Multifunctionality of woody vegetation on sheep and beef cattle farms in Aotearoa New Zealand

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Attestation of Authorship

I hereby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previously published or written by another person (except where explicitly defined in the acknowledgements), nor material which to a substantial extent has been submitted for the award of any other degree or diploma of a university or other institution of higher learning.

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“Knowledge is not what is memorized, but only what benefits (humanity).”

(Imam As Syafii RA)

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Abstract

There is growing global interest in large-scale tree-planting initiatives as a powerful means of mitigating the effects of climate change while achieving multiple social and environmental goals, including biodiversity conservation, socio-economic benefits, and sustainability. Nevertheless, a number of these initiatives have increased the risk to native forests and biodiversity while expanding the cover of exotic trees. In addition, although the addition of woody vegetation to agricultural landscapes has been recognized globally as a 'nature based solution' for mitigating and adapting to climate change and reversing biodiversity loss, there is still a lack of research on the potential contributions of woody vegetation in agricultural landscapes for enhancing carbon stocks, tree and shrub diversity, and ecosystem multifunctionality. Recent tree planting and carbon credit policies in Aotearoa New Zealand make woody vegetation patches, such as remnant native forests frequently found on sheep and cattle farms, susceptible to conversion into new exotic forests. Few quantitative estimates of the potential carbon stock densities in agricultural landscapes have been made, and as such, estimates have not been incorporated into the quantification and future predictions of the multifunctionality of woody vegetation on farms. This research aimed to (1) evaluate how existing tree planting initiatives can contribute to biodiversity and carbon objectives, (2) examine how existing woody vegetation in sheep and beef cattle farms could contribute to climate change mitigation, and (3) quantify multifunctional landscape contributions under possible future land-use change scenarios where existing native woody vegetation is protected and new trees are added. Using Aotearoa New Zealand's One Billion Trees program as a case study, I offered ten recommendations for how large-scale tree planting should be modified to prevent unintended consequences for native species while simultaneously meeting our carbon and biodiversity objectives. I then characterized and quantified carbon stocks and multifunctionality from woody vegetation on three case-study farms in northern, central and southern Aotearoa New Zealand, and showed that native-dominated woodland and shrubland communities had higher multifunctionality scores than exotic-dominated woodland communities. Finally, I developed a spatially-explicit simulation model to compare and contrast the multifunctionality outcomes of a range of native woody vegetation restoration and revegetation scenarios in the three Aotearoa New Zealand case study landscapes. The modelling results demonstrated the significance of understanding the distinctive characteristics of each landscape, as different scenarios of woody vegetation restoration and gully revegetation will create outcomes that are highly dependent on the initial quantities, types, and configurations of woody vegetation and gully distributions in each landscape. The study illustrated how land management decision-making can better consider a combination of co-benefits and trade-offs to identify the most appropriate actions for achieving multifunctionality in agricultural landscapes. This thesis significantly improves our understanding of the potential contribution of woody

vegetation on sheep and beef cattle farms in Aotearoa New Zealand to climate change mitigation and biodiversity conservation.

Chapter 1 General introduction

1.1 Climate Change, Carbon, and Biodiversity

There is a growing global awareness concerning the adverse impacts of climate change resulting from greenhouse gas emissions. Human activities, primarily due to the release of greenhouse gases, have undeniably triggered global warming. By 2011-2020, the global surface temperature had surged by 1.1°C above the 1850-1900 levels (IPCC, 2007). During 2010-2019, global greenhouse gas emissions continued to rise, driven by unsustainable energy use, land use changes, diverse lifestyles, and consumption patterns across regions, countries, and individuals (IPCC, 2007, 2022b). Global net anthropogenic greenhouse gas emissions comprise CO_2 from fossil fuel combustion and industrial processes (CO_2-FFI), net CO_2 from land use, land-use change, and forestry (CO_2-LULUCF), methane (CH_4), nitrous oxide (N_2O), and fluorinated gases (HFCs, PFCs, SF_6, NF_3) (Penman et al., 2003).

