The Impacts of Brush Packs on Landscape Function in a Paddock in Temperate, Mesic Southeastern Australia

by

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Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

Alexander Harrison Date: 21/11/2024

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Abstract

This study investigated the impacts of brush packs on landscape function, focusing on nutrient cycling at a degraded, heavily grazed paddock with a temperate, mesic climate in southeastern Australia. It addressed a gap in the existing research which has largely focused on the impacts of brush packs in semi-arid and arid environments. Brush packs, constructed from *Leptospermum* branches, were established in 2018 during a period of drought to restore function at a dysfunctional, heavily grazed, mostly bare ground site.

The impacts of brush packs on function were assessed at both the landscape-scale and fine-scale using the Landscape Function Analysis (LFA) method, and direct measurements of nutrient cycling: soil respiration and carbon levels. Soil chemical properties, pH and electrical conductivity (EC), were also measured to determine if brush packs had any influence on these important soil properties. Based on previous research, it was expected that brush packs would increase LFA scores in all three indices: stability, infiltration, and nutrient cycling, as well as increase soil respiration rates and carbon levels. It was also expected that brush packs would not change pH or EC levels.

A total of twenty brush packs were established randomly across five transects, with six additional transects established as controls. LFA measurements were taken in 2018, both before and after brush pack establishment, and again in 2019 and 2024. In 2024, measurements of soil respiration, carbon levels, and chemical properties were taken, with carbon and chemical properties compared across four depth intervals (0-1, 1-3, 3-5, and 5-10cm) and carbon fractionated into three fractions (particulate organic carbon, aggregate carbon, and mineral-associated organic carbon).

Unexpectedly, brush packs gave no improvements in function at the landscape-scale relative to control (pasture) transects. At the fine-scale, they showed improvements in all three indices relative to control interpatches. However, brush packs did not improve soil respiration and carbon levels relative to control patches. During the study both brush pack and control patches experienced significant functional improvements due to substantial pasture growth driven by a rare three-year La Niña event. The natural recovery across the site likely overshadowed any additional benefit provided by the brush packs. No significant differences in pH or EC levels were observed.

These findings indicate brush packs may have limited utility as a rehabilitation technique in resilient, mesic landscapes. LFA data collected one year after the brush packs were established and before the three-year La Niña event drove natural recovery suggests brush packs likely had a head start in terms of functional recovery. If conditions were to shift towards dysfunction, such as during a period of drought and intense grazing the relative functionality of the brush packs may increase, if they persist. Therefore, future research could explore whether the brush packs in this study offer lasting benefits to landscape function under drought conditions.

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List of acronyms and abbreviations

NSW	New South Wales
ANU	Australian National University
TMI	The Mulloon Institute
TTRP	Trigger-Transfer-Reserve-Pulse (model)
LFA	Landscape Function Analysis
SSA	Soil surface assessment (indicators)

km	Kilometres
m	Metres
cm	Centimetres
mm	Millimetres
ppm	Parts per million
µmol	Micromoles
s	Seconds
mg	Milligrams
%VWC	Volumetric water content
µS	Microsiemens
EC	Electrical conductivity
PVC	Polyvinyl chloride
CO_2	Carbon dioxide
TC	Total carbon
TOC	Total organic carbon
TIC	Total inorganic carbon
POC	Particulate organic carbon
AggC	Aggregate carbon
MAOC	Mineral-associated organic carbon
DOC	Dissolved organic carbon
ANOVA	Analysis of variance
LMM	Linear mixed-effects model
EMM	Estimated marginal mean
df	Degrees of Freedom
CI	Confidence interval
Std. Error	Standard error
Std. Dev.	Standard deviation
p/p-value	Probability value

Glossary and Terms

Brush pack:	The term ‘brush pack’ broadly refers to a landscape rehabilitation technique which aims to stabilise soil, reduce erosion, and promote vegetation regrowth by creating a favourable environment for plants. It involves placing and packing branches or brush along the contours of bare, gently sloping areas to capture and retain resources such as soil, water, and plant detritus.
Landscape Function:	Landscape function refers to the fundamental processes that enable a landscape to capture, store, and retain resources such as soil, water, and organic matter. This definition is further explored and expanded upon in Chapter 2: Literature Review.
Landscape-scale:	Effects that extend beyond the brush packs to influence the surrounding landscape, examined at a resolution suitable for evaluating the overall landscape.
Fine-scale:	Direct, localised effects occurring immediately beneath the brush packs, examined at a resolution suitable for the evaluation of individual brush packs.
Main Site:	The Main Site for this study is in a 68-hectare paddock located in Mulloon, NSW, at coordinates: 35.26746111° S, 149.61894167° E.
Reference Site:	The Reference Site is located in an area of least disturbed woodland approximately 3 kilometres west of the Main Site, on the west side of Mulloon Creek in Mulloon, NSW, at coordinates: 35.26076111° S, 149.58511111° E

Chapter 1: Introduction

This study examined the impacts of brush packs on landscape function with a specific focus on nutrient cycling at a degraded, heavily grazed, mostly bare ground site with a temperate, mesic climate in southeastern Australia. It followed a study by Tongway and Ludwig (1996) who found brush packs restored function in degraded semi-arid woodlands of eastern Australia. Given the potential of brush packs to improve landscape function, this study aimed to see if similar outcomes could be achieved in a temperate, mesic grazing landscape which differs significantly from the semi-arid conditions previously studied by Tongway and Ludwig (1996).

Brush packing involves packing branches or brush along the contours of bare, gently sloping areas to capture and retain resources such as soil, water, and plant detritus. In 2018, twenty brush packs were established at a heavily grazed and mostly bare ground sheep camp within a paddock located in Mulloon, NSW. The brush packs were constructed with material sourced from the routine clearing of nearby *Leptospermum* species. The brush packs were constructed by Australian National University students in collaboration with The Mulloon Institute through a formal partnership as part of their Mulloon Rehydration Initiative (Peel et al., 2022).

Improvements in function were assessed using the Landscape Function Analysis (LFA) method (Tongway & Hindley, 2004) in 2018, 2019, and finally in 2024. Initial LFA results in 2024 indicated brush packs had improved the nutrient cycling index. To investigate this in detail, direct measurements of soil respiration and carbon levels, two key indicators of nutrient cycling (Tongway & Hindley, 2004), were taken. Measurements of soil pH and electrical conductivity (EC) were also taken to assess any potential impacts of brush packs on the chemical properties of the soil.

Tongway and Ludwig (1996) found brush packs improved stability, infiltration, and nutrient cycling, with increased soil respiration and carbon levels. They also found no changes in pH or EC levels. Based on these results it was expected that brush packs would improve LFA scores, soil respiration, and carbon levels, even in a temperate, mesic climate, and they would not impact pH or EC levels.

1.1 Research Aims and Hypotheses

This study aimed to assess the impacts of brush packs on landscape function (and hence their potential to restore degraded landscapes), with a specific focus on nutrient cycling, at a degraded, heavily grazed, mostly bare ground site with a temperate, mesic climate in southeastern Australia. This study examined both the fine-scale and landscape-scale impacts of brush packs. At the fine-scale, it examined the direct, localised effects occurring immediately beneath the brush packs. At the landscape-scale, it examines whether these localised effects extend beyond the brush packs to impact the surrounding landscape. While positive impacts at the fine-scale level are beneficial, their overall impact is limited if these effects do not extend throughout the broader landscape. By understanding both scales, this research

aims to provide a comprehensive assessment of the effectiveness of brush packs as a rehabilitation technique in restoring landscape function within the context of this degraded site.

At the landscape-scale, LFA scores for brush pack transects were compared with control transects (at the 'transect-level'). At the fine-scale, LFA scores and direct measurements (soil respiration, carbon, pH, EC) for brush pack patches were compared to interpatches (at the 'zone-level').

Based on the results of Tongway and Ludwig (1996), brush packing was expected to improve LFA index scores of landscape function at both the landscape-scale and fine-scale. Improvements in soil respiration and soil carbon levels underneath brush packs were also expected, with no expected changes in the chemical properties of the soil.

1.1.1 Aims

1. Assess the landscape-scale impact of brush packs on landscape function:
    1.1. Evaluate changes in LFA scores at the transect-level.

2. Assess the fine-scale impact of brush packs on landscape function:
    2.1. Evaluate changes in LFA scores at the zone-level.
    2.2. Evaluate changes in nutrient cycling by directly measuring:
        2.2.1. Soil respiration rates.
        2.2.2. Soil carbon levels.
    2.3. Evaluate changes in soil pH and EC levels.

1.1.2 Hypotheses

1. Brush packs will improve landscape function at the landscape-scale:
    1.1. Brush packs will improve LFA scores at the transect-level.

2. Brush packs will improve landscape function at the fine-scale:
    2.1. Brush packs will improve LFA scores at the zone-level.
    2.2. Brush packs will improve nutrient cycling through increased:
        2.2.1. Soil respiration rates.
        2.2.2. Soil carbon levels.
    2.3. Brush packs will not change pH or EC levels.

1.2 Thesis Outline

This thesis is organised into the following chapters:

Chapter 2: Literature review

- Explores and defines landscape degradation and landscape function

- Defines and synthesises the literature on brush packs
- Explains the choice of method

Chapter 3: Methodology and Methods

- Details the study sites and research design
- Describes the methods used to collect and analyse data

Chapter 4: Results

- Presents the results

Chapter 5: Discussion

- Discusses the results
- Draws upon literature
- Considers implications and limitations
- Suggests directions for future research

Chapter 6: Conclusion

- Summarises key findings

Chapter 2: Literature Review

2.1 Landscape Degradation

2.1.1 Problem Introduction

Landscape degradation is a significant global issue. It leads to a decline or loss of ecosystem processes that support biodiversity, land productivity, and ultimately human well-being (Dobson et al., 1997; Eldridge & Delgado-Baquerizo, 2017; Kertész & Krecek, 2019; Smiraglia et al., 2016). The causes of landscape degradation stem from both natural and human-induced factors, including severe climate events (often intensified by human-induced climate change), agricultural expansion, deforestation, and unsustainable land use practices (Bai et al., 2008; Blaikie & Brookfield, 2015). As the Earth's population continues to grow, pressures on landscapes will intensify, further exacerbating landscape degradation linked to land use changes and human activity (Jha & Bawa, 2006; Meyer & Turner, 1992).

In Australia, extensive areas of land are dedicated to livestock grazing. The introduction and expansion of grazing has had many impacts primarily due to poor grazing management leading to overgrazing (Eldridge & Delgado-Baquerizo, 2017). This has led to soil degradation, erosion, compaction, and significant declines in natural vegetation cover, all of which reduce landscape function and biomass production (Eldridge et al., 2016; McIntyre & Tongway, 2005). These issues are further exacerbated during periods of drought (McKeon et al., 2004).

In some cases, if grazing disturbances are removed, landscapes can gradually recover due to their partially intact function or regenerative capacity (Drewry, 2006). However, if disturbance continues or intensifies (e.g. through ongoing grazing from both domestic livestock and wild herbivores or compounding events such as grazing plus drought) the damage to regenerative capacity may push the landscape across a critical threshold where it enters a new degraded state (Groffman et al., 2006; López et al., 2011). In this new state, the landscape lacks the capacity to heal itself and return to its previous state. To repair function after this threshold has been crossed, costly restoration efforts are necessary (Groffman et al., 2006; Hobbs & Harris, 2001; McIntyre & Lavorel, 2007). Therefore, intervention before this threshold is essential to avoid more costly restoration later.

To address these challenges, strategies that restore landscape function in grazing lands before landscapes cross a critical threshold of degradation are necessary. However, developing restoration strategies requires a clear understanding of how landscape degradation is defined and assessed.

2.1.2 Defining and Assessing Landscape Degradation

There is considerable variation in how landscape degradation is defined and assessed (Gibbons & Freudenberger, 2006; Hobbs, 2016; Ruiz-Jaen & Aide, 2005; Tongway & Ludwig, 2011). In general, most authors use one or more of the following three aspects of the biotic (e.g. vegetation) and abiotic

(e.g. soil) elements of the landscape: (1) composition, (2) structure, and (3) function (Eldridge et al., 2016; Noss, 1990; Ruiz-Jaen & Aide, 2005; Society for Ecological Restoration International Science & Policy Working Group, 2004; Thackway & Lesslie, 2008).

Composition includes the taxonomic array (species composition) and diversity of species (species richness) present in the landscape; structure includes the type and spatial arrangement of vegetation cover in the landscape; and function, as described in the previous section, includes the capacity of the landscape to capture and retain resources (Society for Ecological Restoration International Science & Policy Working Group, 2004). Each of these aspects overlap and are interconnected; structural and compositional shifts often impact function, e.g. vegetation cover or patchiness increases landscape function, and different species can function differently such as seasonal differences between annual and perennial plants (Briske et al., 2005; Eldridge et al., 2016). The assessment and prioritisation of these three aspects often depends on the values or goals applied to the landscape.

The definition of landscape degradation is heavily value-dependent as it inherently involves a comparison: degraded relative to what? (Gibbons & Freudenberger, 2006). This definition depends on the value system applied, which is influenced by many factors including intended land use and ecological context (Hobbs, 2016). For example, a pasture used for grazing may be considered heavily degraded from a compositional perspective compared to a Nature Reserve due to limited species diversity, however, from a functional perspective it may be performing well as a pastoral system if it is successfully capturing and retaining resources. Therefore, the prioritisation or weight given to each aspect; composition, structure, and function, ultimately depends on the applied value system.

This study focused on the functional aspect of landscape degradation, as brush packs aim to restore landscape function. Landscape function refers to the fundamental processes that enable a landscape to capture, store, and retain resources such as soil, water, and organic matter (Tongway & Ludwig, 2011). These processes, including stability, infiltration, and nutrient cycling, are driven by the biotic (e.g. vegetation) and abiotic or physical (e.g. soil) elements of the landscape (Tongway & Hindley, 2004). Landscape function provides the capacity for landscapes to produce biomass, including vegetation, from rainfall and other resource inputs (Bastin et al., 1993). Thus, when a landscape is functional, it is self-renewing or autogenic (Society for Ecological Restoration International Science & Policy Working Group, 2004), hence function is equated with the regenerative capacity of the landscape (Thackway & Freudenberger, 2016). Restoring a landscape's ability to retain resources and restore itself after disturbance is the first step in rehabilitating a degraded landscape (Tongway and Ludwig, 2011, Hobbs and Harris, 2001, Whisenant, 1999). Brush packs aim to restore this landscape functionality and regenerative capacity, helping to kickstart the natural regeneration of structure, composition, and further improvements in function.

Although assessing all three aspects (composition, structure, and function) provides a more reliable and comprehensive assessment of landscape degradation (Ruiz-Jaen & Aide, 2005; Society for Ecological Restoration International Science & Policy Working Group, 2004), it also increases the complexity, time, and cost. Focusing solely on function reduces time and cost while still assessing the most fundamental processes in a landscape (Tongway & Ludwig, 2011).

Reference sites provide useful benchmarks for comparison when evaluating landscape function (Gibbons & Freudenberger, 2006; Society for Ecological Restoration International Science & Policy Working Group, 2004). They are ideally undisturbed or minimally disturbed landscapes that represent a best-case scenario, in terms of landscape health (Ruiz-Jaen & Aide, 2005). To appropriately contextualise assessments of degradation and set rehabilitation goals reference sites should be in close proximity to and exposed to similar conditions as the site being assessed (Hobbs & Harris, 2001; Ruiz-Jaen & Aide, 2005; Society for Ecological Restoration International Science & Policy Working Group, 2004). The Reference Site used in my study meets these criteria (Section 3.1.2).

2.2 Restoring Landscape Function

Efforts to restore landscape function have a long history, with people intuitively attempting to fix eroding landscapes long before formal documentation existed (Heede, 1976; Trimble, 1985). For at least the past few centuries, simple practices such as using rock or brush structures to control gully erosion have been commonly employed worldwide (Gellis et al., 1995; Guyassa et al., 2018). More sophisticated techniques including contour ripping in the United States and live fascines in France and China have also been applied (Didier et al., 2023; Gifford et al., 1977).

One such technique that has been used to restore function is the brush pack. The term brush pack broadly refers to a landscape rehabilitation technique which aims to stabilise soil, reduce erosion, and promote vegetation regrowth by creating a favourable environment for plants. It involves placing and packing branches or brush along the contours of bare, gently sloping areas to improve landscape function by capturing and retaining resources, such as soil, water, and plant detritus (Tongway & Ludwig, 1996).

Similar terms in the literature describe various types of erosion barriers, including “cut-shrub barriers”, “contour branch barriers”, “brushwood dikes”, “brushwood fences”, and others (Aristeidis & Vasiliki, 2015; Eichmanns et al., 2021; Fernández et al., 2011; Marques & Mora, 1998). Some of these barriers are strikingly similar to brush packs, some share a few key characteristics, and some differ vastly. Table 2.1 displays the range of terms used for similar techniques and highlights the inconsistent nomenclature in the literature.

Table 2.1 Summary of similar terms in the literature for brush pack-like rehabilitation techniques that use fine-medium woody debris to restore landscape function. Terms are grouped by primary objective, but there is some overlap in planned or

unintended secondary objectives or co-benefits where techniques may serve additional purposes in various contexts.

Primary Objective	Similar Terms Used	References
Interception of overland flows, trapping of soil, water, and litter	• "branch bundles" • "branch piles" • "brush fencing" • "brush packs" • "brush-packs" • "brushpacking"	(Kimiti et al., 2017; Milton & Coetzee, 2022; Naude, 2017; Pelser, 2017; Tongway & Ludwig, 1996; van den Berg & Kellner, 2005)
Grazing exclusion	• "brush packs"	(Koch et al., 2021; Kwaza et al., 2020)
Post-fire erosion control	• "branch piles" • "brushwood dikes" • "contour branch barriers" • "cut-shrub barriers" • "erosion barriers" (made from branches)	(Aristeidis & Vasiliki, 2015; Fernández et al., 2019; Fernández et al., 2011; Marques & Mora, 1998; Myronidis et al., 2010)
Riverbank and oversteepened slope stabilisation	• "brush layers" • "fascines" • "live palisades" • "wattle fencing"	(Didier et al., 2023; Petrone & Preti, 2010; Polster, 2002; Sotir & Fischenich, 2001)

The brush packs used in this study were not designed as large-scale, robust structures aimed at trapping large amounts of sediment in the context of post-fire erosion control or riverbank and oversteepened slope stabilisation (see examples in Table 2.1). Nor were they designed exclusively as grazing exclosures, such as those established with sparsely arranged, thorny branches with no ability in themselves to trap sediment (see examples in Table 2.1). Instead, they were designed to act as small-scale, simple semi-permeable barriers or sieves that gently intercept, capture, and retain resources from overland flows (Ludwig et al., 1994; Tongway & Ludwig, 1996).

In semi-arid landscapes, concentrating resources in patches is essential due to their limited availability; if rainfall and nutrients were evenly spread, overall productivity would be lower than if these resources were concentrated in patches (Ludwig et al., 1994; Tongway & Ludwig, 1994). Because of this, most subsequent studies on brush packs have focused on restoring function in semi-arid or arid grazing lands, where limited resources make resource capture and retention essential (Milton & Coetzee, 2022; Naude,

2017; Pelser, 2017; van den Berg & Kellner, 2005). However, it is unclear whether this technique will be equally effective in other landscapes, particularly those where resources are less limited and more abundant. To address this uncertainty, this research, conducted in collaboration with The Mulloon Institute, investigated the use of brush packs in a temperate, mesic landscape in southeastern Australia.

Beyond their primary objective of intercepting and capturing resources from overland flows, brush packs also offer several ancillary benefits to landscape function (Naude, 2017; Smith et al., 2007; Tongway & Ludwig, 1996). They provide:

- Immediate soil cover and protection against rainsplash erosion when placed on bare ground.
- Organic matter input by dropping litter.
- Protection from grazing, preventing grass from being grazed back to bare ground.
- Shelter and food for macroinvertebrates.

2.3 How Brush Packs Restore Function

A landscape can be viewed as a system where resources flowing into, out of, and within it are mediated by ecological processes. These processes and flows determine the system's gains and losses, and thus its overall landscape function. Brush packs aim to restore landscape function by increasing the capacity of a landscape to capture and retain resources such as soil, water, and organic matter, thereby increasing gains within the system (Tongway & Ludwig, 1996, 2011). This concept is illustrated by The Trigger-Transfer-Reserve-Pulse (TTRP) Model (Figure 2.1), presented in Tongway and Ludwig (2011), which is a useful conceptual framework for understanding general landscape systems.

