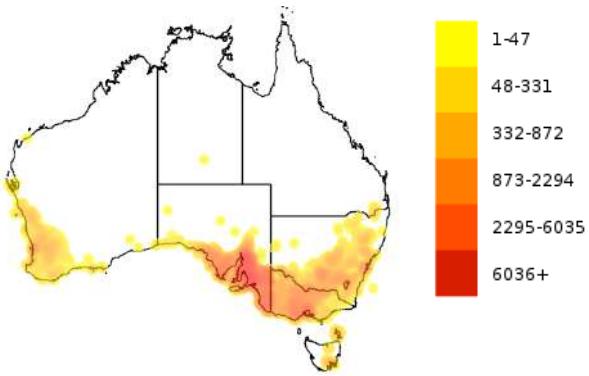
Scientific name: ***Polypogon monspeliensis***
Annual Beardgrass





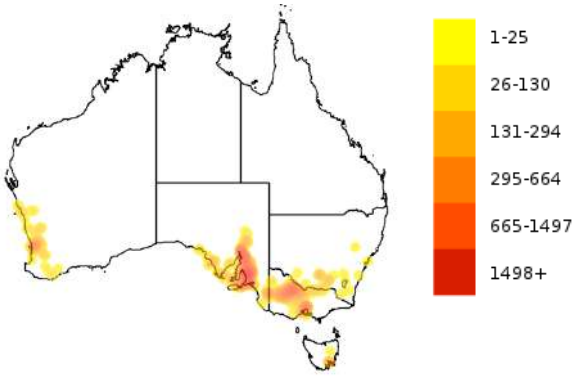
Scientific name: ***Avena barbata***
Bearded Oats





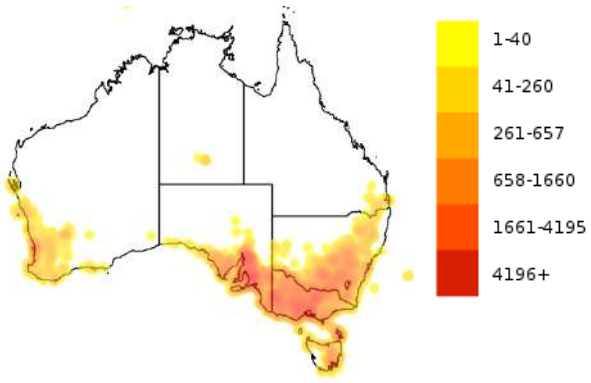
Scientific name: ***Brachypodium distachyon***
False Brome





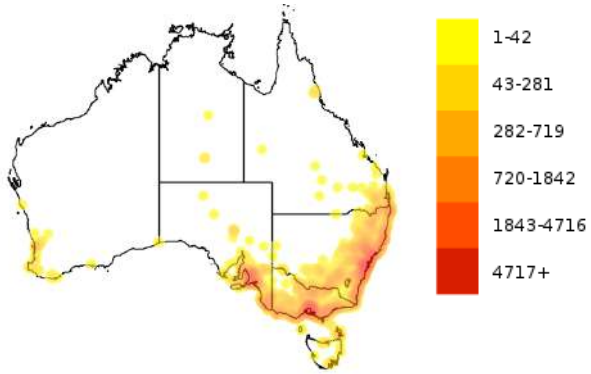
Scientific name: ***Bromus diandrus***
Great Brome





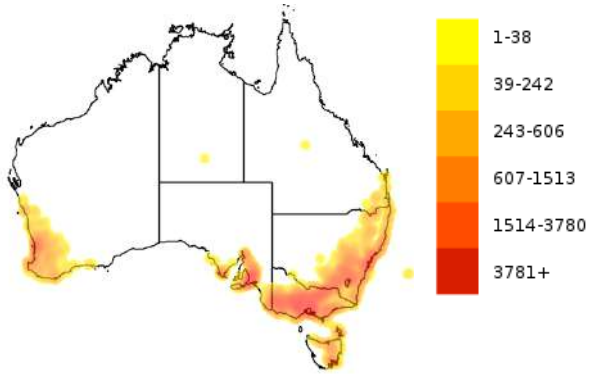
Scientific name: ***Cenchrus clandestinus***
Kikuyu





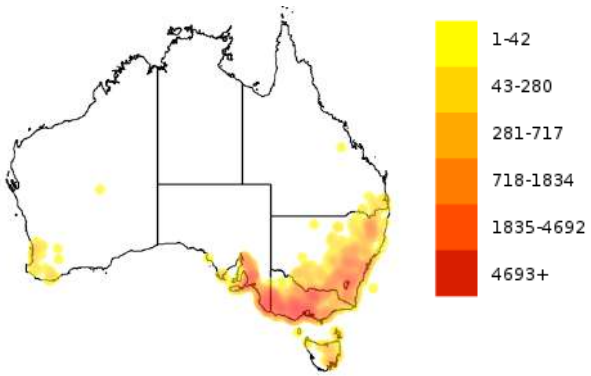
Scientific name: ***Briza maxima***
Large Quaking-Grass