Climate change has far-reaching impacts, evident in extreme weather events worldwide such as more frequent warm days and nights, heatwaves, heavy precipitation, droughts, intense tropical cyclones, and rising sea levels (IPCC, 2007). These changes have significant consequences for human activities. For instance, the increase in greenhouse gases leads to global warming, raising surface temperatures and intensifying the hydrological cycle by enhancing evaporation. Elevated global temperatures also mean higher atmospheric water-holding capacity, exacerbating naturally occurring droughts, such as those triggered by El Niño. With climate change, these droughts are likely to worsen, setting in more swiftly, causing plants to wither earlier, and potentially becoming more extensive and prolonged. Dry land absorbs more solar radiation, heightening heatwaves and increasing the risk of wildfires (Lovejoy & Lee, 2005). These changes have widespread adverse consequences, affecting food and water security, human health, economies, and societies, leading to significant losses and damage to both natural ecosystems and human society (IPCC, 2022b).

Carbon dioxide (CO_2) is a significant anthropogenic greenhouse gas that has experienced an 80% increase in annual emissions between 1970 and 2004 (IPCC, 2007). Human activities, such as fossil fuel use and land-use changes, have led to substantial increases in atmospheric concentrations of CO_2, methane (CH_4), and nitrous oxide (N_2O) since 1750. By 2005, atmospheric concentrations of CO2 (379ppm) and CH4 (1774ppb) far exceeded natural levels observed over the past 650,000 years(IPCC, 2007). Fossil fuel use is the primary contributor to the rise in CO_2 concentrations, while agriculture and fossil fuel use are major factors in the increase of CH_4 (IPCC, 2007).

One of the key components of understanding climate change lies in comprehending terrestrial carbon dynamics. Terrestrial carbon plays a central role in the intricate dynamics of climate change. This significance arises from the fact that terrestrial ecosystems, encompassing forests, soils, and wetlands, are closely linked to land-based systems that absorb atmospheric carbon dioxide (CO_2). However, the escalating global temperatures are reducing the Earth's ability to absorb CO_2, allowing a larger fraction of anthropogenic emissions to linger in the atmosphere. This reduced CO_2 absorption by terrestrial and oceanic systems amplifies the impacts of climate change. The warming-induced changes in terrestrial and oceanic CO_2 absorption create a positive feedback loop, resulting in larger atmospheric CO_2 increases and worsening climate change (IPCC, 2007). The consequences of global warming are profound for terrestrial biological systems, causing shifts in seasonal events like earlier leaf unfolding, bird migration, and animal breeding, and leading to poleward and upward movements in plant and animal species' ranges. These changes exert a significant influence on biodiversity, soil health, and ecosystem stability. Climate change diminishes carbon absorption in terrestrial ecosystems by accelerating the decomposition of organic matter, increasing the release of carbon dioxide into the atmosphere. Higher temperatures, prolonged droughts, altered precipitation patterns, and more frequent wildfires stress plants and ecosystems, hindering their ability to absorb carbon dioxide through photosynthesis. Thawing permafrost in regions like tundra further releases stored carbon. These factors collectively reduce Earth's capacity to absorb carbon dioxide, allowing more to linger in the atmosphere and exacerbate climate change. Furthermore, changes in land-based carbon storage can affect greenhouse gas levels in the atmosphere, thereby influencing climate patterns, with certain ecosystems like tundra, boreal forests, mountain regions, Mediterranean-type environments, and tropical rainforests being particularly vulnerable due to their sensitivity to warming, reduced rainfall, and declining precipitation.