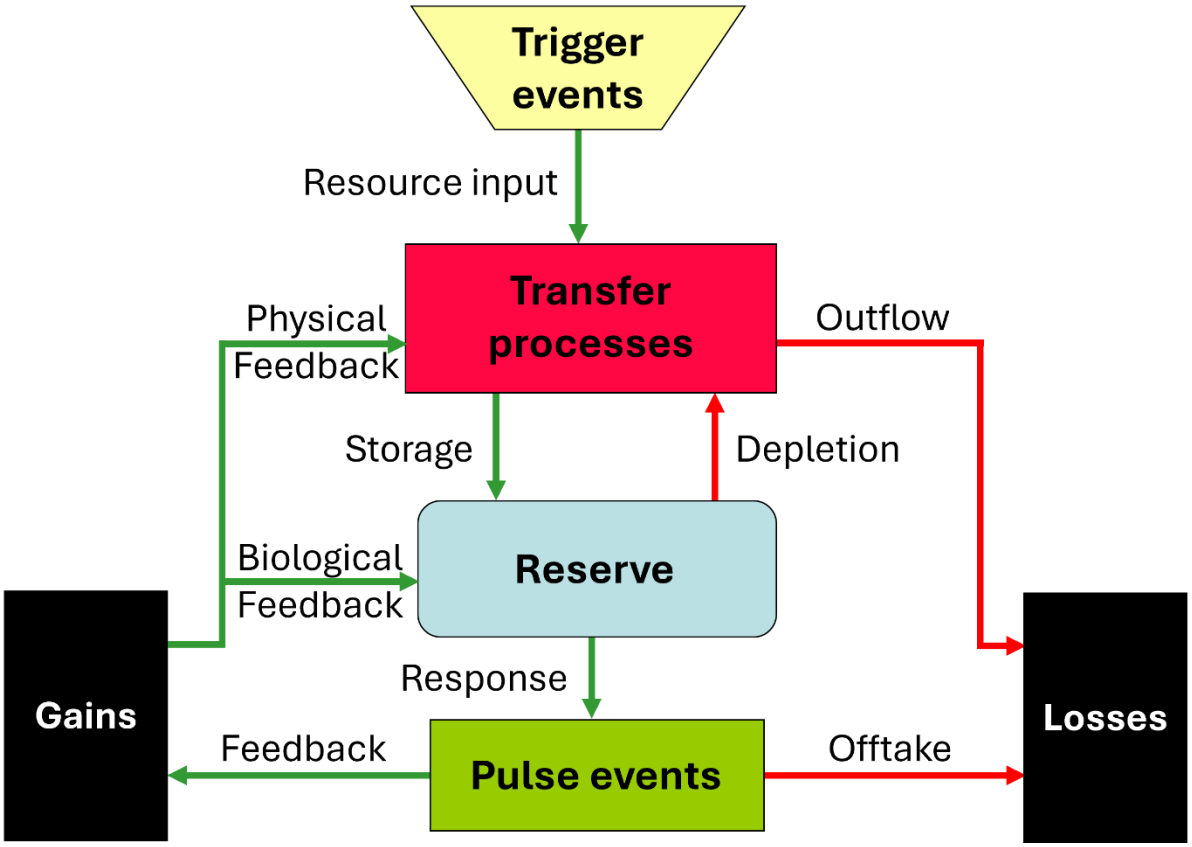

Figure 2.1: The Trigger-Transfer-Reserve-Pulse (TTRP) Model is a conceptual framework for understanding general landscape systems. It illustrates the flow of resources into, out of, and within a landscape. It consists of four components: (1) Trigger events that input resources, (2) Transfer processes that direct resources into storage or losses, (3) the Reserve where resources are stored in the system, and (4) Pulse events generated from the resources stored in the reserve. Gains (green arrows) represent resource capture and retention, while Losses (red arrows) represent resource loss or leakage. The balance between Gains and Losses determines whether a landscape is functional or dysfunctional. Modified from Figure 2.12; Tongway and Ludwig (2011).

The core of the TTRP Model is divided into four main components:

(1) **Trigger events** which provide *resource input* into the system (e.g. rainfall). These trigger various (2) **Transfer processes** which are either lost through *outflow* (e.g. runoff) or gained and *stored* in the (3) **Reserve** (e.g. infiltration). The reserve is the reservoir of resources stored in the landscape and includes resources stored in both the biotic (e.g. vegetation) and abiotic (e.g. soil) elements of the landscape. Resources can be lost from the reserve through *depletion* transfer processes (e.g. erosion), or if sufficient resources and other favourable conditions are present, they can generate a *response* via (4) **Pulse events** (e.g. vegetation growth). Some of what is produced by a pulse event is lost through *offtake* (e.g. grazing), but some is cycled back through the landscape through *feedback* processes. Feedback processes include *biological feedback*, such as gains in biomass and composition which

remain in the reserve, and *physical feedback*, such as increased vegetation patches, structure, and function which increase the capacity of the reserve.

Bookending the TTRP core are the gains and losses of resources from the system. Gains (represented by green arrows) show the successful storage and retention of resources within the landscape, whereas losses (represented by red arrows) show the transfer of resources out of the landscape. Gains and losses from a landscape system fluctuate over time, driven by varying conditions and disturbances such as droughts and periods of high precipitation. In a functional landscape, these fluctuations balance out over the long-term (Figure 2.2a). However, in a dysfunctional landscape they become imbalanced, with losses outweighing gains (Figure 2.2b).

Disturbances such as overgrazing can increase losses through offtakes out of the landscape system. This, in turn reduces the feedback and gains back into the system and diminishes the landscape's ability to capture and retain resources during future trigger events. Over the long-term, this leads dysfunction through the imbalance of gains and losses in the system.

Brush packs aim to restore this balance by improving physical feedback within the system, primarily through intercepting and capturing resources from overland flows. Which in turn increases the feedback and gains back into the system and increases the ability of the landscape to respond to future trigger events such as rainfall. This can help prevent the long-term imbalance of gains and losses which leads to dysfunction.

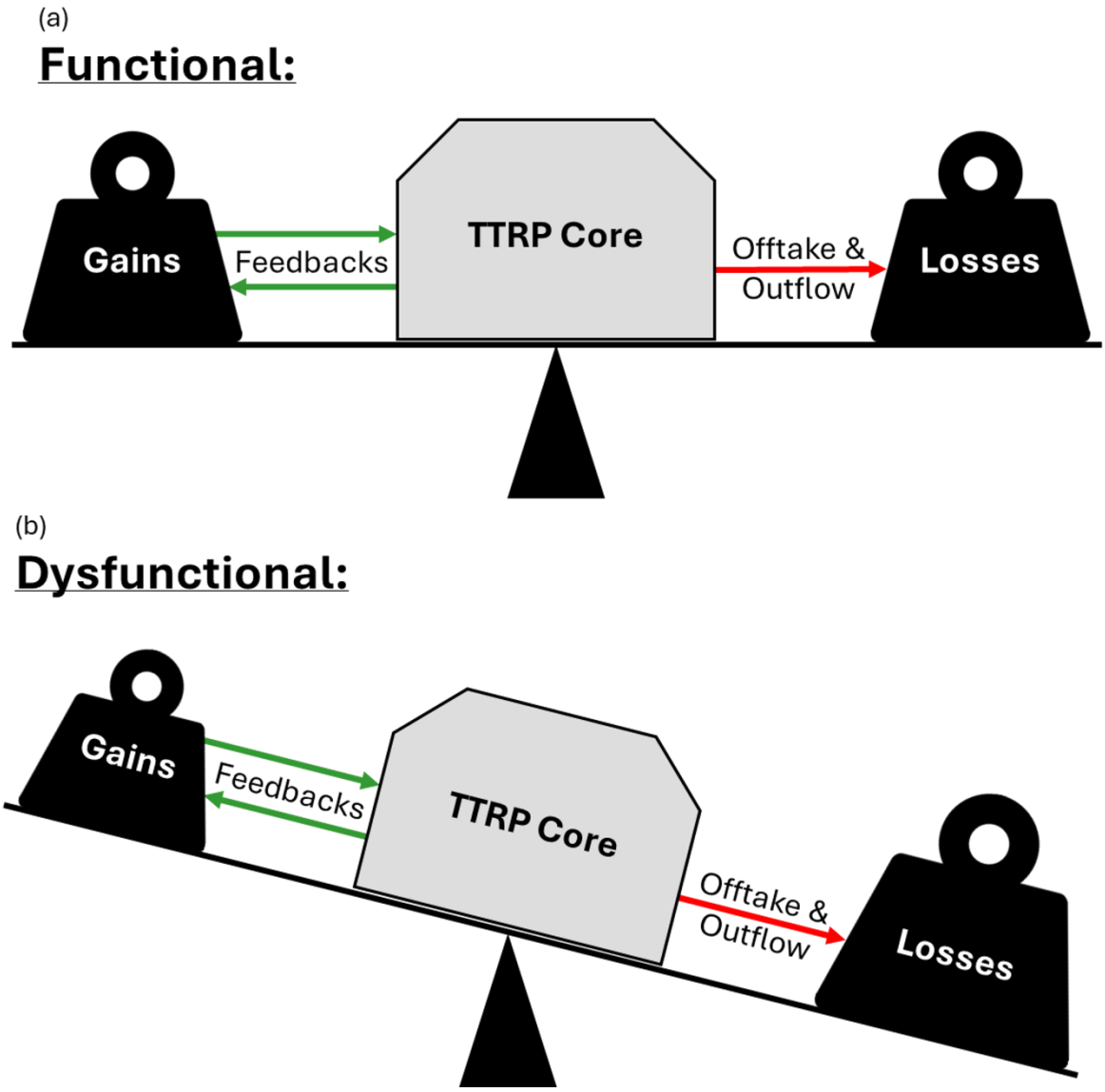

Figure 2.2 Long-term balance of TTRP Model. A landscape system shown in states of (a) balance and proper function when gains and losses are equal. And (b) imbalance and dysfunction when losses outweigh gains, and the landscape is leaking resources. Modified from Figure 2; Ludwig and Tongway (2000).

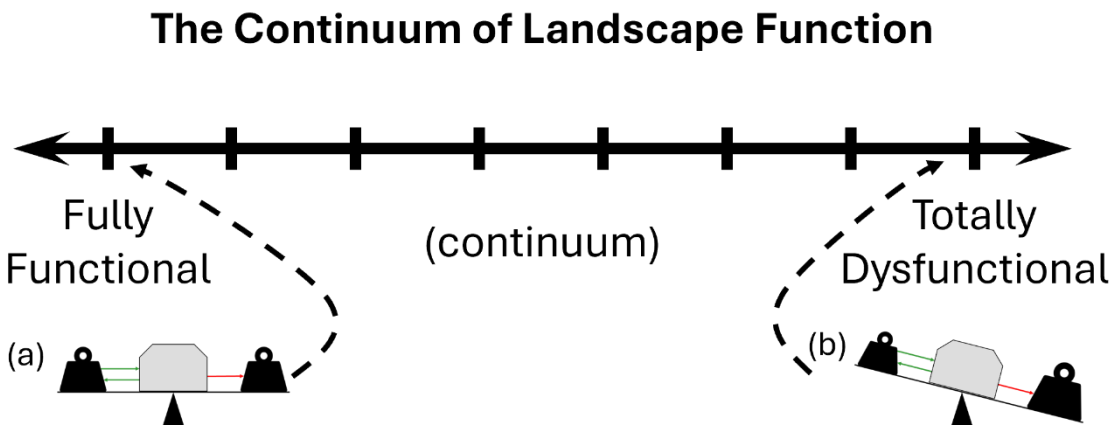

Figure 2.3 Landscape function exists on a continuum. Subtle fluctuations in resource flows due to varying conditions and disturbances cause gradual shifts toward function or dysfunction. Modified from Figure 3; Ludwig and Tongway (2000).

It is important to note that landscape function exists on a continuum. Multiple ecological processes and opportunities for resource flows contribute to landscape function (Figure 2.1). Each of these processes fluctuate at different rates and magnitudes in response to varying conditions and disturbances (Tongway & Ludwig, 2011). These fluctuations can cause subtle shifts in landscape function, resulting in a continuum of landscape function. Hence, Figure 2.2a and Figure 2.2b do not represent a binary system, instead they exist at opposite ends of a sliding scale from functional to dysfunctional (Figure 2.3). Recognising landscape function as a continuum allows for the monitoring of these subtle shifts either towards dysfunction from disturbances such as overgrazing or progress towards function through rehabilitation efforts such as establishing brush packs (Tongway & Ludwig, 2011).

While recognising that landscape function exists on a continuum accurately reflects its nature and provides a scale by which rehabilitation progress can be measured, it is also useful to recognise critical thresholds or tipping points along this continuum. Thresholds represent changes or transitions between states, which, once crossed, are often difficult and costly to reverse (Groffman et al., 2006; Hobbs & Harris, 2001; McIntyre & Lavorel, 2007). The most significant threshold is where landscape function has deteriorated to such an extent that regenerative capacity is crippled (Archer & Stokes, 2000; Briske et al., 2006; Hobbs & Harris, 2001; Tongway & Hindley, 2000). At this ‘functional threshold’, even if disturbance is removed, the landscape lacks the capacity to heal itself or spontaneously return to its former functional state. (Archer & Stokes, 2000; Hobbs & Harris, 2001; Tongway & Hindley, 2000; Whisenant, 1999). Understanding threshold points allows managers to anticipate and avoid difficult-to-reverse change (Groffman et al., 2006). Brush packs can help restore functionality in landscapes that have crossed this threshold, but they can also be used as a preventative measure to avoid crossing it.

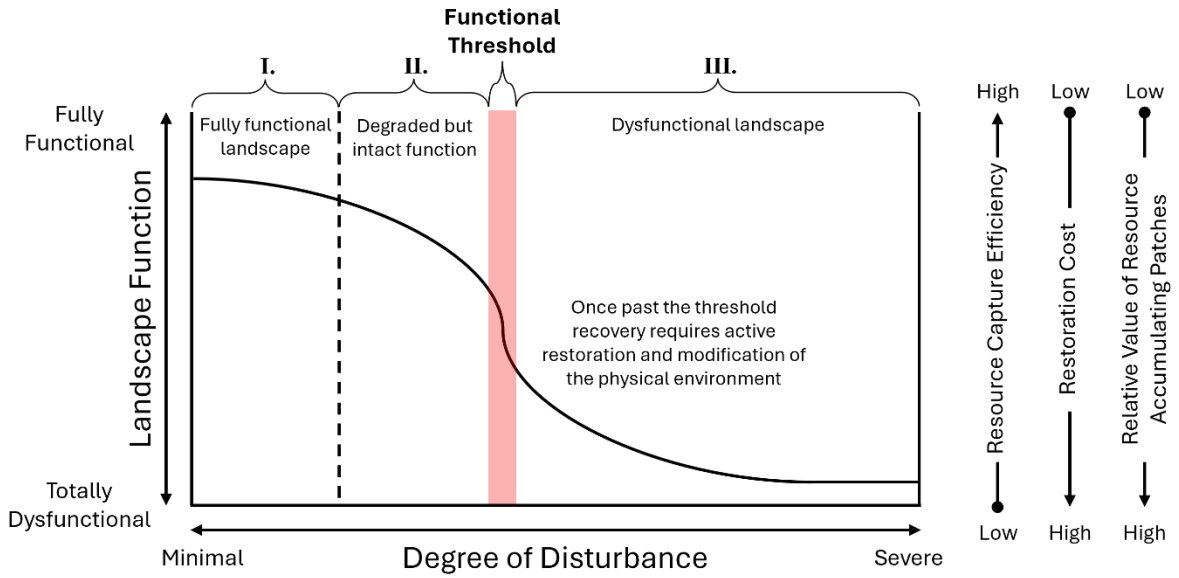

Figure 2.4 Conceptual diagram illustrating the continuum of landscape function with three states: (I) fully functional, (II) degraded but functional, and (III) dysfunctional. The 'functional threshold' marks the tipping point beyond which the landscape loses its capacity for unassisted recovery. Once this threshold is crossed, passive restoration becomes ineffective, active restoration is required, resource capture efficiency is low, restoration cost is high, and the relative value of resource-accumulating patches is high. Modified from Figure 1.1; Whisenant (1999) & Figures 3 and 4; (Tongway & Hindley, 2000).

Figure 2.4 illustrates the continuum of landscape function with three main states: a fully functional state (**I.**), an intermediate state with degraded but intact function (**II.**), and a dysfunctional state (**III.**). The 'functional threshold' is the tipping point before the dysfunctional state. Once crossed, the landscape has lost the capacity to spontaneously return to its previous state. The 'functional threshold' concept is widely recognised in the literature (Arnalds & Archer, 2000; Briske et al., 2006; Hobbs & Harris, 2001; Li et al., 2013; López et al., 2011; Tongway & Ludwig, 2011; Whisenant, 1999).

After this threshold:

- Resource capture efficiency is low across the landscape (Whisenant, 1999).
- Restoration cost is high, and passive restoration is ineffective leaving only active restoration (Briske et al., 2006).
- The relative value of resource-accumulating patches is high (Whisenant, 1999). With landscape function severely compromised, the relative value any single patch that captures and retains resources increases. Whereas in a functional landscape where resource capture efficiency is high, the relative value of each patch is proportionally lower. This also applies to brush packs which act as resource-accumulating patches.

If disturbances at a landscape in State **II.** continue or intensify, the landscape will cross the threshold and transition into **State III.**, a dysfunctional state. This transition is highly undesirable. Tongway and

Hindley (2000) suggest two management options for landscapes in State **II.**, either remove the disturbance until the ecosystem can heal itself back to State **I.** or introduce a mechanism to improve resource capture efficiency. Tongway and Ludwig (1996) demonstrated one such mechanism is brush packs.

2.4 Justification of Approach to Measuring Landscape Function

The LFA method (Tongway & Hindley, 2004), which specifically examines landscape function, was selected as the primary approach for this study because it is rapid, inexpensive, and requires minimal technical expertise. This made it especially well-suited for the student-led data collection in 2018. Also, as this method does not involve sampling or laboratory analysis, large amounts of data could be collected while minimising both time and cost expenditure.

Furthermore, the LFA method is well-studied and widely accepted. Its three indices and 11 soil surface indicators are shown to correlate with soil function across a wide range of environments (Eldridge et al., 2020; Ludwig et al., 2023; Maestre & Puche, 2009; McIntyre & Tongway, 2005; Read et al., 2016; Tongway & Hindley, 2004; Zucca et al., 2013). Numerous other studies have also applied the LFA method to assess landscape rehabilitation efforts (de Luna et al., 2022; De Simoni & Leite, 2019; McDonald et al., 2018; Munro et al., 2012; Pelser, 2017; Read et al., 2016). Details on the application of LFA in this study are provided in Section 3.1.1.

Direct measurements of soil respiration and carbon levels were selected to further investigate differences observed in the LFA nutrient cycling index. These measures were used by Tongway and Hindley (2004) to verify the nutrient cycling index. In their study on brush packs Tongway and Ludwig (1996) also measured soil respiration and carbon levels, allowing for comparison with their findings. Numerous other studies have also employed these direct measurements to assess landscape rehabilitation efforts (Banning et al., 2008; Ingram et al., 2005; Oelbermann et al., 2015; Santini et al., 2019; Shi et al., 2020; Wen et al., 2018). Details on the measurement of soil respiration and carbon levels are provided in Sections 3.3.2 and 3.3.3.1, respectively.

Chapter 3: Methodology and Methods

3.1 Study Sites

3.1.1 Main Site

The Main Site for this study was in a 68-hectare paddock located in Mulloon, NSW, at coordinates: 35.26746111° S, 149.61894167° E (Figure 3.1). It has a temperate, mesic climate. Over the past 10 years it has had a mean annual rainfall of 732mm, ranging from a minimum of 210mm to a maximum of 1253mm. With average monthly temperatures from ranging from 11–27 °C in January and -1–11 °C in July (Appendix 1 – Home Farm Climate Data). The site has a gentle slope of 5° with a west-facing aspect (280°). In 2018, the Main Site was a heavily grazed, mostly bare ground sheep camp showing severe signs of erosion in some places (Figure 4.1). At the present day, 2024, the Main Site shows no signs of erosion and has no bare ground (Figure 4.3). It has a mix of perennial and annual grasses including *Danthonia* species and *Bothriochloa macra*, with some sparse *Eucalyptus* species trees across the site.

The grazing pressure has changed significantly over the duration of the study. From the 1970s until 2021, the paddock supported a consistent flock of 300 sheep, this number was maintained through lambing. After 2021 the size of the flock gradually decreased to 100 in early 2024, after which all sheep were removed. Cattle were introduced to the site in 2021 and have been there since. The cattle herd size has varied, at minimum a dozen, but up to 30 (Cantwell, pers. comm. 2024). There was evidence of cattle activity at the Main Site in 2024. There was some activity from wildlife and feral animals including kangaroos, rabbits, pigs, and wombats visible at and around the site over the timespan of the study. Overall, the grazing pressure from domestic stock decreased during the study.

