Effect of grazing strategies on soil health in pastures, and potential use of spiders as biological indicators

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Abstract

Regenerative grazing, involving high-density herds moved frequently to new pastures with extended recovery periods, allows plants more time to produce soil exudates, while trampled residues create a protective mulch layer.

This study examined if adopting regenerative grazing practices produces significant changes in soil health indicators compared to non-adopting sites. A grazing score based on herd density, pasture length, and recovery time measured the adoption of regenerative grazing practices. Ground-dwelling spiders were used as a novel indicator of soil health in New Zealand pastoral farms, specifically in relation to regenerative farming practices. Other indicators, including earthworm abundance, soil aggregate sizes, soil carbon levels, pasture diversity, pasture brix levels, soil nutrients, and habitat diversity, were also considered.

There were no significant differences in soil health indicators between "regenerative" and "conventional" farms. Secondary analysis exploring the relationship between grazing score and soil health indicators also yielded no significant findings. Pasture slope had an unexpectedly positive effect on some indicators, and habitat diversity had a greater impact on spider abundance than the grazing score. Canonical analysis confirmed the previous findings and identified significant relationships between slope, earthworm abundance, and soil sulphur.

In conclusion, adopting regenerative grazing practices did not yield significant changes in soil health indicators compared to non-adopting sites. Spiders, which were primarily introduced species, were not effective indicators, due to their low abundance in pasture habitats. The study recommends measuring multiple soil biological health indicators collectively to comprehensively assess agricultural practices' impact on soil health.

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1 Introduction

As agricultural practices expand in area around the world and production on those lands intensifies, it is becoming increasingly evident that the ecological life-supporting systems that support agricultural production are deteriorating at a rapid rate under the pressure (Rapport 2003). At the heart of this deterioration is the degradation of natural environments when they are transformed to biologically simplistic agricultural systems to provide natural resources to humans (Foley *et al.* 2005). The strain ecosystems are under now is clear to ecologists around the world – as stated by Foley *et al.* (2011) "*just as our collective land-use practices are degrading ecological conditions across the globe, humanity has become dependent on an ever-increasing share of the biospheres resources*."

The rapid transition of the landscape from forests to simplistic agricultural systems has been particularly evident in New Zealand (Pannell *et al.* 2021), where pastoral agriculture now predominates (Caradus *et al.* 2021). In response to this pressure, new methods of agricultural production are necessary that integrate beneficially with native ecosystems, while being able to produce a sustainable yield of resources to meet the needs of landowners (Dudley and Alexander 2017).

Regenerative agriculture practices have received increased attention for their role in re-shaping current land management practices through actions that aim to restore the health and resilience of farmed landscapes while providing a viable livelihood for farmers and preserving native biodiversity and the function of associated natural ecosystems (Bauddh *et al.* 2020).

In grazing systems, adopting regenerative agriculture as a method of holistic grazing pioneered by Alan Savory is most commonly practiced (Burns 2020). This philosophy is based upon the management of livestock to mimic the grazing of large herds of herbivores on natural grasslands of North America (Nordborg 2016). The application of this grazing management philosophy is especially well suited in sensitive landscapes with erratic rainfall, where small differences in the management of livestock can have dramatic impacts on the landscape (Wezel 2017; Butterfield *et al.* 2019).

To justify the costs of adopting different management practices seeking to achieve the goals of regenerative agriculture, quantitative assessments are required to guide management practices and to report back to stakeholders (Wang *et al.* 2021a). The biological health of the soil is increasingly seen as an important aspect of quantifying the *'regenerative'* aspects of agricultural management practices (Sherwood and Uphoff 2000), due to having the long history of quantifiable measures (Klute 1986) and due to being the most identifiable characteristic shared by all farm sites (Juerges and Hansjürgens 2018).

Soil assessments were historically more focused upon production (Industries 2017), physical, or chemical aspects (Klute 1986). Recently, there has been an increased need to include measures of soil biological health (Kremer and Hezel 2013), and recent advances in the understanding of the function of soil ecosystems have aided the development of more complex indicators to quantify this (Timmis and Ramos 2021). However, much research is still required to identify suitable indicators of soil biological health (Menta and Remelli

2020), especially in respect to assessment that can be carried out without extensive expertise and resources (Hou *et al.* 2020). This is especially true in New Zealand, where agricultural landscapes are highly modified and are now dominated by a simplified range of introduced species (MacLeod 2022).

Research suggests that spiders are strongly associated with land use and habitat structure (Costall 2012), and their diversity and abundance can be used as good bioindicators of ecosystem and soil health (Schwerdt *et al.* 2018). Being keystone terrestrial invertebrate predators at the top of the soil community food web (Topping and Lovei 1997), spiders could, therefore, act as indicator species to represent changes throughout the soil food web (Scott *et al.* 2006). They also have appeal due to their larger size, diversity, widespread distribution, and ease at which they can be sampled with pitfall traps (Sherley 2016). Spiders live on the surface of the soil, so are also not likely to be as strongly impacted by different soil types and hydrological conditions as true soil-living biota, such as earthworms (Coombe 2001).

In this study, spider abundance was investigated alongside other established soil health indicators, on six New Zealand pastoral farms, three of which have adopted regenerative grazing practices and three have self-identified as ‘conventional’. Established indicators of soil health measured included the abundance of earthworms, the mean size of soil aggregates, soil carbon content, and pasture diversity. Pasture brix levels were also measured, as they are commonly used as indicators of soil and pasture health by regenerative grazing practitioners.

This study had three main goals: (1) to assess the impact of regenerative grazing practices on indicators of soil biological, (2) to assess the relationship between a regenerative grazing practices to biological indicators of soil health score and (3) to assess the validity of spider abundance as a bioindicator of soil biological health.

2 Literature Review

2.1 Impact of Agriculture in New Zealand

Before humans reached New Zealand, it was largely covered in temperate and sub-tropical forest (Norton *et al.* 2020). With the arrival of the first Polynesian people to New Zealand between 1250 and 1300 AD, there followed a period of widespread forest clearance by fire, which could be seen in the pollen and charcoal deposits within the plant macrofossils (Gibbs 1980; Perry *et al.* 2014). The clearance of forest by fire resulted in large areas of the eastern and southern New Zealand being transformed to scrub (Perry *et al.* 2014), while wetter areas in the west and north were retained as forest. The remaining tracts of native forest were then subjected to more intensive burning and felling in the second wave of European settlers in the late 18th and early 19th centuries, associated with clearing forest and scrub and establishing pastures (Forbes *et al.* 2020). The result of this was a rapid transition of New Zealand ecosystems from 90% to less than 20% forest cover within a few hundred years (Newnham *et al.* 2018).

Broadscale clearance of land for agriculture only started 150 years ago – far more recent than in other western countries (Schipper *et al.* 2007). The change in the New Zealand landscape is striking due the speed of change that occurred and the dramatic difference between the diverse forests into pastures dominated by only a few exotic species (Pawson 2018). By 1930, 40% of New Zealand's land area had been converted to pastures for agriculture (Tilman 1999), with most of this area within the lushest lowland podocarp forest and swamp areas with the highest biodiversity (Newnham *et al.* 2018). This transition was made possible through the logging of high value timbers and burning what was left to establish exotic pasture grasses (Tilman 1999).

2.1.1 Development of Pastures

The conversion of fertile lowland forests to agriculture in NZ follows the trends of other developed nations but differs in that pastures, instead of crops, were the dominant form of agriculture (Schipper *et al.* 2007). The preference for animal agriculture supported on pastures was due to the initial lush growth of pastures, mild climate and consistent rainfall, combined with good overseas markets for meat and wool fuelled by the industrial revolution in Britain and shortage of supply due to world wars (Perry *et al.* 2014). The favourable conditions for pasture growth in New Zealand's enables 95% of the diet requirement of livestock to be supported from pasture grazing (Pawson 2018; Norton *et al.* 2020).

Grazing lands have been transformed around the world by the introduction of non-native grass species and application of herbicides and fertilisers (Baskaran *et al.* 2009). This intensification has been supported by the conversion of biodiverse small-scale farming systems into large-scale and intensive monoculture systems (Dudley and Alexander 2017). A major factor in fuelling this intensification of agriculture is an increase in the annual global fertiliser consumption, which in 2020 is estimated to be an average use 146.4 kg of fertiliser per hectare of arable land (Bank 2023).

According to Stats NZ, (StatsNZ 2021) annual fertiliser consumption in New Zealand (measured by the amount of nitrogen and phosphorus-based fertilisers applied to agricultural land) has increased by 629 %, from 62,000 to 452,000 tonnes, between 1991 and 2019. This is illustrated in Figure 1 below.

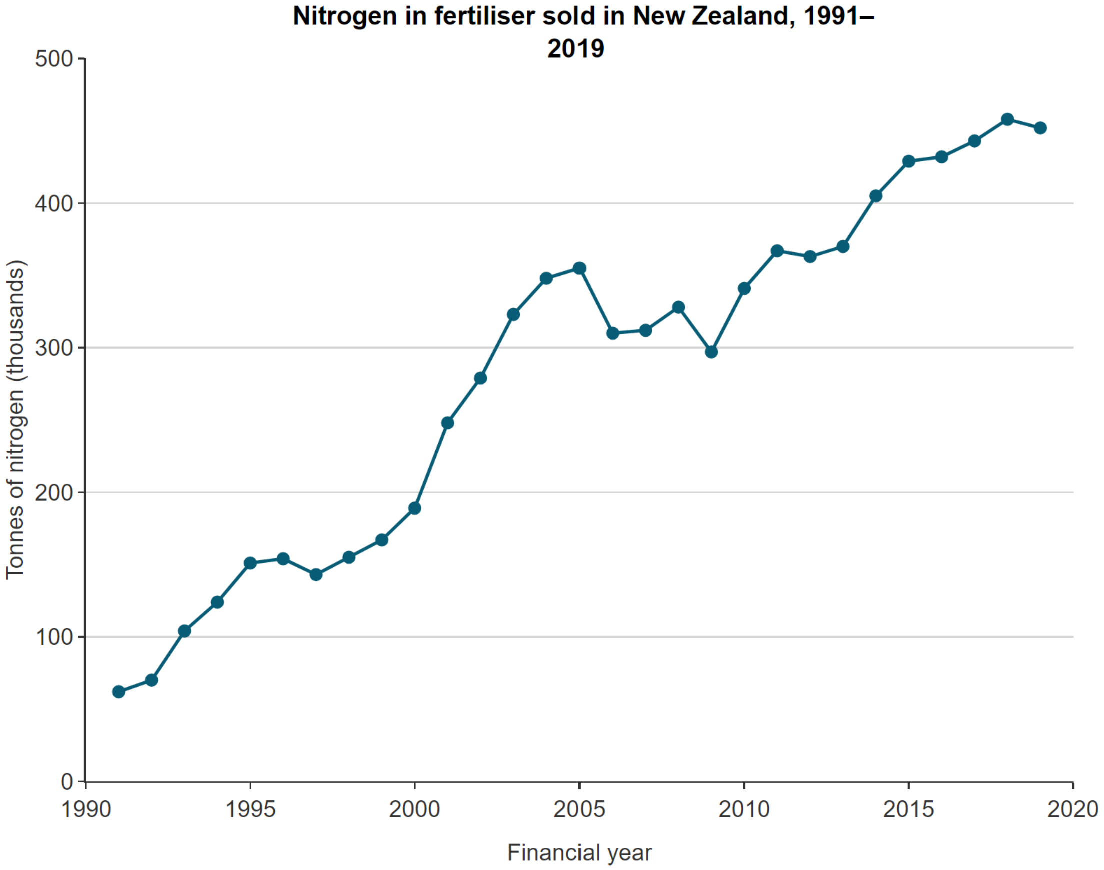


Figure 1. The increase in fertiliser consumption in New Zealand between 1990 and 2020, as measured by thousands of tonnes of nitrogen fertiliser sold (StatsNZ 2021)

The decline in soil fertility, following agricultural conversion of landscapes, was counteracted by the adoption of superphosphate fertilisers in the 1950's and the ability to access remote farms through aerial topdressing application (Perry *et al.* 2014). The use of fertilisers aims to enhance soil fertility, leading to increased plant production and improved quality, which has implications for food production, exports, and agricultural intensification (Caradus *et al.* 2021).

The increased use of fertilisers within the last few decades coincides with an increased land area being used for dairy cattle farming rather than sheep and beef farming (Parfitt *et al.* 2006). Dairy farms have the highest fertiliser application rates (see Figure 2), followed by sheep and beef farms and grain-growing farms (StatsNZ 2021). Superphosphate was the primary fertiliser used, followed by urea (StatsNZ 2021).

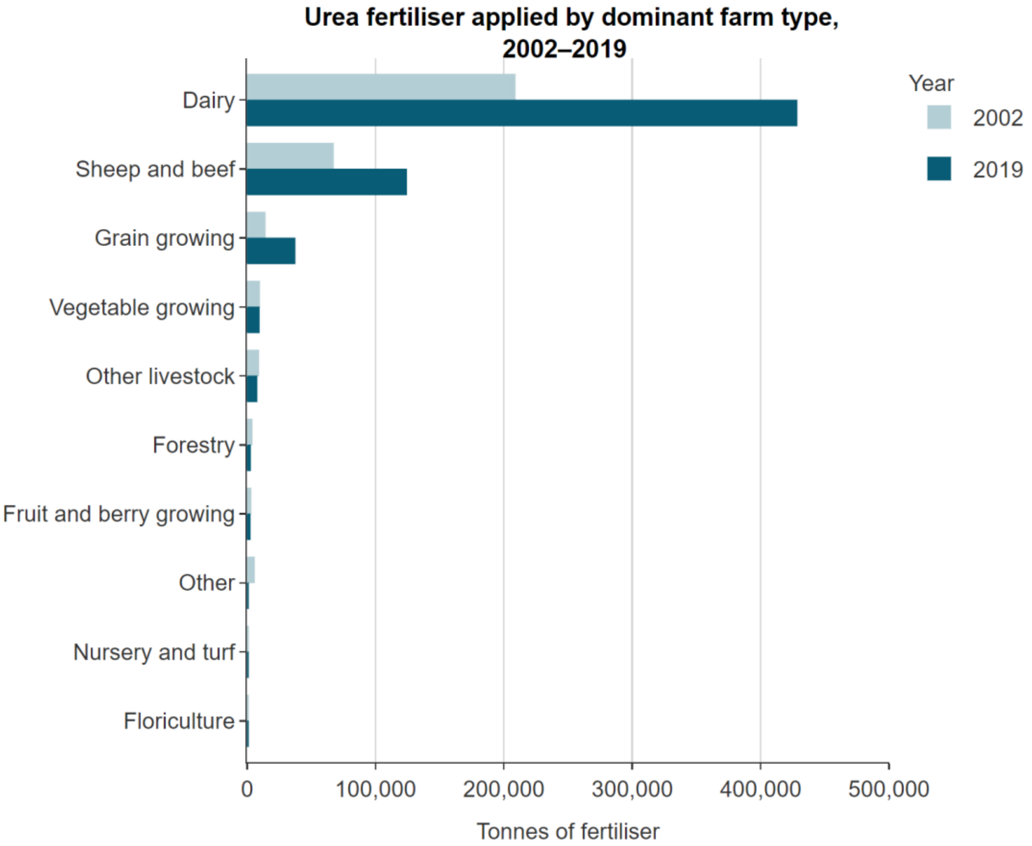


Figure 2. Relative consumption of fertiliser in different agricultural industries in New Zealand in 2002 and 2019 (StatsNZ 2021).

The increased application of fertilisers, combined with concentrated affluent in dairy milking sheds led to an increase in water pollution during this time (Dudley and Alexander 2017), resulting in eutrophication of lakes, rivers and streams and degradation of habitat for native aquatic species (Joy 2015). The 2004 '*Growing for Good'* report by the NZ Parliamentary Commission for the Environment found that intensive farming was leading to a serious decline in national water quality in New Zealand (Brown and Roper 2017). Water quality has since continued to decline in many areas, with 46% of NZ freshwater lakes now considered in poor or very poor ecological health (Ministry for the Environment 2023).

Due to an increasingly competitive export market, the intensification of pastoral agriculture in New Zealand has continued to increase since the 1970's, driven by increased use of fertilisers and water use (Pawson 2018). This has enabled more milk, lambs, wool, and meat to be produced from the same land area (Schipper 2017). In recent decades there has been a boom in dairy production, with New Zealand now the world's largest exporter of dairy products and supplying one third of global dairy trade (Baskaran *et al.* 2009). Unusual amongst, OECD countries, half of New Zealand's greenhouse emissions come from agriculture (OECD 2017).

The intensity and scale of dairy farming in New Zealand is driven by the desire to retain its global competitive advantage as an export producer (Baskaran *et al.* 2009). Part of the pressure around the intensification of agriculture is due to the necessity to compete in a global marketplace against international competitors that often receive heavy subsidies from their governments to support production (Ministry for Primary Industries 2017). By contrast, New Zealand has the lowest level of agricultural subsidies in the OECD (Industries 2017). Instead, farmers in New Zealand operate as any other business, with the extra financial constraints of not being able to adjust prices of goods sold, as milk prices are set by Fonterra (Statista Inc 2020). This requires NZ farmers to maintain viability within the open and competitive market through maximising productivity gains and intensification (Manderson *et al.* 2007). This drive for more intensive production is further reinforced by government goals to increase agricultural productivity and profitability from $32 billion in 2012 to $64 billion by 2025 and from 30 to 40 % of GDP (Brown and Roper 2017). This would require farmers to farm at an even more intensive rate (Brown and Roper 2017).

2.1.2 Impact on Soil Health

The soils in New Zealand, which initially supported good pasture growth (due to the nutrient supply from clearing and burning the forest), were rapidly depleted (Newnham *et al.* 2018). This was due, in part, to the high levels of soil erosion due to removal of forests that stabilised soils, combined with the high rainfall on often rolling farmland terrain (Perry *et al.* 2014). Other factors were the destruction of the existing ecosystems, with their soil-building and nutrient-recycling organisms (Gibbs 1980), and the characteristic tendency of NZ soils to have shallow topsoil and deep clay-based subsoil (Tilman 1999). This shallow topsoil was particularly prone to erosion once the protective forest cover was removed (Donovan 2022).

Soil erosion continues to have a large impact on the productivity of agricultural landscapes in New Zealand and it is estimated that approximately 200-300 million tonnes of soil are carried out to sea every year (Gibbs 1980; Haggerty and Campbell 2008), a rate well above the global average for soil loss (Haggerty and Campbell 2008). The economic cost of hill country erosion is estimated to cost New Zealand between $100 and $150 million per year (Dominati and Mackay 2013). Further, the agricultural landscapes that replaced the native ecosystems lack the biological functionality to rebuild the lost topsoil (Churchman and Velde 2019). Continued land clearance, over-grazing and grazing marginal land all continue this decline in quantity and quality of topsoil available (Churchman and Velde 2019).

2.1.3 Loss of Biodiversity

Globally, conservative estimates are that up until 2010, agriculture has accounted for a 60% decline in terrestrial biodiversity (Kok *et al.* 2018). These losses are projected to increase under the pressure of climate change and further habitat loss (Kok *et al.* 2018). A major challenge facing modern agriculture production is to improve soil health (Thangavel and Sridevi 2017), increase native biodiversity (Dudley and Alexander 2017) and integrate beneficially with surrounding natural habitats (Martin *et al.* 2019). Some go as far as aspiring to replicate the functions of equivalent natural ecosystems to create agricultural ecosystems resilient to disturbances and requiring much less inputs to sustain production (Hoy 2015).

In New Zealand, where broadacre grazing is the predominant form of pastoral agriculture it has been claimed by some that the impact of agriculture is less severe and already aligned to a global trend shifting towards '*regenerative agriculture'* (Neumann 2021). However, the biodiverse temperate forests that covered the majority of low-elevation NZ landscapes have now been largely replaced by almost a monoculture of ryegrass-dominated pastures (Meurk and Swaffield 2000). This simplification of the landscape leads to a loss in biodiversity and function of ecosystems, reduction in population viability in native wildlife (England *et al.* 2020) and loss of species (Krebs and Bach 2018).

Despite the impact of agriculture on natural habitats, relic pockets of natural habitat form important refuges to preserve biodiversity (Meurk and Swaffield 2000). In New Zealand agricultural hill country, many steep gullies provide important refuges of native habitat. Pannell *et al.* (2021) estimates that up to 25% of native vegetation cover occurs in hill country farms. Restoring these areas to enhance biodiversity and increasing the diversity of farming systems to provide alternative habitat and vegetation corridors through farmland are important strategies for enhancing biodiversity on agroecosystems (Monjardino *et al.* 2010).

2.2 Potential Role of Regenerative Agriculture

New Zealand enjoys an international reputation for '*clean and green'* agricultural production, with many overseas consumers seeking out New Zealand agriculture products due to the perceived benefit to the environment from production on these farms (Brown and Roper 2017). This is in stark contrast to the dramatic loss of biodiversity (MacLeod *et al.* 2022), transformation of natural habitat (Meurk and Swaffield 2000) and pollution of waterways (Joy 2015) that has been documented in creating these farming systems.

The dairy industry in New Zealand has received increasing criticism domestically due to its environmental impacts (Baskaran *et al.* 2009), which is supported by increasing pressures for the environmental footprint to be reduced (Norton *et al.* 2020). In a survey of different New Zealand agriculture sectors by Grelet *et al.* (2021), these concerns were expressed as a demand for more research on the impact of farming on water quality, the impact of intensive production-focused farming systems based on monocultures, the lack of biodiversity and lack of value added to produced commodities.

Over the last few decades, NZ farmers have become more aware of the need to become more sustainable, through more efficient use of agrochemicals, fertilisers and other management tools. The increased awareness of the environmental impact of agriculture in New Zealand has led to greater efficiencies in production and a reduction in chemical inputs (Brown *et al.* 2019). In many cases, these changes in management practices also lead to greater economic gain through a reduction in inputs (Brown *et al.* 2019). This has coincided with government-level initiatives such as the target of being carbon-zero by 2050 (Forbes *et al.* 2020).

One theme that has emerged in recent years in response to adopting more sustainable agricultural practices is regenerative agriculture. The goal of regenerative agriculture is to not only make agricultural practices more sustainable (Krebs and Bach 2018), but also to regenerate ecological functions back into farm systems to improve their health and resiliency (Hes *et al.* 2018). The regenerative approach seeks to improve the function of farm systems by modelling them upon the patterns seen in equivalent natural systems (Svec *et al.* 2012; Schreefel *et al.* 2020; Tommy *et al.* 2021). The momentum behind adopting regenerative agriculture is the increasing recognition that conventional methods of farming are doing harm to our environment, to livestock, and to people (Hes *et al.* 2018).

Developing a formal definition of regenerative agriculture is a work in progress. It has been variously described as an agricultural system which aims to "*increase soil health and promote biodiversity while producing nutritious food profitably"* (Fenster et al. 2021), or "*as a scientific discipline as well as an agricultural practice and movement, with the goal to create a more sustainable agriculture*" (Krebs and Bach 2018). At the heart of regenerative agriculture approaches is the desire to integrate agricultural systems with natural systems – so that both can continue in perpetuity in a healthy and resilient state (Svec *et al.* 2012).

New Zealand's agricultural industry is showing considerable interest in the adoption of regenerative agriculture and branding of produce in alignment to this ethos of farm production (Beef&LambNZ 2021). This is in recognition of regenerative agriculture being aligned to global trends shifting towards food production that has a favourable impact on the environment (Grelet et al. 2021) and produces nutrient dense food to optimise human health and wellbeing (LaCanne and Lundgren 2017). Adopting regenerative agriculture practices may offer producers an opportunity to target premium overseas markets (Neumann 2021), such as already seen in the land to market initiative by the Savory Institute operating in New Zealand (LandtoMarket 2022) This requires constant innovation for the sector to minimise environmental impact from production – while also maintaining high production rates (Baskaran *et al.* 2009).

Grelet *et al.* (2021) indicates that New Zealand regenerative practitioners also perceive this approach as bringing real benefit to improving the health of waterways, preventing loss of topsoil, and counteracting the impact of drought. There is also a national-level interest in converting to regenerative agriculture practices – as demonstrated by NZ Primary Sector Council's adopting a regenerative mindset within their vision and strategy (Ministry for Primary Industries 2021).

The main context of the regenerative agriculture is to restore degraded farmland back to a biologically functional state, while sustaining a land-based income to those dependent upon the provision of resources from that land for their livelihood (Krebs and Bach 2018). The key outcomes from adopting this philosophy are to improve soil health, enhance ecosystem services, improve water quality, increase carbon sequestration and improve livestock and farmer wellbeing (Svec *et al.* 2012). This is supported by Fenster *et al.* (2021), who observed an improvement in biodiversity, soil health, water infiltration rates, and economic metrics associated with adopting regenerative farming practices.

Geissdoerfer *et al.* (2016) identified five core principles to regenerative agriculture design:

- *A human-centred approach.*

• A strong integration of experimenting with artefacts.

• Collaboration in multidisciplinary teams.

• An integrative and holistic view on complex problems.

• A characteristic six-step process of 'understand', 'observe', 'define', 'ideate', 'prototype', and 'test'.

The adoption of regenerative design requires a sequential analysis of a site based upon good observation and research, and application of that data to farm-scale actions that restore the health and resiliency of the farm system (Butler *et al.* 2007; Dhakal and Kattel 2019). Understanding the ecology of each site (Giller *et al.* 2021), the impact of market factors (Faiza Akhtar *et al.* 2015), climate, availability of resources and goals of land users (Gilbert Lenssen *et al.* 2013) are all need to be incorporated into the matrix of analysis for developing a robust management strategy (Krebs and Bach 2018).