The Agriculture, Forestry, and Other Land Use (AFOLU) sector contributed 13-21% of global anthropogenic greenhouse gas (GHG) emissions between 2010 and 2019, with net CO_2 emissions resulting in a source of +5.9±4.1 GtCO2eq/yr (IPCC, 2022b). Deforestation accounts for 45% of AFOLU emissions. The IPCC report highlights the significant near-term mitigation potential of the Land Use, Land-Use Change, and Forestry (LULUCF) sector, emphasizing reduced deforestation as the most impactful measure, followed by carbon sequestration in agriculture and ecosystem restoration. Sustainable crop and livestock management, soil carbon sequestration, agroforestry, and biochar also contribute to emissions reduction.

Within terrestrial ecosystems, carbon is distributed among various reservoirs, namely living vegetation (both above and below ground), deceased organic matter (encompassing litter and woody debris), and soil (Penman et al., 2003). This collection of reservoirs is commonly referred to as the three terrestrial carbon pool (Penman et al., 2003). The process of photosynthesis enables vegetation, including trees, plants, and other forms of green cover, to sequester carbon dioxide from the atmosphere. The deposition of organic matter, such as deceased plant material in the form of leaves and branches, plays a significant role in augmenting soil organic matter content through the process of decomposition. The process of decomposition of woody waste, including fallen trees and branches, contributes to the accumulation of carbon within the soil over a period of time. Soil organic carbon is of considerably significance, as it is twice as large as the atmospheric carbon pool (Penman et al., 2003). Understanding the dynamics and interactions inherent in these carbon pools is of utmost importance in the scope of carbon storage management and climate change mitigation. As an illustration, the practice of afforestation, which involves the planting of trees in regions that were previously not a forest, and reforestation, which entails the replanting of trees in areas that have been deforested, serve to enhance carbon sequestration of vegetation biomass (Jia et al., 2019; Shukla et al., 2019). Furthermore, the implementation of sustainable land management methods has the potential to enhance the carbon sequestration capabilities of plants and soil, thereby assuming a crucial role in international endeavors aimed at mitigating climate change (IPCC, 2022a). Recognizing the primary influence of terrestrial carbon in driving climate change, it becomes imperative to acquire dependable estimations pertaining to the quantities of organic carbon that may be effectively sequestered by vegetation and soil.

Biodiversity encompasses the collective representation of species, habitats, and genetic diversity that exist on our planet (Lovejoy & Lee, 2005). Biodiversity plays a crucial role in maintaining the stability and resilience of ecosystems by offering vital services, including but not limited to pollination, nutrient cycling, disease regulation, and climate regulation (Brockerhoff et al., 2003). Biodiversity is continually transformed by a changing climate, but the current climate change by anthropogenic greenhouse gas emissions threatening to accelerate the loss of biodiversity already underway to other human stressors. Human development has transformed and fragmented the natural landscape on which biodiversity depended, creating altered conditions and “islands” of isolated habitats (Earn et al. 2000). The synergy between ‘normal’ stress like habitat fragmentation and altered climate- poses a new challenge to conservation (Lovejoy 1992, Hannah et al. 2002 a). With species being increasingly isolated in fragments, a rapidly changing climate will force migration to find suitable habitat in altered climatic future. Scientific concern has grown regarding the loss and fragmentation of natural ecosystems due to expanding and intensifying human land use, leading to altered ecological processes and rapid declines in biodiversity.

The relationship between biodiversity and climate change is significant. Climate change has the capacity to disturb ecosystems by its impact on temperature and precipitation patterns, hence causing changes in habitats and the distribution of species (Lovejoy & Lee, 2005; Pereira et al., 2010). These changes have a direct influence on the accessibility of resources, exert an influence on the dynamics of species interactions, and possess the potential to result in the extinction of vulnerable species (IPCC, 2022b; Lovejoy & Lee, 2005). Moreover, some extreme events such as wildfires and hurricanes can be attributed to climate change (IPCC, 2007). These events have the potential to cause significant damage to ecosystems, posing an additional threat to biodiversity. The phenomenon of climate change is anticipated to exert a significant influence on various human systems, leading to alterations that frequently manifest as heightened effects on biodiversity (Lovejoy & Lee, 2005).