The soil is derived from a granite base, with a sandy loam texture and is classified a Kurosol according to the Australian Soil Classification (State Government of NSW and NSW Department of Climate Change, 2024). Throughout the site small fragments of charcoal were present in the soil up to a depth of 10cm. These are likely remnants of weed control burns from 15+ years ago (Cantwell, pers. comm. 2024).

3.1.2 Reference Site

The Reference Site was established in 2024, to serve as a benchmark of a functional, least disturbed landscape to compare with the Main Site. The importance of reference sites is discussed in Section 2.1.2.

The Reference Site is located in an area of least disturbed woodland approximately 3 kilometres west of the Main Site, on the west side of Mulloon Creek in Mulloon, NSW, at coordinates: 35.26076111° S, 149.58511111° E (Figure 3.1). It has the same climatic conditions and soil as the Main Site, though

the soil at the Reference Site is derived from a mixed sedimentary base (State Government of NSW and NSW Department of Climate Change, 2024). The site has a gentle slope of 5° and an east-facing aspect (90°). A mix of tree and grass species are present, with *Eucalyptus mannifera* as the dominant tree and *Rytidosperma pallidum* as the dominant grass in that area (Thackway, 2019). While it has not been subject to clearing or livestock grazing since the 1980s, it has a history of clearing and livestock grazing spanning at least the past century (Hazell, pers. comm. 2024).

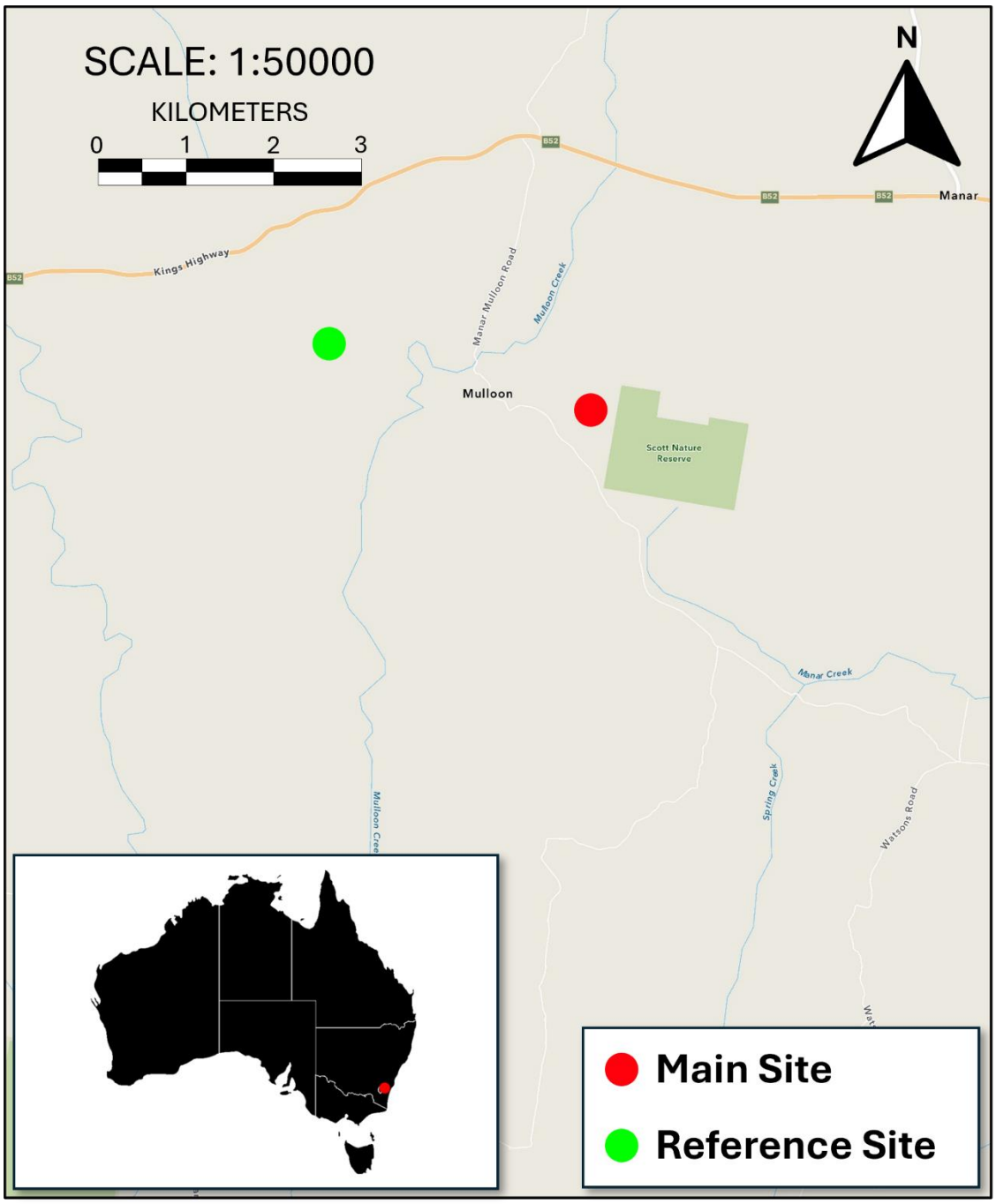

Figure 3.1 Site map. The Main Site and Reference Site are located in Mulloon, NSW, just a few kilometres from Kings Highway.

3.2 Experimental Design

This research was conducted through a formal partnership between the Australian National University (ANU) and The Mulloon Institute (TMI) as part of their Mulloon Rehydration Initiative (Peel et al., 2022).

In 2018, at the Main Site, 6 transects were established by ANU students under the guidance of David Freudenberger (ANU) and Luke Peel (TMI). Each transect was 50m long, running downslope, and spaced approximately 10m from each other to minimise differences caused by landscape position. The start and finish of each transect was marked with a hardwood peg and aluminium numbered tag. Initial (pre-brush pack) LFA measurements were conducted on all 6 transects. After these initial measurements, 5 of the 6 transects were designated to receive brush packs and final (post-brush pack) LFA measurements were taken from the 5 transects. The remaining transect remained untreated to serve as a control.

Between 3 to 5 brush packs were established at random points along each brush pack transect. A total of 20 individual brush packs were installed at the site. The material used consisted of freshly pruned branches collected from the routine clearing of nearby *Leptospermum* species, which were then transported approximately 100m by foot to the site. The brush packs were packed into the ground and secured with wooden stakes. They were also constructed with the branches laying perpendicular to the transect, or along the contour to maximise interception of overland flows. They had heights of 40-70cm, widths (perpendicular to the transect) of 6-10m, and lengths (along the transect) of 1.5-2m. Due to installation being carried out by students, there was minor variation in the heights, widths, lengths, number, and spacing of brush packs along each transect.

In 2018, at the Main Site, 6 control and 5 brush pack transects, a total of 11 transects, were measured by ANU Fenner students trained in LFA during the Environmental Field School course (ENVS2018). In 2019, at the Main Site, David Freudenberger and Luke Peel took LFA measurements from 1 control, and 3 brush pack transects.

In 2024, at the Main Site, Alexander Harrison and David Freudenberger, established 5 additional control transects, along with 6 reference transects at the Reference Site. All transects newly established in 2024 maintained the same 50 metre length, 10 metre spacing, and downslope orientation as in 2018. In 2024, the study included a total of 6 control, 5 brush pack, and 6 reference transects. LFA measurements were taken at each transect. Further details of the LFA measurements are provided in Section 3.3.1.

Also in 2024, 20 ‘control patches’ were established at random points in interpatch zones along the control transects to match the 20 individual brush packs placed at random points along brush pack transects in 2018. Each brush pack was classified as its own patch. These patches were established for fine-scale measurements. In total there were 20 control and 20 brush pack patches as part of this study, all located at the Main Site. Measurements of soil respiration and nutrient and chemical properties were

randomly taken from 10 control and 10 brush pack patches. Further details on these measurements are provided in Sections 3.3.2 and 3.3.3, respectively.

Measurements were taken in 2018, 2019, and 2024 (Table 3.1). Personnel availability limited the frequency, breadth, and depth of these measurements, particularly in 2019 where only 4 transects were measured.

Table 3.1 Summary of measurements conducted across years at the study sites. Measurements include Landscape Function Analysis (LFA), soil respiration, and soil nutrient and chemical properties including carbon (C), pH, and electrical conductivity (EC).

Date	Measurement Type	Details	Fieldwork Personnel
Sep, 2018	LFA	• 6x control transects • 5x brush pack transects	ANU students, David Freudenberger, Luke Peel
May, 2019	LFA	• 1x control transect • 3x brush pack transects	David Freudenberger, Luke Peel
Apr, 2024	LFA	• 6x control transects • 5x brush pack transects • 6x reference transects (located at Reference Site)	Alexander Harrison, David Freudenberger
Jul, 2024	Soil Respiration	• 10x control patches • 10x brush pack patches	Alexander Harrison
Aug, 2024	Soil Nutrient and Chemical Properties (C, pH, EC)	• 10x control patches • 10x brush pack patches	Alexander Harrison, Peter Sharp

3.3 Data Collection

3.3.1 Landscape Function Analysis

Measuring and interpreting soil properties can be labour intensive, technically demanding, and costly, often requiring specialised knowledge, sampling, and laboratory analysis. This problem was addressed by using the Landscape Function Analysis (LFA) method, as it offers a rapid, inexpensive assessment that requires minimal technical expertise (Tongway & Hindley, 2004; Tongway & Ludwig, 2011), making it especially well-suited for the student-led data collection in 2018. See Section 2.4 for further justification of this approach.

The LFA method uses 11 simple soil surface assessment (SSA) indicators to calculate 3 synthetic soil surface indices: (1) Stability; the ability to resist erosion and reform after disturbance, (2) Infiltration; the capacity of the soil to absorb and retain water, and (3) Nutrient Cycling; how efficiently organic matter is cycled back into the soil, see Figure 3.2 (Tongway & Hindley, 2004; Tongway & Ludwig, 2011).

Each transect was divided into individual zones which were identified based on a distinct change in function as compared to the prior zone. Zones were also then classified as either patch or interpatch, following the definitions of Tongway and Hindley (2004). Patch zones are areas that capture and retain resources like sediment, detritus, and water, while interpatch zones allow these resources to flow through or escape. In this study, patch zones included e.g. brush packs and coarse woody debris, whereas interpatch zones consisted of e.g. bare ground, litter, and sparse grass.

The width, start, and end points of each zone along the transect were recorded to estimate the area each zone occupied. For each zone, all 11 SSA indicators were measured at least three times, where possible, to capture natural variability. These measurements were taken from a 1 metre query area that was representative of the broader zone being measured.

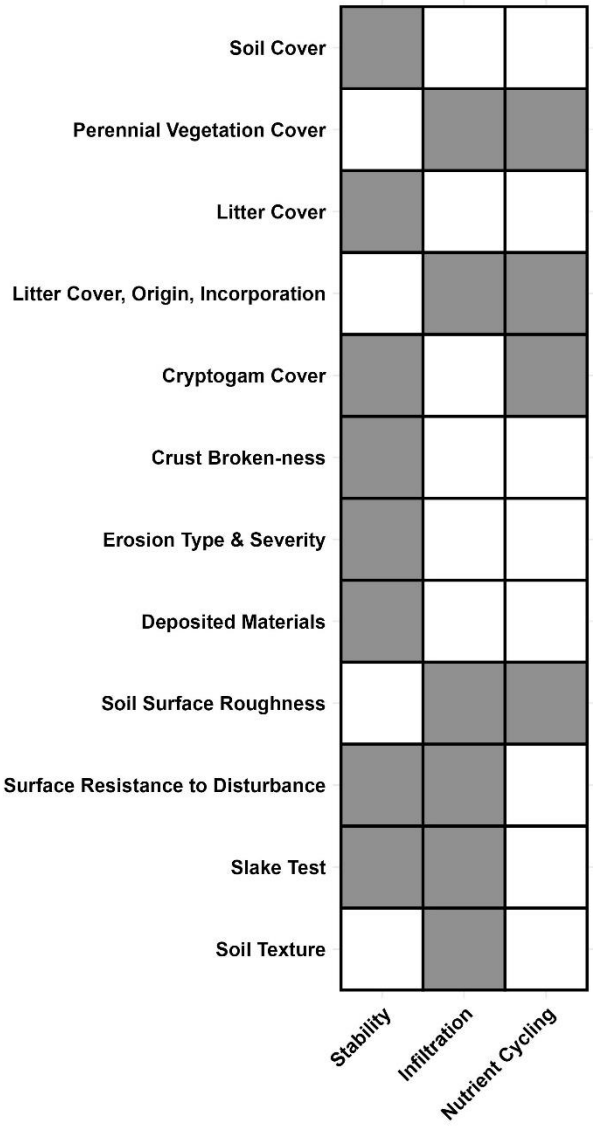

Figure 3.2 The soil surface assessment (SSA) indicators that contribute to the calculation of each of the three indices: Stability, Infiltration, and Nutrient Cycling. Grey boxes indicate that the SSA indicator contributed to the corresponding index, while white boxes indicate no contribution. Modified from Tongway and Hindley (2004).

After completing the analyses of the 2024 LFA data, results showed scores for the nutrient cycling index were significantly higher at brush pack patches compared to interpatches (Figure 4.8). This difference was found to be driven by a few key SSA indicators: soil surface roughness, litter cover, and litter incorporation (Figure 4.9). All of which contribute to the nutrient cycling index (Figure 3.2). To explore this difference in nutrient cycling further, soil respiration and carbon levels were investigated, following Tongway and Hindley (2004), who used these measures to verify the nutrient cycling index.

3.3.2 Soil Respiration

Soil respiration was measured through soil carbon dioxide (CO_2) efflux, it includes the CO_2 released from microbial and root respiration (Maier et al., 2011). Soil respiration was measured in this study due to differences in SSA indicators contributing to the nutrient cycling index, such as litter cover and incorporation, observed in the initial LFA measurements. Numerous studies have also demonstrated a correlation between the LFA nutrient cycling index, litter input, and soil respiration (Bréchet et al., 2018; Han et al., 2015; Maestre & Puche, 2009; Setyawan et al., 2011; Tongway & Hindley, 2004; Wei & Man, 2021).

Soil CO_2 efflux was measured in units of micromoles per square meter per second (μmol m^{-2} s^{-1}) using a LI-COR, LI-6400 Soil CO_2 Flux Chamber System. The ambient CO_2 level was set at 410ppm, as this was what was consistently measured at the site, and the measurement range (Δ) was set as ±10ppm. Wind-induced pressure changes can impact the CO_2 efflux reading (Healy et al., 1996), to minimise this, the device was shielded whilst taking measurements. One month before data collection, polyvinyl chloride (PVC) collars (5cm long, 11cm diameter) were inserted into the soil to prevent CO_2 flushing caused by soil disturbance (Wang et al., 2005). Variations in collar insertion depth can affect CO_2 efflux readings (Hui-Mei et al., 2005), to prevent this, the PVC collars were inserted consistently 2cm into the soil. 10 random control and 10 random brush pack patches were selected from the 20 available for measurement. All measurements were taken between 10am and 3pm to minimise the influence of diurnal fluctuations on soil respiration. Control and brush pack patches were alternated throughout each day to reduce the risk of confounding time-of-day effects on the comparison between treatments. Data collection occurred over three consecutive days (July 23–25) under consistent climatic conditions including temperatures and time since rainfall to minimise environmental variability and its impact on CO_2 efflux measurements. Each day measurements were taken at 20 patches, 10 control and 10 brush pack, with three collars per site and three cycles or repetitions per collar. On the 23rd of July one brush pack patch was missed due to power supply failure.

Soil temperature and moisture were also measured as covariates using a Spectrum Technologies, FieldScout TDR350 Soil Moisture Meter as they are known to influence CO_2 efflux (Maier et al., 2011; Moyano et al., 2012). For each site, temperature and moisture were recorded as the average of three separate measurements taken within 10cm of each collar to account for spatial variability. Temperature was recorded in degrees Celsius (°C), and moisture was recorded as percent volumetric water content (%VWC).

3.3.3 Soil Nutrient and Chemical Properties

To analyse and compare the nutrient and chemical properties of the soil a total of 10 control and 10 brush pack patches were randomly selected from the 20 available sites for each treatment type. Soil samples were taken at depth intervals of 0-1, 1-3, 3-5, and 5-10cm to capture the distribution of carbon

throughout the topsoil profile. Samples were taken from such shallow depths based on the assumption that any change in carbon levels over the timeframe of this study would occur near the soil surface, following Tongway and Ludwig (1996). Samples were collected carefully using a 75mm flat paint scraper. Each depth interval was collected by making a square shape and then scraping off a couple of millimetres of soil at a time, with care to keep the corners and middle of the square at an even level, as to not collect material from another depth interval. At each site, three samples were collected at each depth, spaced 1 metre apart within each patch. These samples were then combined (bulked) together to account for spatial heterogeneity, reducing the significance of random spatial variation. The samples were collected over two days, 29th July and 5th August, spaced within a week of each other. 5 control and 5 brush pack patches were sampled on each day. This sampling method aimed to minimise the risk of confounding effects due to differing sample dates.

After field sampling, the soil samples were refrigerated overnight to slow microbial processes that could alter nutrient and chemical properties. The next day, they were dried at 40°C for at least two days, see Appendix 2 – Chain of Custody (soil samples). This was done to prevent further microbial activity until the samples could be analysed.

Once dry, the soil samples were ground and sieved to <1mm. Although <2mm sieving is standard practice, the high plant matter content (e.g., sticks, leaves, and roots), particularly in brush pack soils at the 0-1cm depth, necessitated a finer sieve. Sieving to <1mm helped reduce this non-representative plant matter, which would otherwise contribute to sample heterogeneity skewing the results of nutrient and chemical analysis. This approach also meant the samples were more representative of the soil itself, which is what was this study intended to measure. These samples were then used for subsequent analysis, the methods of which are described in the following sections.

3.3.3.1 Soil Carbon

Soil carbon was measured in this study due to differences in SSA indicators contributing to the nutrient cycling index, such as litter cover and incorporation, observed in the initial LFA measurements. Numerous studies have also demonstrated a correlation between the LFA nutrient cycling index, litter input, and soil carbon levels (Eldridge & Delgado-Baquerizo, 2018; Eldridge et al., 2020; Reynolds et al., 2018; Setyawan et al., 2011; Tongway & Hindley, 2004; Xu et al., 2013; Xu et al., 2021).

Soil samples were fractionated following the method described by Buss et al. (2021). Soil was separated into three carbon pools: particulate organic carbon (POC), aggregate carbon (AggC), and mineral-associated organic carbon (MAOC). These pools have different levels of carbon stability. With POC being the least stable as it consists primarily of easily decomposable plant matter, AggC having medium stability as it includes carbon stored in aggregates and somewhat protected from decomposition, and MAOC being the most stable as it includes carbon that is bound to mineral particles and protected from

decomposition (Abramoff et al., 2018; Georgiou et al., 2022; Hemingway et al., 2019; Peng et al., 2017; Robertson et al., 2019) .

The purpose of fractionation was to isolate plant matter (e.g., sticks, leaves, and roots) in the POC fraction, which has very high carbon levels that could otherwise skew overall carbon measurements and overshadow smaller but meaningful differences in the soil carbon of the other more stable pools (AggC and MAOC). Also, analysing the carbon levels in the AggC and MAOC fractions provides insight into the stability or longevity of the carbon stored in the soil (Buss et al., 2021). The total carbon content of the original (unfractionated) soil was also measured.

The start weight of the original soil and final weights of each fraction were recorded each time the fractionation was completed. There were two minor deviations from the original method: (1) the soil was sieved to <1mm instead of <2mm, and (2) the rapid recalcitrance test to further separate AggC and MAOC into labile and recalcitrant fractions was not performed. See Figure 3.3 for an outline of the method with deviations from Buss et al. (2021). Unfortunately, the total carbon of the dissolved organic carbon (DOC) fraction could not be measured due to untimely equipment failure.

Soil total carbon was analysed using a Skalar, PRIMACS™ SNC-100 combustion analyser. Both the original (unfractionated) soil samples and three fractionated pools were assessed. Each time the combustion analyser was run, the original sample mass and detected carbon mass was recorded.

To halve the time and cost associated with the analysis, only total carbon was measured. This is because the combustion analyser cannot measure total organic carbon alone and requires two separate measurements of total carbon and total inorganic carbon to calculate total organic carbon (Equation 3.1).