A main criticism of regenerative farming is the decreased economic production of these systems (Howarth *et al.* 2022a), with research by Howarth *et al.* (2022b) supporting this view and demonstrating a lower economic performance of a sample of regenerative farms in New Zealand. However, Fenster *et al.* (2021) counters this argument by raising the importance of developing a regenerative farming system that has a sufficient diversity of complementary enterprises to be financially resilient and profitable in the face of adversity. They noted that the most regenerative sites were associated with integrating a variety of livestock into cropping operations, which resulted in an improved ecological niche diversity that supported a greater number of species and benefitted farm economic resiliency through improved plant management techniques (Fenster *et al.* 2021).

Biggs *et al.* (2012) developed a set of principles to guide the development of regenerative agricultural systems – with the goal of restoring ecosystem services in managed landscapes. They considered how the dynamics between cultural and ecological factors can restore the health and resiliency of natural systems. The seven principles proposed by Biggs *et al.* (2012) are outlined below to give a context of how principle-driven actions are at the heart of regenerative agriculture.

- **Diversity and Redundancy:** This describes how the stability of an ecosystem increases due to a balanced diversity of different species that carry out a wide variety of roles. Stability is also increased by having several species carrying out similar roles within an ecosystem – so that the loss of individual species is not likely to result in a loss of function within that ecosystem.

- **Connectivity:** This describes how connection between species within an ecosystem or between neighbouring ecosystems facilitates an exchange of energy, organic matter information and other resources that support ecosystem functions and social processes. This supports the continued viability of populations, facilitates dispersal of species and further increases the stability of those ecosystems.

- **Slow variables and Feedbacks:** This describes how agricultural ecosystems consist of variables that change and interact on a range of timescales. Slow variables are those that create the environmental context in which living things interact – such as soil formation, tectonic plate movement and climate. Fast variables are the interactions between living things and between living things and their environment. Feedbacks are then interactions between these variables that self-reinforce to compound and build upon the effect they have (positive feedback) or dampens (negative feedback) subsequent changes of the same type. These feedback cycles can create self-reinforcing dynamics that are hard to break and increase ecosystem stability. In contrast, to the self-reinforcing feedbacks, other cycles build up pressure in that ecosystem – that when released can create a sudden change in that system.

- **Understanding social-ecological systems as complex adaptive systems:** This recognises how managers of an eco-agricultural landscapes need to adopt a mindset that is sensitive to the complex function of the variables of that system and the dynamics within it. This approach requires us to continually learn and experiment to adaptively manage uncertainty, disturbance, and surprise rather than attempt to eliminate it.

- **Learning and Experimentation**: This recognises that to manage the complexity of agricultural ecosystems requires constant learning to modify existing knowledge or acquire new knowledge, behaviours, skills, values, or preferences. The need for learning is based on the assumptions that knowledge is always incomplete, and that uncertainty, change, and surprise are inevitable in complex systems. Monitoring provides information about changes in a system, whereas experimentation involves the active manipulation of processes and structures to observe and compare outcomes.

- **Broaden Participation:** There is a growing awareness that a broader participation in the learning process is required than what has traditionally been delegated to specialist agencies and universities. This will result in a greater involvement and acceptance of changes in proposed land management practices. Some key features of this are to increase transparency of information sharing and to seek feedback from all parties to guide management.

The principles used above by Biggs *et al.* (2012) were reinforced by Grelet *et al.* (2021), who describes regenerative agriculture farms operating as complex living systems that require an adaptable and context-specific approach. Grelet *et al.* (2021) reinforces the journey posed by adopting regenerative agriculture practices, that recognises *“the importance of the context of a given farm or farmer (including strengths and limiting factors), goal-based planning, and the exploration of new tools/practices while not abandoning the safe and familiar.”*

2.3 Regenerative Grazing Strategies

Livestock grazing is one of the dominant form of agricultural land use, with 25% of the global land surface devoted to production (Asner *et al.* 2004). Globally, the impact of grazing has received increasing scrutiny as the true cost of production is better understood and linked to clearance of rainforests and desertification (Asner *et al.* 2004) and pollution of waterways (Joy 2015). The Food and Agriculture Organization of the United Nations (Conant 2010) estimates that 0.7 to 1.2 billion ha globally (20-35% the world's permanent pastures) are degraded. In extreme cases, especially in fragile areas with sporadic rainfall (Savory and Butterfield 1998), this can result in loss of grasslands and conversion to arid desserts, a process called desertification (Asner *et al.* 2004). Researchers speculate that this process is responsible for the formation of deserts in many arid regions that previously supported human populations (Boissoneault 2017).

In the past several decades, there has emerged an increasing interest in adopting grazing management to avoid overstocking, utilise fertilisers more efficiently and avoid a negative impact upon the environmental (Wang *et al.* 2021b), associated with adopting regenerative grazing practices. Briske *et al.* (2008) describes the goal of regenerative approaches to ensure the organic matter in pasture soils are restored through grazing management aiming to maximise the activity of pastures to sequester carbon, and that this can be achieved by managing grazing practices to:

- Increase pasture productivity.
- Improve pasture species composition.
- Give key pasture species more recovery time between grazing events.
- Reduce animal grazing selectivity to improve pasture diversity.
- Ensure more uniform animal distribution and grazing pressure.

Savory and Butterfield (1998) suggest a method of holistic grazing to manage pasture systems, which has become the focus of many regenerative grazing strategies (Massy 2017). Holistic grazing is based on planned rotational grazing that *'mimics nature'* with the aim of sequestering carbon and retaining soil water (Johnson 2021). This is achieved by supporting the development of a positive feedback loop between pasture plants and soil organisms (Webster 2018), where plant growth is optimised in a diverse pasture, allowing plants the maximum ability to provide exudates through their roots to soil organisms (bacteria and fungi) (Zhu *et al.* 2020), so that those soil organisms in turn can thrive and provide the plants with trace minerals to support their growth (Schutter and Dick 2002).

By supporting the positive feedback loop between pastures and soil organisms, Machmuller *et al.* (2015) claims that regenerative approaches to land management can result in these systems acting as a carbon sink and sequester up to one-third of the annual increase in the atmospheric CO_2. A recent study by the UK Food, Farming and Countryside Commission (Xavier Poux *et al.* 2021) has drawn similar conclusions and claimed that a national transition to *'agroecology'* method of farming will enable UK to reduce its agricultural emissions by 55-70%. Ghosh and Mahanta (2014) suggest that the ability of grazing management to stimulate pasture growth and promote more effective nutrient cycling to sequester carbon into the soil as a proven fact.

The holistic method of grazing management was proposed by Allan Savory to act as a decision-making framework for utilising local resources within the context of local conditions to manage livestock in a manner that supports the revitalisation of grazing systems (Savory and Butterfield 1998). Xavier Poux *et al.* (2021) describe the outcome of these and other methods of agroecology as acting as good stewards of natural resources, preserving soil health, preserving biodiversity, and creating landscapes that support human health. McSherry and Ritchie (2013) supports these claims and provides evidence of high levels of below-ground carbon accumulating when cropping systems convert to regenerative grazing.

The claims of 'holistic grazing' have been criticised widely due to the lack of credible examples demonstrating their effectiveness compared to more conventional grazing systems (Nordborg 2016), and due to the amount of land required to provide sufficient food from these systems at the expense of loss of natural habitat due to their inherent inefficiency (Monbiot 2022). Nordborg (2016) presented a critical review of the literature on holistic grazing and showed that there are few studies that demonstrate a positive effect from adopting holistic grazing that are *'approved'* by the Savory Institute. Of these, only six studies conducted measurements of farm systems, which only reported small effects. Nordborg (2016) concluded that these studies demonstrated that holistic grazing could be an example of good grazing management, but that there was no evidence that it performed better than alternative well-managed grazing methods.

However, the significance of these studies supporting holistic grazing was probably downplayed. Ferguson *et al.* (2013), for example, compared 18 holistically managed farms with 18 conventional farms and reported significant differences, with the holistically managed farms having higher soil respiration, deeper topsoil, increased earthworm presence, more tightly closed herbaceous canopies, and marginally higher forage availability. Recent research from Wang *et al.* (2021b) also reports upon the crucial role of managing stocking rate and developing sustainable grazing strategies to help improve long-term ecological and economic sustainability of grazing systems. In support of (Savory and Butterfield 1998), Wang *et al.* (2021b) identify the key importance of providing adequate recovery time from grazing and adopting longer post-herbivory recovery periods in each paddock in management of grazing systems.

2.4 Quantifying Soil Health

With the increase in soil degradation in the last decades, there is a pressing need for quantitative soil health assessment to assist the sustainable management of agricultural soils, as it supports important ecosystem functions (Dennis *et al.* 2012). Lehmann *et al.* (2020) describe soil health as *"the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals and humans, and connects agricultural and soil science to policy, stakeholder needs and sustainable supply-chain management."* Dennis *et* *al.* (2012) describes soil as a vital living system, with Kremer and Hezel (2013) building on this idea to describe the functions of soil to support plant growth attributed primarily to ecological characteristics associated with biological diversity and ecosystem stability.

Stevenson *et al.* (2022) sought to clarify the definition of soil health further by describing the following physical, chemical and biological indicators of soil health:

- Soil cycles nutrients effectively.
- Provides good aeration to promote root growth.
- Regulates the flow of water and rainfall in the water cycle.
- Reduces runoff and erosion.
- Resilient to drought, heavy rainfall events, and temperature extremes.
- Resilient to disease and pest problems.
- Contains a well-rounded microbial community.

Scientists have been historically uneasy to embrace the concept of *'soil health'* due to the challenges of defining it with universally quantifiable assessment measures (Lobry de Bruyn 1997; Lehmann *et al.* 2020). Lehmann *et al.* (2020), for example, describes challenges with adopting universal soil health measures due to *"soil heterogeneity, the site-specific nature of soil management and the varying ecosystem services that have sometimes conflicting or competing needs".* Lehmann *et al.* (2020) proposed that these challenges could be overcome by using a set of multiple indicators to quantify soil health and provide a more robust assessment. Davidson *et al.* (2021) suggested that these soil health assessments might be best carried out in alternative years, due to the cost, time and need for specialist resources to obtain that information.

Pearce and Venier (2006) describe monitoring a few indicator species as an intuitively appealing method of measuring the ecological sustainability of land management, due to the difficulty of sampling a broader range of species. McGeoch *et al.* (2002) suggests that species with a more generalist habitat preference ('*detector species*') would be more effective as indicator species of ecological change than more specialised species with more sensitive habitat requirements, because they act as indicators of general habitat changes rather than indicating loss of specific specialist niches. It is also more likely that these detector species are more abundant in modified farmland and therefore are more useful as bioindicators within this context (McGeogh 1998; McGeoch *et al.* 2002).

Pearce and Venier (2006) describes the principal job of biological indicators as capturing underlying changes in ecosystem function and integrity. Suitable indicator species should demonstrate a biology that is sensitive to the impacts of land management on the function of ecosystems (Niemi Gerald *et al.* 1997; Rousseau *et al.* 2013; Schwerdt *et al.* 2018). The diversity of key natural enemy groups like ladybugs or lacewings, for example, could be used as a biological indicator of soil health in crops and also have the benefit of control of pest invertebrates (Pearce and Venier 2006). Biological indicators have further benefit in that they represent changes in biodiversity that relate to the ability of agroecosystems to maintain functions, provide services, and resist disturbances (Buzhdygan *et al.* 2020).

Bruyn (1997) notes that more caution is required before adopting bioindicators of soil health and suggests that much more basic research is required. In particular, he raises the concern of farmers themselves being able to use a bioindicator that can easily and reliably be used to monitor their soil sustainability and the difficulty of current bioindicators requiring too much technical expertise, associated expenses, time to conduct these assessments and difficulty with interpreting results.

Pearce and Venier (2006) noted a limitation in adopting biological indicators of soil health was the ability to be easily measured in the field and demonstrate changes in soil biological communities. Lehmann *et al.* (2020) extended on these concerns, describing the limitation of biological indicators to address the complex demands upon soil to support vital living systems, agricultural production, and animal and human welfare.

Lehmann *et al.* (2020) note that a soil health indicator should satisfy the following four main criteria:

- Be relevant to soil health.
- Be relevant to other ecosystem functions and services (plant production, water quality, human health, and climate control).
- Be sensitive - changing detectably and quickly without being reflective of merely short- term oscillations.
- Be practical - measured cheaply and with a short turnaround time, and informative for management.

Similar criteria for biological indicators were earlier identified by Pearce and Venier (2006):

- Indicators must be feasible and cost-effective to sample.
- Be easily and reliably identified.
- Be functionally significant.
- Respond to disturbance in a consistent manner.

It has been highlighted in several recent international workshops on sustainable agriculture that soil macrofauna have high potential as indicators of soil health due to their importance in regulating soil processes which are vital to the continued formation of soil and protection against soil degradation (Anderson, 1995). For example, predatory ground beetles and spiders have been widely recommended as biological indicators of soil health (Pearce and Venier 2006; Scott *et al.* 2006; Schwerdt *et al.* 2018).

2.5 Spiders as Biological Indicators of Soil Health

Research suggests that spiders are strongly associated with habitat structure, contain a range of useful generalist detector species and are sensitive to the impact of land use (Topping and Lovei 1997; Pearce and Venier 2006; Curtis *et al.* 2022).

Coombe (2001) performed a review of literature to determine the suitability of spiders as suitable ecological indicators. They found that spiders met most criteria, including:

- a high diversity and abundance
- a widespread distribution
- easy sampling and sorting
- relatively low random fluctuation in population sizes and community composition
- a range of dispersal abilities
- measurable response to habitat change
- good representation of diversity in other taxa.

This is supported by Bromham *et al.* (1999) who note an important advantage of spiders as indicators of ecological impacts of land use is the scale at which spiders interact with their habitat, their importance within native ecosystems, their sensitivity to land use and that they can be easily sampled in large numbers.

Further support is provided by Lakshmi and Joseph (2017) who describe the direct involvement of spiders in soil functions due to their contribution to the soil food web, which makes them important indicators sensitive to land management practices. In addition, the short life spans of spiders provide a meaningful indication of the impact of agriculture management practices on population numbers (Bromham *et al.* 1999). Spiders with moderate dispersal ability are more likely to indicate local impacts of land use (Bromham *et al.* 1999). Another advantage of using spiders as biological indicators is how common they are in agricultural environments (Curtis *et al.* 2022).

Spiders are often ignored or under-sampled in biodiversity and conservation assessments because of their large diversity, small size and lack of taxonomic guides (Curtis *et al.* 2022). Another limitation was raised by Curtis *et al.* (2022) who note that the species richness of spiders in agricultural environments is quite low.

2.5.1 Sampling Spiders

Pitfall traps are a common method for sampling spiders, along with emergence traps, sweep netting, suction sampling, leaf litter extraction, ground searching, and beating (Curtis *et al.* 2022). Curtis *et al.* (2022) demonstrate that the most effective method for sampling spiders in pastures was pitfall traps and Bromham *et al.* (1999) demonstrated that pitfall traps are reliable indicators of spider diversity and abundance in pastures. Pitfall trapping has been widely adopted as an efficient means of collecting spiders in field sites (Sherley 2016; Bergeron *et al.* 2018), including within pastures (Topping and Lovei 1997). In New Zealand, a study by Curtis *et al.* (2022) showed that pitfall traps were the best method for catching spider species. They compared several techniques, and most species were caught using pitfall traps (26 out of a total of 28 species).

Pitfall trapping has been widely adopted as an efficient means of collecting spiders in field sites (Sherley 2016; Bergeron *et al.* 2018), including within pastures (Topping and Lovei 1997). Curtis *et al.* (2022) demonstrated that out of four sampling methods used to quantify the biodiversity of spiders for biodiversity assessment in New Zealand pastures, pitfall traps and ground hand collection were the two most effective methods at representing spider diversity (86.6% and 95.4% of total species diversity respectively). Pitfall trapping is not a reliable measure of absolute abundance but is best used as an indicator of spatial or temporal changes in invertebrate populations (Bromham *et al.* 1999).

The likelihood of an invertebrate falling into a pitfall trap depends upon its body size, foraging behaviour and mobility (Bromham *et al.* 1999), as well as the surrounding vegetation density and type (Coombe 2001). Thomas *et al.* (2006) demonstrated in field data and simulation models that vegetation density in agricultural habitats also had an important role in pitfall trap data.

The timing of collecting invertebrates is another important factor – to obtain a representation of population size at a suitable time of year (Duchesne *et al.* 1997; Bromham *et al.* 1999; Pearce and Venier 2006) and to sample when the number of adults within the population make species identification most feasible. Studies performed in late autumn are more likely to have a larger representative sample of mature adults that are easier to identify (Pearce and Venier 2006).

2.6 Other Biological Indicators of Soil Health

2.6.1 Earthworms

Earthworms as alternative soil biological health indicators are explored here for their potential to be incorporated into the study as positive biological control to spider abundance and diversity and to be combined to create a more robust measure of soil biological health. Pearce and Venier (2006) note that the value of bioindicators is increased when using *"a team of bioindicators to provide a multi-scaled and holistic assessment of sustainability."* Pearce and Venier (2006) recommend that spiders could be combined with other biological indicators to demonstrate changes more accurately across a variety of spatial scales and habitat types.

Earthworms are well represented in pasture systems (Köhler *et al.* 2014) and have a demonstrated response to changes in agricultural practices (Pérès *et al.* 2011). Earthworms can be considered *'ecosystem engineers'* (Capowiez *et al.* 2014), with their burrowing activities resulting in the deeper penetration of water and air into the soil (Köhler *et al.* 2014). The physical activities of earthworms (soil burrowing and organic matter burial) also have a large impact upon the physical, chemical and biological soil processes (Capowiez *et* *al.* 2014). This activity results in increased vigour of plant growth (Capowiez *et al.* 2014) and improves the habitat to support other soil organisms (Pérès *et al.* 2011).

It is often claimed that through their activities (soil burrowing and organic matter burial) earthworms exert a huge influence on physical, chemical and biological soil processes but in most cases the clear quantitative demonstration of this influence is still to be assessed (Capowiez *et al.* 2014). Through burrowing, earthworms can contribute to the rapid transfer of water, air and solutes to deeper soil layers and these animals are now recognised as '*soil ecosystem engineers'* (Capowiez *et al.* 2014). Earthworms are considered good environmental indicators candidates since they are well represented in the soil system in terms of density, they respond to a variety of environmental and ecological factors (such as changes in agricultural practice), and they can be considered as an indicator of soil functioning due to their strong impact on soil (Pérès *et al.* 2011). Köhler *et al.* (2014) showed that there was a higher diversity and abundance of worms when arable land was managed by organic methods and/or incorporated trees.

Each functional group represents a distinct ecological category that differs in their burrowing and feeding behaviour (Schon *et al.* 2022). Epigeic earthworms live and feed on organic matter near the soil surface, endogeic earthworms burrow extensively throughout the topsoil and form semi-permanent burrows, and anecic earthworms are the largest and form deep, semi-permanent burrows that open to the soil surface (Schon *et al.* 2022). The presence of each group enhances the recycling of organic matter in the soil and the availability of the nutrients within that organic matter to plants (Schon *et al.* 2022). When all these groups are all present, the recycling of organic matter in pastoral agriculture soils is considered to function at a more optimal level (Schon *et al.* 2022). All the species within these groups are exotic, with the rich diversity of New Zealand's native earthworms (nearly 200 species) being confined to native forests and forest fragments (Schon *et al.* 2022). Schon *et al.* (2022) recommends a target earthworm abundance of >400 ind./m2 in New Zealand pastoral soils, within which it would be desirable to have all three ecological groups of earthworms present, to ensure soil functioning is maximised.

Earthworms have a demonstrated response to a variety of environmental and ecological factors and can be considered as indicators of soil functioning due to their strong impact on soil (Pérès *et al.* 2011). An example of their response to farming methods was demonstrated by Köhler *et al.* (2014) who showed that there was a higher diversity and abundance of earthworms when arable land was managed by organic methods and/or incorporated trees.

2.6.2 Pasture Diversity

Measures of pasture sward composition have been used as indicators of pasture condition (Ludwig *et al.* 2000) and the diversity of pastures (using a species richness index) can build upon this to act as a metric for landscape health (Xu *et al.* 2019). A more diverse pasture is assumed to improve pasture multifunctionality and stability (Xu *et al.* 2019).

When investigating pasture diversity, researchers use plant functional groups to simplify the complexity of species present (Nicholas *et al.* 1998). The purpose of using plant functional groups (instead of plant species) is to provide a deeper understanding of ecological processes and community dynamics (Duckworth *et al.* 2000). Plant functional groups are derived from traits based on species morphology, physiology and/or life history, depending on the aims and scale of the research (Duckworth *et al.* 2000). The use of plant functional groups simplifies ecological systems to support the analysis of changes in the plant communities in response to land use changes (Nicholas *et al.* 1998) or farm management practices (Romera *et al.* 2017).

Zavala-Hurtado *et al.* (1996) support the use of morphological characteristics to organise plants into functional groups, noting the importance of representing behavioural strategies to adapt to environmental pressures as the variables of most significance.

Leishman and Westoby (1992) used multivariate clustering methods to see what groupings of plant species would emerge. The analysis by Leishman and Westoby (1992) identified 43 traits that could be used to organise plants into functional groups; comprising eight vegetative, nine life-history, 15 phenology and 11 seed-biology characteristics. Zavala-Hurtado *et al.* (1996) support the use of morphological characteristics to organise plants into functional groups, noting that morphological characteristics represent strategies of plants to cope with environmental pressures.

Understanding the effects of grazing intensity on plant diversity and ecosystem functions can inform better grazing management practices. Grazing acts as a disturbance on pasture communities, that has been demonstrated to influence pasture diversity through preferential selection of more palatable species (Lavorel *et al.* 1998), or through over-grazing resulting in modification of abiotic conditions that favour more resilient species (Teague and Kreuter 2020). Xu *et al.* (2019) conducted a field experiment to examine the impact of grazing intensity on alpine grassland ecosystems, that demonstrated changes in plant community composition with increasing grazing intensity. The ecosystem multifunctionality (EMF) was highest under no grazing and lowest under heavy grazing (Xu *et al.* 2019). Grazing, therefore, has a direct impact upon pasture diversity in ways that has been demonstrated to modify the resiliency of pastures (Dahal *et al.* 2020).

2.6.3 Pasture Brix Reading

Brix readings are commonly used in horticulture to assess the health and vigour of plant growth (Kleinhenz and Bumgarner 2012) and within the fruit and vegetable industry for quality checks to meet import/export standards (Balsom and Lynch 2008). A brix reading is obtained using a refractometer that a glass prism to measure the refractive index of a liquid (Balsom and Lynch 2008). The refractive index of the liquid is then dependent upon how many solutes are dissolved in that liquid (Lemus and White 2014).

A brix meter refractometer is calibrated to give a percentage value of the dissolved sucrose to water ratio in a solution, relative to 20°C (Balsom and Lynch 2008). For example, a brix measurement of 25% means there is 25 grams of soluble content and 75 grams of water in 100 grams of solution (Balsom and Lynch 2008). A measurement will be affected by all soluble compounds in a solution, as they all have an effect on the refractive index (Balsom and Lynch 2008).

Brix readings of pastures have been popular tools in the regenerative farming community in New Zealand to measure pasture quality (Rowarth 2020). Brix readings are a measure of the solutes in plant sap and are claimed to indicate the health and rigour of pastures and consequently their value as forage for ruminants (Lemus and White 2014). Forage crops with a higher reflective index will have a higher sugar, protein and mineral content and a greater density (Lemus and White 2014). Research indicates that live-weight gains of livestock in New Zealand can be improved by using high sugar grasses (Edwards *et al.* 2007). Lemus and White (2014) indicate that brix readings of pastures can be interpreted using the following guideline: <3 % very poor, 4-7 % poor-moderate, 8-12% good and >13 % excellent.

Elevated brix readings in pastures are correlated with improved feed quality (Lemus and White 2014). Higher sugar and solute levels in plant sap lead to higher brix readings, may indicate greater nutrient production by the plants (Lemus and White 2014). This likely enhances the ability of pastures to release root exudates and establish stronger relationships with soil organisms, although further investigation is needed.

While brix readings serve as a tool for some farmers to monitor pasture quality and soil health, it is important to note that their validation is limited. Scientific studies primarily exist in the context of horticulture, monitoring the sugar levels of fruits and vegetables, which may not directly correspond to pasture health.

2.6.4 Soil Carbon

Carbon is the main element present in soil organic matter, on average making up 58% by weight (Livesley *et al.* 2021). Soil organic carbon is a vital component of productive agriculture (Thangavel and Sridevi 2017; Zhu *et al.* 2020), which influences many soil characteristics including nutrient and water holding capacity, nutrient cycling and stability, improved water infiltration and aeration (Ferrarini and Bini 2012). The carbon in soil is created by the decomposition of organic matter (Zhou *et al.* 2020). Studies have shown that soils with higher carbon result in increased agricultural productivity due its positive role in maintaining soil health, raising fertility, reducing erosion, and encouraging soil biota (Tunlid *et al.* 2022). Light soils with high sand content have less soil carbon compared to medium to heavy soils with more clay particles, which have larger surface areas (The University of Waikato 2015).

The transformation of organic residues into humus is the mechanisms by which carbon is stored in the soil (Han *et al.* 2021). These humic compounds play an important role by improving soil structure and porosity, resulting in better drainage and deeper root growth of plants (Oldfield *et al.* 2019). The creation of humus is mediated by microbial communities that cycle soil nutrients and break down organic matter (Kristine 2013) and requires nitrogen, phosphorus and sulphur and other elements in smaller quantities (Han *et al.* 2021). The activity of soil microbes, in turn, improves soil structure through forming soil aggregates (Zhou *et al.* 2020). High levels of organic carbon help to maintain agricultural production through its positive role in maintaining soil health, raising fertility, reducing erosion and acting as a positive feedback to encourage the development of the soil biota (Ferris and Tuomisto 2015). This process is referred to as carbon sequestration and plays a major role in storing atmospheric carbon in the ground (Tunlid *et al.* 2022).