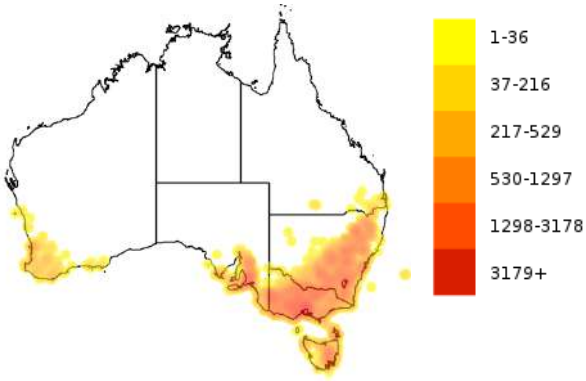
Scientific name: ***Phalaris aquatica***
Phalaris





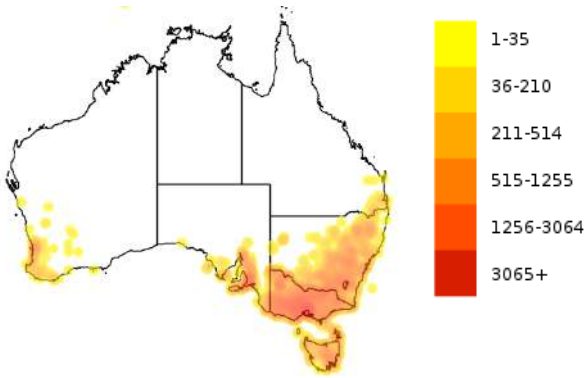
Scientific name: ***Bromus hordeaceus***
Soft Brome





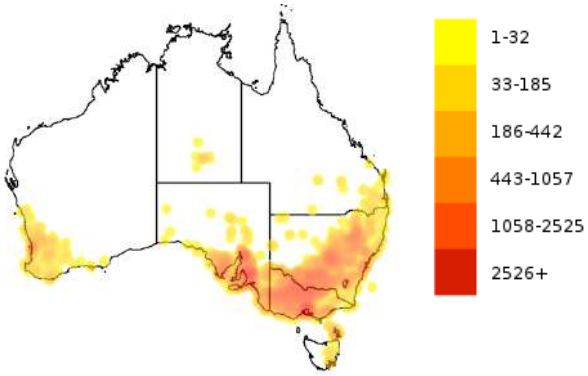
Scientific name: ***Vulpia bromoides***
Squirrel-Tail Fescue





Scientific name: ***Lolium rigidum***
Wimmera Ryegrass

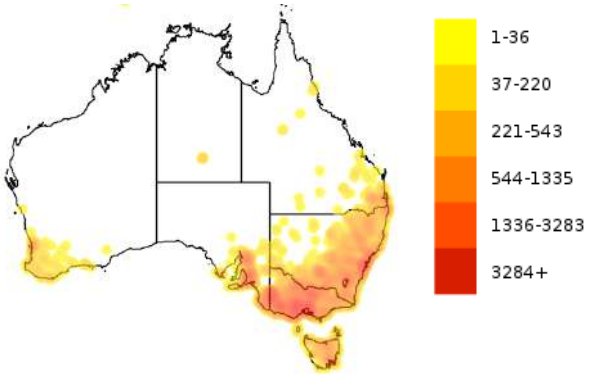




Family: **POLYGONACEAE**

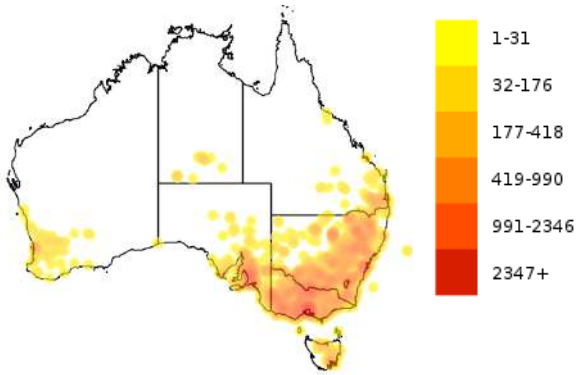
Scientific name: ***Rumex crispus***
Curled Dock





Scientific name: ***Polygonum aviculare***
Prostrate Knotweed

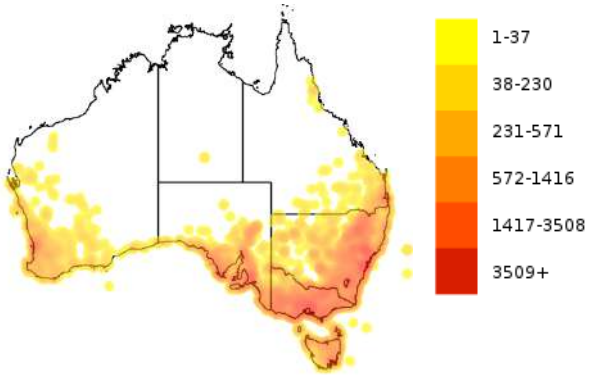




Family: **PRIMULACEAE**

Scientific name: ***Lysimachia arvensis***
Pimpernel





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Erigeron bonariensis
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Helminthotheca echioides
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Lactuca serriola
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Dittrichia graveolens
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Hirschfeldia incana
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Medicago polymorpha
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Trifolium subterraneum

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Malva parviflora

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Kickxia elatine

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Plantago lanceolata

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Polypogon monspeliensis

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Avena barbata

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Brachypodium distachyon

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Bromus diandrus

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Cenchrus clandestinus

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Briza maxima

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Phalaris aquatica

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Bromus hordeaceus

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Vulpia bromoides

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Lolium rigidum

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Rumex crispus

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Polygonum aviculare

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Lysimachia arvensis

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