Equation 3.1 Total organic carbon (TOC) from total carbon (TC) and total inorganic carbon (TIC) calculation

$$TOC = TC - TIC$$

The soil at the Main Site is non-calcareous, with no expected inputs of inorganic carbon during the study, as there are no inputs from the chemical or physical weathering of a calcareous parent material (Lorenz & Lal, 2022). Therefore, any observed changes in total carbon solely reflect changes in total organic carbon, driven by organic carbon inputs from brush packs and other decaying plant matter in the landscape.

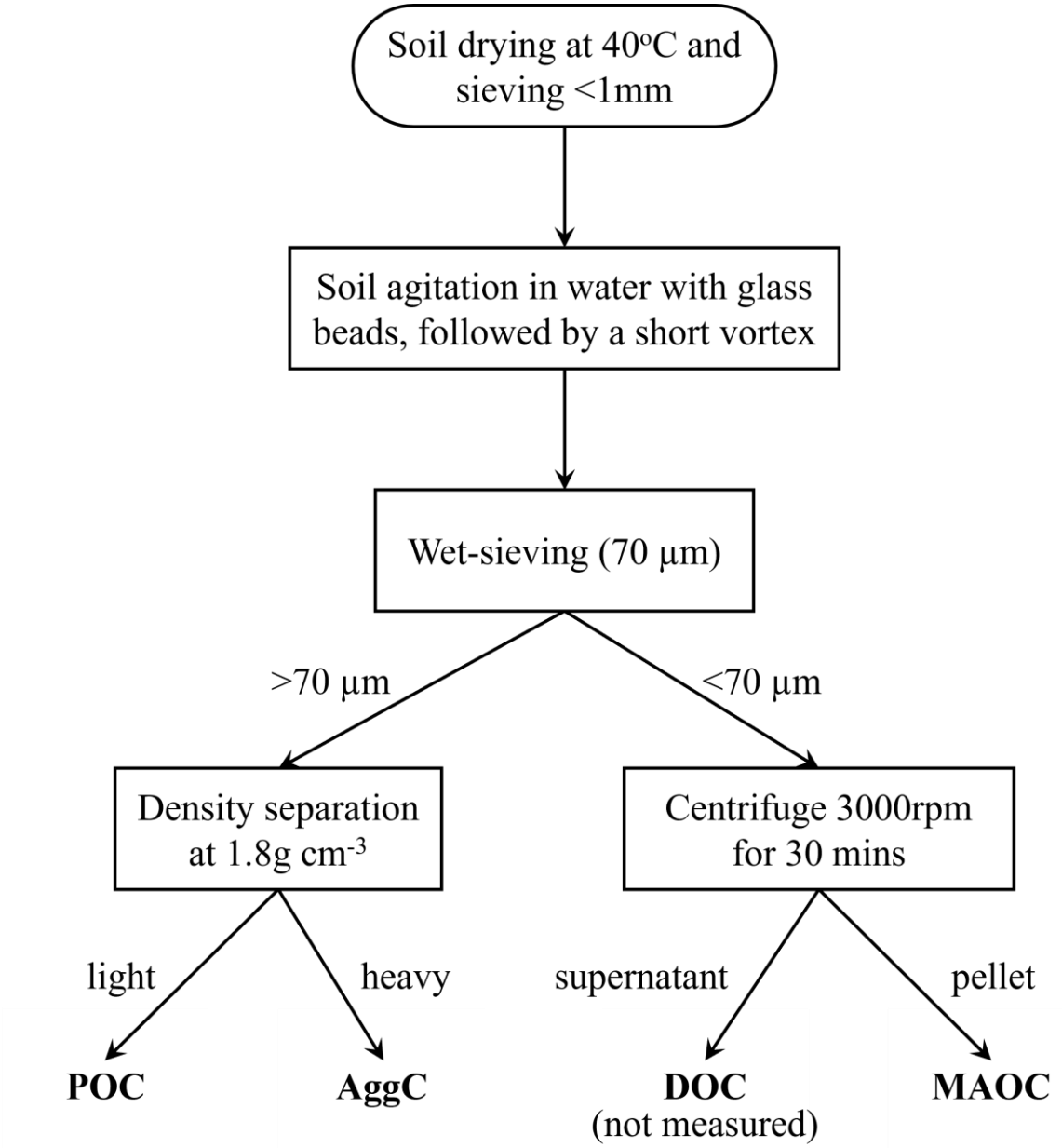

Figure 3.3 Flowchart of soil carbon fractionation method used in this study. The soil sample was separated into four fractions: particulate organic carbon (POC), aggregate carbon (AggC), mineral-associated organic carbon (MAOC), and dissolved organic carbon (DOC). Modified from Buss et al. (2021). Notable deviations in this study from the method outlined by Buss et al. (2021) include initially sieving the soil to <1mm instead of <2mm, and not performing the rapid recalcitrance test.

3.3.3.2 Soil pH and Electrical Conductivity

Soil pH and EC were measured to see if there were any differences in the chemical properties of the soil between control and brush pack patches. pH and EC were measured using a TPS WP-81 pH, Conductivity, and Temperature Kit, following the method outlined by Rayment and Lyons (2010). The

soil samples were thoroughly mixed with Milli-Q ultrapure water at a 1:5 soil-to-water ratio. EC was recorded in microsiemens per centimetre (µS/cm).

3.4 Data Analysis

All data was initially compiled and organised in Microsoft Excel, version 2410, where summary statistics for LFA measurements, such as means and standard errors, were also calculated. All complex data analyses including modelling and significance testing were performed using R Statistical Software, version 4.4.1 (R Core Team, 2024). The following packages were used: 'dplyr', 'rstatix', 'ggplot2', 'lme4', 'emmeans', and 'performance' (Bates et al., 2015; Kassambara, 2023; Lenth, 2024; Lüdecke et al., 2021; Wickham, 2016; Wickham et al., 2023).

To test for significant differences all data were subjected to an ANOVA using the base 'aov()' function in R (R Core Team, 2024). Due to the nature of the soil respiration and litter incorporation data, different approaches for significance testing were required. These are detailed in Sections 3.4.2 and 3.4.1.2, respectively.

Significance levels from all tests are indicated by the number of asterisks: * = $p < 0.05$, ** = $p < 0.01$, and *** = $p < 0.001$

3.4.1 Landscape Function Analysis

Before any calculations were made, there were a few errors in the 2018 student data that required amendment. These included minor adjustments, such as changing soil cover scores for litter zones from 5/5 to 1/5, following the method laid out by Tongway and Hindley (2004), which states leaf litter should not contribute to the soil cover score. Major corrections were also made, such as addressing inconsistencies between the zones recorded on control and brush pack transects which undermined the comparability of the two transect types. Further details on these amendments can be found in Appendix 3 – LFA Data & Edits Record.

Once the data had been 'cleaned' of errors, the many different zone types that had been recorded were also aggregated. This involved grouping terms such as bare ground and scald as just bare ground, or grassy tussock and grass sward as just grass sward. These changes made the data from each transect comparable. All aggregated zones can be found in Appendix 3 – LFA Data & Edits Record.

Overall LFA index scores (including averages and standard errors) for each transect and zone were calculated using the 11 SSA indicators and zone dimensions measured in the field. These calculations were performed in Microsoft Excel using the formula detailed by Tongway and Hindley (2004).

To assess whether brush packs improved function at the landscape-scale, the data was analysed at the transect-level. And to assess whether brush packs improved function at the fine-scale, the data was analysed at the zone-level.

3.4.1.1 Transect-level Analysis

Each transect was treated as one repetition for that transect type. Data was grouped by transect type (control, brush pack, reference) and year measured (2018, 2019, 2024).

To assess the contribution of brush packs compared to other zones on the transect-scale the zones were aggregated into three groups: interpatch, brush pack, and other patches. Then the average percentage contribution of each zone group to the LFA indices was compared between 2018 and 2024 brush pack transects (Figure 4.5). The average extent, measured in length (m), width (cm), and area (m^2) of brush pack patches was also compared between 2018 and 2024 to better understand their contribution to transect-level function (Table 4.1). The 2019 data was excluded from these comparisons as the same number of transects and brush packs were not measured. Due to the structure and context of the data, statistical significance testing was not performed.

To assess whether brush packs improved function at the transect-level, average LFA index scores were compared between control and brush pack transects across all three years (Figure 4.6). And to investigate how the function of both transects changed over the years, average LFA index scores for control and brush pack transects were compared across consecutive years (2018 to 2019 and 2019 to 2024) (Table 4.2). An ANOVA was performed to test for significant differences.

To assess whether brush packs improved function at the transect-level relative to that of the Reference Site, average LFA index scores were compared between reference and brush pack transects in 2024 (Figure 4.7). An ANOVA was performed to test for significant differences.

3.4.1.2 Zone-level Analysis

Each SSA entry for every zone (with multiple entries per transect) was treated as one repetition for that zone type. Data was grouped by zone type (interpatch, brush pack) and year measured (2018, 2019, 2024).

To assess whether brush packs improved function at the zone-level, average LFA index scores were compared between interpatch and brush pack patches across all three years (Figure 4.8). And to investigate how the function of the areas changed over the years, average LFA index scores for interpatch and brush pack patches were compared across consecutive years (2018 to 2019 and 2019 to 2024) (Table 4.3). An ANOVA was performed to test for significant differences.

It was observed that the significant differences in LFA index scores at the zone-level were being driven by only a select few SSA indicators. These included soil cover, soil surface roughness, litter cover, and litter incorporation. The average values and standard errors of these four indicators were compared between interpatch and brush pack patches across all three years (Figure 4.9). To investigate how these four indicators changed over the years, scores in each indicator for interpatch and brush pack patches were compared across consecutive years (2018 to 2019 and 2019 to 2024) (Table 4.4). An ANOVA

was performed to test for significant differences in the soil cover, soil surface roughness, and litter cover indicators. However, due to the nature of the litter incorporation data being categorical rather than quantitative an ANOVA could not be performed. Instead, Fisher's exact test was used, as it is better suited for small sample sizes and data with uneven distributions as it does not make assumptions about the underlying distribution of the data, unlike other options such as a chi-squared (χ^2) test. This test was performed using the 'rstatix' package (Kassambara, 2023).

3.4.2 Soil Respiration

The CO_2 efflux data was first 'cleaned'. During the initial stages of data analysis certain cycles (each cycle being one measurement repetition or datapoint) were removed due to abnormality and extreme deviations from the rest of the data. These cycles exhibited unusually large CO_2 efflux values, with some exceeding 10µmol m^{-2} s^{-1}. The LI-6400 measures three cycles per collar, and significant deviations within these short cycles, which are otherwise expected to remain mostly consistent, indicate measurement error. These abnormalities were attributed to poor connections between the LI-6400 and the collar or potential leaks in the collar itself, which would artificially spike the CO_2 efflux readings. To avoid possible pseudoreplication, all measurement cycles for each collar were averaged, and the resulting mean was treated as a single repetition or datapoint.

After cleaning and organising the data, the effects of brush packs on soil respiration were analysed using a linear mixed-effects model (LMM) to account for covariates. Multiple models with different combinations of fixed, random, and interaction effects were made using the 'lme4' package (Bates et al., 2015). All models were compared using the 'performance' package (Lüdecke et al., 2021), and the model with the lowest Akaike information criterion was selected. The final model included treatment, moisture, and the interaction between treatment and moisture as fixed effects, with site (patch) measured as a random effect to account for spatial variation in respiration. Temperature was dropped as a fixed effect because it did not significantly improve model fit with the data. This approach was based on consultation with an advisor from the ANU Statistical Support Network.

To compare soil respiration between control and brush pack patches, estimated marginal means (EMMs) were calculated (Figure 4.10) using the 'emmeans' package (Lenth, 2024). The EMMs were calculated with 95% confidence intervals at the average moisture level of all datapoints. To test for significant difference between the EMMs, a pairwise comparison was performed using the 'emmeans' package (Lenth, 2024). Then the predicted relationship between CO_2 efflux and moisture was plotted with 95% confidence intervals (Figure 4.11).

To determine the significance of each fixed effect in the model, p-values were calculated from t-tests on their respective estimated coefficients (Table 4.5). The degrees of freedom used were calculated using Satterthwaite's approximation to account for the mixed model structure, as is common practice with LMMs. Standard deviations of the random effects were calculated to quantify the variability

attributed to differences between sites and assess the contribution of site-level variation to the overall model (Table 4.6). These calculations were performed using the base 'summary()' and 'confint()' functions in R (R Core Team, 2024), in conjunction with the 'lme4' package (Bates et al., 2015).

When the analysis was repeated without excluding the abnormal datapoints, the interpretation of the results did not differ. The results of the 'uncleaned' data set are included in Appendix 4 – Soil Respiration Raw Data and R Code.

3.4.3 Soil Nutrient and Chemical Properties

3.4.3.1 Soil Carbon

Following Buss et al. (2021), to standardise the results, the percentage carbon in sample was calculated based off the initial sample weight and the total milligrams (mg) of carbon detected by the combustion analyser (Equation 3.2).

Equation 3.2 Percentage carbon in sample calculation

$$Carbon\ in\ sample\ (\%) = \left(\frac{Total\ detected\ carbon\ (mg)}{Initial\ sample\ weight\ (mg)}\right) \times 100$$

For the unfractionated soil the carbon in *sample* (%) is equal to the carbon in *soil* (%). This is not the case for the three soil fractions. To further standardise the results, the carbon in sample (%) for each fraction was converted to carbon in soil (%) as well, using their percentage weights, see Equations 3.3 and 3.4. This allows for a more straightforward comparison across fractions and with the unfractionated soil.

Equation 3.3 Percentage weight of fraction calculation

$$Weight\ of\ fraction\ (\%) = \left(\frac{Mass\ of\ fraction\ (g)}{Final\ mass\ of\ sample\ (g)}\right) \times 100$$

Equation 3.4 Percentage carbon of soil calculation

$$Carbon\ in\ soil\ (\%) = Carbon\ in\ sample\ (\%) \times \left(\frac{Weight\ of\ fraction\ (\%)}{100}\right)$$

The data was grouped by treatment, fraction and depth interval. Means and standard errors were calculated, and an ANOVA was performed between treatments of the same fraction and depth interval to determine if there were any significant differences between groups.

3.4.3.2 Soil pH and Electrical Conductivity Levels

No transformation of the pH and EC data was necessary. The data was grouped by treatment and depth interval. Means and standard errors were calculated, and an ANOVA was performed to determine if there were any significant differences between treatments at the same depth interval. Outliers were calculated using the 'ggplot2' package (Wickham, 2016).

Chapter 4: Results

4.1 Overview of Site Changes and Environmental Context

Conditions at the Main Site improved considerably over the study period, as shown in Figures 4.1, 4.2, and 4.3, which display photos of the same transect in 2018 (both pre- and post-brush pack placement) and from 2024.

Figure 4.1 Transect at Main Site in 2018 pre-brush pack placement. Same transect as in Figures 4.2 and 4.3. Photo courtesy of David Freudenberger.

Figure 4.2 Transect at Main Site in 2018 post-brush pack placement. Same transect as in Figures 4.1 and 4.3. Photo courtesy of David Freudenberger.

Figure 4.3 Transect at Main Site in 2024. Same transect as in Figures 4.2 and 4.3. Photo courtesy of David Freudenberger.

Additional photos of the Main Site over the years can be found in Appendix 5 – All Site Photos.

Over the study period, a rare ‘triple-dip’ La Niña event occurred from August 2020 to March 2023. This led to a large amount of rainfall at the Main Site, significantly higher than in the years leading up the study (Figure 4.4).

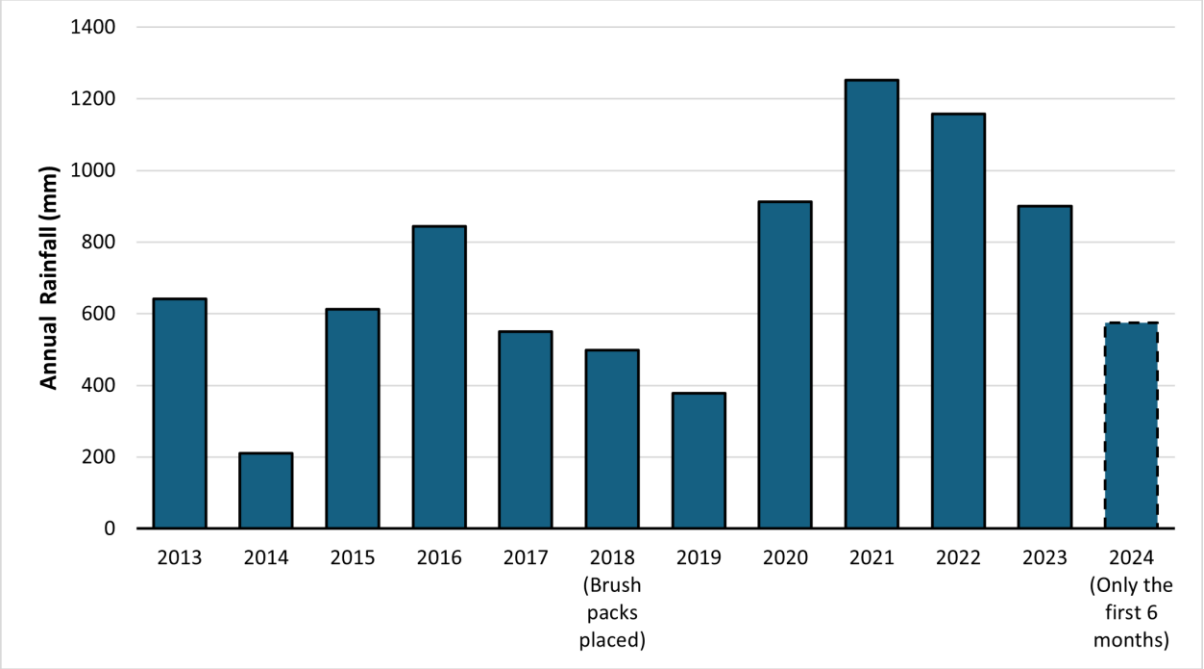

Figure 4.4 Annual rainfall in millimetres recorded from 2013 to 2024, covering five years before and six years after the installation of brush packs in 2018. The 2024 value represents rainfall for only the first six months. Data was sourced from the weather station at Mulloon Home Farm (2006–July 2020) and the new Weather Maestro station (from August 2020), with any data gaps filled from nearby stations. Data kindly compiled by Chris Inskeep, (TMI). Data available in Appendix 1 – Home Farm Climate Data.

The large amount of rain (Figure 4.4) and the reduced grazing pressure (Section 3.1.1) has resulted in significant pasture growth at the Main Site (Figure 4.3).

4.2 Landscape Function Analysis

4.2.1 Transect-level Analysis

The functionality of the interpatch zones had the greatest influence on transect-level function, contributing most to the overall LFA scores (Figure 4.5). Although interpatch zones had lower LFA scores than brush pack or other patch zones, their larger area across the transect resulted in the greatest impact.

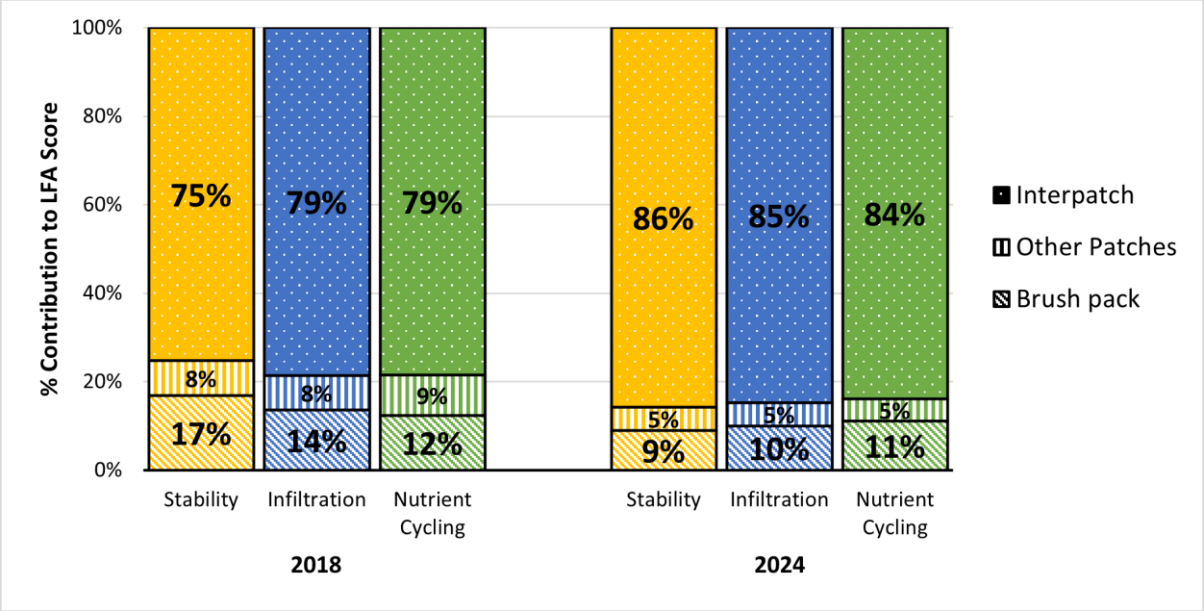

Figure 4.5 Percentage contribution of different zone groups (interpatch, other patches, and brush pack) to Landscape Function Analysis (LFA) index scores for 2018 and 2024 brush pack transects. Each bar shows each of the three indices: stability (yellow), infiltration (blue), and nutrient cycling (green) and is stacked by zone groups. Each bar is stacked by zone group, interpatch at the top, followed by other patches, and brush pack at the bottom. Each bar represents the proportional contribution of these zones to the overall LFA score for a given index. Significance testing was not conducted due to the nature and structure of the data.