Soil carbon levels are considered a useful indicator of the sustainable management of grazing lands (Ludwig et al. 2000) through providing a link between livestock activities and landscape function. Well managed perennial grasslands can act as significant carbon sinks because they can store large amounts of carbon in vegetation, root systems and stable aggregates for long periods of time (Lin *et al.* 2020). Many policies on sustainable agriculture management now include soil carbon as a key indicator of impacts of land use (Ostle *et al.* 2009). Some soils, like peat soils, can have particularly high carbon content (40-60% carbon (Waikato 2015)) and care must be taken therefore to compare equivalent soil types when drawing conclusions about the impact of grazing practices on soil carbon levels.

2.6.5 Soil Aggregates

Soil aggregates have a major influence upon the physical structure of soil (Mangalassery *et al.* 2013), protect against soil erosion (Hashimi *et al.* 2020) and increase agricultural productivity (Ćirić *et al.* 2012). Soil aggregates are typically formed by binding microbial polysaccharides with smaller soil particles such as silt and clay (Mangalassery *et al.* 2013), with larger macro-aggregates typically forming around plant roots and coarse organic fragments (Mangalassery *et al.* 2013). Many bacteria produce a layer of polysaccharides or glycoproteins that coat the surface of soil particles, which cements them to form stable aggregates (Sheng *et al.* 2020). These aggregates are like the soil building blocks and result in increased air circulation, nutrient retention, water infiltration and water holding capacity of the soil (Han *et al.* 2021).

The formation of aggregates is an important aspect of carbon sequestration in soil (Xiao *et al.* 2021). By binding carbon up into aggregates, this carbon is protected from degradation by oxidation and microbial activity (Hashimi *et al.* 2020). This protects the soil carbon from breakdown by decomposing organisms or catalytic enzymes (Sheng *et al.* 2020). These aggregates are however highly vulnerable to land degradation associated with adverse farming practices (Han *et al.* 2021).

Soil aggregates are sometimes used as an indicator of soil health due to their influence on improving soil structure, fertility, and productivity (Zhou *et al.* 2020). Incorporation of perennial forage rotations and improved grazing practices have been shown to increase the formation of macro-aggregates in soil (Lin *et al.* 2020). Perennial grasslands can be significant carbon sinks because they can store large amounts of carbon in vegetation, root systems and stable aggregates for long periods of time (Lin *et al.* 2020). Degradation of soil will result in an increase in the proportion of smaller sized aggregates (Mangalassery *et al.* 2013). Therefore, an increase in the mass of larger aggregate as measured by the dry mean weight diameter (dMWD) is an indicator of an improvement in soil health.

Many studies have been conducted on the effects of land use change on aggregate stability, but few focus upon dry aggregate size distribution (dASD) and the factors affecting it (Ćirić *et al.* 2012). Dry ASD is one of the major physical characteristics of soil, and it strongly affects soil fertility and its resistance to erosion and degradation. It is also considered to be an indicator of soil structure (Ćirić *et al.* 2012). One of the most common indices of dASD is dry mean weight diameter (dMWD). High values of dMWD usually indicate high water permeability and air capacity but lower erodibility of soil (Ćirić *et al.* 2012).Wacha *et al.* (2018) demonstrated that strongly aggregated soil will have a higher mean weight diameter and that these were both characteristics of an undisturbed soil.

Incorporation of perennial forage rotations and improved grazing practices have been shown to increase the formation of macro-aggregates in soil (Lin *et al.* 2020). Due to the role aggregates play in enhancing soil physical properties, storing nutrients and carbon, and enhancing soil habitat to support soil biology (Han *et al.* 2021), the measurement of aggregates is considered an important indicator of soil fertility, sustainability, and productivity of agricultural systems (Zhou *et al.* 2020).

2.7 Other Variables of Interest

There are a wide variety of factors within a farming system that can act as confounding variables in any study trying to determine a cause and effect relationship (Pourzand and Bell 2021). Three key landscape variables considered here are landscape diversity, slope and soil nutrients and their potential impact upon soil health.

2.7.1 Soil Nutrients

New Zealand backcountry pastures are characterised by low fertility (Lopez and Kemp 2016). Soil nutrient management plays a crucial role in supporting pasture production (Lyttle 2018). Implementing effective nutrient management practices helps optimize nutrient use efficiency and reduce environmental impacts, while also improving pasture health and forage quality (Lyttle 2018).

Nutrient management should be approached holistically, considering the interplay between soil, pastures, and environmental factors (Fertiliser Association of New Zealand 2018). Integrated nutrient management strategies that combine soil testing, precision fertilisation, and organic matter management can help optimise nutrient availability, minimise nutrient losses, and promote long-term soil fertility. When fertilisers are correctly applied they provide essential nutrients such as nitrogen, phosphorus, potassium, sulfur and micronutrients that balance nutrient levels in the soil and support a more diverse range of pasture species (Williams and Haynes 1990). Fertilisers also contribute to the accumulation of soil organic matter, improving soil structure, water-holding capacity, and nutrient availability, which further supports the growth of diverse plants (Morgan 2013). Fertilisers should be applied at the right time, in the right amounts, and in the most appropriate forms to support optimal plant growth while minimizing the potential for nutrient runoff or leaching into waterways (Drewry *et al.* 2022).

2.7.2 Landscape Diversity

Landscape diversity may offer an alternative explanation to gazing as a major landscape variable that could impact soil health and biological diversity. The agricultural landscape acts as a mosaic of habitats, with different agriculture systems at different stages of maturity and interspersed with a variety of remnant refuge habitats (Martin *et al.* 2019). The configuration (size, shape, and spatial arrangement) and composition of these habitats is a key factor impacting biodiversity on farmland (Martin *et al.* 2019). Landscape diversity directly influences genetic and species diversity through providing suitable habitat to support the niches of each species (Herzog *et al.* 2017).

Agroecosystems are frequently characterized as barriers to dispersal of invertebrates and many vertebrates in the landscape, and often provide poor habitat availability (Santos *et al.* 2021). When agricultural landscapes increase in size and intensity of management, remnant habitat patches become both reduced in size and more distant from each other, leading to an increase in local extinction rates, with no ability for populations to be recolonization if the dispersal capacities of the involved species are low (Herzog *et al.* 2017). Martel *et al.* (2019) demonstrated that habitat fragmentation was a major cause of loss of biodiversity in agroecosystems due to inability of species to disperse between habitat remnants. Habitat refuges are often embedded in a matrix of more intensively used agricultural land, that offers little habitat islands to sustain biodiversity (Hendrickx *et al.* 2007; Martel *et al.* 2019). The connectivity between remaining suitable habitat fragments (landscape connectivity is defined as the degree to which the landscape facilitates or impedes species movements) is therefore considered particularly critical for species survival (Martel *et al.* 2019).

The configuration and arrangement of landscapes (in addition to their composition) (proportion of land-use types), is increasingly suggested as a key factor in determining biodiversity and associated ecosystem services within agricultural landscapes (Martin *et al.* 2019). Martin *et al.* (2019) demonstrated that complex agricultural landscapes offer a wider variety of important edge habitats that increase dispersal opportunities for species between habitat patches. Overall, arthropods were most abundant in landscapes that combine high edge density with high proportions of semi-natural habitat (Martin *et al.* 2019). Hendrickx *et al.* (2007) measured land-use intensity, landscape structure and habitat diversity, and investigated the impact of these factors upon the diversity of wild bees, carabid beetles, hoverflies, true bugs and spiders. They discovered that increased land use intensity resulted in a decrease in species richness of all groups investigated (Hendrickx *et al.* 2007). Hendrickx *et al.* (2007) emphasize the importance of conserving diverse agricultural landscapes that incorporate habitat refuges to protect biodiversity and avoid local extinction processes in highly fragmented landscapes.

Belfrage *et al.* (2015) tested 18 habitat descriptors to evaluate the habitat status of a farm and to track changes occurring due to modified land use and management, and proposed a core set measure based upon the following:

i. Four descriptors to measure structural composition and configuration of farms (Habitat Richness, Habitat Diversity, Patch Size, and Linear Habitats).
ii. Three descriptors addressing specific habitat types (Crop Richness, Shrub Habitats, and Tree Habitats).
iii. One interpreted descriptor (Semi-Natural Habitats).

Farm habitat maps can provide ground truth information for regional and global biodiversity monitoring, where agro-biodiversity conservation usually operates via habitat restoration and conservation (Herzog *et al.* 2017)

Belfrage *et al.* (2015) found that small farms had significantly higher on-farm landscape heterogeneity than large farms and that there was a strong positive relationship between on-farm landscape heterogeneity and the number of breeding birds, butterflies, and herbaceous plant species. Belfrage *et al.* (2015) also found that the complexity of single habitats is important for biodiversity, with meandering ditches with overgrown grasses having a higher diversity of insects, birds, and reptiles than straight ditches with overgrown grasses.

2.7.3 Slope

Slope has been demonstrated in several studies to have a significant impact upon pasture production and soil health in New Zealand pastures (Zhang *et al.* 2005; Lopez and Kemp 2016; Mackay-Smith *et al.* 2021; Donovan 2022). New Zealand backcountry is characterised by rolling or steep terrain (Donovan 2022), which can lead to increased erosion (Donovan 2022), and loss of soil nutrients (Zhang *et al.* 2005).

Lopez and Kemp (2016) provided an overview of hill country pastures in New Zealand, with a specific focus on the role of slope. They highlighted that hill country pastures are characterized by steep slopes and naturally low soil fertility. The study emphasized that slope plays a crucial role in shaping the landscape and influencing the abundance and production of different pasture species. Variations in slope gradient were found to affect soil organic matter, nutrient pools, pH, physical properties, and biological activity, ultimately influencing herbage mass production. This was supported by Zhang *et al.* (2005) who investigated the abundance of functional groups in pastures and identified slope and soil Olsen P as the most important factors influencing the abundance of certain functional groups.

Dymond *et al.* (2006) focused on the impact of slope on landslide susceptibility and erosion in New Zealand's hill country. The study emphasised the importance of reforestation and improved vegetation management on steep and moderate slopes to mitigate future land slips. These studies highlight the significant impact of slope on a range of factors (plant diversity, soil fertility and soil quality) which will all have a significant impact upon soil health.

2.8 Summary

Research on identifying robust biological indicators of soil health that can effectively assess the relative impact of adopting regenerative farming methods on soil health is vital for advancing sustainable agriculture practices. While there has been significant progress in understanding the role of soil biology in evaluating soil health, there are still notable gaps in the current body of knowledge:

1. **Limited Focus on Novel Indicators**: Existing research often concentrates on well-established soil health indicators such as earthworms, microbial communities, and soil carbon content. There is a need to explore and validate new, innovative indicators like spider abundance that can provide a broader understanding of soil health.
2. **Sparse Attention to Regenerative Grazing**: Despite the growing interest in regenerative agriculture practices, there is a scarcity of studies that specifically investigate the impact of adopting regenerative grazing methods on soil health. Existing research predominantly revolves around soil health in conventional farming systems.
3. **Limited Integration of Landscape Variables**: Soil health research often overlooks the broader landscape context. There is a need for more comprehensive studies that examine the interplay between landscape diversity, slope, and soil nutrients in the context of regenerative farming.
4. **Inconsistent Methodologies**: The methodologies for assessing soil health indicators vary widely across studies, making it challenging to compare and synthesize findings. Standardized methods and protocols are required to ensure consistency and reliability in assessing soil health.

Addressing these research gaps will enhance our understanding of how regenerative farming methods impact soil health and provide practical insights for farmers seeking to transition to more sustainable and ecologically sound agricultural practices. It will contribute to the development of science-based recommendations for regenerative farming and foster more widespread adoption of these approaches to improve soil health and overall agricultural sustainability.

3 Study Objectives and hypotheses

This study aimed to examine the potential impact of regenerative grazing practices on biological measures of soil health, specifically focusing on spider abundance as a new indicator of soil biological health. It was hypothesized that the adoption of regenerative grazing would result in positive changes in these biological indicators.

The study also investigated the relationships between soil biological health indicators and other variables, including grazing score, landscape habitat diversity, slope, and soil nutrients. The hypotheses were structured as follows:

Hypothesis 1 - Grazing treatment comparison: Differences in soil health indicators were predicted between sites implementing regenerative grazing practices and those using conventional grazing. Biological indicators of soil health were used to assess these differences.

Hypothesis 2 - Relationship between soil health indicators, grazing score, and landscape variables: A quantitative relationship was hypothesized between biological measures of soil health and a grazing score, which represented the alignment of farms to holistic regenerative grazing strategies. It was expected that this relationship would be more important compared to the relationships of biological measures of soil health with slope, landscape habitat diversity, and soil nutrients.

Hypothesis 3 - Similar response of spiders to other indicators: Spider abundance was anticipated to exhibit a trend similar to other biological measures of soil health in response to grazing score, habitat diversity, slope, and soil nutrients.

To address these research questions and hypotheses, the study employed the space-for-time substitution approach. This approach involved comparing different sites that represent different stages of adopting regenerative grazing practices as a substitute for a more preferred approach of measuring a site over multiple years and observing changes over time. This approach was taken due to the limited timeframe available to complete the study. The advantage of this is that it allowed multiple soil health and landscape variables to be included in a short term study (Dominati *et al.* 2021). However, a more ideal approach would have been to measure these variables at equivalent sites and observe changes over multiple years (Grelet and Lang 2021). This would better account for any confounding variables that are common to farming systems and make comparative studies between farm sites difficult (Pourzand and Bell 2021).

4 Methods

4.1 Field Site Selection

Study sites were selected based on results from a site selection survey (see Appendix 1) sent to farmers in the Taranaki, Whanganui, Manawatu, and Hawkes Bay regions of New Zealand in December 2021. These present areas that could be easily accessed during two sampling visits in April and May 2022 from Whanganui. This survey aimed to identify a cluster of sites from a similar region, undertaking the same farming activity on similar sized farms, but differed by having either a regenerative or conventional approach to grazing management. The survey was distributed to potential farmers through a Facebook advertisement (see Appendix 2) which invited participants with farming interests within the target regions. The Massey University human ethics approval number for this study is NOR 22/07.

The aim of the survey was to identify a region where the study could be located on farms as similar as possible. By locating farm sites within the same region, the impact of differences from soil types and climate was sought to be minimised. From 85 respondents throughout the Taranaki, Whanganui, Manawatu, and Hawkes Bay regions, 6 sites from Hawkes Bay were determined to be most similar. These sites were all large (270-800 hectares), predominately raising cattle for beef production and were in hilled backcountry areas. Three of these sites self-identified as following regenerative grazing practices and three as following conventional grazing practices.

Figure 3 below shows the location of all sites in the Hawkes Bay region. Conventional grazing sites (sites 1-3) are highlighted in a dark green and regenerative grazing sites in a light green (sites 4-6).

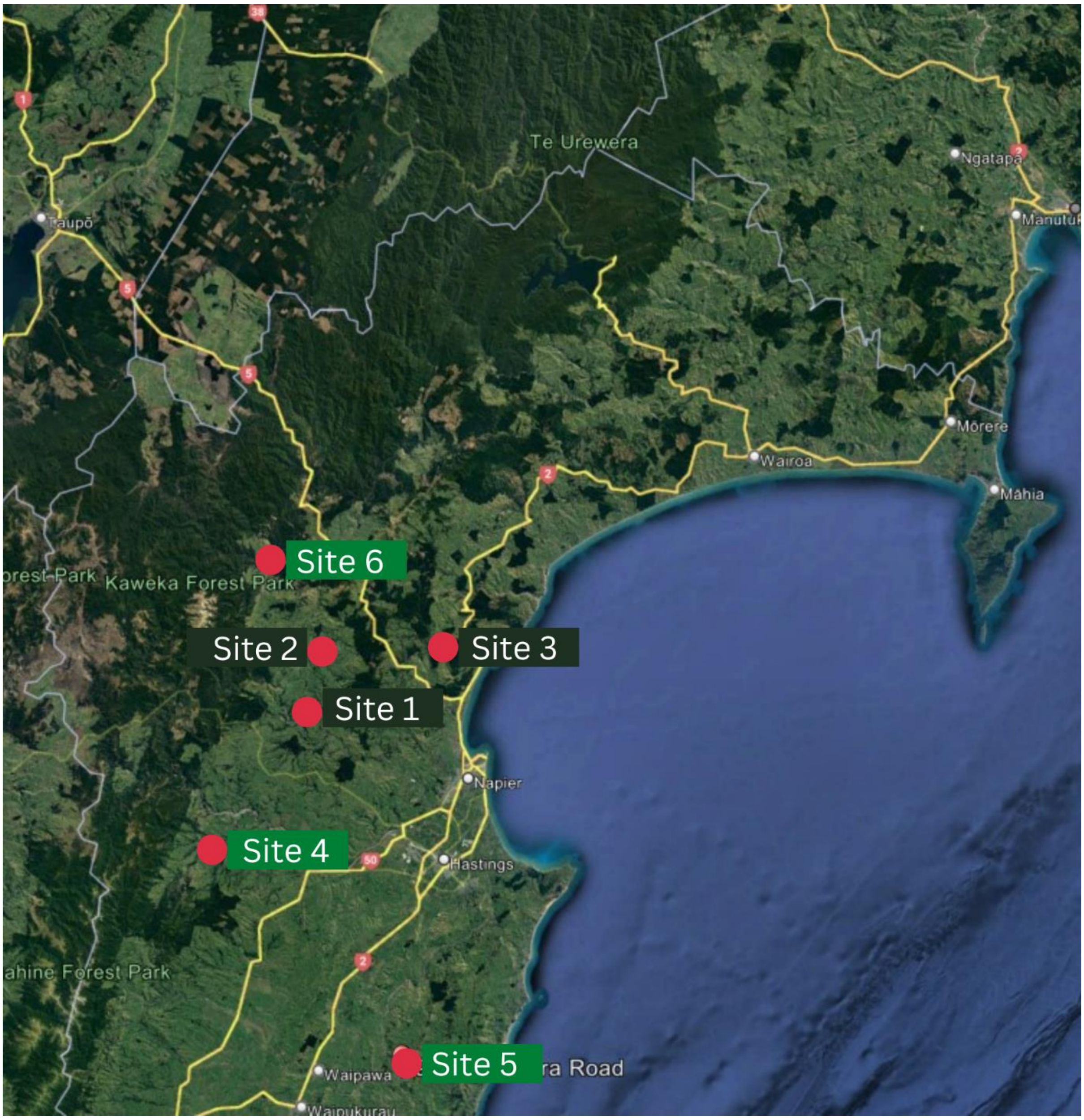


Figure 3. Field sites for study from the Hawkes Bay region. Image retrieved from Google Earth (28/11/2022) and modified in Canva. Conventional grazing sites (sites 1-3) are highlighted in a dark green and regenerative grazing sites in a light green (sites 4-6).

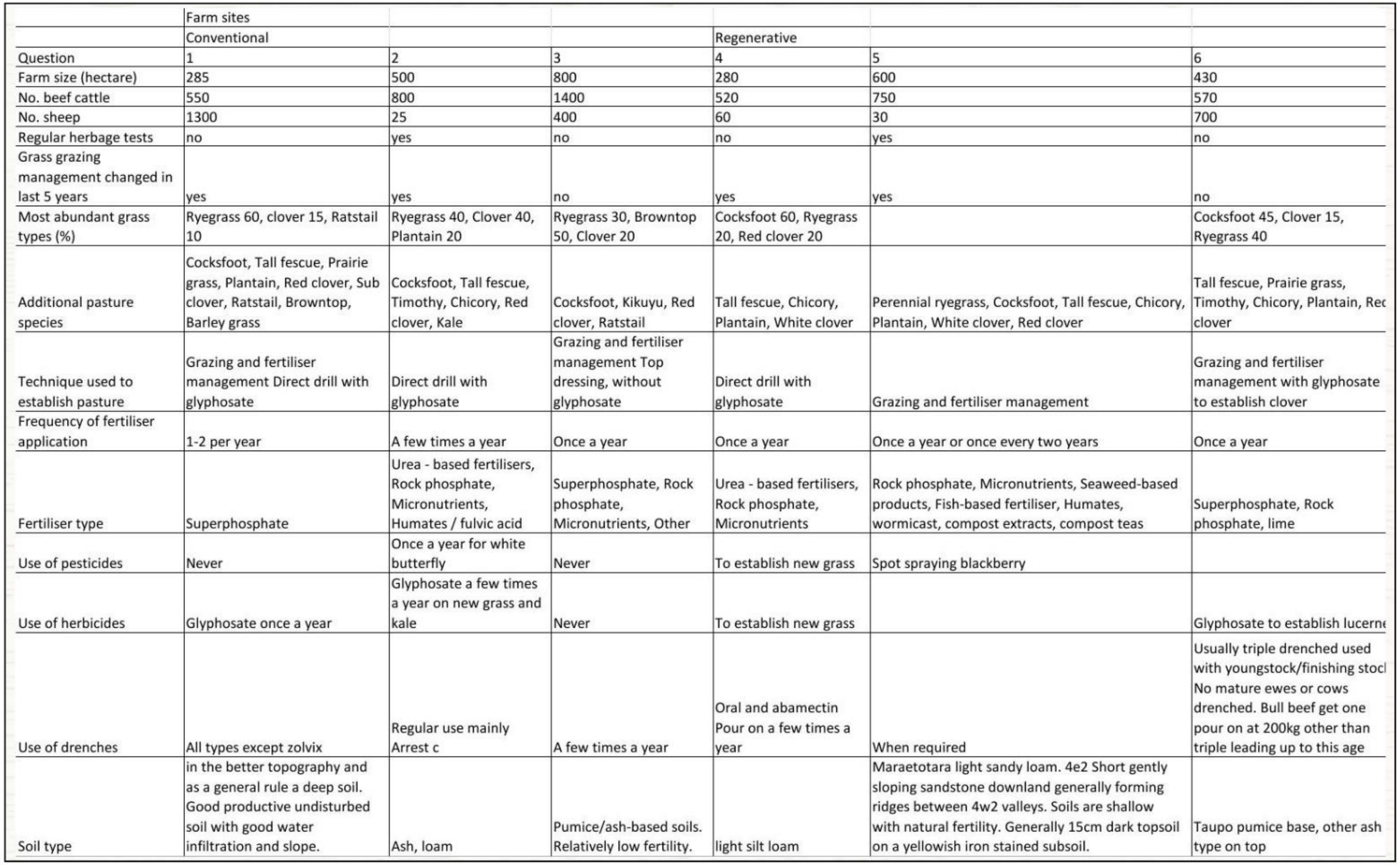
The study specifically focused on beef raising farms due to their relatively less intensive production compared to dairy farming, resulting in lower chemical inputs. In addition, beef farming was commonly practiced in backcountry farms that tended to have preserved patches of native forest (Meurk and Swaffield 2000). Both preserving native biodiversity and minimizing chemical usage are significant considerations for farms embracing regenerative agriculture.

The differences between conventional and regenerative practices may be more distinct in dairy farming, where the farming intensity and use of chemical inputs are typically higher (Baskaran *et al.* 2009). However, in the context of beef farming, the argument can be focused on the comparison between conventional and regenerative grazing practices.

Conventional beef farming commonly employs extensive grazing, allowing animals to graze for longer durations in larger areas. In contrast, regenerative grazing involves a higher stocking rate and shorter grazing periods, enabling more intensive utilization of the pasture (Gosnell *et al.* 2020).

The six farms selected for the study from results of the site selection survey (see Appendix 1) are summarised in Table 1 below. This Table shows the size of the farm, type of farming activity, techniques used to establish the pasture and what agrichemicals are applied to maintain the pasture. These results provide a reference for interpreting results between sites, to see if size and type of farm and use of agrichemicals may have had an impact on the other results collected.

Table 1. Summary of farms used in this study, data from the site selection survey. This information provides an overview of management practices on the farm sites. All sites are beef and sheep grazing farms in Hawkes Bay, NZ. Sites 1-3 are the conventional grazing sites and sites 4-6 are the regenerative grazing site.