Biodiversity plays an important role in the mitigation and adaptation of climate change. Diverse ecosystems possess enhanced capabilities in the sequestration of carbon dioxide from the atmosphere, serving as an effective measure in moderating the greenhouse effect. Specific species and genetic traits within biodiversity naturally adapt to climate changes, enhancing ecosystem resilience. In biodiverse tropical regions, resource-poor farmers turn to natural resources as a fallback income source. This practice may increase forest exploitation during climate-related crop failures, as natural forest species are more resilient to climate change (Lovejoy & Lee, 2005). Preservation of biodiversity is thus not only imperative for the well-being of our planet but also fundamental to our approaches in addressing the challenges presented by climate change. The intricate interplay between carbon cycles, ecosystem processes, soil health, and biodiversity conservation highlights the complexity of these issues. Measurement techniques for carbon sequestration, ecosystem processes, and biodiversity assessments are essential for accurate data collection (IPCC, 2022a; Jia et al., 2019). Additionally, understanding the relationships between carbon sequestration, native and non-native biodiversity, ecosystem services, multifunctionality, and habitat conservation is crucial for holistic climate change mitigation strategies. Conservation scientists also emphasize the significance of maintaining or restoring landscape connectivity as a crucial strategy for climate adaptation and conserving global biodiversity. This approach is essential for helping ecosystems adapt to changing climates and is widely recommended for ecological adaptation efforts (Theobald et al., 2012).

1.2 Nature Based Solutions Approach

Nature-based solutions (NbS) have emerged as a holistic approach to meeting the central challenges of mitigating and adapting to climate change, protecting biodiversity, and ensuring human well-being. Defined as solutions to societal challenges involving working with nature, NbS encompass a wide range of actions, from protecting and managing natural ecosystems to incorporating green and blue infrastructure in urban areas and applying ecosystem-based principles to agricultural systems.(Seddon et al., 2019) This comprehensive approach acknowledges the interdependencies of the 17 social, environmental, and economic goals outlined in the United Nations (UN) Sustainable Development Agenda. Despite the importance of addressing the synergies and trade-offs between these goals, practical implementation has often fallen short, hindering progress towards meeting these objectives by 2030.

The IPCC Climate Change and Land Report underscores the pivotal role of land use change mitigation, highlighting terrestrial ecosystem stewardship as crucial for limiting the global temperature rise to 1.5°C (Shukla et al., 2019). In the face of these challenges, Nature-based Solutions (NbS) offer an integrated solution (Seddon et al., 2019, 2021). NbS provides a cost-effective means to address both climate mitigation and adaptation challenges. Initiatives, such as restoring natural forests in upper catchments or planting trees in urban areas, exemplify NbS, offering dual or multiple benefits, from flood prevention and carbon sequestration to improved air quality and recreational opportunities(Cohen-Shacham et al., 2016). These approaches have been endorsed by influential bodies such as the IPBES Global Assessment and the Intergovernmental Panel on Climate Change (IPCC), highlighting their significance as key action tracks in addressing the challenges posed by climate change and biodiversity loss (Seddon et al., 2020). NbS promotes activities such as afforestation/reforestation and avoiding deforestation, which play a pivotal role in mitigating greenhouse gas emissions. Recognizing potential pitfalls of common climate change mitigation, such as monocultures and low diversity plantations that compromise long-term carbon storage and biodiversity, a well-designed NbS incorporates diverse native species and adheres to social safeguards.