Brush packs persisted over the duration of the study showing some shrinkage in extent as they decayed and broke down (Table 4.1).

Table 4.1 Average dimensions and area of brush pack patches in 2018 and 2024, based on measurements from 20 individual patches.

	Average length (m)	**Average width (m)**	**Average area (m^2)**
2018	1.73	8.03	13.93
2024	1.27	7.39	9.36

No significant differences were found in any LFA index between control and brush pack transects over 2018, 2019, and 2024, except for the infiltration index in 2019 ($p < 0.05$), see Figure 4.6.

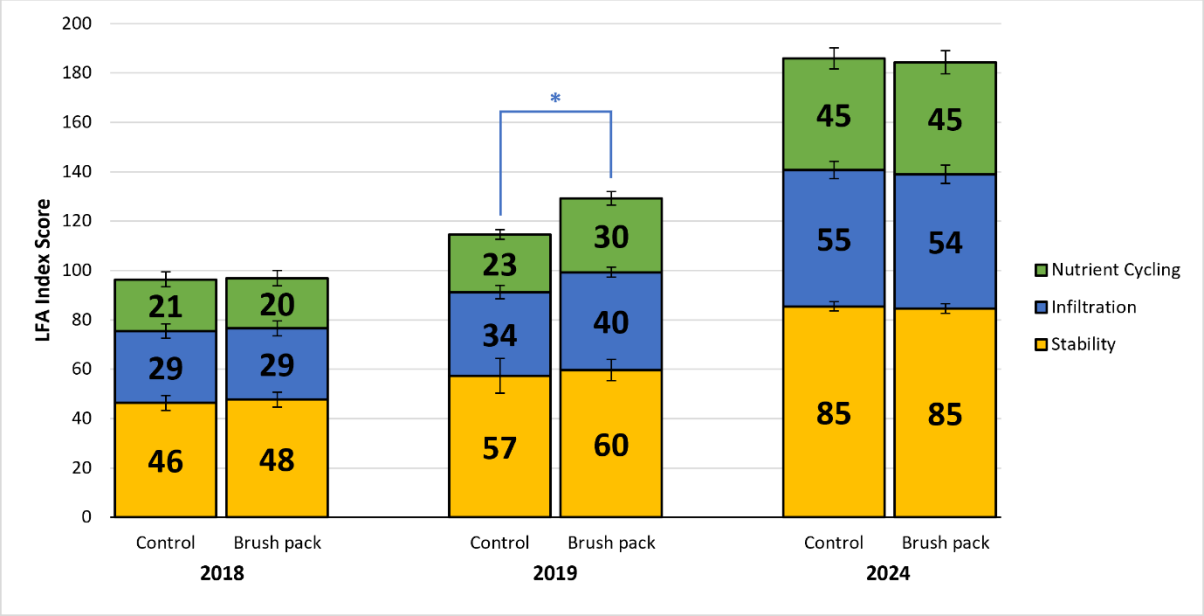

Figure 4.6 Transect-level **c**hanges in Landscape Function Analysis (LFA) index scores across three years (2018, 2019, and 2024) comparing control and brush pack transects. The scores are stacked by the three indices: stability (yellow), infiltration (blue), and nutrient cycling (green). Error bars represent the standard error of the mean. Significant differences between transects within the same year, determined by ANOVA, are indicated by color-coded asterisks with corresponding lines: yellow for stability, blue for infiltration, and green for nutrient cycling. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Over the study period, function improved significantly ($p = <0.05$ to <0.001) for both control and brush pack transects, with an increase in the overall LFA scores for all indices except for the infiltration and nutrient cycling indices for control transects across 2018 and 2019, and 2019 and 2024 (Table 4.2).

Table 4.2 Results of transect-level significance testing for changes in Landscape Function Analysis (LFA) index scores over time (2018 to 2019 and 2019 to 2024) for control and brush pack transects. The indices assessed include stability, infiltration, and nutrient cycling. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Transect Type	Comparison	LFA Index	p-value
Control	2018 to 2019	Stability	**0.03 ***
		Infiltration	0.42
		Nutrient Cycling	0.53
	2019 to 2024	Stability	**<0.001 *****
		Infiltration	**<0.001 *****
		Nutrient Cycling	**<0.001 *****

Brush pack	2018 to 2019	Stability	**<0.001 *** **
		Infiltration	**0.02 ***
		Nutrient Cycling	**0.01 ***
	2019 to 2024	Stability	**<0.001 *** **
		Infiltration	**0.005 ****
		Nutrient Cycling	**0.008 ****

In 2024, compared to brush pack transects at the Main Site, transects at the Reference Site had significantly higher LFA index scores ($p < 0.05$) for infiltration and nutrient cycling compared to brush pack transects (Figure 4.7). There was no statistical evidence that the stability index scores were different.

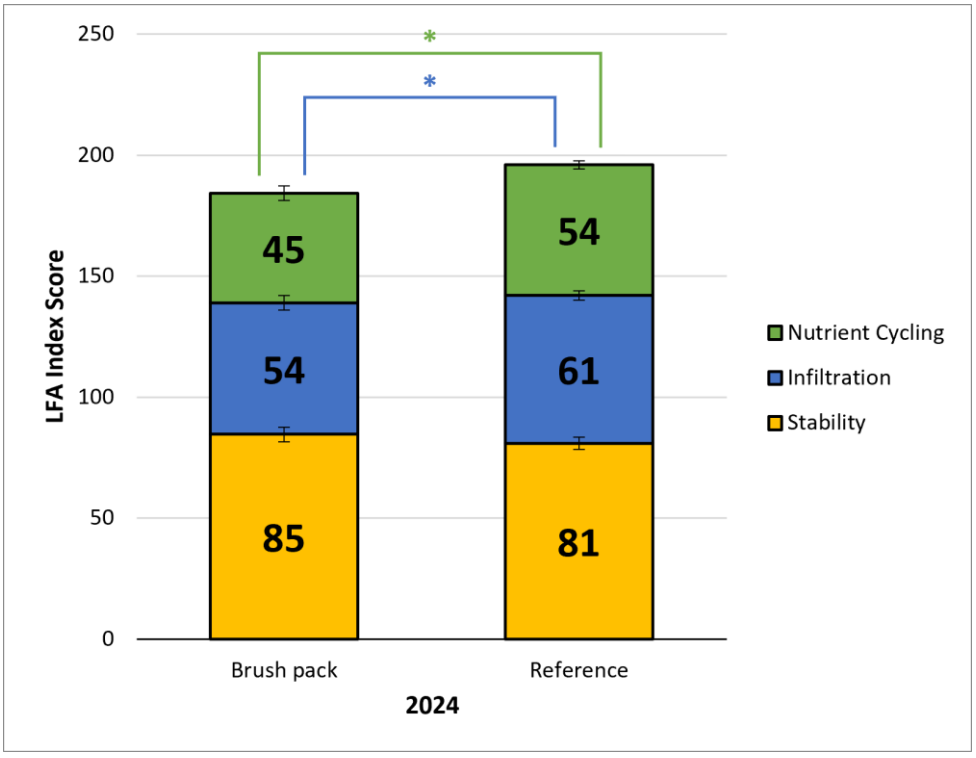

Figure 4.7 Transect-level Landscape Function Analysis (LFA) index scores for brush pack and reference transects in 2024. The scores are stacked by the three indices: stability (yellow), infiltration (blue), and nutrient cycling (green). Error bars represent the standard error of the mean. Significant differences between transects within the same year, determined by ANOVA, are indicated by color-coded asterisks with corresponding lines: yellow for stability, blue for infiltration, and green for nutrient cycling. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

4.2.2 Zone-level Analysis

As shown above in the transect-level analysis, brush pack patches had minimal influence on overall LFA scores (Figure 4.5), and there were few significant differences between control and brush pack transects over the years (Figure 4.6 and Table 4.2).

However, at the zone-level, brush pack patches demonstrated greater function compared to interpatches. In 2018, brush pack patches had significantly greater stability index scores compared to interpatches ($p < 0.001$). In both 2019 and 2024, significant differences were observed across all three indices (all $p < 0.001$), with brush pack patches showing greater function in each case (Figure 4.8).

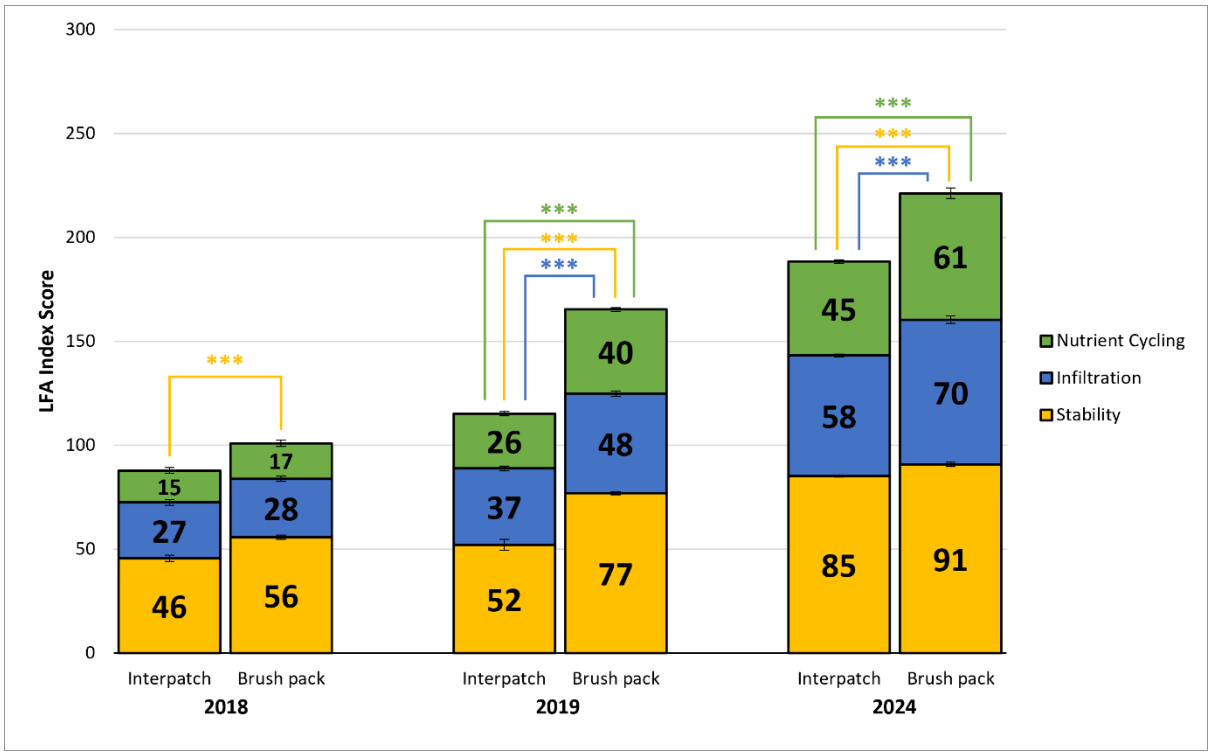

Figure 4.8 Zone-level Landscape Function Analysis (LFA) index scores for interpatch and brush pack patches across three years (2018, 2019, and 2024). The scores are stacked by the three indices: stability (yellow), infiltration (blue), and nutrient cycling (green). Error bars represent the standard error of the mean. Significant differences between patches within the same year, determined by ANOVA, are indicated by color-coded lines with corresponding asterisks: yellow for stability, blue for infiltration, and green for nutrient cycling. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Over the study period, function improved significantly ($p < 0.001$) for both interpatch and brush pack patches, with increasing LFA scores across 2018 and 2019, and 2019 and 2024 (Table 4.3). However, the stability score for interpatches between 2018 and 2019 was only marginally significant ($p = 0.048$).

Table 4.3 Results of zone-level significance testing for changes in Landscape Function Analysis (LFA) index scores over time (2018 to 2019 and 2019 to 2024) for control and brush pack patches. The indices assessed include stability, infiltration, and nutrient cycling. The number of asterisks indicates the significance level: * = p < 0.05, ** = p < 0.01, *** = p <0.001.

Zone Type	Comparison	LFA Index	p-value
Interpatch	2018 to 2019	Stability	**0.048 ***
		Infiltration	**<0.001 *****
		Nutrient Cycling	**<0.001 *****
	2019 to 2024	Stability	**<0.001 *****
		Infiltration	**<0.001 *****
		Nutrient Cycling	**<0.001 *****
Brush pack	2018 to 2019	Stability	**<0.001 *****
		Infiltration	**<0.001 *****
		Nutrient Cycling	**<0.001 *****
	2019 to 2024	Stability	**<0.001 *****
		Infiltration	**<0.001 *****
		Nutrient Cycling	**<0.001 *****

Upon further investigation of the data, only a few key SSA indicators were found to drive the differences in LFA index scores between interpatch and brush pack patches. The key indicators included soil cover, soil surface roughness, litter cover, and litter incorporation, which together cover all three of the LFA indices (Figure 3.2). These indicators were the only ones that differed significantly between interpatch and brush pack patches, while the others showed minimal differences (Figure 4.9).

Soil cover was the only SSA indicator to change significantly ($p < 0.001$) immediately after the brush packs were established in 2018. Soil cover remained at a constant max score of 5/5 for brush pack patches across all three years and was still significantly higher than at interpatches in 2019 ($p < 0.001$) but not in 2024. Brush pack patches had significantly ($p < 0.001$) higher scores than interpatches in soil surface roughness in 2019 and 2024, litter cover in 2019 and 2024, and litter incorporation in 2024 only.

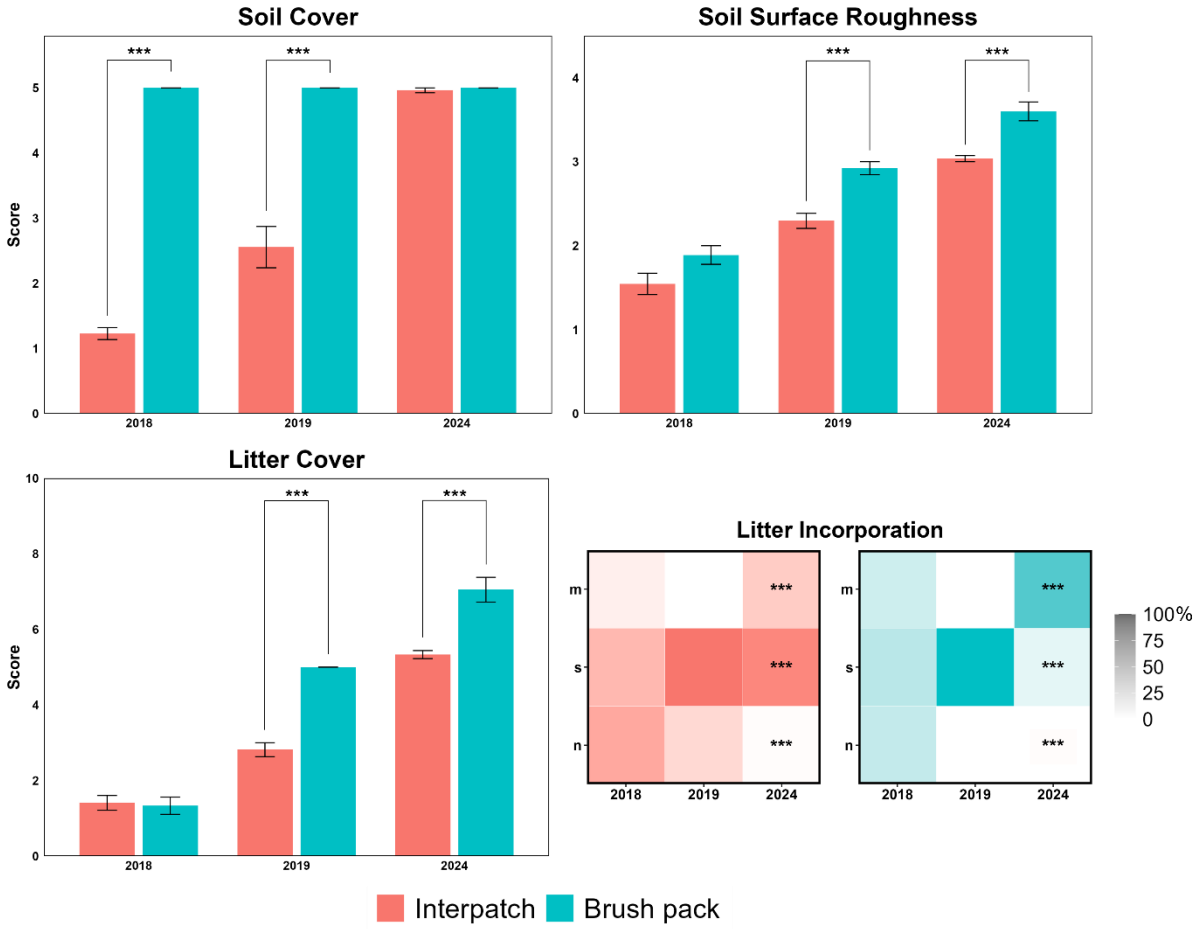

Figure 4.9 Zone-level comparisons of the four key Soil Surface Assessment (SSA) indicators between interpatch and brush pack patches across three years (2018, 2019, and 2024). The bar plots show scores for soil cover (max score = 5), soil surface roughness (max score = 4), and litter cover (max score = 10), with error bars representing the standard error of the mean. The heatmap shows the litter incorporation scores, categorised as nil (n), slight (s), and moderate (m) incorporation, displayed as percentages of the total number of observations for that year. Extensive (e) incorporation was omitted for clarity, as there were no observations with extensive incorporation. Significant differences between interpatch and brush pack patches within the same year, determined by ANOVA, or by Fisher's exact test for litter incorporation, are indicated by asterisks. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Over the study period, all four of these SSA indicators improved significantly ($p = <0.05$ to <0.001) for both interpatch and brush pack patches (Table 4.4). Except for soil cover for brush pack patches which showed no change as it was already at the max score (5/5) in 2018.

Table 4.4 Results of zone-level significance testing for changes in Soil Surface Assessment (SSA) indicator scores over time (2018 to 2019 and 2019 to 2024) for interpatch and brush pack patches. The indicators assessed include soil cover, litter cover, soil surface roughness, and litter incorporation. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Zone Type	Comparison	SSA Indicator	p-value
Interpatch	2018 vs 2019	Soil Cover	**<0.001 *** **
		Soil Surface Roughness	**<0.001 *** **
		Litter Cover	**<0.001 *** **
		Litter Incorporation	**0.01 ***
	2019 vs 2024	Soil Cover	**<0.001 *** **
		Soil Surface Roughness	**<0.001 *** **
		Litter Cover	**<0.001 *** **
		Litter Incorporation	**<0.001 *** **
Brush pack	2018 vs 2019	Soil Cover	No change
		Soil Surface Roughness	**<0.001 *** **
		Litter Cover	**<0.001 *** **
		Litter Incorporation	**0.001 ****
	2019 vs 2024	Soil Cover	No change
		Soil Surface Roughness	**<0.001 *** **
		Litter Cover	**<0.001 *** **
		Litter Incorporation	**<0.001 *** **

4.3 Direct Measurements of Nutrient Cycling

Initial LFA measurements revealed brush pack patches had significantly higher scores in the nutrient cycling index and for SSA indicators such as litter cover and incorporation. To further investigate this difference, soil respiration and carbon measurements were taken with the expectation that they would both be higher at brush pack patches. This follows Tongway and Hindley (2004), who used these direct measurements to verify the nutrient cycling index.

4.3.1 Soil Respiration

Soil respiration was measured through soil CO_2 efflux.

A linear mixed-effects model was used to analyse CO_2 efflux, accounting for the covariate moisture. Surprisingly, the difference in CO_2 efflux between treatments was not statistically significant, however, variability in CO_2 efflux was higher at control patches (Figure 4.10).