	Farm sites					
	Conventional			Regenerative		
Question	1	2	3	4	5	6
Farm size (hectare)	285	500	800	280	600	430
No. beef cattle	550	800	1400	520	750	570
No. sheep	1300	25	400	60	30	700
Regular herbage tests	no	yes	no	no	yes	no
Grass grazing management changed in last 5 years	yes	yes	no	yes	yes	no
Most abundant grass types (%)	Ryegrass 60, clover 15, Ratstail 10	Ryegrass 40, Clover 40, Plantain 20	Ryegrass 30, Browntop 50, Clover 20	Cocksfoot 60, Ryegrass 20, Red clover 20		Cocksfoot 45, Clover 15, Ryegrass 40
Additional pasture species	Cocksfoot, Tall fescue, Prairie grass, Plantain, Red clover, Sub clover, Ratstail, Browntop, Barley grass	Cocksfoot, Tall fescue, Timothy, Chicory, Red clover, Kale	Cocksfoot, Kikuyu, Red clover, Ratstail	Tall fescue, Chicory, Plantain, White clover	Perennial ryegrass, Cocksfoot, Tall fescue, Chicory, Plantain, White clover, Red clover	Tall fescue, Prairie grass, Timothy, Chicory, Plantain, Red clover
Technique used to establish pasture	Grazing and fertiliser management Direct drill with glyphosate	Direct drill with glyphosate	Grazing and fertiliser management Top dressing, without glyphosate	Direct drill with glyphosate	Grazing and fertiliser management	Grazing and fertiliser management with glyphosate to establish clover
Frequency of fertiliser application	1-2 per year	A few times a year	Once a year	Once a year	Once a year or once every two years	Once a year
Fertiliser type	Superphosphate	Urea - based fertilisers, Rock phosphate, Micronutrients, Humates / fulvic acid	Superphosphate, Rock phosphate, Micronutrients, Other	Urea - based fertilisers, Rock phosphate, Micronutrients	Rock phosphate, Micronutrients, Seaweed-based products, Fish-based fertiliser, Humates, wormicast, compost extracts, compost teas	Superphosphate, Rock phosphate, lime
Use of pesticides	Never	Once a year for white butterfly	Never	To establish new grass	Spot spraying blackberry	
Use of herbicides	Glyphosate once a year	Glyphosate a few times a year on new grass and kale	Never	To establish new grass		Glyphosate to establish lucerne
Use of drenches	All types except zolvix	Regular use mainly Arrest c	A few times a year	Oral and abamectin Pour on a few times a year	When required	Usually triple drenched used with youngstock/finishing stock No mature ewes or cows drenched. Bull beef get one pour on at 200kg other than triple leading up to this age
Soil type	in the better topography and as a general rule a deep soil. Good productive undisturbed soil with good water infiltration and slope.	Ash, loam	Pumice/ash-based soils. Relatively low fertility.	light silt loam	Maraetotara light sandy loam. 4e2 Short gently sloping sandstone downland generally forming ridges between 4w2 valleys. Soils are shallow with natural fertility. Generally 15cm dark topsoil on a yellowish iron stained subsoil.	Taupo pumice base, other ash type on top

Due to emphasis on how grazing management impacts soil biological health indicators, the type of soil was not analysed beyond what was self-reported by the farmers. However, this would be recommended in a larger study.

In Hawke's Bay, the fertility of the backcountry soils can vary depending on soil type, management practices, and environmental conditions (Noble 1985). The region's soils are generally moderately fertile, with good drainage properties suitable for agricultural activities (Pohlen 1937). Organic matter addition and soil testing are important for improving and managing soil fertility effectively (Drewry *et al.* 2022). Over 100 distinct soil types have been identified in Hawkes Bay, and classified into groups such as Rendzina, Podsol, and Brown Loam, based on their development stage and parent rock characteristics (Pohlen 1937). Factors that impact upon soil fertility include erosion, nutrient depletion and acidification, loss of organic matter, compaction, and contamination with toxic substances, all of which can be exacerbated by deforestation, unwise land use practices, and the absence of sustainable land management techniques (Environment 2021).

Table 2 (below) shows the value scoring from the site selection survey (see Appendix 1), used to obtain a more refined view of the adoption of different farming methodologies and philosophises. These results are useful to see if farm sites adopting conventional grazing methods (sites 1-3), differ from those adopting regenerative grazing methods (sites 4-6) in terms of the value they place on commonly perceived indicators of high-quality pastures. Sites scoring above 50 could be considered to have a high emphasis on managing their site to optimise production of high-quality pastures.

Table 2. Summary of results from the site selection survey (Appendix 1) question on perceived value the farmer places (1 low – 10 high) on different indicators of high-quality pastures. All sites are beef and sheep grazing farms in Hawkes Bay. Sites 1-3 are the conventional grazing sites and sites 4-6 are the regenerative grazing sites.

Farm site	soil health	shade	pasture diversity	pasture recovery	pasture residual	rye grass leaf stage	pasture quality	Total Score
1	6	8	9	9	8	6	8	54
2	10	5	8	8	8	5	10	54
3	8	0	7	8	8	4	9	44
4	9	6	7	10	4	4	6	46
5	9	5	7	9	8	8	8	54
6	6	3	7	9	8	1	6	40

The selected sites (farms) were then sent a second survey, so-called grazing survey (see Appendix 4) to create a grazing score based upon the farmers' response to questions about specific grazing practices. This enabled an analysis of the correlation between the grazing score and measures of soil biological health, in addition to a comparison between regenerative and conventional sites. This attempts to capture a more nuanced approach to analysing the impact of grazing practices than the binary conventional vs. regenerative approach.

Table 3 (below) presents the results from the grazing score survey (see Appendix 4 for this survey). The purpose of this survey was to create a score based upon self-reported management practices by the site farmer. Each grazing management variable is associated with a score out of 5, with higher scores being aligned to management practices associated with adoption of regenerative grazing (holistic grazing) techniques.

Table 3. Summary of results from grazing score survey, used to assess how closely each farm site follows management practices associated with adoption of holistic grazing principles. All sites are beef and sheep grazing farms in Hawkes Bay. Sites 1-3 are the conventional grazing sites and sites 4-6 are the regenerative grazing sites.

Farm site	Stock density	Grazing duration	Recovery time	Residual length	Total Grazing Score
1	1	2	3	2	8
2	3	3	3	2	11
3	1	1	1	2	5
4	5	4	3	3	15
5	5	5	4	3	17
6	5	2	4	3	14

Sampling was conducted in late April and early May 2022, when there is most likely to be a higher number of adult spiders in the population and soil moisture from autumn rains would likely enable a good comparison to earthworm populations. Studies performed at this time are also more likely to encounter more stable weather in New Zealand.

4.2 Grazing Score

The scoring system used in this study has been adapted and extended based upon a grazing scoring system for recognising adoption of regenerative agriculture that was described by Fenster *et al.* (2021). Fenster *et al.* (2021) used an 8-point scoring system based upon survey where farmers received 0-2 points depending on their response to questions on using drenches, stock density, rotation frequency and rest periods of pastures. This scoring system was based on regenerative practices and was strongly associated with key metrics defining regenerative systems, including soil health, biodiversity promotion, and profitability (Fenster *et al.* 2021). The benefit of this approach is that it creates a continuum to represent the level to which regenerative practices were adopted on grazing farms. This enables a more refined method of representing the extent to which regenerative grazing practices were adopted, instead of the binary regenerative or conventional categorisation.

In this study, the survey sent to farms selected to participate in the study (see Appendix 4: Grazing Score Survey) assessed grazing management practices: the density of cattle per hectare of land they are allowed to graze (stock density), average time cattle are left grazing in a paddock (grazing length), time pasture is rested between grazing (rotation length) and length of pasture residuals (length of grass) after grazing.

These practices were selected to represent regenerative management practices that seek to mimic the traits of natural herds of grazers that are tightly bunched by predators and move frequently across grasslands in search of fresh grazing (Savory and Butterfield 1998). This is replicated within a farm practicing regenerative agriculture with a high-density herd moved frequently (normally once a day) to fresh grazing by dividing up a paddock using mobile electric fencing and allowing a long recovery time before pastures are grazed again.

Each of the four grazing variables that were scored were an average of the last 2 years, to provide an average of variation across seasons and changes in grazing practices (if regenerative grazing practices were recently adopted).

The questions from the survey are shown below:

Question 1. What is your stock density per hectare in land they are allowed to graze when temporary fencing is used? (stock density).

- 1 point = 0-20 cattle per hectare
- 2 points = 20-30 cattle per hectare
- 3 points = 30-40 cattle per hectare
- 4 points = 40-50 cattle per hectare
- 5 points = 50+ cattle per hectare

Question 2. What is the average time cattle are left grazing in a paddock? (grazing length).

- 1 point = 5 days or more
- 2 points = 4 days
- 3 points = 3 days
- 4 points = 2 days
- 5 points = 1 day or less

Question 3. How long is pasture rested (on average) between grazing? (rotation length).

- 1 point = less than a week
- 2 points = between 1-3 weeks
- 3 points = between 3-6 weeks
- 4 points = between 6-9 weeks
- 5 points = between 9-12 weeks

Question 4. How long are pasture residuals (length of grass) after grazing?

- 1 point = 0-5 cm
- 2 points = 5-10 cm
- 3 points = 10-20 cm
- 4 points = 20-40 cm
- 5 points = 40-60 cm

4.3 Pasture Diversity

The organisation of plant functional groups in this study is guided by the work of Leishman and Westoby (1992) who identified plant functional groups based on sets of traits common to species (e.g., morphological, physiological, environmental responses). The number of pasture species present in backcountry pastures is relatively small and stable, featuring mostly perennial ryegrass, brown top and white clover (Lopez and Kemp 2016) and therefore a large number of functional groups would not be required.

In this study the following functional groups were established that aim to represent types of species that are either closely related, or perform similar ecological roles:

- Annual grasses
- Perennial grasses
- Clovers
- Broad-leaf pasture plants
- Small-leaf pasture herbs

The sampling of pastures followed procedures described by Mackay-Smith *et al.* (2022). At each farm site, five sampling plots were selected that were approximately in the middle of the 100m radius area used to determine habitat diversity. The paddock this sampling site was in was also representative of the farmers response to the grazing survey. Pasture samples were obtained at each of the five sample sites on the same day, using a 1m^2 metal quadrat (20cm x 50cm). The quadrat samples were taken every 15m along a transect line (see transect lines in landscape maps in Appendix 6) that run parallel to a fence line, (the same transect line along which other variables samples were collected). The pasture samples were cut to ground level from within the quadrat and stored in a labelled paper bag in the fridge at 4^{o}C for up to two weeks until they were processed.

Pasture species were sub-sampled by splitting each sample in half, mixing that, and then taking another half. Using this technique, ¼ of each 1m^2 quadrat sample was then hand sorted and identified to family level. Each sorted group was then individually wrapped in paper, labelled, and then placed back in the paper bag with the other ¾ of each sample.

The paper bags of each sample were then placed in a 60 ^{o}C oven for 1 week to dry samples. The dry weight of each sample was then recorded, in addition to the dry weight of each pasture type sample. A Simpson diversity index was calculated based upon the relative weight (kg) of each functional group per hectare at each sample site.

4.4 Pasture Brix

The technique in this study follows that described by Lemus and White (2014) and Balsom and Lynch (2008). At each farm site, five separate brix readings were obtained every 15m along a transect line, in a 3m radius around each pitfall trap (see next section and site maps in Appendix 4). A handful of pasture grasses and herbs representing the greatest range of species present was obtained at each sampling site. This sample was rolled in the palm of the hand to help break up plant tissue and soften the sample for 1 minute and then pressed through a standard garlic crush press to obtain a few drops of liquid from the sample. This liquid sample was placed on the sample plate of the refractometer and the refraction of light measured. At each farm site, five brix readings were obtained, and the average brix value compared between farm sites.

4.5 Abundance and Diversity of Spiders

Spiders were collected in this study using pitfall traps as commonly used to assess spiders' abundance (Pawson *et al.* 2008; Bergeron *et al.* 2018; Aldebron *et al.* 2020). On each farm site, five pitfall traps were set, spaced approximately 15m apart, along a transect line (see site maps in Appendix 4) that ran parallel to a fence line (to aid in locating the traps on the return trip to the farm site). The traps were placed approximately 5m inside these fence lines to avoid areas that were frequently trampled by stock, which may impact the study results. The traps were left in the field for two weeks as recommended by Bromham *et al.* (1999). Farm sites were visited, and traps set up at the end of April 2022 and collected two weeks later at the start of May 2022.

Pitfall traps used in this study were based upon design described by (Pawson *et al.* 2008). A circular, 680 ml polypropylene plastic container of 100 mm diameter was buried to ground level, with a square metal plate fitted on top to avoid any stock damaging the trap by standing on top of it. A 10cm wide strip of metal was cut from each end of each metal cover to create a peg that could be pushed into the ground and secure the cover in place. This follows the New Zealand Department of Conservation guidelines for invertebrate pitfall traps (Sherley and Stringer 2016). A 50% monoethylene glycol (antifreeze) solution was used as a preservative. Samples were subsequently transferred into 70% alcohol for storage prior to counting. Spiders were identified to family level using a Nikon SMZ-IB Dissecting microscope, but due to overall low numbers only the total abundance data were analysed.

4.6 Abundance and Diversity of Earthworms

The earthworm sampling procedure in this study followed that described by Schon *et al.* (2022). Earthworm sampling was carried out in May 2022 (at the same time as pitfall traps were collected), when autumn rains and cooler weather bring worms closer to the surface (Schon *et al.* 2022). Following the method in Schon *et al.* (2022) earthworm samples were turf samples with dimensions of 20 x 20 x 20 cm (20 cm3), collected using a straight-sided spade. The soil sample was placed on a tarpaulin next to the hole for hand sorting. Hand-sorting has been shown to be an efficient means of assessing earthworm abundance as well as species richness (Gutiérrez-López *et al.* 2016).

The soil was broken up manually and all worms collected were transferred to a container for identification on site. In this study the total number of worms was recorded and the diversity of functional groups, using the key provided by (Schon *et al.* 2022) to identify epigeic, endogeic and anecic worm functional groups in New Zealand pastures. Each sample was sorted for 20 minutes, with particular care taken to break up soil around the root hairs of the grasses. All soil and worms were then returned to their hole.

At each farm site five separate samples for earthworms were collected, 15m apart along a transect line (see site maps in Appendix 4) that ran parallel to a fence line. Earthworm samples were collected 2m away from where the pitfall traps were placed to collect spider samples.

4.7 Soil Carbon Content

At each farm site, five separate soil samples were obtained. Each sample was obtained using a 10 cm diameter metal corer to a depth of 10cm - representing a 385.85 cm^3 volume of soil following the techniques described by Kremer and Hezel (2013). Soils samples were collected every 15m along a transect line established at each farm site and then stored in plastic bags at 4°C for two weeks before being processed for analysis.

The soil carbon content was measured using loss-on-ignition (Zhu *et al.* 2020). Each soil sample was broken down to its natural structure and then dried for 48 hours at 65 °C. The dried soil was then hand sieved through a 2000 micrometre sieve to obtain a fine medium for analysing the carbon content of the soil. Any soil debris larger than this was discarded. A 10g subsample of the dried and sieved soil samples was then placed into dried porcelain jars within a muffle furnace set to 400°C for 16 hours. The weight loss of the soil after burning in the furnace was recorded and multiplied by 0.58 to obtain an estimate of the carbon content of each soil sample.

4.8 Soil Aggregates

In this study, dry mean weight diameter (dMWD) was determined using the calculation described by Ćirić *et al.* (2012), where then sum weights from each aggregate size class were multiplied by x_i (the mean diameter of each size fraction) and w_i (the weight percentage of each aggregate sample in mm)

$$\text{dMWD} = \sum_{i=1}^{n} x_i \, w_i$$

At each farm site, five separate soil samples were obtained every 15m along a transect line established at each farm site. Each sample was obtained using a 10 cm diameter metal corer to a depth of 10cm - representing a 385.85 cm^3 volume of soil following the techniques described by Kremer and Hezel (2013). These samples were in addition to those collected for carbon and nutrient analysis. Soil samples were collected and then stored in plastic bags at 4°C for two weeks before being processed for analysis. Each sample is broken down to its natural structure and then dried at 65 °C. The dried soil was then hand sieved through a nest of eight interlocking sieves to separate the dried soil samples into different aggregate sizes. This corresponded to separating the soil sample into aggregate particle sizes of 8000, 4000, 2000, 1000, 500, 250, 125, 63 micrometres and the residual particles less than 62 micrometres, following (Bedel *et al.* 2018). The sample retained on each sieve was weighed and the weight recorded. Results are reported for % aggregate size, cumulative % aggregate size, and mean weight diameter.

4.9 Soil Nutrients

At each farm site, three separate soil samples were obtained for analysis of soil nutrients. Each sample was obtained using multiple cores from a 10 cm diameter metal corer to a depth of 10cm, until a sample size of approximately 500g was obtained following the method described by (Mackay-Smith *et al.* 2022). Soils samples were collected every 15m along a transect line established at each farm site and then stored in plastic bags at 4°C for two weeks before being processed for analysis.

Soil samples were then sent to a testing laboratory (Hills Laboratories in Hamilton) where they were analysed for total potentially available nitrogen (Total N), pH, Olsen phosphorus (Olsen P) and concentration of sulphur, potassium, calcium, magnesium, sodium (using 1M Neutral ammonium acetate extraction followed by ICPO-OES).

4.10 Landscape Diversity

There are a wide variety of potential descriptors of landscape diversity and the measure selected should be adapted to the unique features of the site and be representative of landscapes found in the area (Herzog *et al.* 2017). For this study, habitat diversity was measured as the land area (m^2) belonging to each of the following categories: pasture, exotic scrub, exotic trees, native scrub, native trees and wetlands. These categories account for all habitat types encountered in the study areas and the surrounding countryside and typify the habitat of rural backcountry Hawkes Bay. This categorisation distinguishes between native and exotic tree and shrub species, which is important because of the habitat specificity of many New Zealand invertebrates to native forest remnants (Costall 2012), especially New Zealand native spiders (Coombe 2001). Other studies have also adopted habitat categories to analyse habitat diversity on farms (Dennis *et al.* 2012; Herzog *et al.* 2017).

The percentage of area not covered in grass was used in this study as a representation of landscape diversity (see Appendix 7), as these other habitat types were considered most important for providing habitat refuges to spiders that would impact upon their abundance and diversity (Hendrickx *et al.* 2007; Martin *et al.* 2019).

Vegetation type was assessed from high resolution 2D images compiled in the Drone Deploy App (see Appendix 7), from hundreds of individual photos taken from a Phantom 3 drone flying across the site along transect lines and taking overlapping photographs from a 100m elevation.

4.11 Slope

Slope was measured in this study through analysis of topographical changes at each farm site using the VectorWorks programme. Within this programme topographical information was imported from Google Earth and converted to a landscape map to show elevation changes (see Appendix 5: slope Maps). Using this approach, a 100m radius landscape map of each farm site was constructed in Vectorworks, around where the samples sites were located. Within each 100m radius landscape map, 6 transect lines were taken in VectorWorks and the elevation change and slope of each transect line was recorded (see raw data in Appendix 5.6).

5 Statistical Analysis

This section describes the statistical analysis for the four hypothesis explored in this study. Statistical analysis was performed using SAS software (Version 9.4).

5.1 Hypothesis 1 - Grazing treatment comparison:

A Nested ANOVA was performed with farm site (1 to 6) as the random factor and treatment (regenerative or conventional grazing management) as the fixed factor. The measures of soil health compared between treatments were spider abundance, earthworm abundance, pasture diversity, pasture brix, the mean weight diameter of soil aggregates and the soil carbon concentration.

5.2 Hypothesis 2 - Relationship between soil health indicators, grazing score, and landscape variables:

There is a quantitative relationship between biological measures of soil health and a grazing score that represents the individual alignment of farms to holistic regenerative grazing strategies, and it was expected that this relationship is more significant than the effect of landscape variables (i) slope, (ii) habitat diversity and (iii) soil nutrients.

To investigate this hypothesis, a multiple correlation analysis was conducted, considering the grazing score, slope, habitat diversity, and soil nutrients in relation to measures of soil health at the farm sites. The inclusion of slope and habitat diversity as alternative sources of variation aimed to determine if they have a stronger impact on soil health measures compared to the grazing management score. Soil nutrients were considered due to their known influence on certain aspects of soil health. By examining the grazing score, derived from the grazing survey, the study aimed to avoid oversimplifying farms into regenerative or conventional categories, instead focusing on understanding how changes in grazing management may affect soil health measures.

5.3 Hypothesis 3 – Spiders will show a similar response to other indicators:

The relationship between the variables recorded at each site was performed using a canonical ordination analysis. For this analysis all measures were normalized by converting the mean to 0 and SD to 1.

The SAS code for a canonical ordination analysis is limited to a maximum of 12 variables and each of these variables must also have the same number of samples (replicates). Since soil nutrients were included in the SAS canonical ordination analysis and only 3 replicate samples of soil nutrients were obtained, other experimental variables included in this analysis also needed to be reduced to 3 replicates. To do this, the first 3 replicates were used for the canonical ordination analysis (for variables with more than 3 replicates).

For this analysis the 3 replicates of the following variables were included:

- Slope
- pH
- Soil Potassium
- Soil Calcium
- Soil Total Nitrogen
- Soil Phosphorus
- Soil Carbon
- Total number of spiders
- Total number of earthworms
- Pasture diversity
- Pasture Brix

6 Results

6.1 Hypothesis 1 - Grazing treatment comparison:

There was no significant effect of grazing management on soil health indicators, but treatment*site interaction was significant for all variables except spider abundance (which was overall low) (Table 4).

The results in table 4 show there were no farm sites that showed consistently higher measures of soil health compared to other sites. Instead, all sites demonstrated variability, high measures of some soil health variables and low measures in other soil health variables.

Table 4. Results of Nested Mixed ANOVA for the effect of treatment (tmt, conventional or regenerative grazing management) on farm sites 1-6; sites 1-3 were conventional grazing (Conv, C) and sites 4-6 were regenerative grazing (Reg, R). Letters a,b,c within each row indicate significant difference between sites (LSD test, α = 0.05) where Site(Tmt) interaction was significant. Site(Tmt) groups are further highlighted by colouring values in the lowest range (letter a, coloured red), middle range (letter b, blue) and highest range (letter c, green).

	Treatment Means				Site Means							
	Conv	Reg	SE	p-value Tmt	C*1	C*2	C*3	R*4	R*5	R*6	SE	p-value Site(Tmt)
Soil Carbon (%)	14.01	13.2	2.29	0.45	17.57^{c}	13.35^{b}	11.10^{b}	17.35^{c}	8.17^{a}	14.38^{b}	1.31	<0.001
Soil aggregate size (dMWD).	3.33	3.5	0.45	0.26	3.19^{b}	3.31^{b}	3.49^{b}	3.38^{b}	4.64^{c}	2.47^{a}	0.18	<0.001
Spider abundance	2.4	2.2	0.57	0.76	2.2	2.4	2.6	3.0	0.6	3.0	0.79	0.21
Earthworm abundance	17.27	13.27	10.46	0.16	41.8^{c}	5.0^{a}	5.0^{a}	5.0^{a}	5.0^{b}	29.8^{b}	3.37	<0.001
Pasture diversity (Simpson's index)	0.52	0.52	0.12	0.96	0.52^{b}	0.32^{a}	0.71^{c}	0.35^{a}	0.45^{a}	0.77^{c}	0.5	<0.0001
Pasture Brix	7.51	7.09	0.75	0.19	7.14^{b}	7.86^{b}	7.52^{b}	7.52^{b}	8.64^{c}	5.12^{a}	0.38	<0.0001

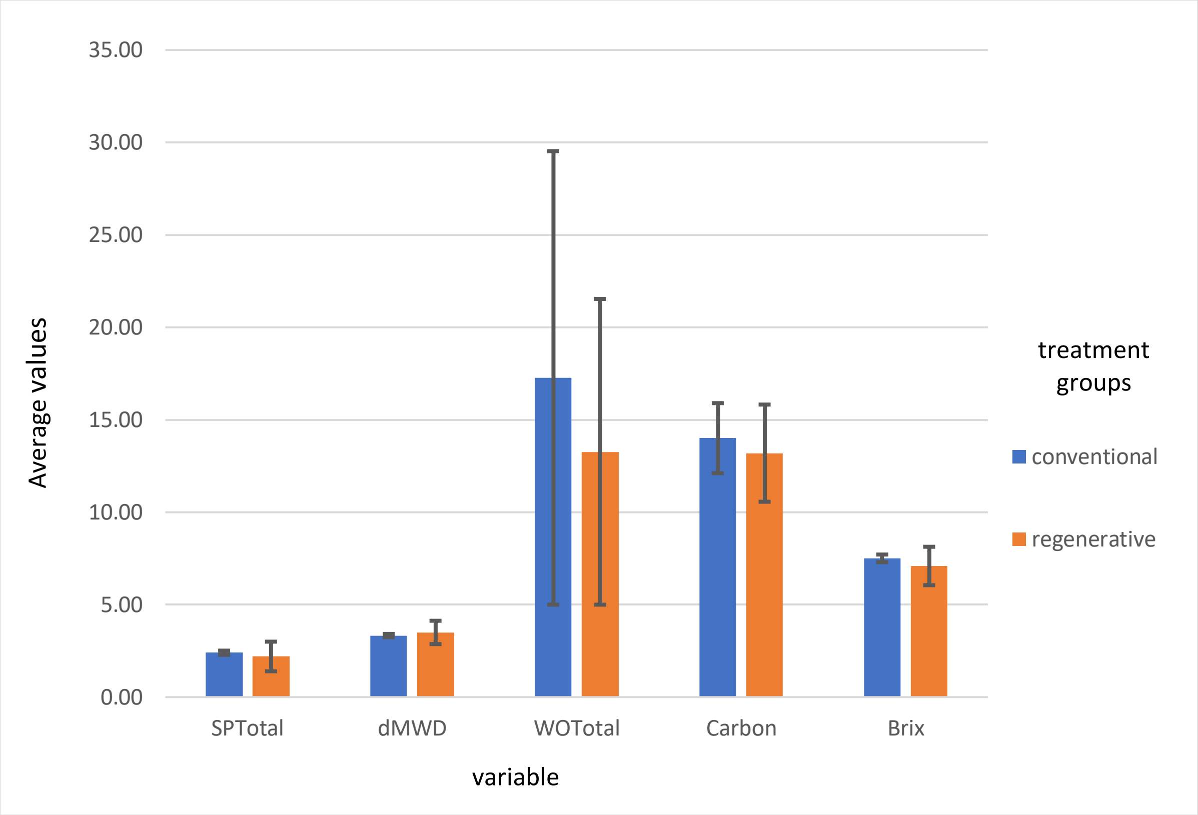


Figure 4. Comparison of average values of soil biological health variables measured for the two treatment groups (conventional grazing and regenerative grazing sites) with SE bars. There are three replicate sites per treatment group. Soil health variables are total number of spiders (SPTotal), total number of earthworms (WOTotal), the mean diameter of each size fraction of soil aggregates (dMWD), the percentage of soil carbon (Carbon) and the refractive index from a brix reading of pasture grasses (Brix).