NbS not only contribute significantly to mitigating climate change and enhancing climate change adaptation but also serve as effective tools in tackling the biodiversity crisis. By focusing on the protection, restoration, and sustainable management of natural ecosystems, NbS promote biodiversity conservation through habitat restoration and the creation of green corridors. These solutions also contribute to ecological connectivity, supporting wildlife migration and genetic diversity. Moreover, NbS encourages sustainable resource use, preventing overexploitation, and engages local communities in conservation efforts, fostering coexistence between humans and wildlife for mutual benefit. NbS also promote safeguarding ecosystems, promoting habitat restoration, enhancing ecological connectivity, encouraging sustainable resource use, and engaging local communities, offering holistic approaches to conserving biodiversity and fostering the resilience of natural ecosystems.

1.3 Sustainable Agricultural Landscapes: Climate Change, International Policy, Tree Planting, and Nature-Based Solutions

Climate change poses a serious threat to natural, social, and economic systems (IPCC, 2022b). Innovative strategies are needed to mitigate anthropogenic greenhouse gas emissions and manage inevitable climatic hazards, making climate change a priority for the land-based sectors

(IPCC, 2019; Millar et al., 2007). The majority of 'The National Determined Contributions' (NDCs), which are climate change initiatives commitments submitted by 193 countries to the United Nations Framework Convention on Climate Change (UNFCCC), contain country-specific commitments in the land-based sectors. These follow the explicit call of The Paris Agreement to use the full range of land-based mitigation options (United Nations Framework Convention on Climate Change, 2015). The preservation of existing forests, forest restoration, and tree planting are regarded as highly effective mitigation strategies in the land-based sectors (Bastin et al., 2019; Chazdon et al., 2016; Griscom et al., 2017; Lewis, et al., 2019). These, along with protecting the woody vegetation that already exists (Lewis, et al., 2019), will help make up for significant losses from historical deforestation emissions and sequester current carbon emissions (Grassi et al., 2017). Global partnerships, such as Forest Landscape Restoration (The Bonn Challenge, 2020) and the One Trillion Trees initiative by the World Economic Forum (World Economic Forum initiative, 2020), have initiated large-scale initiatives to plant trees around the world. The preservation of existing forests, forest restoration, and tree plantings will also promote "Nature-based Solutions" (NbS) for combating climate change (Griscom et al., 2017). NbS have been defined as measures to maintain, sustainably manage, and restore natural or modified ecosystems, which effectively and adaptively address societal concerns, while simultaneously providing benefits for human welfare and biodiversity (Cohen-Shacham et al., 2016). These efforts can decrease floods and erosion, enhance water and soil quality, increase biodiversity and landscape amenity, and produce lumber and other fibers (e.g. Lin et al., 2013; Mace et al., 2012). As a result of climate change, these environmental benefits are anticipated to become even more important in the future (Brown & Brown, 2018).

Despite being marketed as forest restoration, the majority of the current projects feature extensive tree planting, with the majority of the species planted being commercial species such as *Eucalyptus* spp. for paper, or *Hevea braziliensis* for rubber (Lewis , et al., 2019). Although extensive tree planting can have a variety of benefits, there are currently concerns regarding the unfavorable outcomes of such activities. Some countries have experienced unfavorable results from the successful large scale tree plantings, for example, in China, expanding forest cover has increased environmental degradation (Cao et al., 2020) and in Japan there was an oversupply of timber from afforestation (Knight, 1997). Large-scale tree planting initiatives may result in the loss of other ecosystems crucial for conservation; transform diverse native vegetation into monoculture plantations; have a detrimental impact on biodiversity and ecosystem functions on grasslands, savannas, and open-canopy woodlands; alter natural landscape configuration; initiate the introduction or increased abundance of invasive plants and pest animals; decrease the amount of water available; and increase the risk of fire from larger and more connected forested areas when planting fast-growing species in high-density stands (Fleischman et al., 2020; Holl & Brancalion, 2020; Lewis, et al., 2019; Veldman et al., 2019; Zastrow, 2019). Although there has been research in this area (e.g. Bremer & Farley, 2010; Di Sacco et al., 2021; Lindenmayer et al., 2012), more investigation is required to determine which actions will be more feasible for landowners in achieving the goal of mitigating climate change, while avoiding the unfavorable results associated with large-scale-tree planting.