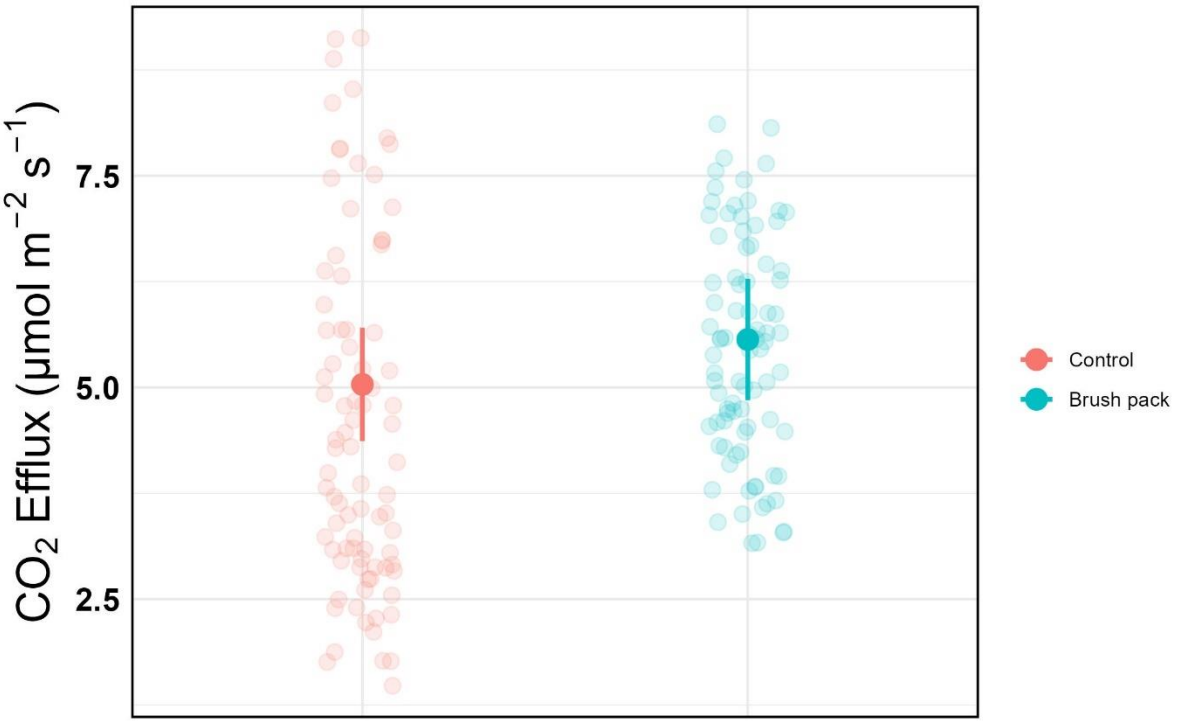

Figure 4.10 Estimated marginal means (EMMs) of CO_2 efflux (µmol m^{-2} s^{-1}) for control (red, EMM = 5.04) and brush pack (blue, EMM = 5.57) treatments, with individual measurements shown as transparent background points (ranges: control = 1.48-9.13, brush pack = 3.16-8.11). Solid vertical lines represent 95% confidence intervals (4.37-5.71 for control and 4.85-6.28 for brush pack). EMMs were calculated from a linear mixed-effects model for each treatment at the average moisture level of 17.64 (%VWC) across all data points. A pairwise comparison of the EMMs revealed a p-value of 0.2710, indicating that the difference between treatments was not statistically significant.

All the fixed effects used in the model had statistically significant effects (estimated coefficients), see Table 4.5. The main effect of both the brush pack treatment and moisture was negative. However, there was a large, positive interaction effect between the brush pack treatment and moisture. Together, the main effects and this interaction effect result in higher estimated CO_2 efflux at brush pack patches under high moisture conditions and lower under low moisture conditions.

Table 4.5 Summary table of the fixed effects estimates from the linear mixed-effects model. Includes the estimated coefficients (Estimate), standard errors (Std. Error), degrees of freedom (df), 95% confidence intervals of the estimated coefficient (Estimate CI 2.5% and Estimate CI 97.5%), and p-values (p-value) for the intercept (representing the control treatment at moisture = 0), brush pack treatment (TreatmentB), moisture (Moisture), and the interaction between treatment and moisture (TreatmentB:Moisture).

Fixed effects:	Estimate	Std. Error	df	Estimate CI 2.5 %	Estimate CI 97.5%	p-value

(Intercept)	8.72	1.25	21.25	6.34	11.15	**<0.001 *** **
TreatmentB	-4.39	1.80	26.66	-7.89	-1.01	**0.02 ***
Moisture	-0.21	0.06	21.27	-0.33	-0.09	**0.003 ** **
TreatmentB: Moisture	0.28	0.10	28.96	0.09	0.48	**0.01 ***

Figure 4.11 shows the predicted relationship between CO_2 efflux and moisture levels for control and brush pack treatments. The interaction effect from Table 4.5 is visible with CO_2 efflux increasing as moisture rises in brush pack patches, while the opposite occurs in control patches. The 95% CIs for both treatments largely overlap suggesting there isn't a significant difference in CO_2 efflux at different moisture levels.

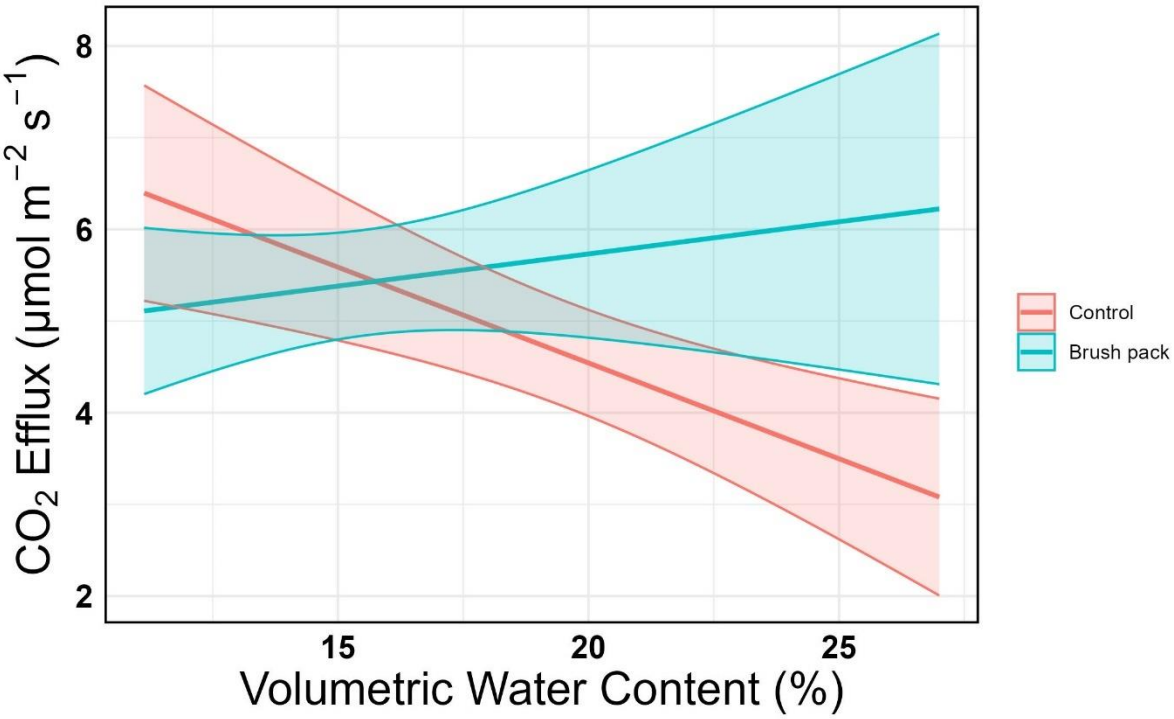

Figure 4.11 The predicted relationship between CO_2 efflux (µmol m^{-2} s^{-1}) and moisture levels (%VWC) for control (red) and brush pack (blue) treatments, based on a linear mixed-effects model, is plotted across the range of recorded moisture levels for all data points. The centre of the plot aligns with the average moisture level across all data points of 17.64 %VWC, and the lines intersect, before this point, at 15.74 %VWC. The shaded regions represent 95% confidence intervals.

The model's residual standard deviation was large, relative to the range of efflux values observed, and statistically significant, with confidence intervals not crossing zero (Table 4.6). This indicates some variability was not explained by the model. The residual standard deviation is also larger than that of site (patch), this indicates a greater variability within-patches than between-patches.

Table 4.6 Summary table of the random effects estimates from the linear mixed-effects model. Includes the variance (Variance), standard deviation (Std. Dev.), and 95% confidence intervals of the standard deviation (Std. Dev. CI 2.5% and Std. Dev. CI 97.5%) for the random effect of site (SiteName) and residual variability (Residual).

Random effects:	Variance	Std. Dev.	Std. Dev. CI 2.5 %	Std. Dev. CI 97.5%
SiteName	0.66	0.81	0.43	1.11
Residual	1.74	1.32	1.18	1.48

4.3.2 Soil Carbon

Soil carbon was measured across four depth intervals to capture the distribution of carbon throughout the soil profile.

Surprisingly, no significant differences in the unfractionated soil carbon were observed between control and brush pack patches (Figure 4.12). Carbon levels generally decreased with depth.

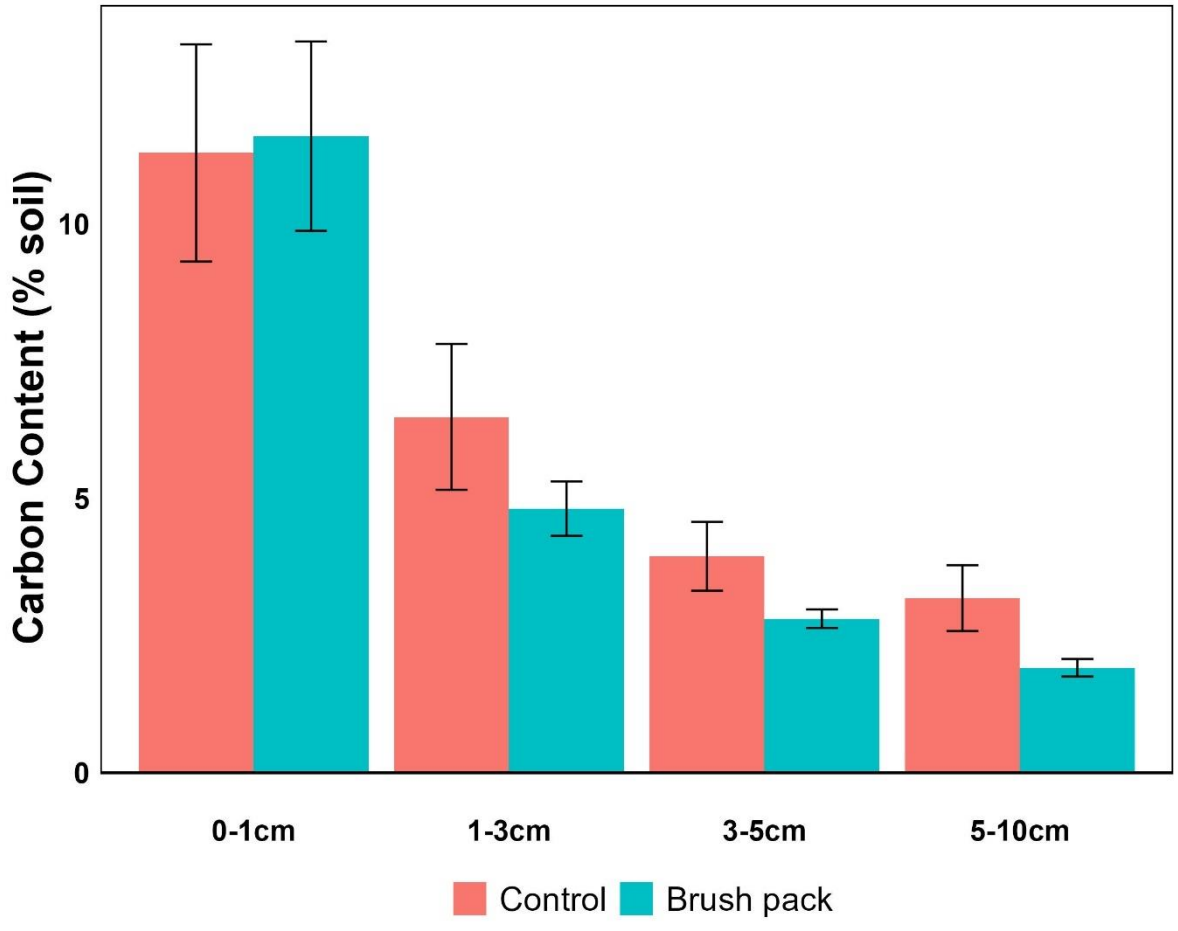

Figure 4.12 Unfractionated soil total carbon content as a percentage of total soil (% soil) by depth (0-1cm, 1-3cm, 3-5cm, and 5-10cm) for control (red) and brush pack (blue)

treatments. Error bars represent the standard error of the mean. No significant differences between treatments within a depth interval were observed.

Soil was fractionated to isolate plant material which could skew overall carbon measurements and provide insight into the stability of carbon in the soil.

Significant differences in carbon content were observed for the AggC fraction at depths 3-5 and 5-10cm, where control patches had more carbon ($p < 0.05$) (Figure 4.13). No significant differences were observed in any other fractions or depth intervals. Carbon generally declined with depth in the POC and AggC fractions, while it increased with depth in the MAOC fraction. The POC fraction exhibited the greatest variability. Although the difference was not statistically significant (except for the two AggC depths), carbon levels were generally higher at control than brush pack patches.

Summary tables presenting the means, standard errors, and p-values for the carbon data, grouped by treatment, fraction, and depth, are provided in Appendix 6 – Carbon Summary Data.

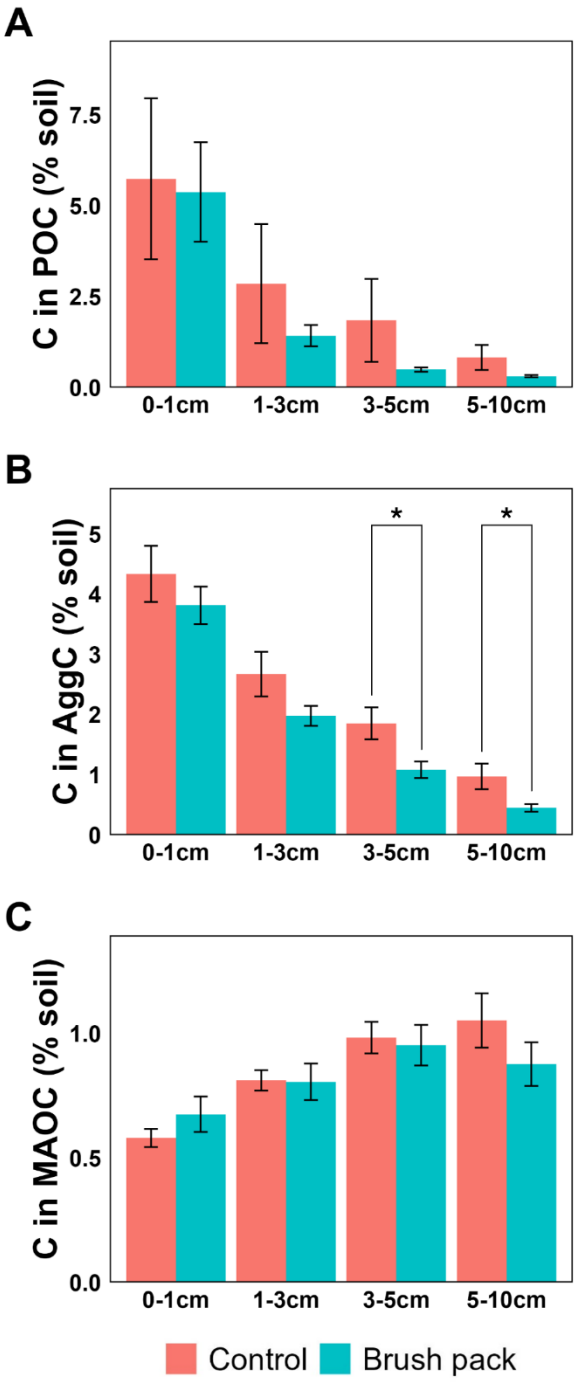

Figure 4.13 Fractionated soil total carbon contents as a percentage of total soil (% soil) across different depths (0-1cm, 1-3cm, 3-5cm, and 5-10cm) for control (red) and brush pack (blue) treatments. Fractions include (A) particulate organic carbon (POC), (B) aggregate carbon (AggC), and (C) mineral-associated organic carbon (MAOC). Significant differences between treatments within the same fraction and depth interval, as determined by ANOVA, are indicated by asterisks. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

4.4 Soil Chemical Properties

4.4.1 Soil pH and Electrical Conductivity

Soil pH and EC were measured to see if there were any differences in the chemical properties of the soil between control and brush pack patches.

A significant difference in pH was observed at the 1-3cm depth, where brush pack patches showed slightly a higher (less acidic) pH (Figure 4.14). No significant differences were found at other depths. pH variability was generally greater at control patches, as indicated by a wider interquartile range and the presence of an outlier at the 0-1cm depth interval.

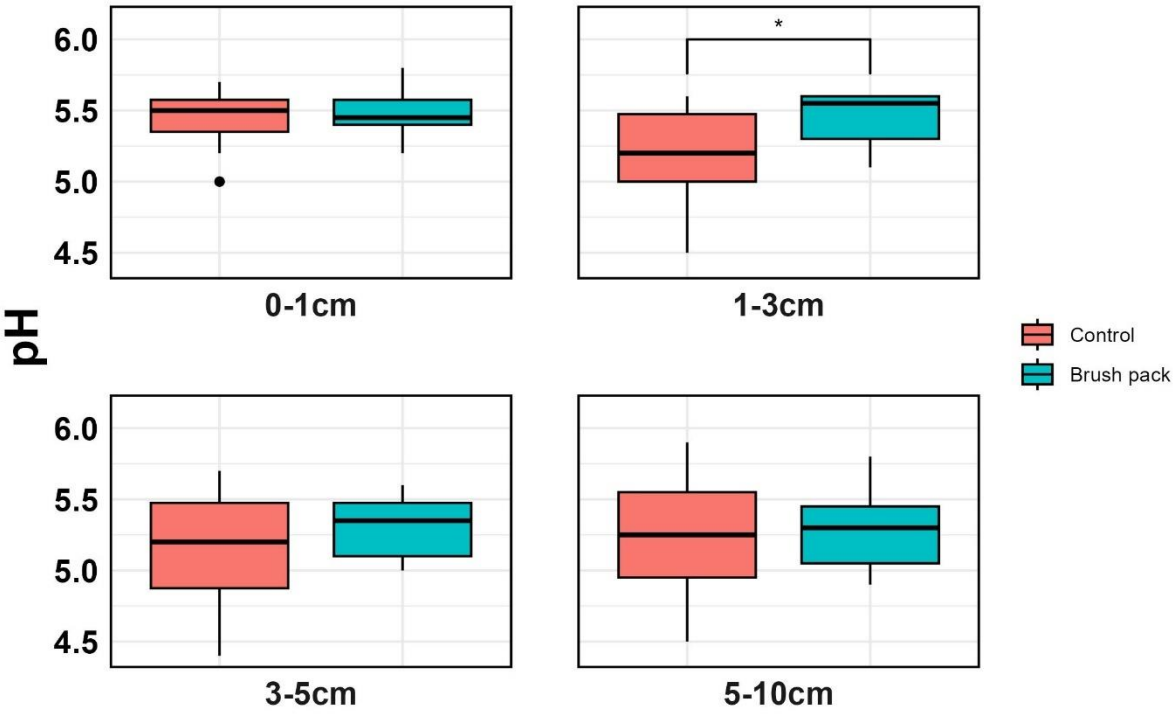

Figure 4.14 Soil pH across different depths (0-1cm, 1-3cm, 3-5cm, and 5-10cm) for control (red) and brush pack (blue) treatments. Box plots show the median, interquartile range, and outliers. Significant differences between treatments within a depth interval, determined by ANOVA, are indicated by asterisks. The number of asterisks indicates the significance level: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

No significant differences in EC were found at any depth (Figure 4.15). Greater variability in EC was generally observed for control patches, as indicated by wider interquartile ranges compared to brush pack patches and multiple outliers. Two control patches exhibited particularly high EC values, corresponding to the two outliers visible across all four depth intervals.