Figure 4 above indicates the difference in average values of the biological indicators of soil health between the conventional and regenerative farm sites. This figure reinforces what was shown in in that statistical analysis shown in Table 4, that there was no significant difference in the average recorded values of soil biological health between farm sites practicing regenerative and conventional farming. The raw data used for the analysis is shown in Appendix 5.

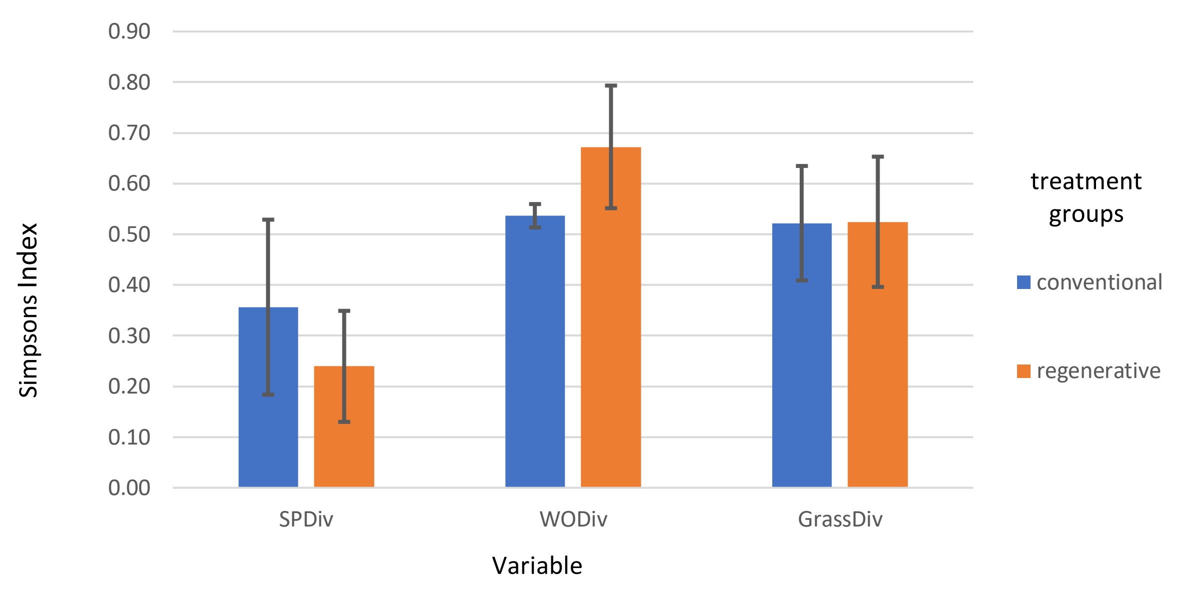


Figure 5. Average values of the Simpsons diversity index of functional groups for soil biological health measures with SE bars. Soil biological health variables measured for the two treatment groups (conventional grazing and regenerative grazing sites). There are three replicate sites per treatment group. Soil health variables shown are Simpsons diversity index of spider families (SPDiv), Simpsons diversity index of earthworm functional groups (WODiv), and Simpsons diversity index of pasture functional groups (GrassDiv).

Figure 5 compares average values of the biological measures of soil health between the regenerative and conventional farming sites, this time looking at differences in the calculated Simpson's Diversity Index of the diversity of spider families, earthworm functional groups and pasture plant functional groups.

Due to the low number of spider and worms recorded from the sites it was decided to base the statistical analysis (Table 4) on differences of the total number of earthworms and spiders. Figure 5 above also reinforces the lack of significant difference in the data between the treatment groups.

Unlike the diversity data for spiders and earthworms, pasture diversity was used and included in the statistical analysis in table 4. The raw data for this is shown in Appendix 5.4.

6.2 Hypothesis 2 - Relationship between soil health indicators, grazing score and landscape variables:

The multiple regression analysis demonstrated that there was no significant correlation ($P<0.05$) between the measures of soil health and the grazing score (Table 5, Table 6).

Soil carbon had a significant and positive correlation with habitat diversity ($P = 0.014$) (Table 5). Table 5 also shows a highly significant positive correlation between slope and four measures of soil health: earthworm abundance ($P = 0.0012$), pasture diversity ($P = 0.0013$), aggregate size ($P = 0.0001$) and pasture brix ($P = 0.005$).

Table 5. P-values (Pr>t) for the multiple regression analysis testing relationships between landscape variables (grazing score, habitat diversity and slope) and soil health indicators (abundance of spiders, abundance of earthworms, soil carbon, pasture diversity, soil aggregate size and pasture brix). Significant relationships are highlighted in bold, α=0.05, the sample size was five replicates per treatment.

Soil Health indicators	**Grazing score**	**Landscape Diversity**	**Slope**
Spider abundance	0.42	0.32	0.45
Earthworm abundance	0.97	0.103	**0.0012**
Soil carbon (%)	0.68	**0.014**	0.22
Pasture diversity (Simpson's index)	0.23	0.05	**0.0013**
Soil aggregate size (dMWD)	0.999	0.63	**0.0001**
Pasture Brix	0.63	0.83	**0.005**

Table 6. P-values (Pr>t) for the multiple regression analysis testing relationships soil nutrients and measures of soil health (total number spider, total number of worms, soil carbon, pasture diversity, aggregate size and pasture brix). Significant relationships are highlighted in bold, α=0.05, the sample size was three replicates per treatment.

Soil Health measures	**Total N**	**Olsen P**	**Sulphur**
Spider abundance	**0.055**	0.20	0.083
Earthworm abundance	0.969	0.22	**0.012**
Soil carbon (%)	0.6781	0.12	0.24
Pasture diversity (Simpson's index)	0.68	**0.04**	0.37
Soil aggregate size (dMWD)	0.42	**0.03**	**0.012**
Pasture Brix	**0.003**	**0.058**	0.25

The analysis in Table 6 demonstrates that soil Nitrogen (Total N) had a significant impact upon spider abundance (P=0.055) and very significant impact upon pasture brix (P=0.003), while soil Phosphorus (Olsen P) had a significant impact upon pasture diversity (Simpson's diversity index) (P=0.04), soil aggregate size (dMWD) (P=0.03) and pasture brix (P=0.058), and finally soil sulphur had a significant impact upon earthworm abundance (P=0.012) and soil aggregate size (dMWD) (P=0.012).

The raw data for this analysis is shown in Appendix 5.5 (soil nutrients), Appendix 5.2 (soil aggregate size) and Appendix 5.6 (slope, brix and soil carbon).

6.3 Hypothesis 3 – Spiders will show a similar response to other indicators:

The hypothesis was that spider abundance will show a similar trend in response to the (i) grazing score, (ii) habitat diversity, (iii) slope and (iv) soil nutrients, as other biological measures of soil health.

Figure 6 shows the results of canonical variation analysis with patterns in the relationships between experimental variables. The CAN 1 variable, explaining 83% of variability, demonstrates a strong positive correlation between pasture diversity, abundance of worms, soil sulphur and slope, and that this correlation was strongest at site 6. These variables were most important in explaining differences between sites. This concurs with the research question 2 results, which showed that a steeper slope at sites correlated positively with a higher number of earthworms, a higher pasture diversity and (in addition) a higher soil sulphur content. The CAN 2 variable (explaining 11% of variability) shows a positive correlation with total number of spiders, Total N, Olsen P and pH.

This analysis demonstrates that there was not a positive correlation between the abundance of spiders and other biological indicators of soil health. However, there was a strong positive correlation between the worm abundance and grass diversity, that was strongly associated to site 6 (as demonstrated by CAN 1 relationships). There was also a strong negative correlation between soil carbon and pasture brix that was not strongly linked to any site (as demonstrated by CAN 2 relationships).

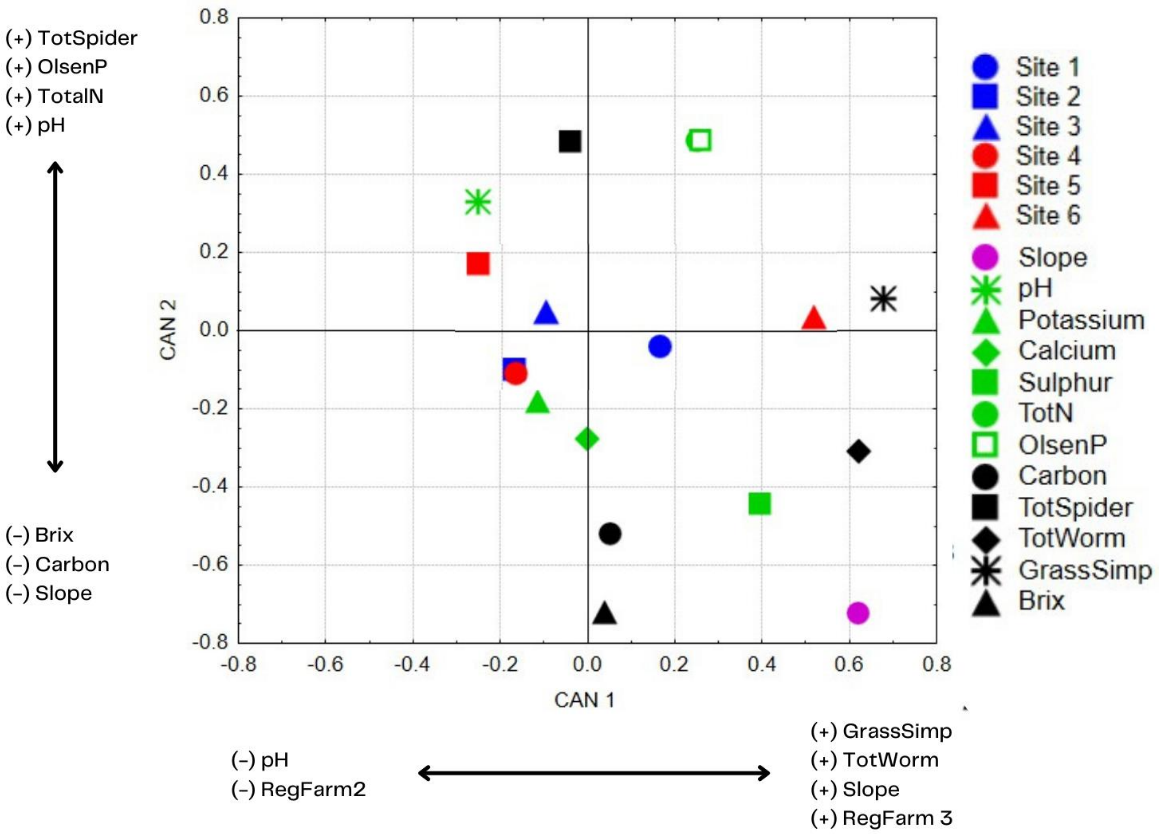


Figure 6. Results of canonical variation analysis demonstrating relationships between experimental variables (each with 3 replicates) and farm sites. Experimental variables were measures of soil biological health - earthworm abundance (TotWorm), spider abundance (TotSpiders), soil carbon % (C), pasture diversity (GrassSimp) and pasture brix (Brix) and measures of soil chemistry - pH, K, Ca, S, Total N (TotN), Olsen P (P), pH. Study sites are also mapped on this graph, with sites 1-3 grazed conventionally and sites 4-6 grazed following regenerative or holistic grazing principle. A landscape variable (slope) was also included. The variables that are clustered together on the extremes of the axis have the most significant relationship to each axis and the most significance in explaining experimental variability.

7 Discussion

7.1 Hypothesis 1 - Grazing treatment comparison:

This study's results demonstrated that farm sites that have adopted a holistic grazing strategy and self-identify as practicing regenerative farming did not show a significant difference in biological indicators of soil health. Consequently, the main hypothesis that regenerative grazing would result in a positive impact on soil health can be rejected.

7.2 Hypothesis 2 - Relationship between soil health indicators, grazing score and landscape variables:

This hypothesis explored the relationship between the grazing score developed from a survey of farmers and changes in the biological indicators of soil health. This was then compared to the relationship between habitat diversity, slope, and soil nutrients to the biological indicators of soil health to see the relative impact of the grazing score compared to other landscape variables.

7.2.1 Grazing Score

The analysis of the relationship between the grazing score and biological indicators of soil health (spider abundance, earthworm abundance, soil aggregate size, soil carbon, pasture diversity, and pasture brix), revealed no significant relationship. This finding suggests that the grazing score, which represents the degree of alignment with holistic regenerative grazing strategies, does not directly correlate with these specific measures of soil health.

7.2.2 Landscape Diversity

The relationship between landscape diversity and the biological measures of soil health was also explored as an alternative to the grazing score and for comparison. This was due to previous studies indicating that habitat diversity has a significant impact on spider abundance and diversity (Martin *et al.* 2019; Ball 2022). In agricultural landscapes, the configuration and composition of habitats are crucial for biodiversity conservation, providing suitable niches for species (Herzog *et al.* 2017). However, agroecosystems often limit habitat availability, hindering invertebrate and vertebrate dispersal (Vandewalle *et al.* 2010). As agricultural landscapes intensify, fragmented habitats result in increased local extinction rates and reduced connectivity (Herzog *et al.* 2017; Martel *et al.* 2019). Diverse landscapes with semi-natural habitats are vital for supporting biodiversity (Martin *et al.* 2019).

In this study, there was no significant relationship between landscape diversity and spider abundance, earthworm abundance, soil aggregate size, or pasture brix, but there was a significant relationship between habitat diversity and soil carbon (P=0.01) and pasture diversity (P=0.05). This demonstrates that habitat diversity has had a more significant impact on the biological measures of soil health than the grazing score, but that this impact has not been universally observed.

The lack of a significant relationship between landscape diversity and spider abundance may result from the scale with which landscape diversity was measured (across 100m radius area). Studies suggest that the impact of landscape diversity may be more meaningfully assessed on a larger land area. Belfrage *et al.* (2015) measured farm heterogeneity at farm level scale and surrounding landscape level and indicated that environmental factors on a farm scale proved to be subordinate to environmental factors at the landscape scale in explaining the total variation in local species composition. The impacts from surrounding land use were also explored by Dauber *et al.* (2005), who analysed landscape diversity in a 250m radius around pitfall traps set to catch invertebrates. They discovered that land use from the surrounding landscape accounted for 40% of the variance of carabid species richness.

7.2.3 Slope

Slope was explored as another landscape variable that may result in significant changes in the biological indicators of soil health. The study found highly significant relationships between slope and earthworm abundance (P=0.0012), pasture diversity (P=0.0013), soil aggregate size (P=0.0001) and pasture brix (P=0.005). Slope, therefore, had a much stronger impact upon the biological measures of soil health than either the grazing score or habitat diversity. These were all positive correlations; meaning that an increase in slope also resulted in an increase in the abundance of earthworms, pasture diversity, soil aggregate size and pasture brix. These results contrast with findings from (Cosgrove and Field 2016) who reported that an increase in slope resulted in a decrease in soil health.

There is no clear reason for the positive correlation between slope and biological indicators of soil health, but some possible explanations are:

- **Microclimate and Moisture:** Slopes can create variations in microclimate and moisture availability. In some cases, slopes may enhance water movement and drainage, reducing waterlogging and promoting better soil health (Bennie *et al.* 2008).

- **Organic Matter Accumulation:** On gentle slopes, organic matter can accumulate due to erosion of soil in steeper slopes uphill of the sample area (Chaplot *et al.* 2009).

- **No Soil Erosion:** Steep slopes are more susceptible to erosion, which can lead to the loss of topsoil and nutrients, but if the soil erosion rates are low and appropriate soil conservation practices are implemented soil health may not be significantly impacted by erosion (Becker *et al.* 2022).

The canonical variation analysis (Figure 6) demonstrated that the farm site that was causing this significant effect of slope was farm site 6, which was a site practicing regenerative farming and located adjacent to the Mohaka River in the foothills of the Kaweka Forest. This contrasted strongly with the other study sites which were all less steep, but also much further away from significant areas of natural habitat. The impact associated with slope from the farm site 6 may, therefore, instead be due to the surrounding areas of natural habitat. To obtain a clearer understanding of this relationship it would be advisable to record slope over a larger area and include a greater range of sites in the analysis.

7.2.4 Soil Nutrients

Soil nutrients were investigated as another variable that may impact biological soil health indicators. The relationship between soil nutrients and soil health variables was explored using a canonical analysis (Figure 6). None of the soil nutrients contributed to CAN1 (explaining 61.8% of all experimental variability) which showed a positive correlation between slope and two biological indicators (earthworm abundance and pasture diversity). CAN 2 (explaining 21.8% of all experimental variability) showed a positive link between Total N, Olsen P) and the abundance of spiders. Sulphur had a negative contribution to CAN2.

Due to the significant effect of Nitrogen, Phosphorus and Sulphur, the possible correlation between these soil nutrients and the biological indicators of soil health are explored below. The relationship between soil nutrients and biological indicators of soil health was however not a major focus for the study, so the interpretation of this result will not be explored in detail. However, the study has demonstrated that there are significant positive correlations between soil nutrients and biological indicators of soil health, and this would be worth exploring in greater detail within another study.

Nitrogen and phosphorus are both crucial plant nutrients which support healthy root development, nutrient uptake, and overall plant nutrient utilization (Razaq *et al.* 2017). The link between these nutrients and spider abundance is not clear but may be due to the improved vigour of plant growth providing a dense habitat for spiders. Also, while much has been said about the damage to the heath of natural environments when agrichemical fertilisers are used in excess, the level of soil nutrients is shown in this study to correlate positively with at least two biological indicators of soil health (Table 6).

Sulphur is a component of amino acids, proteins, and enzymes, and it contributes to the synthesis of chlorophyll (Narayan *et al.* 2022). Adequate levels of soil sulphur support plants by enhancing the ability to take up and utilize other essential nutrients like nitrogen, phosphorus, and potassium (Narayan *et al.* 2022). This improves overall nutrient availability in the soil and promotes healthy plant growth (Narayan *et al.* 2022). Sulphur is also important for microbial activity in the soil, by supporting the health of these soil organisms this, in turn, can improve soil health (Chaudhary *et al.* 2023). However, high sulphur levels have also been shown to be toxic to earthworms (Soil Quality Institute 2001).

7.3 Hypothesis 3 – Spiders will show a similar response to other indicators:

This study aimed to assess the suitability of ground-dwelling spiders as biological indicators of soil health in New Zealand pastoral farms, particularly regarding any changes in soil health associated with adopting regenerative farming practices. The ability of spiders to act as biological indicators of soil health was compared to other biological indicators of soil health (earthworm abundance, soil carbon levels, soil aggregates, pasture brix, and pasture diversity) to provide a comprehensive understanding of the impact of grazing practices on soil health. This hypothesis investigated if spiders show a similar response to other biological indicators of soil health in relation to farm sites with different grazing scores.

This was investigated in part with Hypothesis 2, and in part with a canonical ordination analysis (Figure 5). The analysis showed that spider abundance contrasted strongly with the other biological indicators of soil health. Spider abundance had no significant relationship with grazing score, type of management (conventional or regenerative), or any landscape or soil variables. Spider abundance contributed little to the % variability among sites (Table 4, Figure 6).

The results demonstrated that spiders do not show a similar response to land use impacts as other biological indicators of soil health and the hypothesis that they would was, therefore, rejected.

7.4 Spiders as bioindicators of soil health

There is limited research on the use of spiders as bioindicators of soil health in New Zealand (Curtis *et al.* 2022). Most of the research on spiders in New Zealand has focused on their taxonomic diversity and biogeography, with little attention paid to their use as bioindicators (Ball 2022). Therefore, the lack of information on spider ecology and their responses to changes in soil health may limit their effectiveness as bioindicators.

There are a wide range of potential factors that may impact upon the results obtained in this study. Understanding and anticipating the impact of management practices on complex systems, such as agroecosystems, can be challenging. Reductionistic scientific thinking frameworks are effective for understanding simplified agricultural systems but may fall short in comprehending the complexity and interconnectedness of highly complex agroecological systems, resulting in *"wicked problems"* for management. Measuring the functioning of complex systems like agroecosystems directly can be difficult and costly due to the size, complexity, and interactions involved.

Some possible explanation for the lack of a significant response from spiders to grazing practices are the population structure of spiders in New Zealand, the sampling methodology, the lack of impact from grazing differences or the study design – which are each explored further here.

In New Zealand the spiders most commonly encountered in pastures are mostly of European origin, with the money spiders (Linyphiidae) being the most abundant and commonly encountered (Topping and Lovei 1997). Topping and Lovei (1997) compared NZ spider biodiversity and abundance to similar pastures in England and found both sites were dominated by money spiders (Linyphiidae) with high dispersal ability, but New Zealand sites only had half the number of species due to limitations on how many species successfully colonised New Zealand from agricultural imports.

This study confirms what was reported by (Topping and Lovei 1997), with the largest percentage of spiders being caught in the Linyphiidae family (money spiders) at 56%, followed by the Lycosidae family (wolf spiders) at 42%. These studies in combination with the results of this study suggest that New Zealand pastures provide a poor habitat for native spiders and only a few introduced species can thrive within this environment. The diversity of spiders is therefore likely to be a poor measure of soil health in pastures due to the low variety of species that occur in this habitat. This is supported by Topping and Lovei (1997) who demonstrated that in grazed pastures in New Zealand the species richness steadily increased with decreasing intensity of grazing (from high intensity grazing to abandoned pasture). Native spider species, in particular, were seldom encountered in grazed farmland, but dominated spider communities in forest and native tussocks (Topping and Lovei 1997).

The impact of agricultural habitats on spider diversity may relate to habitat availability (Coombe 2001), with litter type and depth being important factors for providing hunting and foraging niches and for protection from predators and desiccation (Pearce and Venier 2006). Bromham *et al.* (1999) investigated how spider abundance and diversity was impacted by farm habitat type and found that spiders were caught in greater numbers in the grazed sites than within the woodland, but that the diversity of spiders was higher in the woodland. Bromham *et al.* (1999) study suggests that woodland provides a habitat better suited to a more diverse range of spiders, but more spiders may have been caught in pastures due to their increased mobility in these habitats.

Another factor that may influence the ability of spider’s act as soil health indicators is the likelihood of a spider falling into a trap. The likelihood of this happening would depend upon its mobility, activity levels, size and foraging behaviour, as well as the physical surroundings of the trap (Bromham *et al.* 1999). Bromham *et al.* (1999) demonstrated that pitfall traps are reliable indicators of spider diversity and abundance in pastures by comparing numbers collected from a grazed pasture site and comparing them to those collected within ungrazed woodland. Sherley (2016) describes pitfall traps as being ideal for obtaining an approximate index or relative abundance (amongst highly mobile ground-dwelling arthropods) and for measuring changes in abundance if sampled at the same time of year across multiple years. However, they caution that these changes are indices and do not reflect the actual relative abundance of species in the habitat (Sherley 2016).

Another challenge of using spiders as biological indicators is that the identification of spider species can be challenging due to their complex morphology, high diversity, and the lack of taxonomic expertise (Schwerdt *et al.* 2018). Misidentification of spider species can lead to incorrect conclusions about the soil health status (Curtis *et al.* 2022). In the case of this study, spider abundance was used, instead of diversity, due to the low numbers of spiders encountered and low variability. Therefore, in this case at least, spider taxonomical issues did not play a role in the results obtained.

7.5 Lack of impact from regenerative grazing

Holistic grazing is commonly used as part of the framework for adopting regenerative agriculture on pasture farms in New Zealand (Rowarth *et al.* 2020). This practice aims to mimic the natural movement patterns of grazing animals to improve soil health and ecosystem function (Savory and Butterfield 1998). It has gained attention worldwide, including in New Zealand, due to its potential to restore degraded pastures and enhance soil fertility (Sherren and Kent 2019). Holistic grazing has been claimed to promote increased organic matter in the soil through diverse plant growth, enhance nutrient cycling through even grazing pressure and distribution of animal waste, reduce soil compaction through rotational grazing, promote biodiversity in both plants and soil microbial communities, and improves water quality by preventing soil erosion and runoff (Gosnell *et al.* 2020). While there are several studies that examine the impacts of holistic grazing on soil health, specific references to New Zealand pastures are limited. Furthermore, the ecological processes and responses to grazing intensity may vary along bioclimatic gradients, and the understanding of ecosystem responses to managed grazing remains limited and fragmented (Nordborg 2016).

The farming community has expressed mixed opinions about regenerative agriculture, with some arguing that existing rotational grazing practices are already aligned with the goals of regenerative agriculture in pastoral systems (Beef&LambNZ 2021). However, other common grazing management practices that are not aligned with the principles of regenerative farming, such as short pasture recovery time, prolonged grazing, use of fertilizers and agrichemicals, pollution of waterways, soil erosion, and loss of native biodiversity, highlight the need for alternative approaches (Baskaran *et al.* 2009). This study would, however, support the view that changing grazing practices alone are insufficient to result in a significant difference in levels of soil health compared to conventionally grazed farms in New Zealand.

7.6 Study design

While the analysis did not reveal a significant relationship between the grazing score and the biological indicators of soil health, it is important to interpret these findings within the limitations of the study. Further research, including a broader range of variables and a larger sample size, is necessary to fully understand the potential effects of grazing practices on soil health and to explore other aspects that may also influence the relationship between grazing management and biological indicators of soil health. The absence of a significant relationship does not necessarily imply that there are no effects of grazing practices on soil health. Other indirect or long-term impacts that were not measured in this study, such as changes in soil structure, nutrient cycling, or microbial communities, could still occur. Additionally, the effects of grazing practices on soil health may vary depending on specific site characteristics, regional conditions, or the duration of the grazing management practices.