The integration of woody vegetation into agricultural landscapes as a management method for a sustainable landscape (Harvey et al., 2008) is gaining popularity in response to global campaigns to restore and plant trees. The presence of woody vegetation within agricultural landscapes provides an opportunity for climate change adaptation and mitigation, as well as for the development of multifunctional landscapes (Nair & Garrity, 2012). A 'multifunctional landscape' is one that simultaneously includes a diversity of ecosystem functions (Manning et al., 2018). For example, woody vegetation within agricultural landscapes is acknowledged for its role in both sequestering carbon and preserving biodiversity, thus providing multiple ecosystem functions (Nair & Kumar, 2011; Norton et al., 2020; Tscharntke et al., 2011). With rising initiatives to integrate trees into agricultural landscapes, there is a growing need for information on the carbon stocks that are held in landscapes outside of continuous forests (Schnell et al., 2015). Numerous studies have been published that provide a summary of the literature on the ecosystem functions provided by woody vegetation in agricultural landscapes, e.g., a global study of biomass on agricultural land (Zomer et al., 2016) and an investigation into the multifunctionality of agricultural land (Case et al., 2020c; Renting et al., 2009). However, further work is required, such as determining how much woody vegetation exists within agricultural landscapes, measuring its area, structure, composition and functions, and in developing strategies to preserve it and improve its multifunctionality.

Despite its widespread distribution, existing woody vegetation in agricultural landscapes is a frequently overlooked source of carbon, and little is known about the carbon stocks in this system or their capacity to store carbon (Rosenstock et al., 2016; Schnell et al., 2015; Zomer et al., 2016). A relatively low carbon stock value (5 t C ha^{-1}) in agricultural land was implemented globally as a general value or tier 1 estimates that was used on estimating biomass carbon stock for countries' national carbon inventory (Intergovernmental Panel on Climate Change, 2014). In addition, attention was largely focused on the carbon that newly planted trees would sequester, rather than how much carbon already existed in the project area. Furthermore, there was insufficient knowledge on the multifaceted nature of existing woody vegetation. More research is needed on accurately estimating the carbon stocks and multifunctionality of the current land cover on agricultural land, in addition to quantifying the trade-offs and co-benefits of other activities beyond carbon sequestration. This understanding will increase the potential for including agricultural landscapes in combating global climate change (Shukla et al., 2019).

There is a growing body of literature that aims to estimate the impacts of various future tree planting scenarios on ecosystem functions and land use change, such as Doelman et al., (2020) on the impacts of afforestation related to food security, Berthrong et al. (2009) on the effect of afforestation on soil carbon and Masera et al., (2003) on the modelling of the carbon sequestered by afforestation projects. These studies have produced alternative predictions for the future potential of global forest cover and ecosystem functions that could result from a global tree-planting initiative. However, because agricultural areas are often excluded from research on the effects of global tree planting, existing woody vegetation in the agricultural landscape, as well as its trade-offs with its ecosystem functions, have been relatively neglected (Doelman et al., 2020). The projected forest expansion through tree planting, such as Bastin et al. (2019), was also criticized for endangering existing native habitats (Lewis, et al., 2019). Moreover, the majority of land use change models either (1) ignored the future spatial configuration that would result from the additional trees and only estimated how much future scenarios would affect carbon storage; or (2) separated production zones and conservation areas, thereby ignoring interactions between the components that can be included to create a landscape that is less fragmented and more spatially connected, and (3) did not adequately consider multifunctionality.