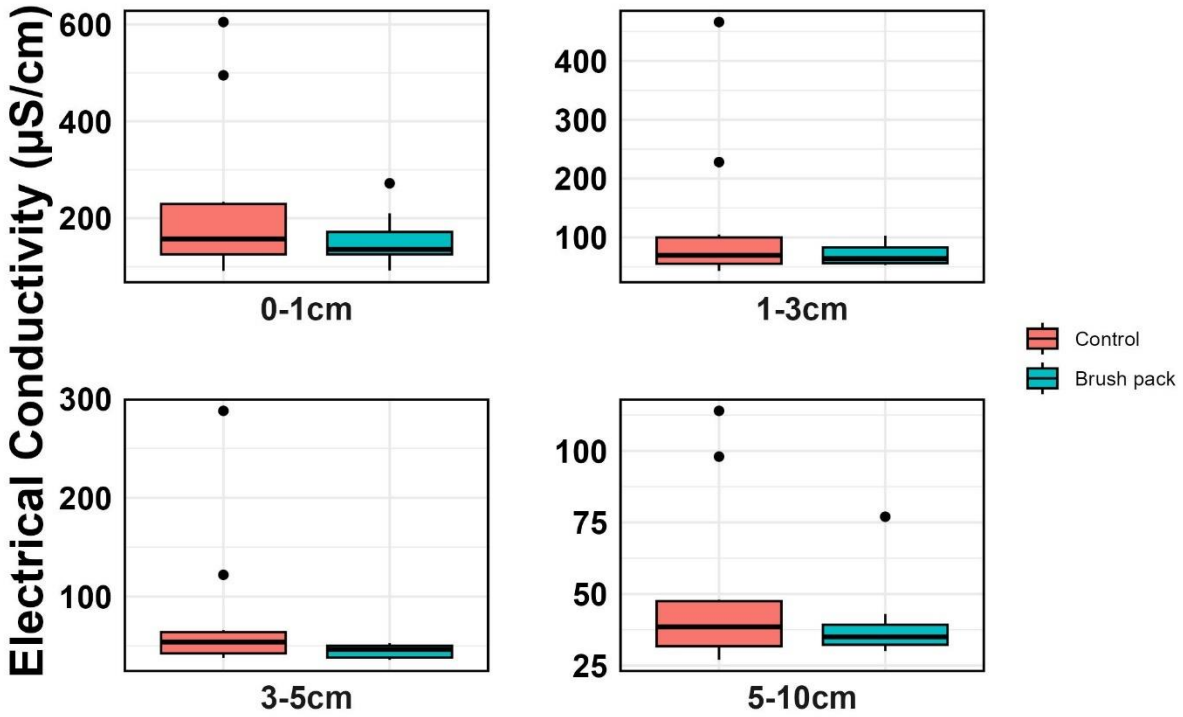

Figure 4.15 Electrical conductivity (µS/cm) across different depths (0-1cm, 1-3cm, 3-5cm, and 5-10cm) for control (red) and brush pack (blue) treatments. Box plots display the median, interquartile range, and outliers. No significant differences between treatments within a depth interval were observed.

Chapter 5: Discussion

Landscape degradation is a significant global issue which leads to a decline of essential ecosystem processes or landscape function. These losses compromise the capacity of landscapes to support biodiversity, maintain productivity, and provide essential ecosystem services. One technique aimed at restoring landscape function is the use of brush packs.

Brush packs can help to restore function in degraded semi-arid woodlands of eastern Australia, as demonstrated by Tongway and Ludwig (1996). Most subsequent studies have also examined brush packs in semi-arid or arid grazing lands (Milton & Coetzee, 2022; Naude, 2017; Pelser, 2017; van den Berg & Kellner, 2005). However, they are yet to be studied in higher rainfall environments. This study addressed this gap by investigating whether brush packs can achieve similar outcomes in the temperate, mesic climate of southeastern Australia.

Brush packs were established by ANU students in 2018 at the Main Site during a period of severe drought, where the site was heavily grazed and mostly bare ground (Figure 4.1). Data on simple soil surface indicators were collected in 2018, 2019, and 2024, alongside more sophisticated measurements of soil properties, including soil respiration and carbon levels, taken in 2024.

This research aimed to assess the impacts of brush packs on landscape function, with a specific focus on nutrient cycling, at a degraded, heavily grazed, mostly bare ground site with a temperate, mesic climate in southeastern Australia. It assessed both landscape-scale and fine-scale impacts, comparing brush packs to control transects and patches. By understanding both scales, it provides a comprehensive assessment of whether localised effects beneath brush packs extend to the surrounding landscape

Unlike in the study by Tongway and Ludwig (1996), over the course of this study the functionality of the landscape improved dramatically driven largely by climatic conditions. The increased rainfall from the three-year La Niña event between August 2020 and March 2023, substantially improved ground cover across the landscape, and in parallel grazing pressure within the property was also reduced (Figure 4.4 and Section 3.1.1). This improvement is evident in the site photographs, where pasture growth increased substantially between 2018 and 2024 (Figures 4.1 and 4.3). While the landscape was initially degraded, it clearly did not cross the functional threshold, as it retained sufficient capacity from natural recover (Figure 2.4). As functionality increases along a continuum, the relative value of resource-accumulating patches such as brush packs decreases (Figure 2.4). The improvement of the entire landscape, including the interpatches, likely overshadowed or reduced the significance of any potential benefit from the brush packs in terms of functionality. This is discussed in detail with respect to each measure of functionality in the following sections.

5.1 Impact at the Landscape-scale

5.1.1 Landscape Function Analysis

To assess whether brush packs improved function at the landscape-scale, LFA scores were first examined at the transect-level. It was expected that brush packs would improve all three LFA indices (stability, infiltration, and nutrient cycling) at the transect-level.

The landscape-level LFA data also suggests the functionality of the whole landscape improved over the course of this study. Low LFA scores in 2018 reflects the initial poor functionality at the Main Site (Figures 4.6 and 4.1). By 2019, some improvement was observed for both control and brush pack transect, although the Main Site was still in drought (Figure 4.4). Control transects, however, did not improve significantly in terms of the infiltration and nutrient cycling indices, while brush packs showed statistically higher infiltration index scores compared to control transects (Table 4.2 and Figure 4.6). By 2024, significant improvements were observed across both transects (Table 4.2), largely driven by the high rainfall input from a rare three-year La Niña event and reduced grazing pressure (Figure 4.4 and Section 3.1.1). These findings align with other studies that have identified rainfall as a strong driver of LFA scores, with increased precipitation leading to significantly higher scores across all three indices (Adel et al., 2022; Read et al., 2016). Furthermore, Adel et al. (2022) found the total number and area of resource-accumulating patches was significantly higher with increased precipitation, a relationship reflected in the dramatic pasture growth observed in this study.

Interpatch zones contributed the most to LFA scores on brush pack transects, both in 2018 and 2024 (Figure 4.5). Although brush pack patches received higher LFA scores than interpatch zones, their smaller area meant their overall contribution to the LFA scores was limited. From 2018 to 2024 the proportional contribution of interpatch zones increased while brush packs patches decreased, reflecting – as Whisenant (1999) suggests, that in functional landscapes the relative value of each resource accumulating patch is proportionally lower (also see Figure 2.4). This was also partly due to a slight reduction in the extent of brush pack patches as they decomposed and shrank over the years (Table 4.1). These findings suggest that brush packs either did not cover a sufficiently large area or the difference in LFA scores between interpatch zones and brush pack patches was not great enough to significantly improve function at the landscape-scale compared to the rate at which function improved across the entire landscape.

The Reference Site was established to evaluate whether brush packs enhanced function at the transect-level relative to a minimally disturbed, functional landscape. Despite significant improvements in functionality at the Main Site, it was still less functional in terms of infiltration and nutrient cycling compared to the Reference Site, although there was no significant difference in stability (Figure 4.7). Over the last four decades, the Reference Site has experienced significantly less offtake through grazing, allowing the soil to accumulate and incorporate more litter leading to significantly higher infiltration

and nutrient cycling scores. It is also important to note, the lack of a significant difference in stability cannot be solely attributed to the brush packs, as it also reflects the significantly increased functionality across the entire Main Site over the course of the study. In a study also conducted in temperate southeastern Australia, Munro et al. (2012) found that between a paddock site (analogous to the Main Site) and a remnant woodland (analogous to the Reference Site), there was no significant difference in the stability index, but higher infiltration and nutrient cycling indices in the remnant woodland. Furthermore, the numerical values Munro et al. (2012) found for all three indices at the paddock and remnant woodland site are comparable to the Main Site and Reference Site, respectively, in this study.

5.2 Impact at the Fine-scale

5.2.1 Landscape Function Analysis

To assess whether brush packs improved function at the fine-scale, LFA index and SSA indicator scores were examined at the zone-level. It was expected that brush packs would improve all three LFA indices (stability, infiltration, and nutrient cycling) at the transect-level.

Despite the minimal influence of brush pack patches at the transect-level, there were many significant differences observed in LFA scores between interpatch and brush pack patches at the zone-level. These differences were driven by four key SSA indicators: soil cover, soil surface roughness, litter cover, and litter incorporation. These differences align with the benefits of brush packs (Naude, 2017; Smith et al., 2007; Tongway & Ludwig, 1996), as outlined in Section 2.2:

- **Soil cover:** Brush packs provide immediate soil cover and protection from rainsplash erosion.
- **Soil surface roughness:** Brush packs act as filters which intercept overland flows and capture detritus such as soil and litter.
- **Litter cover and incorporation:** Brush packs provide a direct source of litter input and trap additional litter washed down from upslope. Additionally, they provide shelter and food for macroinvertebrates which helps incorporate this material into the soil.

In 2018, soil cover was significantly higher at brush pack compared to control patches due to the immediate rainsplash protection provided by brush packs (Figure 4.9). Soil cover contributes only to the stability index (Figure 3.2), which explains the significant increase in stability observed in brush zones relative to interpatch in 2018 (Figure 4.8). No other SSA indicators differed significantly as the brush packs had not yet integrated with the soil. Therefore, there were no significant differences observed for the infiltration and nutrient cycling indices (Figure 4.8). Despite no immediate improvements in soil surface roughness and litter cover, as the brush packs became integrated with the ground and dropped litter, it is likely the scores would have improved shortly after the brush packs were placed, well before the second round of measurements in 2019.

By 2019, brush packs had time to integrate with the soil and drop and trap litter. This led to significantly higher scores in soil surface roughness and litter cover compared to control patches, while the initial difference in soil cover persisted (Figure 4.9 and Table 4.4). These three SSA indicators collectively contribute to all three LFA indices (Figure 3.2), leading to significantly higher scores at brush pack compared to control patches (Figure 4.8).

By 2024, brush packs had even more time to integrate and drop and trap litter. This led to an additional significant difference in litter incorporation, while soil surface roughness and litter cover remained significantly greater at brush pack patches (Figure 4.9). While soil cover remained unchanged at brush pack patches, it increased significantly at interpatches due to substantial pasture growth over the course of the study (Table 4.4). As a result, brush pack patches no longer had significantly higher soil cover scores compared to interpatches (Figure 4.9). Interpatches also showed significant increases in soil surface roughness, litter cover, and litter incorporation, and all three of the LFA indices, reflecting the overall functional improvement across the site (Tables 4.4 and 4.3). Together, soil surface roughness, litter cover, and litter incorporation collectively contribute to all three LFA indices (Figure 3.2), leading to significantly higher scores at brush pack compared to control patches (Figure 4.8). These differences over the years suggest brush packs can have lasting benefits in rainsplash protection, intercepting overland flows, and providing organic matter input. These benefits also improve certain resource flows leading to more gains within the landscape system, as described by the TTRP Model (Figure 2.1):

- **Depletion:** Soil cover reduces soil lost from the reserve through depletion processes such as rainsplash erosion.
- **Physical feedback:** Soil surface roughness increases the landscape's capacity to capture and retain resources from transfer processes, such as overland flows, in the reserve.
- **Resource input and Storage:** Litter input contributes resources to the reserve.

All these lasting benefits can help prevent a long-term imbalance in resource gains and losses, which can lead to dysfunction (Figure 2.2).

Few studies report the LFA scores at the zone-level, as most focus on the overall transect-level scores (Adel et al., 2022; de Luna et al., 2022; Kellner et al., 2022; Pelser, 2017; Zhao et al., 2022). However, among the studies that have reported LFA scores at the zone-level, patch zones consistently score higher than interpatches across all three indices (McDonald et al., 2018; Read et al., 2016; Tongway & Hindley, 2004), similar to the results found in this study.

5.2.2 Direct Measurements of Nutrient Cycling

Direct measurements of nutrient cycling were taken due to initial LFA measurements suggesting brush packs had significantly improved nutrient cycling at the fine-scale. Soil respiration and carbon levels

were measured following Tongway and Hindley (2004) who used these measures to verify the nutrient cycling index.

5.2.2.1 Soil Respiration

To directly assess the impact of brush packs on nutrient cycling, soil respiration was measured through soil CO_2 efflux. It was expected that brush packs would increase the rate of soil respiration. Soil temperature and moisture were also measured as covariates as they are known to influence CO_2 efflux (Maier et al., 2011; Moyano et al., 2012). However, only soil moisture was included in the analysis, as it had a significant effect on CO_2 efflux in this study, whereas soil temperature did not.

The CO_2 efflux values observed in this study are comparable in magnitude and variability with those observed in studies of soil respiration in temperate grasslands (Apostolakis et al., 2022; Bremer & Ham, 2002; Joos et al., 2010; Reinthaler et al., 2021). Control patches displayed higher variability in CO_2 efflux values compared to brush pack patches (Figure 4.10). This may reflect a greater diversity amongst control compared to brush pack patches which were mostly homogenous.

A linear mixed-effects model was used to analyse the data. It showed a large positive interaction effect between brush pack treatment and moisture (Table 4.5). This suggests brush packs increase CO_2 efflux under higher moisture levels. However, across the range of moisture levels observed in this study, the differences in predicted CO_2 efflux between brush pack and control patches were not statistically significant (Figure 4.11).

The literature suggests that the relationship between soil respiration and moisture follows a parabolic pattern, with an optimal moisture level at which CO_2 efflux peaks (Hursh et al., 2017; Moyano et al., 2012; Reichstein et al., 2003). It is unclear whether the range of data in this study has crossed this optimal moisture CO_2 efflux peak. The main effect of moisture was negative (Table 4.5), so CO_2 efflux at control patches decreased with moisture, while CO_2 efflux at brush pack patches increased with moisture due to the large positive interaction effect (Figure 4.11). This contrast means we cannot say with certainty that brush packs will increase CO_2 efflux under higher moisture levels past the range sampled in this study. More data across a broader range of moisture levels is necessary to adequately assess this relationship.

Brush packs did not significantly improve soil respiration rates compared to control patches (Figure 4.10). This result was unexpected, as numerous studies have shown correlations between the LFA nutrient cycling index, litter input, and soil respiration (Bréchet et al., 2018; Han et al., 2015; Maestre & Puche, 2009; Setyawan et al., 2011; Tongway & Hindley, 2004; Wei & Man, 2021), and because Tongway and Ludwig (1996) found brush packs significantly improved soil respiration rates.

Over the study period, the substantial pasture growth across the entire site, led to significant improvements in soil cover, litter cover, and litter incorporation scores at control patches (Figure 4.9 and Table 4.4). The literature suggests soil respiration rates are positively related to vegetation productivity, primarily driven by increased root respiration (Hursh et al., 2017; Raich & Schlesinger, 1992; Reichstein et al., 2003), but also due to increased litter input (Bréchet et al., 2018; Han et al., 2015; Wei & Man, 2021). This substantial growth may have overshadowed or reduced the significance of any impact of brush packs on soil respiration rates.

Unfortunately, without baseline soil respiration data from 2018, it is not possible to determine how significantly these rates have changed over time. Also, data from 2019 would have helped assess whether brush pack patches had an initial advantage before the high rainfall input from the three-year La Niña event that began in 2020.

5.2.2.2 Soil Carbon

To directly assess the impact of brush packs on nutrient cycling, soil carbon levels were measured. It was expected that brush packs would increase soil carbon levels.

Carbon was measured at four depth intervals (0-1, 1-3, 3-5, and 5-10cm) to capture distribution throughout the soil profile. Such shallow depths were sampled based on the assumption that any change in carbon levels over the timeframe of this study would occur near the soil surface, following Tongway and Ludwig (1996). Samples were also fractionated into three different fractions (particulate organic carbon, aggregate carbon, and mineral-associated organic carbon) to isolate plant material and provide insight into carbon stability in the soil.

It is difficult to compare the results of this study to the literature because there are few reports where sampling was across such shallow depth intervals, and no research has combined this with fractionation. Also, the soil was sieved to <1mm prior to analysis, which differs from the standard <2mm used in most studies, and soil carbon is highly variable across different land uses and climates.

Unfractionated soil carbon was particularly high in this study, especially in the 0-1cm intervals, where levels exceeded 10% (Figure 4.12). This was due to large amounts of plant matter at this depth, however, it is possible that some of the root mat was inadvertently included in samples at this depth interval. These results are comparable to a study conducted in temperate grazing lands of eastern Australia by Gibson et al. (2023), where soil organic carbon was found to be 3.68% over a single 0-15cm depth interval. In this study the 5-10cm intervals had similar carbon levels (Figure 4.12). Small fragments of charcoal were found throughout the site and across all depth intervals, which may have artificially enhanced carbon levels, these fragments would have been isolated in the particulate organic carbon fraction.

General depth trends were observed across unfractionated and fractionated soil samples (Figures 4.12 and 4.13). Carbon levels in the unfractionated soil decreased over the 0-10cm range. This occurs due to reduced organic matter input with depth. This trend corroborates with the literature, which although mostly looks at broader depth intervals, shows a decline in soil carbon with depth (Balesdent et al., 2018; Dietzel et al., 2017; Román Dobarco et al., 2023).

Particulate organic carbon levels decreased with depth (Figure 4.13a), as this fraction is primarily derived from plant material, which has less input at depth (Abramoff et al., 2018). The particulate organic carbon fraction also had the highest variability in carbon content, likely because it includes a wide variety of organic materials (Abramoff et al., 2018). Aggregate carbon levels also decreased with depth (Figure 4.13b), due to reduced soil aggregate stability with depth (Le Bissonnais et al., 2018; Pan et al., 2023). However, mineral-associated organic carbon levels increased with depth (Figure 4.13c), as this fraction takes time to be transformed or decomposed from organic matter (Román Dobarco et al., 2023). These patterns align with the relative stability of the carbon fractions, mineral-associated organic carbon is the most stable fraction, while particulate organic carbon and aggregate carbon are more vulnerable to decomposition (Abramoff et al., 2018; Georgiou et al., 2022; Hemingway et al., 2019; Peng et al., 2017; Robertson et al., 2019).

Two significant differences in the aggregate carbon fraction were observed, where control patches had higher aggregate carbon levels than brush pack patches at the 3-5 and 5-10cm depth intervals (Figure 4.13b). These differences were unexpected, particularly as they occurred at the deepest of the four depth intervals. Any difference in carbon levels was expected to occur near the surface. Unfortunately, without baseline carbon data from 2018, it is impossible to determine whether these differences resulted from changing conditions over the course of the study or if the control patches sampled started with higher aggregate carbon levels.

Brush packs did not significantly improve soil carbon levels compared to control patches. This was unexpected considering numerous studies have demonstrated a correlation between the LFA nutrient cycling index, litter input, and soil carbon levels (Eldridge et al., 2020; Reynolds et al., 2018; Setyawan et al., 2011; Tongway & Hindley, 2004; Xu et al., 2013; Xu et al., 2021), and because Tongway and Ludwig (1996) found brush packs significantly improved soil carbon levels.

Over the study period, the entire site experienced substantial pasture growth, control and brush pack patches saw significant improvements in soil cover (only control patches), litter cover, and litter incorporation scores (Table 4.4). This substantial growth led to increased litter production and root-derived carbon inputs, both of which can contribute to higher soil carbon levels (Poeplau et al., 2021; Xu et al., 2021). As a result, soil carbon levels beneath both control and brush pack patches may have approached saturation.

Carbon saturation occurs when the capacity of a soil to store additional organic carbon is limited due to inherent physicochemical characteristics (Six et al., 2002). Once this limit is reached, additional carbon inputs may not lead to further increases in soil carbon (Chung et al., 2008; Chung et al., 2010; Six et al., 2002; Stewart et al., 2007). It cannot be asserted that carbon saturation was reached. This could have been assessed by measuring carbon levels over time and observing whether they tapered off. Unfortunately, carbon levels were not measured in 2018 or 2019. However, as soils approach carbon saturation, the efficiency of carbon storage from new carbon inputs decreases (Chung et al., 2010; Stewart et al., 2007). Therefore, even full saturation was not reached, being close to saturation may have slowed the rate at which the soil could store additional carbon. This may have reduced the significance of any impact that the brush packs had on soil carbon levels.