The grazing score also may not have captured all the relevant factors associated with grazing practices that influence spider abundance. Other aspects such as microclimate conditions, and other land management practices not accounted for in the grazing score could also play a role. This lack of significant difference could be attributed to multiple factors, including the possibility of no impact from grazing practices, insufficient replication numbers, or the sites not being fully comparable due to the presence of confounding factors such as varying soil types. However, all farms except one identified as having an ash loam type soil that is typical of the region. Extra replication across more farms, or more samples on each farm over time would have provided more clarity. In addition, the lack of clear results to prove or disprove a hypothesis is common in ecological studies, where the range of observed values for measured variables often surpasses the expected differences caused by treatments. Therefore, it is imperative to exercise caution when interpreting these findings and recognize the necessity for further research with increased replication to attain more robust and conclusive outcomes.

Despite growing interest in regenerative agriculture, research on its impact on soil health is limited. More comprehensive studies are needed to understand its effectiveness.

8 Conclusion

In conclusion, grazing practices alone do not strongly impact soil health in this study. There was no significant impact from adopting regenerative ‘holistic’ grazing strategy compared to conventional grazing on spider abundance or any other biological measures of soil health). There were no significant relationships between grazing score and biological indicators of soil health. But there was a significant positive correlation between habitat diversity, slope, Total N, Olsen P and some of the soil health indicators. Spiders also did not show a positive correlation to other biological indicators of soil health.

While spiders are important bioindicators, their effectiveness in New Zealand pastures is limited by challenges such as taxonomy, sampling techniques, species composition, and limited research. However, spiders may still play an important role when included with other biological indicators of soil health to monitor the impacts of changing land use practices over time on soil health. Future research should focus on developing taxonomic keys, improving sampling techniques, and studying spider communities' responses to soil health changes.

This research demonstrates that changing grazing management alone is insufficient for improving soil health and increasing biodiversity. In New Zealand, enhancing habitat diversity within farmed landscapes is crucial.

9 Recommendations

There were no strong impacts revealed from this study from which to make an authoritative recommendation for further studies. However, regarding the use of spiders as bioindicators of soil health one strong trend revealed from the literature is that habitat is an important factor impacting spider diversity and this is particularly important for New Zealand spiders that are largely confined to native forest remnants (Topping and Lovei 1997).

In New Zealand, 25% of the remaining indigenous vegetation cover is found on mixed livestock farms located predominantly in hill country. As these habitat fragments are put under increasing strain from being imbedded within an ever more intensively managed agricultural landscape, the populations of invertebrates have declined (Fountain and Wratten 2013). It is imperative to reverse this trend by designing landscapes that accommodate suitable habitat to sustain invertebrate populations.

Over the last 150 years New Zealand has undergone a striking and rapid transformation from ecologically diverse forests to pastures dominated by only a few exotic species. This has resulted in a dramatic decline in invertebrate biodiversity in farmlands and because these habitats dominate lowland areas of New Zealand emphasis should be placed on farming methods that incorporate habitat refuges to support invertebrate biodiversity, and landscape features that enable connectivity between habitat patches to aid dispersal of populations. The farming systems adopted in New Zealand are characterised by the dominant forms of agriculture from the colonial cultures that modified New Zealand to suit their preferred methods of land use. These forms of agriculture may not actually represent the true ecological strengths of New Zealand’s biogeography or when implemented act in ways that harmonise with surrounding natural systems. New Zealand could leverage the country's ecological strengths to produce commodities aligned with its unique context. This requires time, experimentation, and exploration of diverse agroecological systems that support native biodiversity and provide alternative income sources.

Therefore, spiders could have an important use as bioindicators within a farmed landscape in New Zealand, but this role is likely to be more significant if those farmed landscapes incorporated a greater variety of landscapes (especially native forest remnants), and farming systems that better integrate with those native systems to create a greater diversity within the mosaic of a farmed landscape.

One way this might be achieved is by incorporating silvopasture systems within grazed areas in New Zealand, especially in low productivity hill country that currently hosts the greatest amount of forest habitat remnants. Silvopasture is a type of agroforestry that integrates trees with livestock and pasture in the same area, and it is a farming system that has the potential to address some of the key issues of backcountry pastoral farming in New Zealand (Mackay-Smith *et al.* 2021). Silvopastoral systems are inherently complex and result in many ecological, economic, and cultural outcomes within the agricultural system (Jose and Dollinger 2019). This approach can be used to restore invertebrate diversity by providing habitat and food resources for invertebrates. The trees and shrubs in silvopasture systems provide a diverse range of habitats, including shelter, nesting sites, and food resources. This diversity can support a wide range of invertebrate species, including pollinators, decomposers, and predators of pest insects (Jose and Dollinger 2019).

A recommended study would be a multi-year analysis of any improvement in soil health associated with silvopasture plantings of trees on pasture farms compared to similar paddocks (on the same farm) without silvopasture plantings. This study could expand upon the methods of measuring soil health using biological indicators used in the current study but have more measurements (at least four per year) and be performed over multiple years (three are recommended). This would allow a more robust interpretation of how farm management practices and especially incorporating a greater habitat diversity may better achieve the goals of regenerative agriculture in improving soil health and increase biodiversity – while maintaining or improving farm productivity (through more diverse range of productive systems).

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11 Appendices

11.1 Appendix 1 Site Survey

Part 1: Information on Study

Impact of Grazing Management on Soil Health - site selection survey

This survey is part of a postgraduate research project looking at how grazing management impacts upon soil health.

The primary purpose of this survey is to identify suitable study sites in commercial beef / dairy farms.

The information obtained in this survey is confidential and will not be shared with anyone except the student completing the study (Massey University Masters of Philosophy Student: Richard Pedley) and his supervisory team (Drs Maria Minor and Lydia Cranston from Massey University, and Dr Gwen Grelet from Manaaki Whenua - Landcare Research).

Should you be potentially interested in hosting a study site on your farm, we would be grateful if you could complete this survey by the end of February. Richard will follow up with you once the information from the survey has been collected and processed.

This research will involve 2 farm visits from Richard:

- First visit in April 2022: to map study sites, collect detailed information about farm history and management from farm owners / managers, and set pitfall traps for collecting invertebrate samples.

- Second visit in May 2022: to collect pitfall trap samples, perform a pasture diversity survey, record number of earthworms in study sites and collect soil sample to test for formation of soil aggregates.

Thank you for your time completing this survey 😊

For further queries about this research, please contact Richard Pedley
at r.pedley1@massey.ac.nz

Part 2: Contact details

Contact details Please provide your contact details below so that Richard can contact you via email if you have been selected to participate in the study.

Name:

Email:

Part 3: General questions about your farm

How big is your farm (hectares)

Where is your farm located?

- ○ Hawkes Bay
- ○ Manawatu
- ○ Whanganui
- ○ Other

What agricultural activity occur on your farm?
(multiple responses allowed)

- ☐ Effective Dairy platform
- ☐ Dairy grazing
- ☐ Dairy support
- ☐ Beef finishing
- ☐ Beef breeding
- ☐ Sheep breeding
- ☐ Sheep finishing
- ☐ Other

If you are engaged in more than one type of farming activity (indicated above) does this occur in the same paddocks?

- ○ Yes
- ○ No
- ○ Not applicable

Please indicate current animal stock sizes:

Beef cattle

Dairy cattle

Sheep

Deer

Chicken ☐
Other ☐

Do you identify as a practitioner of one of the following farming types? (multiple responses allowed)

- ☐ Biodynamics
- ☐ Holistic grazing
- ☐ Organic agriculture
- ☐ Permaculture
- ☐ Regenerative agriculture
- ☐ No specific identification
- ☐ Other

Part 4: Grazing and pastures

The following questions aim to capture some information about your grazing and pasture management

Please indicate how important are the below items to determine how and when to graze your paddocks (0 not important, 10 very important)

- Soil health
- Shade
- Pasture diversity
- Pasture recovery period between grazing (duration of pasture rest period)
- Amount of pasture left behind after grazing
- Ryegrass leaf stage
- Pasture quality
- Pasture diversity

How often do you move your stock in spring?

- ○ More than once a day
- ○ Once a day
- ○ Every 2-3 days
- ○ less frequently than 3 days

Has your grazing management changed in the last five years?

- ○ Yes
- ○ No

If Yes, how has your grazing management changed?

What are the three most abundant pasture species present and what is the approximate percentage cover of each?

- Species 1 Percent (1-100)
- Species 2 Percent (1-100)
- Species 3 Percent (1-100)

How many of the following species occur species occur in your pastures?

- ☐ Perennial ryegrass
- ☐ Cocksfoot
- ☐ Tall fescue
- ☐ Prairie grass
- ☐ Timothy
- ☐ Kikuyu
- ☐ Chicory
- ☐ Plantain
- ☐ White clover
- ☐ Red clover
- ☐ Phacelia
- ☐ Barley/Oat/wheat
- ☐ other (please list any extra species if known)

What technique was used to establish your pasture?

(multiple choices possible)

- ☐ Grazing and fertiliser management
- ☐ Direct drill
- ☐ Top dressing

- ☐ Round up application
- ☐ Light tillage
- ☐ Full tillage
- ☐ Other

Part 5: Farm Amendments

The following questions address any use of products or irrigation water that might have a direct or indirect impact on soil invertebrates.

Is irrigation used within the grazing area?

- ○ Yes
- ○ No

What is the source of water and its application method?

How often are fertilisers applied on the farm?

- ○ Never
- ○ Once a year
- ○ A few times a year
- ○ Other

Which of the following item do you apply to your pastures regularly?(multiple choices possible)

- ☐ Urea - based fertilisers
- ☐ Superphosphate
- ☐ Rock phosphate
- ☐ Blood and bone fertiliser
- ☐ Micronutrients
- ☐ Seaweed-based products
- ☐ Dilute sea water
- ☐ Fish-based fertilser
- ☐ Humates / fulvic acid

- ☐ Effective microorganisms (commercial products)
- ☐ Other biological products (wormicast, compost extracts, compost teas etc..)
- ☐ Effluent
- ☐ Other

How often are pesticides used on the farm?

- ○ Never
- ○ Once a year
- ○ A few times a year
- ○ Other

What pesticide products are used? (if applicable)

How often are herbicides used on the farm?

- ○ Never
- ○ Once a year
- ○ A few times a year
- ○ Other

What herbicide product is used?

How often are drenches used on the property?

- ○ Never
- ○ Once a year
- ○ A few times a year
- ○ Other

What drench products are used?

11.2 Appendix 2 FaceBook Add

Available online in November 2022

MASSEY UNIVERSITY
TE KUNENGA KI PŪREHUROA
UNIVERSITY OF NEW ZEALAND

Calling on Farmers

Soil Health Study

11.3 Appendix 3 Site Maps

The following maps show the 100m radius area used for determining landscape diversity and where soil health indicator variables Sites 1-3 identified as practicing conventional grazing and sites 4-6 as practicing regenerative grazing management of cattle raised for beef.

Figure A1. Site 1, 100m radius map. Image retrieved from Google Earth (28/11/2022)

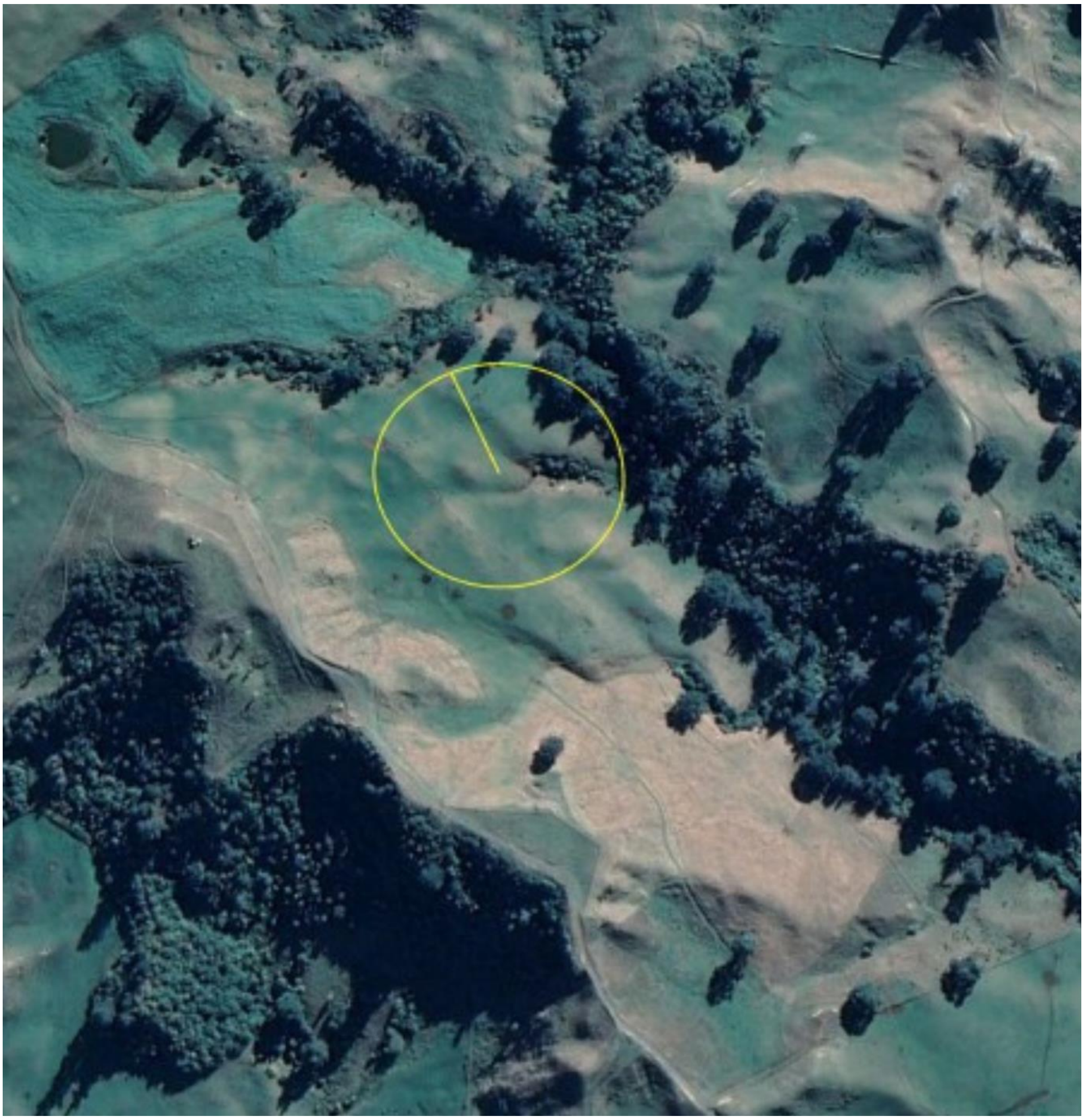

Figure A2. Site 2, 100 m radius map. Image retrieved from Google Earth (28/11/2022)

Figure A3. Site 3, 100 m radius map. Image retrieved from Google Earth (28/11/2022)

Figure A4. Site 4, 100 m radius map. Image retrieved from Google Earth (28/11/2022)

[redacted]

Figure A5. Site 5, 100m radius map. Image retrieved from Google Earth (28/11/2022)

Figure A6. Site 6, 100m radius map. Image retrieved from Google Earth (28/11/2022)

11.4 Appendix 4 Grazing Score Survey

The first round of grazing scores resulted in very close scoring.

- This time I will try to break up the range further to try to create a greater range of scores.
- Please answer as an average of the last 2 years
- Please pick one 1-10 hectare paddock with no (or minimal) impact from sheep to make your responses.

Please remember the survey participants are only known to me and my supervisors and details of the participants will not be included in the final report or any subsequent publications.

One thing I noticed was that the scoring from the last survey was based on recent grazing practices. However, the invertebrate communities sampled will be representative of grazing practices over the last 2 years.

- With that in mind, could you please respond to the following questions as an average over the last 2 years.
- For example, if beef were left in paddocks to graze for 8 days (on average) a year ago and 4 days (on average) this last year - the average grazing time across the last two years would be 6 days.
- Please do a similar average calculation for the following four questions.

Name:

Please include your name below, so that I can match the following responses to your previous responses.

Name ☐

What is your stock density per hectare in the area of land they are allowed to graze when temporary fencing is used?

- ○ 0-10 cattle per hectacre
- ○ 10-20 cattle per hectare
- ○ 20-30 cattle per hectare
- ○ 30-40 cattle per hectare
- ○ 40-50 cattle per hectare

- ○ 50-60 cattle per hectare
- ○ 60-70 cattle per hectacre
- ○ more than 70 per hectare

What is the average time cattle are left grazing in a paddock?

- ○ 1/2 day
- ○ 1 day
- ○ 2 days
- ○ 3 days
- ○ 4 days
- ○ 5 days
- ○ 6 days
- ○ 7 days or more

How long is pasture rested (on average) between grazing? (rotation length)

- ○ less than a week
- ○ between 1-3 weeks
- ○ between 3-6 weeks
- ○ between 6-9 weeks
- ○ between 9-12 weeks
- ○ between 12-15 weeks
- ○ between 15-18 weeks
- ○ more than 18 weeks

How long are pasture residuals (length of grass) after grazing?

- ○ 0-5 cm
- ○ 5-10 cm
- ○ 10-20 cm
- ○ 20-40 cm
- ○ 40-60 cm
- ○ 60-80 cm
- ○ 80-100 cm
- ○ A meter or more

What Soil Type?

Another factor that may impact results is differences in soil type between sample sites.

Could you please indicate in the next question the type of soil in the 1-10 hectare sample paddock we will be using for testing.

If the sample paddock has soil that is fairly typical of the region that would help a lot and make matching soil types between sites easier.

Please describe the soil types in the sampling paddock below

11.5 Sampling Data

11.5.1 Spider abundance and diversity

Table A1. Data collected from pitfall traps on numbers of spiders belonging to different families (Phal = Phalangiidae (European harvestmen), Lyc = Lycosidae (wolf spiders), Lin = Linyphiidae (sheet weavers), Dol = Dolomedes (nurseryweb). Table shows number of spiders collected that belong to each group at each site and the Simpsons diversity index of spider families at each site. Treatment 2 corresponds to sites practicing conventional grazing (x 3 sites) and treatment 1 to sites practicing regenerative grazing (x 3 sites).

			Spider (number)					Simpsons Diversity
Site	Replicate	Treatment	Phal	Lyc	Lin	Dol	SPTotal	SPDiv
1	1	2	0	0	3	0	4	0.40
1	2	2	0	0	2	0	1	0.00
1	3	2	0	0	2	0	0	0.00
1	4	2	0	0	2	0	5	0.05
1	5	2	1	0	3	0	1	0.00
2	1	2	0	1	3	0	3	0.20
2	2	2	0	1	0	0	2	1.00
2	3	2	0	0	2	0	2	1.00
2	4	2	0	1	2	0	2	1.00
2	5	2	0	2	1	0	3	0.20
3	1	2	0	2	3	0	4	0.09
3	2	2	0	0	3	0	1	0.00
3	3	2	0	0	2	0	2	1.00
3	4	2	0	0	5	0	3	0.20
3	5	2	1	0	0	0	3	0.20
4	1	1	3	0	0	0	5	0.05
4	2	1	3	0	0	0	3	0.20
4	3	1	1	0	0	0	2	1.00
4	4	1	2	0	0	0	5	0.05
4	5	1	7	2	1	0	0	0.00
5	1	1	0	6	0	0	0	0.00

5	2	1	0	2	0	0	0	0.00
5	3	1	0	5	0	0	0	0.00
5	4	1	5	2	0	0	0	0.00
5	5	1	0	0	0	0	3	0.20
6	1	1	0	0	3	1	6	0.03
6	2	1	0	0	1	0	2	1.00
6	3	1	1	0	0	0	5	0.05
6	4	1	0	5	0	0	2	1.00
6	5	1	1	0	1	0	0	0.00

11.5.2 Aggregate size

Table A2. Dry weight of aggregates, separated according to different sieve sizes 8000mm (coarse) to <62 mm (very fine) and the calculated dry mean weight diameter (dMWD) of each site. Treatment 2 corresponds to sites practicing conventional grazing (x 3 sites) and treatment 1 to sites practicing regenerative grazing (x 3 sites).

			Soil weight (g) collected in different sized sieves (mm)									Soil size (mm)
Site	Rep	Treat	8000	4000	2000	1000	500	250	125	63	<62	dMWD
1	1	2	21.9	18.1	24.2	18	11.8	8.2	4.3	2.1	1.1	2.94
1	2	2	16.6	18.6	25.3	19.8	15	10.1	5.4	2.6	1.1	2.52
1	3	2	28.1	21.9	22.8	14	7.5	3.6	2.1	1.1	0.6	3.71
1	4	2	18.2	16.9	22.3	17.6	11.8	7.5	4.3	2.1	0.8	2.80
1	5	2	29.9	19.5	18.9	11.1	7.2	3.6	1.8	0.9	0.5	3.97
2	1	2	28.1	24.8	23.4	16.1	14.8	11.9	10	4.7	1.7	2.94
2	2	2	26.3	27.2	24	15.6	11.1	7.7	5.9	3	1.2	3.21
2	3	2	27	23.2	18	13.7	11.6	9.1	6.9	3.9	1.5	3.20
2	4	2	32.8	26.6	21.3	15.8	12	9.2	7.8	4.2	1.6	3.33
2	5	2	38.5	30.1	22.5	14.1	10.1	6.3	4	2.1	0.9	3.85
3	1	2	21.1	18.4	16.5	14.4	14.3	8.8	5.9	3.2	1.6	2.88
3	2	2	23.4	25.2	19	13.2	9.4	5.4	3.5	2.1	0.9	3.39
3	3	2	61	37.7	21.4	14	10.1	7.2	9.4	9.2	2.5	4.08
3	4	2	39.8	21.5	21.3	16	11.8	7.9	6.2	4.1	1.7	3.62
3	5	2	44.9	36.7	26.5	18.4	13.5	8.3	10.4	7.7	1.6	3.50
4	1	1	19	33.3	24.4	15.7	11.7	9.4	4.9	1.7	0.7	2.97
4	2	1	22.8	27.1	25.1	16.5	11.3	9	5.8	2.1	1	3.03
4	3	1	32.8	36	25	15.1	11.5	8.6	5.1	1.9	1	3.51
4	4	1	28.8	34.5	21.9	15.7	11.1	8.6	5.1	2	1.4	3.38
4	5	1	46.6	43.1	28.2	15.2	9.8	6.9	3.6	1.2	0.5	4.02
5	1	1	59.9	37.7	26.3	15	9.6	5.9	3.2	1.5	0.7	4.41
5	2	1	57.2	35.6	24.7	13.3	8.2	4.8	2.6	1.1	0.9	4.50
5	3	1	77.4	45.9	25.5	12	6.6	3.3	1.6	0.9	0.6	5.01
5	4	1	69.7	40.1	29.4	13.5	5.7	2.7	1.2	0.7	0.6	4.85
5	5	1	67	45.7	29	16.2	9.5	6.3	3.8	1.6	0.7	4.45
6	1	1	12.7	7.8	10.9	10	8.6	8	7.9	4.8	1.6	2.38
6	2	1	14.1	8.4	9.9	8	8.4	8.5	10.6	8.3	3.9	2.28
6	3	1	17.6	14.1	14.4	9.9	8.9	8.9	10.3	8	4.2	2.54

6	4	1	11.3	8.8	9.7	15.8	6.4	3.2	4	5.3	2.6	2.47
6	5	1	17.2	14.8	15.7	12.8	9	7.9	7.9	4.9	2.9	2.67

11.5.3 Abundance and diversity of worms

Table A3. Data collected from spade-sized soil samples of worms belonging to different functional groups (EPI = epigeic, END = endogeic, ANE = anecic). Table shows number of worms collected that belong to each group at each site and the Simpsons diversity index of worm functional groups at each site. Treatment 2 corresponds to sites practicing conventional grazing (x 3 sites) and treatment 1 to sites practicing regenerative grazing (x 3 sites).