There has been little research to date that has attempted to combine the spatially-strategic targeting of tree planting and conservation for climate mitigation with other, more biodiversity-oriented, objectives, such as the promotion of green corridors or the enhancement of landscape connectivity. One recent effort to combine carbon storage and biodiversity conservation objectives into a more spatially explicit framework was research conducted in the Republic of Korea (Choi et al., 2022a) that developed and compared national-scale afforestation scenarios to maximize biodiversity and carbon objectives. This lack of research effort is despite the tendency of governments to develop land use policy interventions that are justified on the basis of a variety of objectives including to combine carbon farming and conservation within a more spatially explicit concept of multifunctional landscapes (Gimona & Van Der Horst, 2007). There are now significant opportunities to enhance landscape management by researching how future scenarios of planting new trees into agricultural landscapes affects the existing spatial variability across these landscapes. This is important because each landscape will provide a different set of outcomes, depending on the present spatial configuration and function of vegetation cover.

1.4 Aotearoa New Zealand as a case study

Aotearoa New Zealand's commitment to addressing climate change began with its ratification of the United Nations Framework Convention on Climate Change's (UNFCCC) Kyoto Protocol in

2002, establishing crucial guidelines for greenhouse gas mitigation and climate resilience policies (Ministry for the Environment, 2022c). Building on this foundation, New Zealand further pledged to reduce net emissions by 50% below 2005 levels by 2030 under the Paris Agreement in 2020 (UNFCCC, 2020). The nation took significant legislative steps with the enactment of the Climate Change Response (Zero Carbon) Act in 2019, which set ambitious domestic targets. These targets included achieving net zero emissions for all greenhouse gases except biogenic methane by 2050 (New Zealand Government, 2019). To ensure effective implementation and continuous progress towards these goals, New Zealand established the Climate Change Commission in 2019 (He Pou a Rangi, 2019), reflecting the country's commitment in combating climate change (New Zealand Government, 2019).

In May 2022, New Zealand introduced its first emissions reduction plan, as outlined by the Ministry for the Environment (2022d). This comprehensive plan showcased climate change mitigation efforts spanning seven key sectors: transport, energy and industry, building and construction, agriculture, forestry, waste, and fluorinated gases. Concurrently, within the Climate Change Act, the Climate Change Commission proposed amendments to the emissions pricing mechanism, specifically focusing on the Emissions Trading Scheme (ETS), a system of carbon credits. As per this proposal, agricultural emissions are slated to be priced by 1 January 2025. This development signifies an expanded need for emissions tracking, extending beyond the national level of greenhouse gases inventory report to the UNFCCC. The plan necessitates emissions to be documented at the producer and farm levels. To facilitate this transition, organizations like He Waka Eke Noa or Primary Sector Climate Action Partnership, in collaboration with the Ministry of Primary Industry, have been actively supporting producers (Ministry for Primary Industries (MPI), 2020). Through initiatives such as 'know your numbers', they empower producers with the knowledge to assess their emissions and formulate effective strategies for measurement and management at the farm level (Ministry for Primary Industries, 2020; Ministry for the Environment, 2022a). The Climate Change Response Act 2002 has legislated milestones for 100 percent of farms to measure and document their annual greenhouse gas emissions by 31 December 2022 and 100 percent of farms to have a farm plan to measure and manage their emissions by 1 January 2025 (Ministry for the Environment, 2022a).

In the forestry sector, New Zealand aims to 1) grow the forestry and wood processing industry to deliver more value from low-carbon products, 2) support landowners and others to undertake afforestation, particularly for erodible land, through incentives and reduced costs, and 3) maintain existing forests by exploring options to reduce deforestation and encourage forest management practices that increase carbon stocks in pre-1990 forests. Policy goals to encourage forest establishment and discourage deforestation have been in place since 2013. ETS and several tree planting incentives were set up to promote the establishment of forests and discourage deforestation (Ministry for the Environment, 2022c). Furthermore, the incentives for planting trees aim to increase afforestation rates overall, particularly those for commercial use, which have been steadily decreasing from 1996 to 2008 (Ministry for the Environment, 2022d). In 2018, Aotearoa New Zealand's tree planting incentives were combined into a single, multipurpose program named the One Billion Trees (1BT) Program, with the goal of reaching one billion trees planted by 2028 (Ministry for Primary Industries, 2020c; Ministry for the Environment, 2022a). The 1BT project submission period ended in 2021, and tree planting will continue until 2028 (Ministry for Primary Industries, 2022).