Contrary to the hypothesis, control patches generally exhibited higher carbon levels than brush pack patches (Figures 4.12 and 4.13), although this trend was not statistically significant (except for the aggregate carbon fraction at 3-5 and 5-10cm depth intervals). This may be due to significant variation in the amount of material used for the brush packs, with some packed quite densely. Milton and Coetzee (2022) suggest dense brush packs can exclude light, inhibiting grass growth. Because grass growth leads to increased litter production and root-derived carbon inputs, factors which can lead to increased soil carbon levels (Poeplau et al., 2021; Xu et al., 2021), this may have led to generally higher carbon levels at control patches.

5.2.3 Soil Chemical Properties

Soil pH and EC levels were measured to see if there were any differences in the chemical properties of the soil between control and brush pack patches. It was expected that brush packs would not change pH or EC levels.

The magnitude and variability of pH and EC levels observed in this study are comparable to Bann (2016), who conducted a study on the pH and EC of soils across multiple sites within the Southern Tablelands of NSW. This region shares similar environmental conditions with the Main Site in this study. However, it is important to note, in this study soil samples were sieved to <1mm not the standard <2mm which may limit this comparability.

The pH levels showed no significant differences, except for the 1-3cm depth interval where brush pack patches had a higher (less acidic) pH. This was unexpected, as any influence of brush packs on pH we would expect to see the differences in the uppermost (0-1cm) interval first and then potentially extending down to other intervals. The absence of a difference in the 0-1cm interval means the significant difference at 1-3cm cannot be solely attributed to change driven by brush packs. This difference could be attributed to random pH variability which was higher at control patches.

EC values showed no significant differences. Control patches exhibited higher EC variability, including two control patches with particularly high EC levels. These two sites are visible outliers across all four

depth intervals. The higher variability of pH and EC at control patches may reflect a greater diversity amongst control patches as suggested with the soil respiration results.

High spatial variability in pH and EC is well-documented in southeastern Australia, with bare areas often exhibiting the most extreme variation in both pH and EC levels (Bann, 2016; de Caritat et al., 2011; Semple et al., 2006). EC generally decrease as vegetation cover increases, with more vegetated areas typically showing lower EC levels (Bann, 2016; Barrett-Lennard et al., 2003; Semple et al., 2006). Brush packs may have initially increased pasture growth, as in 2019 they had significantly higher infiltration and nutrient cycling LFA indices, which may have led to less extreme EC levels. There may also be a legacy effect, with control patches starting with higher EC levels and more variable pH and EC levels. Unfortunately, we cannot assert either of these ideas without pH and EC data from 2018 and 2019. An analysis of the pH and salt levels in the *Leptospermum* branches used to construct brush packs could clarify if they are likely to influence pH and EC.

5.3 Implications

This study expands the range of climatic conditions under which brush packs have been studied. While previous research focused on semi-arid and arid climates, this study examined brush packs in a temperate, mesic climate. Improvements in functionality driven by natural recovery and significant pasture growth are comparable to improvements from brush packs in this study. As landscape function increases, the relative value of resource-accumulating patches, such as brush packs, decreases (Whisenant, 1999). Therefore, brush packs may have limited utility as a rehabilitation technique in resilient, mesic landscapes where grazing is sustainable.

Although climatic conditions likely masked some of the impacts, this study demonstrates that brush packs constructed from *Leptospermum* branches, a renewable resource that was already being routinely cleared, can persist for over six years through significant rainfall and continued grazing. This persistence is important, as it means brush packs continue to improve landscape function by intercepting overland flows and capturing and retaining resources such as soil, water, and plant detritus. The longer-lasting an intervention is, the more labour- and cost-effective it becomes.

The 2019 LFA data suggests that brush packs likely had a head start in terms of functional recovery, before the rare ‘triple-dip’ La Niña event facilitated natural recovery. Furthermore, also in 2019, the property manager observed a reduction in erosion at, and downslope from the site compared to other areas (Cantwell, pers. comm. 2024). At the fine-scale brush packs were found to significantly improve several indicators of function. They provided immediate soil cover and protection from rainsplash erosion. They increased soil surface roughness, filtering overland flows and capturing detritus. They also increased litter cover and incorporation by providing litter input through decomposition and trapping material from upslope, and creating habitat for macroinvertebrates, which helped incorporate organic matter into the soil. These improvements in functionality suggest, as a rehabilitation technique,

the brush packs may provide lasting benefit, if they persist, even if conditions were to shift back into a period of drought and heavy grazing were to resume.

5.4 Limitations

There was some variation in the construction of the brush packs, including their height, width, and length. There was also variation in the number and spacing of the brush packs along each transect. This limited the comparability of each brush pack transect and patch, particularly for the 2019 LFA data where not all brush pack transects and patches were measured.

The lack of baseline and short-term soil respiration and carbon data from 2018 and 2019 makes it challenging to distinguish between the impacts of brush packs and natural recovery. It also makes it difficult to contextualise the current data, particularly considering the significant pasture growth observed, as we do not know for certain or to what degree soil respiration and carbon levels improved significantly over the study period.

A significant limitation of this study is that the soil respiration measurements capture only a single snapshot in time. Soil respiration is known to vary significantly across seasons and due to climatic conditions (Bremer & Ham, 2002; Raich & Schlesinger, 1992). Because of this, it cannot be asserted that the results of this study will persist throughout the year. Taking more CO_2 efflux measurements over an entire year and under different climatic conditions would provide a more comprehensive understanding of the relationship between brush packs and CO_2 efflux.

There were also some issues with the model and data used to assess soil respiration in this study. Relative to the range of CO_2 efflux values observed, the model had a high residual standard deviation which indicates significant random variation in CO_2 efflux was not explained by the model (Table 4.6). This could be due to the highly variable nature of CO_2 efflux, or that there is another covariate driving this variability that was not included in the model. Moyano et al. (2012) suggest several soil properties such as bulk density, clay content, and organic content could impact soil CO_2 efflux measurements. Some soil properties that differed, such as pH and EC, or likely differed such as bulk density, were not included in the model. Both pH and EC, although not significantly different, showed greater variability in control patches, similar to the greater variability observed for soil respiration (Figures 4.14, 4.15, and 4.10). Bulk density may also have varied, although it was not measured, particularly if the brush packs excluded livestock during the study, thereby reducing soil compaction. Future models should consider incorporating bulk density, pH, EC, and other soil properties as additional fixed effects.

This study directly measured nutrient cycling due to 2024 LFA data indicating this was improved at brush pack patches. However, the three SSA indicators which lead to this improvement also contribute to the infiltration index. In 2019, at the transect-level, brush pack transects have significantly higher infiltration index scores compared to control transects. Unfortunately, direct measurements of

infiltration could not be completed due to weather conditions (saturated soils). This limited the ability of the study to assess the infiltration aspect of landscape function.

5.5 Future Research

This study found that when the surrounding landscape is functional the relative functionality of brush packs is comparable. However, if conditions were to shift, such as during a period of drought, and landscape functionality was to decrease, the relative functionality of brush packs may increase, if they persist. This leads to the question: Will the brush packs in this study provide lasting benefits to landscape function under drought conditions?

Brush packs were found to have a minimal proportional contribution to landscape function at the transect-level, this limits their impact on function at the landscape-scale. The LFA method calculates this contribution based on two factors: the average LFA scores of all the patches assessed on the transect, and the proportion of the transect area which they occupy. While the average LFA scores cannot be directly altered, increasing the area or extent occupied by brush packs should increase their proportional contribution. This leads to the question: Does increasing the number or physical extent of brush packs increase the significance of their impact on function and the landscape-scale?

A diverse range of materials have been used to construct brush packs. The brush packs used in this study were constructed from locally sourced *Leptospermum* branches, while Tongway and Ludwig (1996) used locally sourced *Acacia aneura* branches. Other studies on brush packs have also utilised various locally sourced materials (Kimiti et al., 2017; Milton & Coetzee, 2022; Naude, 2017; Pelser, 2017; van den Berg & Kellner, 2005). However, there has been no analysis of whether these different materials vary in their durability or impacts on function. This leads to the question: How do different construction materials, such as branches from various species, influence the durability and effectiveness of brush packs in restoring landscape function?

Tongway and Ludwig (1996) found brush packs supported significantly higher numbers of macroinvertebrates compared to control treatments. Other studies have also shown different forms of woody debris can increase macroinvertebrate abundance (Barton et al., 2011; Grodsky et al., 2018; Parkhurst et al., 2022). Brush pack patches had significantly higher LFA scores in litter cover and incorporation in this study. Litter provides shelter and contributes organic matter as a food source for macroinvertebrates, which in turn, increase the nutrient cycling into the soil. This leads to the question: What impact do brush packs have on macroinvertebrates, and does this impact influence landscape function?

Chapter 6: Conclusion

This study aimed to assess the impacts of brush packs on landscape function, with a specific focus on nutrient cycling, at a degraded, heavily grazed, mostly bare ground site with a temperate, mesic climate in southeastern Australia. It addressed a gap in the existing research which has largely focused on the impacts of brush packs in semi-arid and arid environments. Both landscape-scale (transect-level) and fine-scale (zone-level) impacts were assessed to provide a more comprehensive assessment.

Landscape function was first assessed using Landscape Functional Analysis, with the expectation that they would improve scores in all three of the indices: stability, infiltration, and nutrient cycling. Brush packs showed no significant improvements in function at the landscape-scale relative to control transects 6 years after establishment. However, at the fine-scale they showed improvements in all three indices relative to control patches. After this initial assessment revealed significant differences in nutrient cycling, direct measures of soil respiration and carbon levels were taken to further investigate this. Chemical properties, pH and EC, were also measured to see if brush packs had any influence on these soil properties.

Contrary to expectations based on previous studies, brush packs did not significantly improve soil respiration and carbon levels relative to control patches. However, during the study both brush pack and control patches experienced significant functional improvements due to substantial pasture growth driven by a rare three-year La Niña event. The natural recovery across the site likely overshadowed any additional benefit provided by the brush packs. As expected, the brush packs did not influence chemical properties.

These findings suggest that brush packs may have limited utility as a rehabilitation technique in resilient, mesic landscapes. Data taken one year after the brush packs were established and before the three-year La Niña event drove natural recovery suggests brush packs likely had a head start in terms of function recovery. If conditions were to shift towards dysfunction, such as during a period of drought, the relative functionality of the brush packs may increase, if they persist. Therefore, brush packs could offer lasting benefits to landscape function under drought conditions and heavy grazing.

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Appendix 1 – Home Farm Climate Data

This includes all the raw data kindly compiled by Chris Inskeep (TMI). It also includes the transformations I made. The full dataset is too large to be attached to this document and can be made available upon request.

Appendix 2 – Chain of Custody (soil samples)

This includes the sites and dates of where I sampled from. As well as the initial time spent drying in the oven and subsequent drying times once the soil was fractionated for the carbon analyses. It also includes dates of some of the analyses. The full dataset is too large to be attached to this document and can be made available upon request.

Appendix 3 – LFA Data & Edits Record

LFA Data:

The LFA data includes an Excel file for each transect measured in 2018, 2019, and 2024. These files are too large to be attached to this document and can be made available upon request.

LFA Edits record:

This includes some of the edits I made to the LFA data and the rationale behind them. It also includes how I aggregated all the different zone types that were recorded.

Edits to 2018 student data:

- *Students did not record some zones on their second day of measurements so there were erroneous differences between control and brush pack transects when the only difference there should have been was the addition of the brush packs. This significantly impacted the calculation of each index. To solve this problem. I took all the distance, width, and SSA measurements from the post-brush pack data and inserted these as new zones into the pre-brush pack or control data. This meant the only difference between the control and brush pack transect LFA data was the brush packs which was the difference I wanted to measure.*

- All Litter rainsplash/soil cover changed to 1. Litter doesn't count as rainsplash protection.
- All CWD zones rainsplash/soil cover changed to 5. CWD counts as rainsplash protection
- Students recorded inconsistent textures. All textures changed to 3
- Group 4 transect 22-21 forgot to finish filling out the SSA indicators for one of their BP measurements. So, I finished filling it out based on what they had for previous BP measurements (which were all the exact same).
- Group 2 randomly had soil cover as 4 for sparse grass/scale in their pre brushpack data. Their pre and post data for sparse grass/scald was exactly the same except for this erroneous number. So, I changed it to 2 to make their data the same and because it is unlikely that a sparse grass/scald zone would have such high soil cover.
- Group 2 made an error in their post BP data. 70cm width BP patch should be CWD, not BP. I believe they just saw CWD that was already on the ground and decided to call it a brush pack.
- Changed group 3 slake test from 0 to 1 for Litter in post and pre BP as this matches their other data and slake should not have been 0. This also helps the spreadsheet function properly by not ignoring slake test when calculating the indices.
- Fixed group 5 who incorrectly entered landscape organisation data, they had their starts and ends mixed up.
- Surface resist. to disturb. Values that were 2.5 changed to 3 so the spreadsheet would function properly.

Edits to 2019 David and Luke data:

- Transect 22-21. In the landscape organisation sheet I swapped 22.3 with 21.7 because they were in wrong order, larger distance number should not be above smaller distance number.

Zones aggregated:

Old	New
Brushpack	Brushpack
Brushpacks	Brushpack
SCALD	Bareground
Bareground	Bareground
bare with litter	Bareground
Sedge	Sedge
Grass Sedge	Sedge
Grassy tussock	Grass Sward
Grass Sward	Grass Sward
Grassy Tussock/Sward	Grass Sward
Grassy Sward/Tussock	Grass Sward
Interpatch	Sparse Grass
Bare/ Some Grass	Sparse Grass
GRASS	Sparse Grass
Grass sward with litter	Sparse Grass
Sparse Grass	Sparse Grass
Sparse Grass/Scald	Sparse Grass
Bare & Patchy Grass	Sparse Grass
log	Coarse Woody Debris
Coarse Woody Debris	Coarse Woody Debris
Unchanged:	
Litter	Litter
Rock	Rock

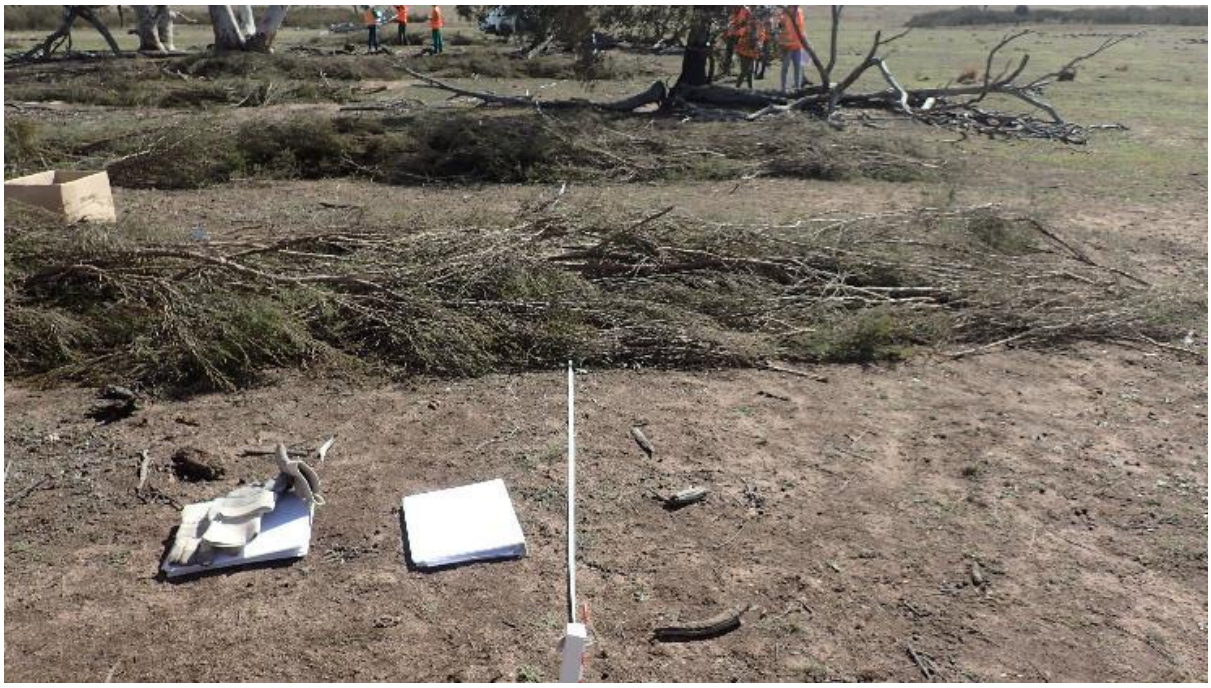
Appendix 4 – Soil Respiration Raw Data and R Code

This includes the raw data from the LI-6400, the cleaned and formatted dataset I fed into R, the uncleaned dataset, and the code used to generate the linear mixed-effects model and graphs presented in this thesis. The full dataset and code are too large to be attached to this document and can be made available upon request.

Appendix 5 – All Site Photos

This includes all of the photos I have of the Main Site and Reference Site. Most of them are courtesy of David Freudenberger. I have attached a few of them here to show the change in site over the years, all of these were taken by David Freudenberger (see if you can spot yourself if you were one of the students!). But there are too many to attach to this document, the rest of the photos can be made available upon request.

Same Transect pre-brush pack 2018, post-brush pack 2018, and in 2024:



Examples of interpatch 2018 vs 2024:

Examples of a brush pack 2018 vs 2024:

Appendix 6 – Carbon Summary Data

These summary tables include the data used to make all the plots presented in the results chapter and the results of all significance testing. An excel file of these and the raw data used to calculate the means and standard errors can be made available upon request.

Table of means, SEs, and p-values between treatments of the same fraction and depth interval:

		Treatment				
		Brush pack		Control		
Fraction	**Depth Interval**	**Avg**	**Std. Error**	**Avg**	**Std. Error**	**p-value**
Bulk/Unfractionated	**0-1cm**	11.5999	1.724476	11.2946	1.979672	0.9087
	1-3cm	4.8129	0.495652	6.4846	1.327946	0.2536
	3-5cm	2.8077	0.169341	3.944333	0.626205	0.0837
	5-10cm	1.9127	0.159993	3.1809	0.59994	0.056
POC	**0-1cm**	5.383838	1.369491	5.747073	2.214556	0.890603
	1-3cm	1.429742	0.291705	2.861961	1.638231	0.400718
	3-5cm	0.496003	0.058593	1.850755	1.142023	0.251537
	5-10cm	0.315493	0.032601	0.830229	0.344654	0.154361
AggC	**0-1cm**	3.814319	0.311335	4.33811	0.465952	0.362321
	1-3cm	1.979864	0.164987	2.672665	0.370225	0.104588
	3-5cm	1.084885	0.13687	1.855517	0.26478	**0.018661***
	5-10cm	0.451312	0.062692	0.974189	0.212595	**0.029824***
MAOC	**0-1cm**	0.676114	0.071324	0.580981	0.036289	0.249972
	1-3cm	0.806545	0.073446	0.812586	0.041168	0.943597
	3-5cm	0.95355	0.081776	0.983816	0.063338	0.773176
	5-10cm	0.877675	0.087707	1.053114	0.109258	0.226529

Table of significance testing between depth intervals of the same treatment and fraction:

Treatment	**Fraction**	**Depth Comparison**	**p-value**
Brush pack	**Bulk/Unfractionated**	**0-1cm vs 1-3cm**	**0.0014****
		1-3cm vs 3-5cm	**0.0012****
		3-5cm vs 5-10cm	**0.0012****
	AggC	**0-1cm vs 1-3cm**	**5.95E-05*****
		1-3cm vs 3-5cm	**0.000569*****
		3-5cm vs 5-10cm	**0.000528*****
	MAOC	**0-1cm vs 1-3cm**	0.218866
		1-3cm vs 3-5cm	0.197738
		3-5cm vs 5-10cm	0.534861
	POC	**0-1cm vs 1-3cm**	**0.011245***
		1-3cm vs 3-5cm	**0.005682****

		3-5cm vs 5-10cm	**0.0149***
Control	**Bulk/Unfractionated**	**0-1cm vs 1-3cm**	0.0588
		1-3cm vs 3-5cm	0.1138
		3-5cm vs 5-10cm	0.3912
	AggC	**0-1cm vs 1-3cm**	**0.011876***
		1-3cm vs 3-5cm	0.089418
		3-5cm vs 5-10cm	**0.018273***
	MAOC	**0-1cm vs 1-3cm**	**0.000514*****
		1-3cm vs 3-5cm	**0.035966***
		3-5cm vs 5-10cm	0.589936
	POC	**0-1cm vs 1-3cm**	0.308793
		1-3cm vs 3-5cm	0.618748
		3-5cm vs 5-10cm	0.403518