			Worms (number)				
Site	Replicate	Treatment	EPI	END	ANE	WOTotal	WODiv
1	1	2	6	11	12	29	0.34
1	2	2	2	24	12	38	0.49
1	3	2	0	33	15	48	0.56
1	4	2	0	27	12	39	0.56
1	5	2	2	35	18	55	0.51
2	1	2	0	6	2	8	0.69
2	2	2	0	4	5	9	0.29
2	3	2	0	2	2	4	0.59
2	4	2	0	1	0	1	0.61
2	5	2	0	2	1	3	0.66
3	1	2	0	9	6	15	0.58
3	2	2	0	1	0	1	0.45
3	3	2	0	4	2	6	0.35
3	4	2	0	0	1	1	1.00
3	5	2	0	2	0	2	0.37
4	1	1	0	9	6	15	0.49
4	2	1	0	1	0	1	1.00
4	3	1	0	4	2	6	0.48
4	4	1	0	0	1	1	1.00
4	5	1	0	2	0	2	1.00
5	1	1	0	9	6	15	0.49
5	2	1	0	1	0	1	1.00
5	3	1	0	4	2	6	0.48
5	4	1	0	0	1	1	1.00
5	5	1	0	2	0	2	1.00
6	1	1	5	25	8	38	0.48
6	2	1	4	16	10	30	0.40
6	3	1	2	6	3	11	0.35
6	4	1	3	17	12	32	0.42
6	5	1	2	25	11	38	0.51

11.5.4 Pasture Diversity

Table A4. Data collected of pasture weight per hectare from pasture quadrat samples that belong to different pasture functional groups (AG= annual grass , PG = perennial grass, CL= clover, BL= broadleaf, H=herbs) at each site and the Simpsons diversity index of worm functional groups at each site. Treatment 2 corresponds to sites practicing conventional *grazing (x 3 sites) and treatment 1 to sites practicing regenerative grazing (x 3 sites). The diversity of pasture functional groups (GrassDiv) was calculated using the Simpsons Diversity Index.*

			Pasture weight (kg/hec)					
Site	Replicate	Treatment	AG	PG	CL	BL	H	GrassDiv
1	1	2	0	545.2	110.4	34.4	55.2	0.56
1	2	2	34.8	341.2	242	0	20	0.43
1	3	2	0	304	109.2	222.4	65.6	0.32
1	4	2	6.4	677.2	201.6	0	96.8	0.53
1	5	2	0	864.4	101.6	0	8	0.80
2	1	2	310.4	176.8	187.6	0	7.2	0.35
2	2	2	347.6	366.4	222	0	5.2	0.34
2	3	2	268	173.6	226.4	129.6	0	0.27
2	4	2	264	322.4	87.6	66	0	0.34
2	5	2	133.6	208	118.4	26.8	0	0.32
3	1	2	0	340.4	61.2	0	76.8	0.55
3	2	2	0	973.2	127.6	10.8	0	0.78
3	3	2	0	1223.6	132.4	0	0	0.82
3	4	2	0	1185.2	211.6	0	0	0.74
3	5	2	0	788	162	0	28.8	0.68
4	1	1	382.4	738.4	165.2	56	0	0.40
4	2	1	288	331.6	59.2	416.4	0	0.31
4	3	1	516.8	407.6	161.2	122.4	0	0.32
4	4	1	333.2	629.2	148.4	299.2	0	0.31
4	5	1	185.6	640.4	88.8	202	0	0.39
5	1	1	476	176.4	261.2	0	0	0.39
5	2	1	989.6	414	48.4	8	0	0.54
5	3	1	913.6	72	270.8	22.4	0	0.56
5	4	1	112	466	442	63.6	0	0.36
5	5	1	0	478.4	420.8	116.8	0	0.41
6	1	1	0	878.8	62	2.4	36.4	0.81
6	2	1	122	710	55.6	0	127.2	0.52
6	3	1	0	960	88	0	0	0.85
6	4	1	0	864	52	0	0	0.89
6	5	1	0	644	58.8	0	20	0.80

11.5.5 Soil Nutrients

Table A5. Data collected on nutrients in soil samples analysed by Hills Laboratory. Measurement units vary depending upon what is reported from Hills Laboratory. Budget constraints meant that only 3 samples could be measured per site. Treatment 2 corresponds to sites practicing conventional grazing (x 3 sites) and treatment 1 to sites practicing regenerative grazing (x 3 sites).

			Soil nutrients						
			me/100g	me/100g	me/100g	me/100g	mg/kg	kg/ha	mg/L
Site	Rep	Treat	Potassium	Calcium	Magnesium	Sodium	Sulphur	Nitrogen	Phosphorus
1	1	2	1.37	10.1	3.16	0.2	13	312	16
1	2	2	1.65	8.5	2.3	0.13	11	360	16

1	3	2	0.91	7.6	1.55	0.17	9	235	9
2	1	2	0.56	12	1.75	0.18	11	255	11
2	2	2	1.03	13.8	2.11	0.08	6	325	8
2	3	2	0.38	8.2	1.4	0.16	5	274	7
3	1	2	0.49	7.5	1.12	0.11	6	247	24
3	2	2	0.62	11.2	2.59	0.11	9	341	32
3	3	2	0.76	10.5	1.97	0.11	8	261	19
4	1	1	1.9	7.1	1.46	0.11	9	189	18
4	2	1	0.76	9.2	1.66	0.13	6	225	13
4	3	1	0.79	10.9	2.13	0.2	10	191	13
5	1	1	0.29	8.8	2	0.39	5	301	32
5	2	1	0.42	8.7	1.76	0.4	6	244	42
5	3	1	0.33	8.7	2.16	0.35	5	334	42
6	1	1	0.38	9.8	0.68	0.18	12	214	15
6	2	1	0.55	11.2	1.71	0.19	10	244	25
6	3	1	0.47	11.4	1.4	0.18	8	253	31

11.5.6 Slope, Pasture Brix and Soil Carbon

Table A6. Average slope in degrees (of 5 slope gradients) at each site and average pasture brix (of 5 brix samples) at each site. Treatment 2 corresponds to sites practicing conventional grazing (x 3 sites) and treatment 1 to sites practicing regenerative grazing (x 3 sites).

Site	Replicate	Treatment	Soil Carbon (%)	Slope (angle)	Brix
1	1	2	14.07	11.03	7
1	2	2	20.14	11.03	5.5
1	3	2	16.39	11.36	7.2
1	4	2	18.63	11.87	8
1	5	2	18.61	9.96	8
2	1	2	11.85	1.12	7
2	2	2	15.09	3.58	6.8
2	3	2	13.17	5.20	8.8
2	4	2	13.48	2.82	8.9
2	5	2	13.15	0.36	7.8
3	1	2	17.63	2.11	7.5
3	2	2	13.81	0.27	6.8
3	3	2	9.04	0.30	6.5
3	4	2	7.62	6.01	9
3	5	2	7.42	3.44	7.8
4	1	1	17.47	2.76	8.1
4	2	1	17.14	4.02	7.2
4	3	1	18.14	5.00	8
4	4	1	17.57	3.56	7.5
4	5	1	14.87	0.98	6.8
5	1	1	8.71	0.69	5.5
5	2	1	10.92	0.65	6.5
5	3	1	6.01	2.03	4
5	4	1	7.48	0.90	4.6

5	5	1	7.71	0.22	5
6	1	1	9.74	26.79	9
6	2	1	20	35.18	8
6	3	1	17.63	38.93	8.8
6	4	1	13.86	40.70	8.4
6	5	1	10.69	41.26	9

11.6 Appendix 5 Slope Maps

Site 1

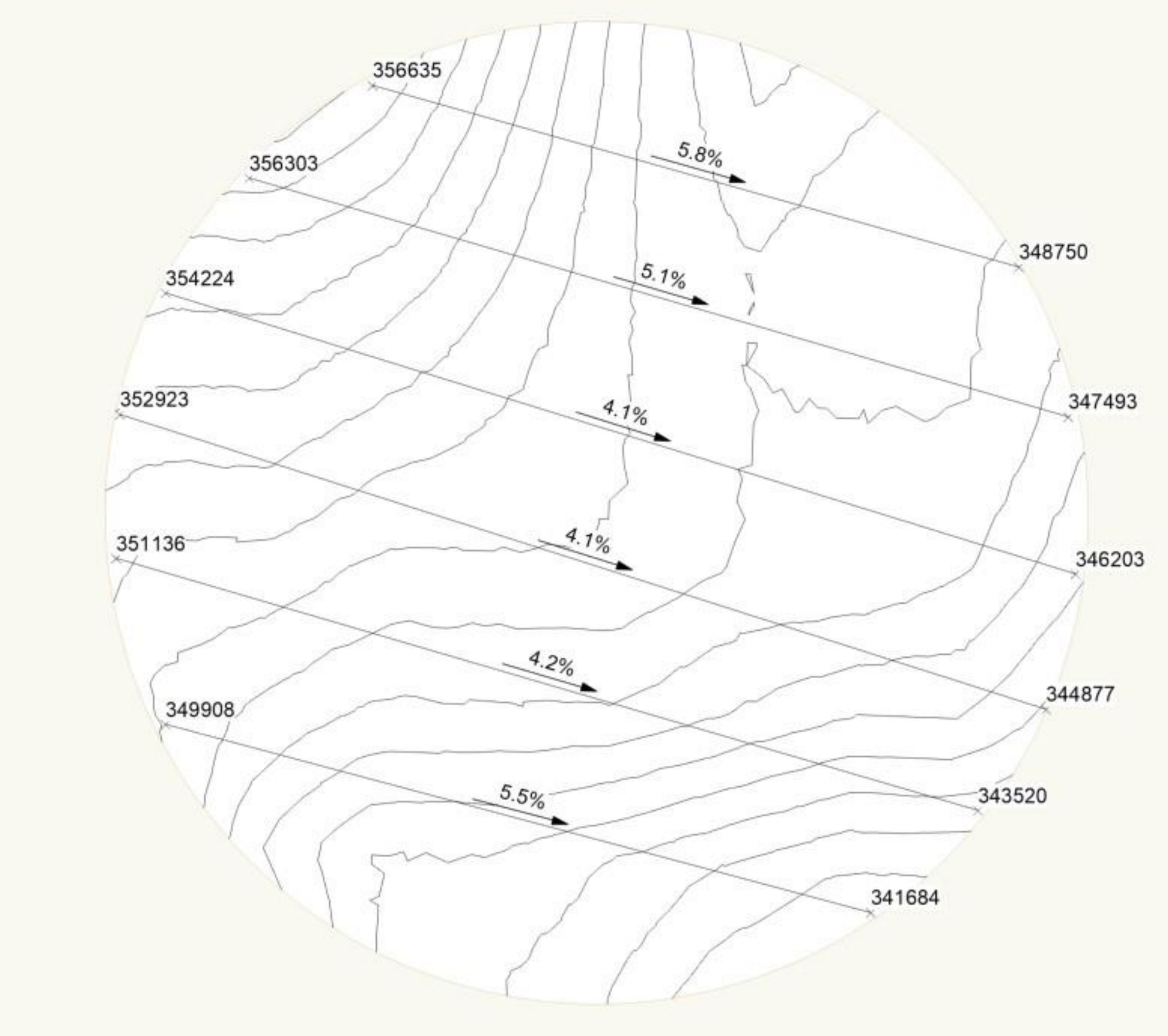

Figure A7. Shows elevation changes (mm) at site 1, between two points on either end of 6 elevation transect lines crossing the 100m radius topographical map and percentage difference between the two elevations. The elevation difference was recorded a slope in the results used in statistical analysis (see raw data in Appendix 5.6).

Site 2

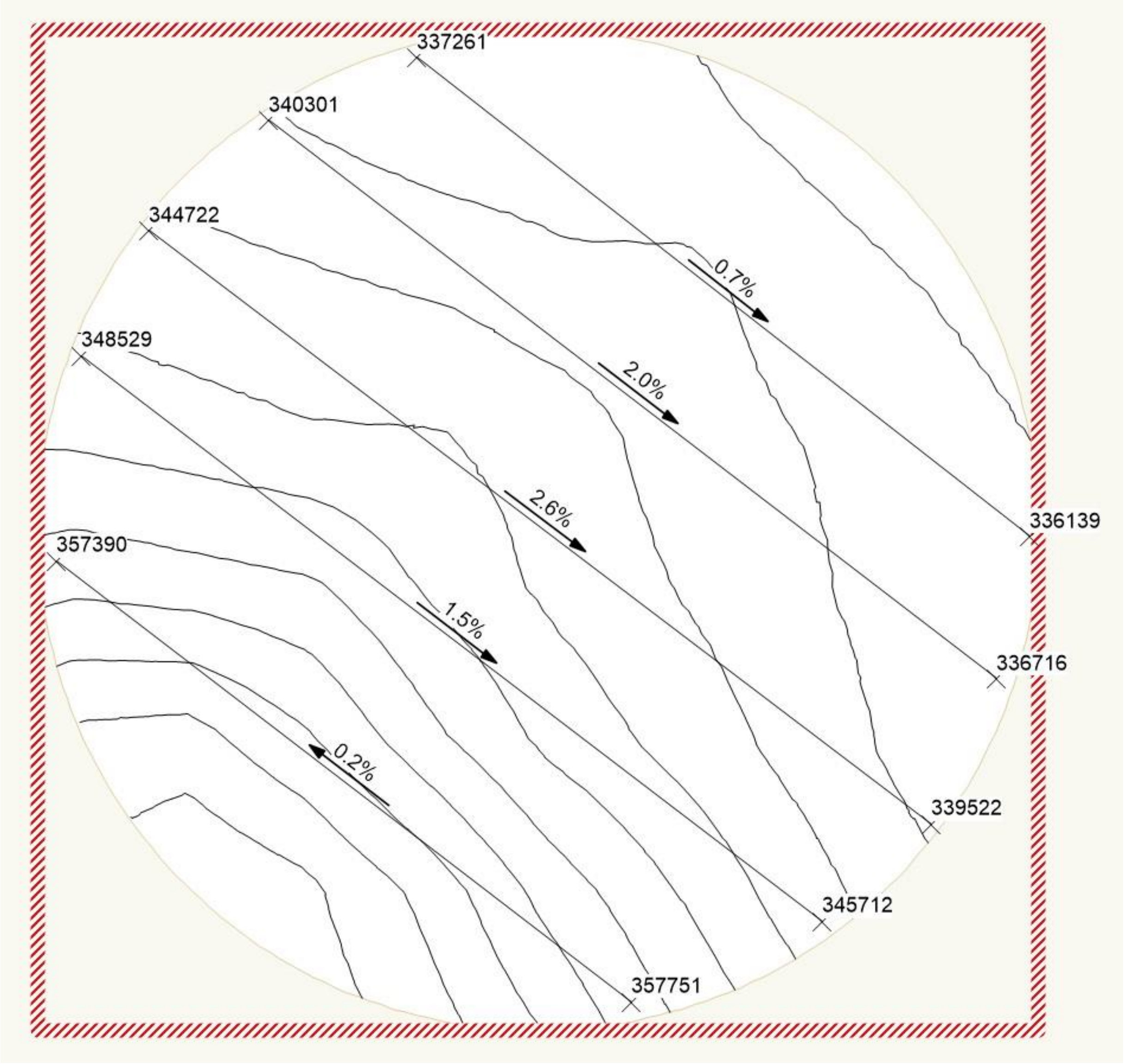

Figure A8. Shows elevation changes (mm) at site 2, between two points on either end of 6 elevation transect lines crossing the 100m radius topographical map and percentage difference between the two elevations. The elevation difference was recorded a slope in the results used in statistical analysis (see raw data in Appendix 5.6).

Site 3

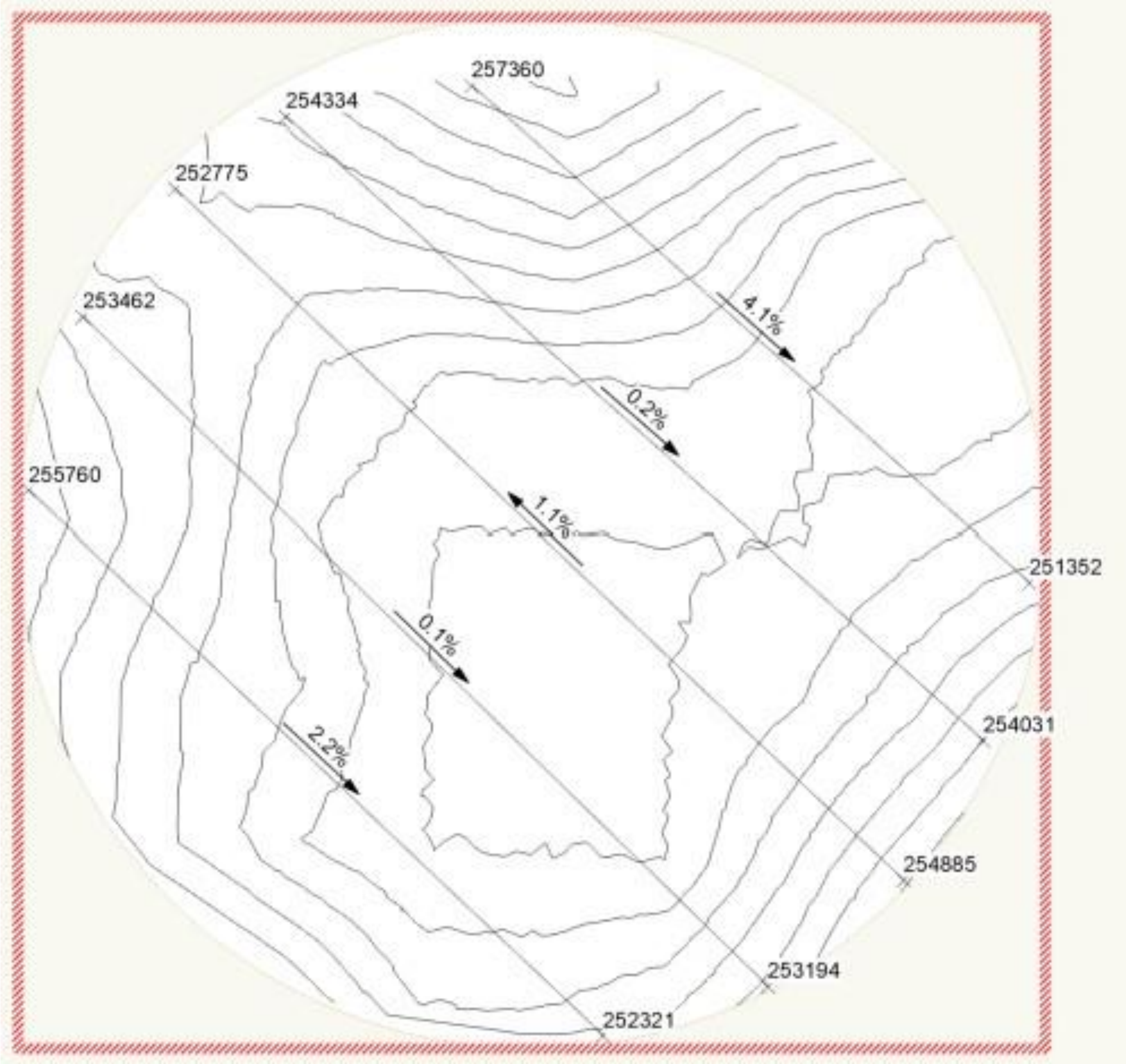

Figure A9. Shows elevation changes (mm) at site 3, between two points on either end of 6 elevation transect lines crossing the 100m radius topographical map and percentage difference between the two elevations. The elevation difference was recorded a slope in the results used in statistical analysis (see raw data in Appendix 5.6).

Site 4

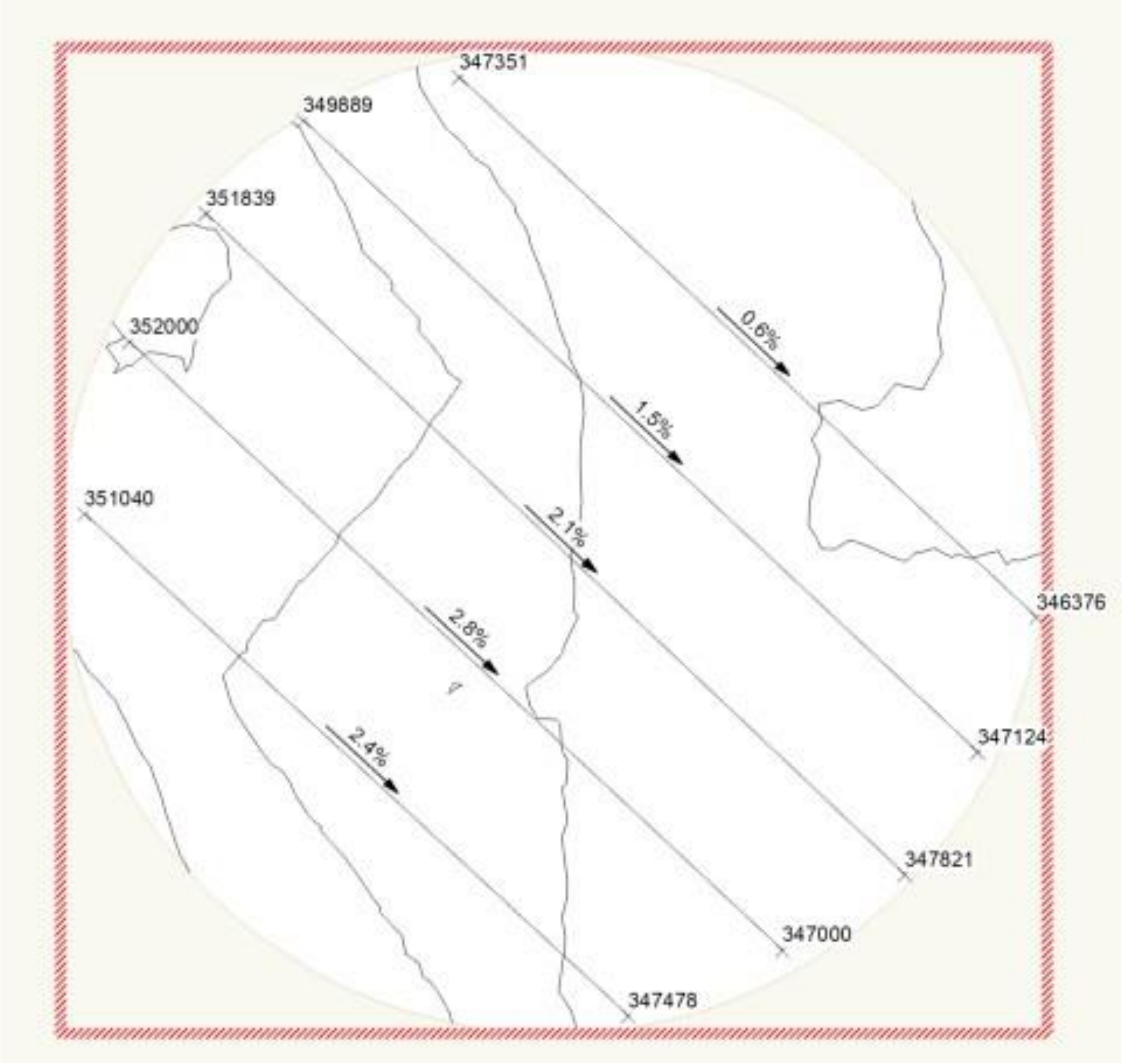

Figure A10. Shows elevation changes (mm) at site 4, between two points on either end of 6 elevation transect lines crossing the 100m radius topographical map and percentage difference between the two elevations. The elevation difference was recorded a slope in the results used in statistical analysis (see raw data in Appendix 5.6).

Site 5

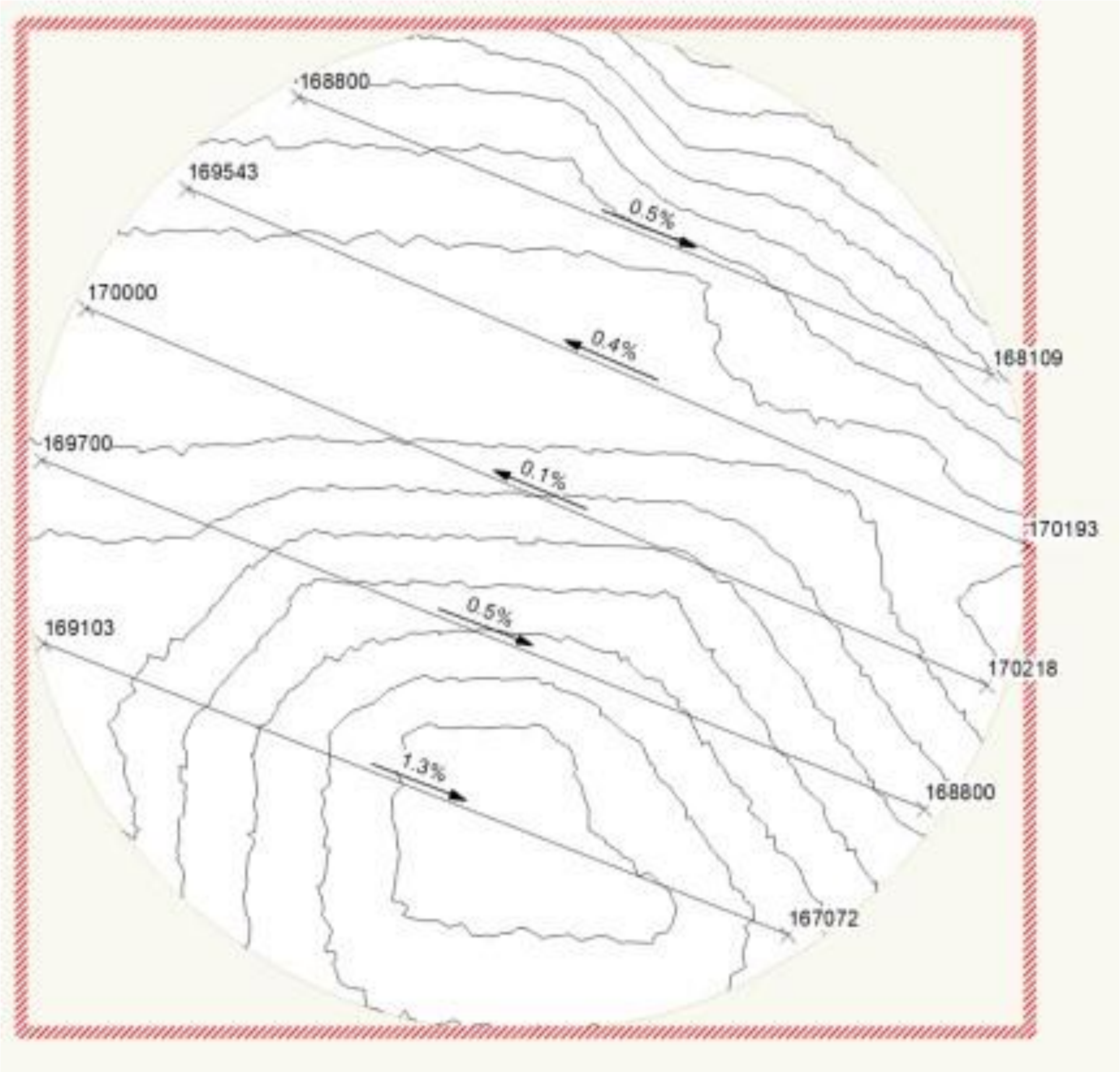

Figure A11. Shows elevation changes (mm) at site 5, between two points on either end of 6 elevation transect lines crossing the 100m radius topographical map and percentage difference between the two elevations. The elevation difference was recorded a slope in the results used in statistical analysis (see raw data in Appendix 5.6).

Site 6

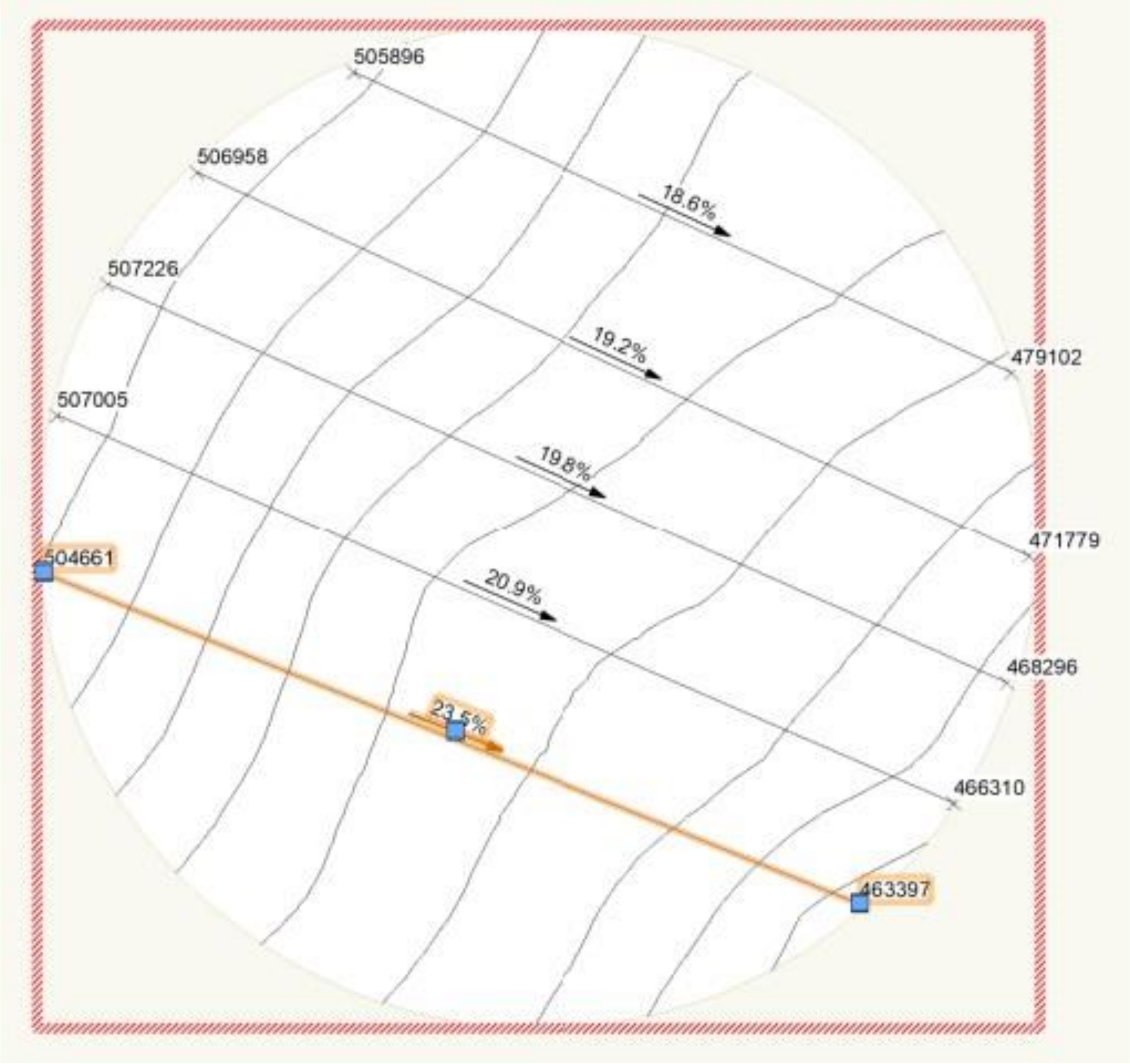

Figure A12. Shows elevation changes (mm) at site 6, between two points on either end of 6 elevation transect lines crossing the 100m radius topographical map and percentage difference between the two elevations. The elevation difference was recorded a slope in the results used in statistical analysis (see raw data in Appendix 5.6).

11.7 Appendix 6 Landscape Maps with Sampling Transect Lines

Each of the following maps shows the different areas covered by different types of vegetation (other than grass) at each site. As any of these non-pasture habitats may provide important habitat to support spider abundance and diversity the proportion of each site not covered in grass was used as a representation of landscape diversity.

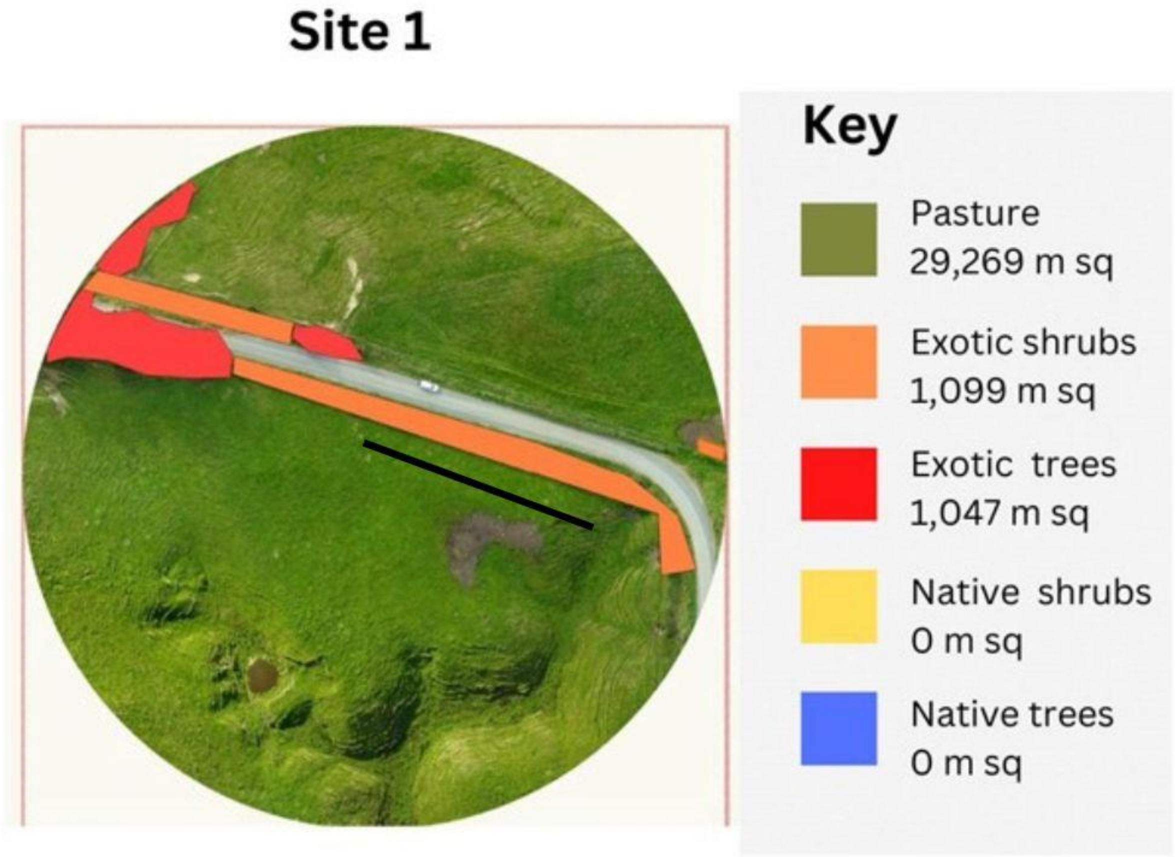

Figure A13. Shows landscape diversity map, with different vegetation types and the area covered by each vegetation type (m2) at site 1. The sampling transect line is also represented by the black line and every 15m along this line earthworm, spider, pasture brix, pasture diversity, soil carbon, soil aggregates, soil nutrients samples were collected (all in the same general vicinity), with five replicate samples of each variable recorded.

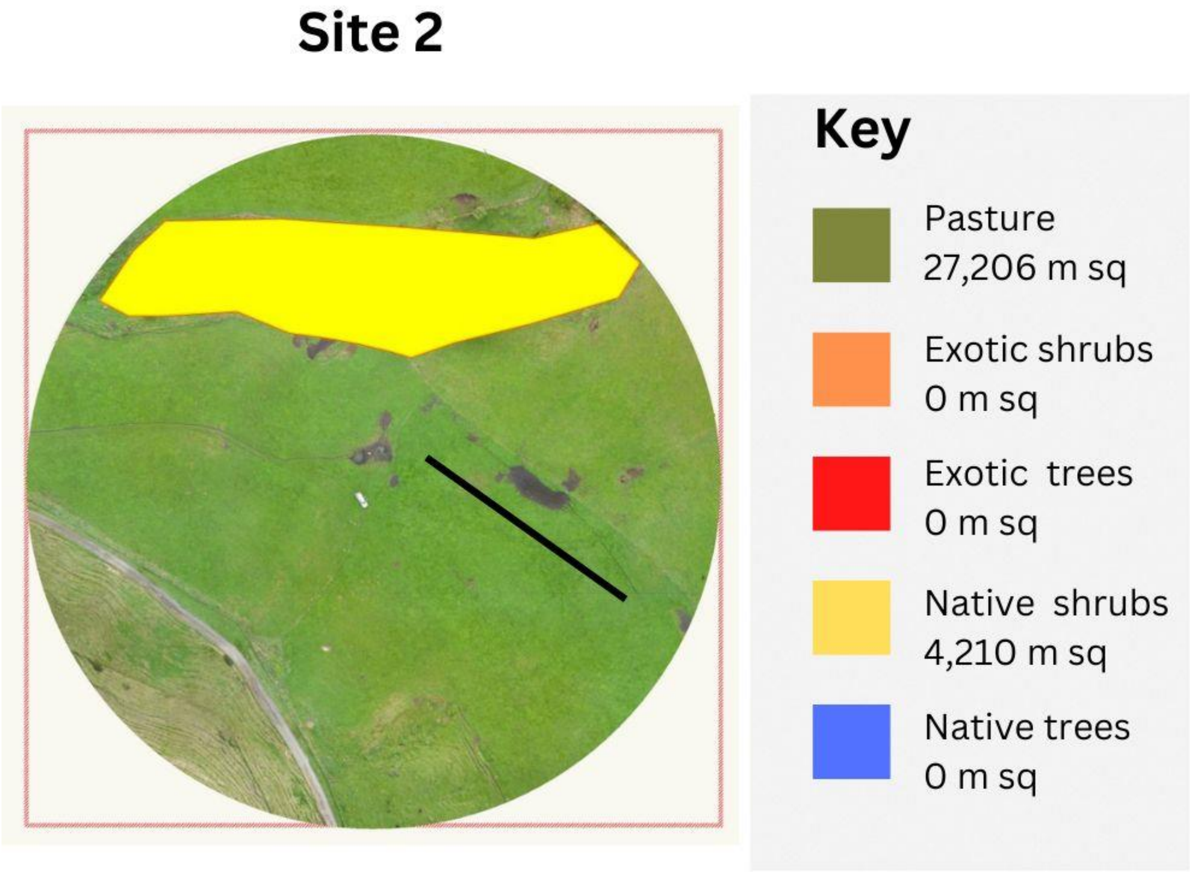

Figure A14. Shows landscape diversity map, with different vegetation types and the area covered by each vegetation type (m2) at site 2. The sampling transect line is also represented by the black line and every 15m along this line earthworm, spider, pasture brix, pasture diversity, soil carbon, soil aggregates, soil nutrients samples were collected (all in the same general vicinity), with five replicate samples of each variable recorded.

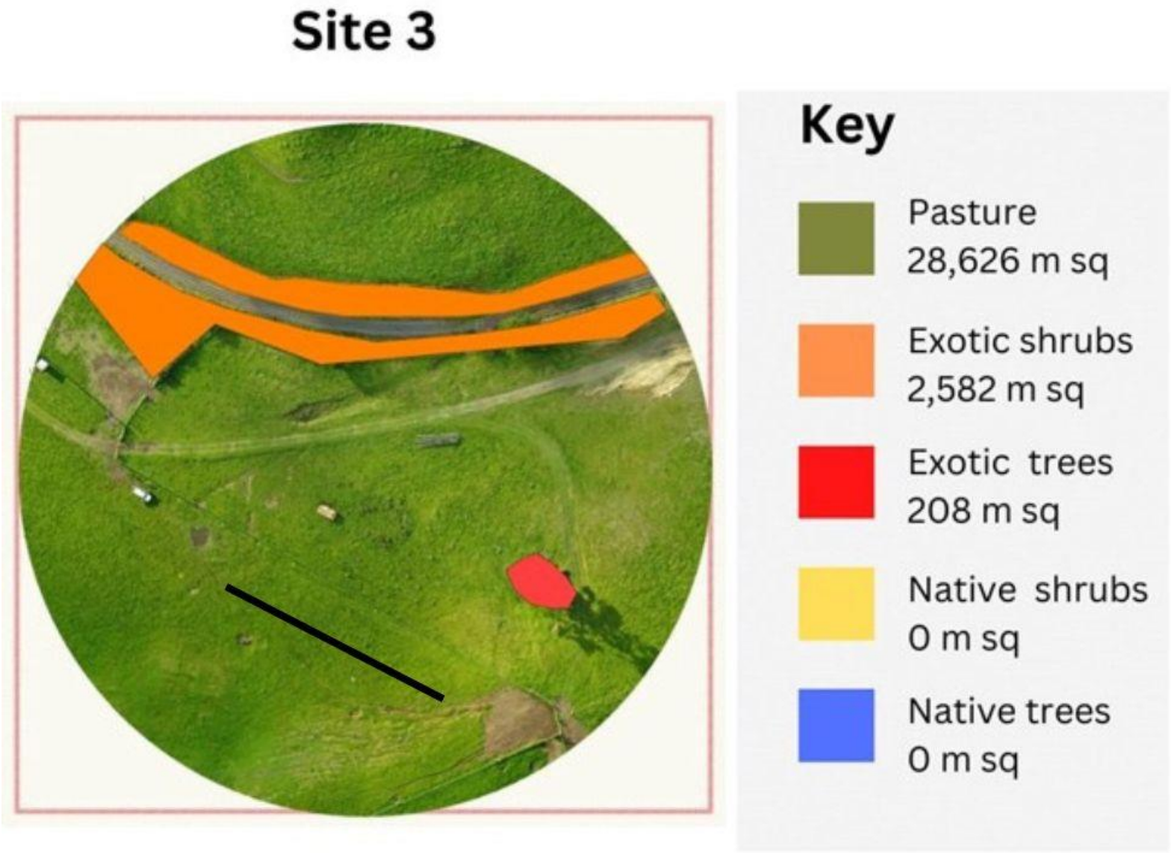

Figure A15. Shows landscape diversity map, with different vegetation types and the area covered by each vegetation type (m2) at site 3. The sampling transect line is also represented by the black line and every 15m along this line earthworm, spider, pasture brix, pasture diversity, soil carbon, soil aggregates, soil nutrients samples were collected (all in the same general vicinity), with five replicate samples of each variable recorded.

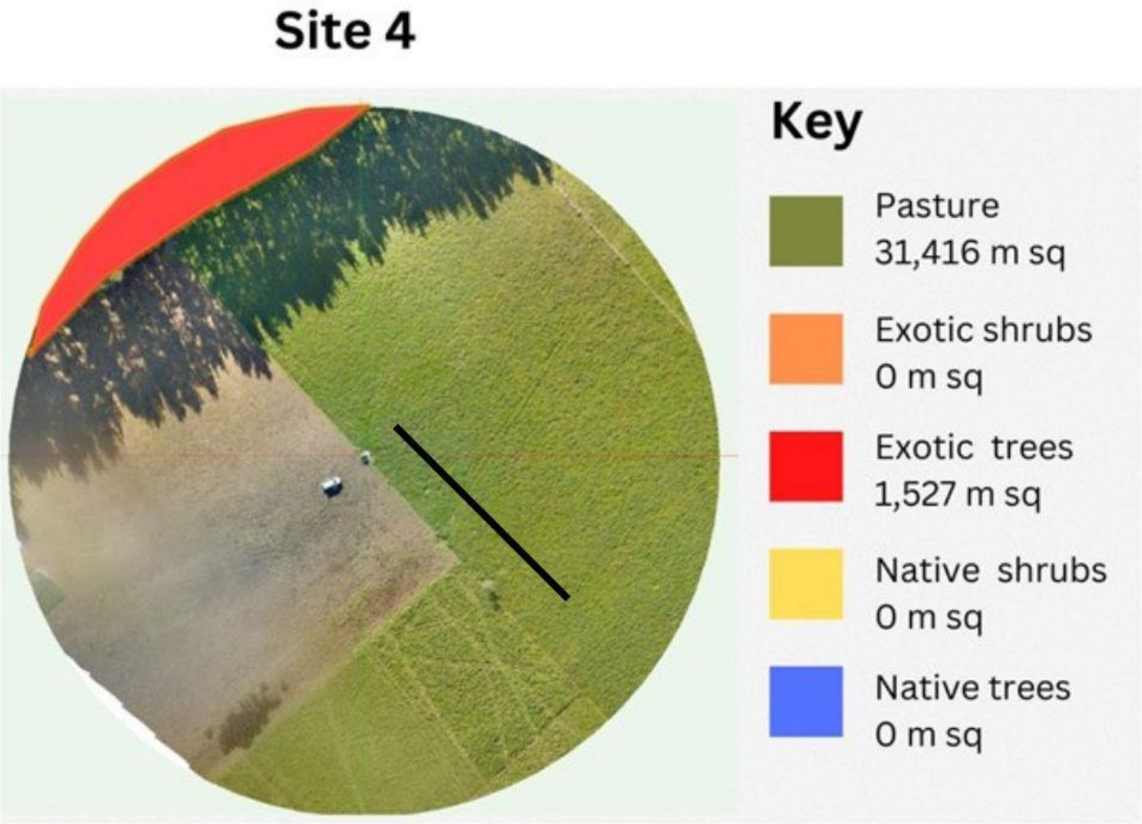

Figure A16. Shows landscape diversity map, with different vegetation types and the area covered by each vegetation type (m2) at site 4. The sampling transect line is also represented by the black line and every 15m along this line earthworm, spider, pasture brix, pasture diversity, soil carbon, soil aggregates, soil nutrients samples were collected (all in the same general vicinity), with five replicate samples of each variable recorded.

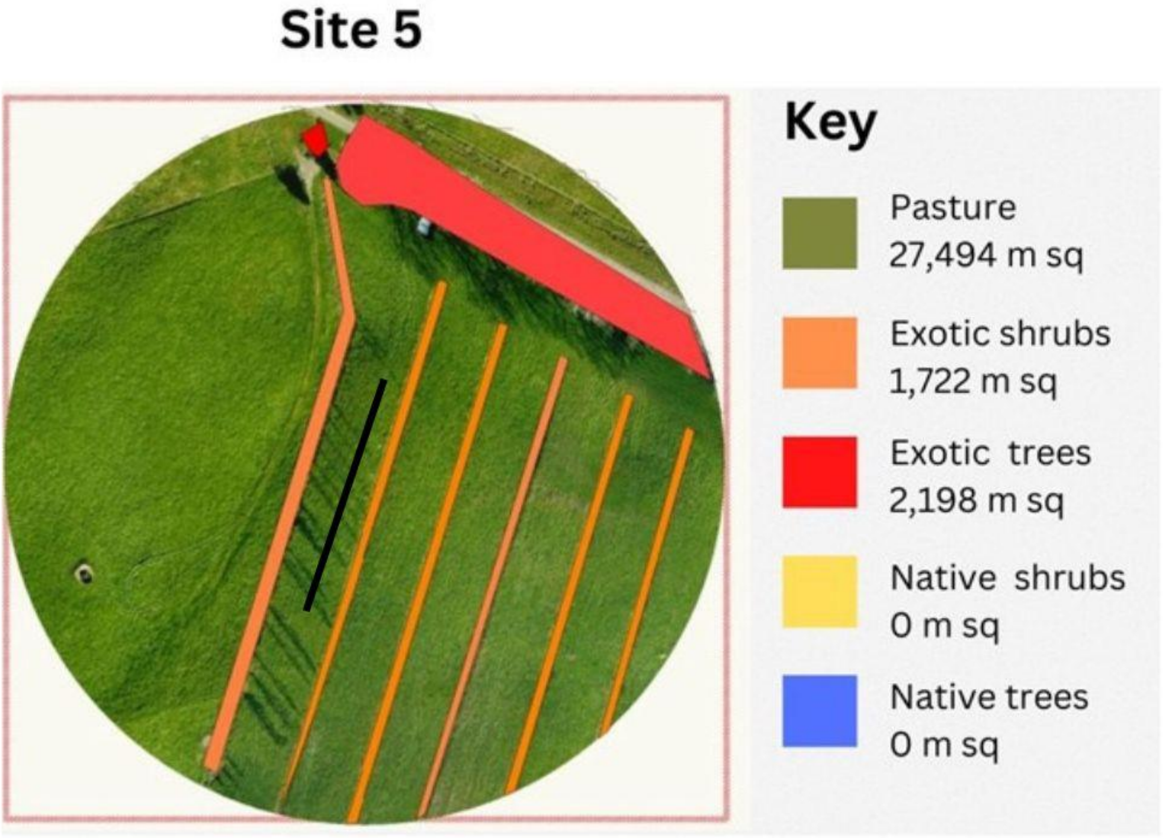

Figure A17. Shows landscape diversity map, with different vegetation types and the area covered by each vegetation type (m2) at site 5. The sampling transect line is also represented by the black line and every 15m along this line earthworm, spider, pasture brix, pasture diversity, soil carbon, soil aggregates, soil nutrients samples were collected (all in the same general vicinity), with five replicate samples of each variable recorded.vicinity), with five replicate samples of each variable recorded

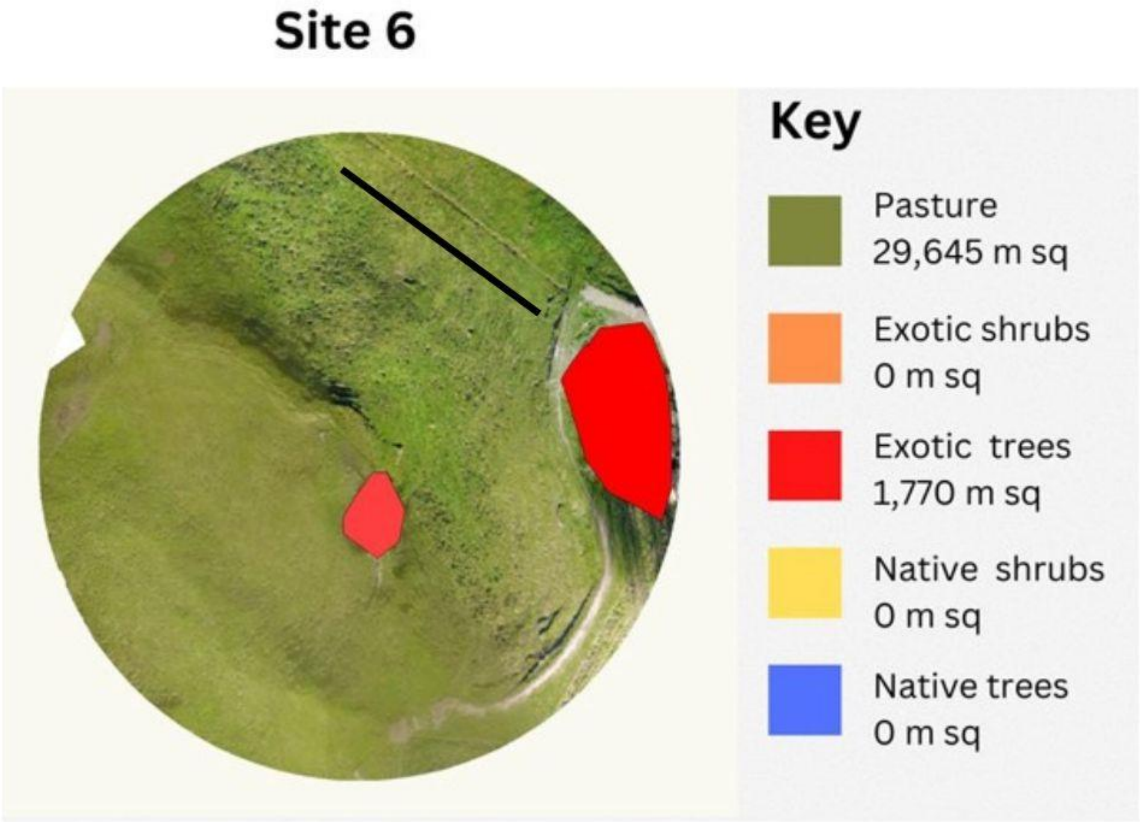

Figure A18. Shows landscape diversity map, with different vegetation types and the area covered by each vegetation type (m2) at site 6. The sampling transect line is also represented by the black line and every 15m along this line earthworm, spider, pasture brix, pasture diversity, soil carbon, soil aggregates, soil nutrients samples were collected (all in the same general vicinity), with five replicate samples of each variable recorded.