In Aotearoa New Zealand's agricultural and forestry landscapes, native vegetation covers a sizable portion of the land: 28.75 percent of an 11,490-hectare area (Pannell et al., 2021). It's important to note that the land, currently utilized for sheep and beef cattle farming, was originally covered by indigenous vegetation before its conversion. Before the arrival of the Polynesian people, Aotearoa New Zealand was predominantly covered by forests, with an estimated original forested area of 97,200 square miles, likely enveloping nearly the entire country (Cumberland et al., 1941). A notable example is the South Island high country, which is presently characterized by tussock-covered landscapes dedicated to sheep and cattle farming. Historically, this area was covered by indigenous vegetation adapted to fluctuating and often dry climates (M. McGlone, 2004; 2010). Over the past 2.5 million years, the region has undergone various environmental conditions, ranging from virtually treeless grasslands during glacial to forested interglacial periods. The high country, particularly during the last 12,000 years, has experienced different vegetation covers in its interglacial state. The initial human settlement in the 13th century brought about widespread fires, resulting in the destruction of 75% of the forest and tall scrub cover in the eastern South Island. European further modified the landscape by introducing fire cycles, leading to alterations in native plant communities. The region's biodiversity and vegetation structure have been significantly impacted by factors such as high stocking rates, rabbit outbreaks, and the introduction of invasive species. Similarly, the historical lowland vegetation was more extensive, exemplified in regions like the North Auckland peninsula with its high average temperatures and infrequent frost. This area featured significant trees such as taraire (*Beilschmiedia taraire*) and the highly prized kauri (*Agathis australis*), both confined to the warmer side of the 550 annual isotherm. The expansive bushland of the North Island, especially in dissected Tertiary regions like Taranaki, the King Country, and inner Wanganaui, comprised a diverse mix of podocarp-dicotyledonous forest, representing the broadest expanse of Aotearoa's forest. This encompassed various tall timber dominants, including podocarps like rimu (*Dacrydium cupressinum*), totara (Podocarpus totara, found in drier habitats), matai (*P. spicatus*), miro (*P. ferrugineus*), and kahikatea (*P. dacrydioides*, in swamp-forest habitats). The forest association, which included more shade-tolerant broad-leaved species like tawa (*Beilschmiedia tawa*), the partly epiphytic kamahi

(*Weinmannia racemosa*), and rata (*Metrosideros robusta*), possibly indicated a climax vegetation state (Cumberland et al., 1941).

In Aotearoa New Zealand, where livestock farming has historically been one of the main forms of agriculture, sheep and beef cattle farms that occupy about 33 % of the land area of New Zealand (Morris, 2013) will likely play a bigger role in mitigating the environmental effects of climate change and the growing emphasis on sustainable agriculture. Approximately 40% of New Zealand's land area is dedicated to sheep and beef farming, highlighting the significance of safeguarding remnant forest patches within these agricultural landscapes for biodiversity conservation (Norton et al., 2019). Currently, lowland native vegetation types often persist as remnants on agricultural land, holding substantial ecological value (Dodd et al., 2011). Woody vegetation is a common part of pastoral farming in Aotearoa New Zealand, especially on sheep and beef cattle farms, which occur as shelterbelts, shade trees, widely spaced trees over pasture, riparian planting, remnant native forests, or more densely planted plots (Mead, 1995; Norton & Reid, 2013; Pannell et al., 2021). Woody cover serves a variety of ecosystem functions, including enhancing farming productivity and preserving biodiversity (Dominati et al., 2019a; Maseyk et al., 2019). However, the co-occurrence of invasive plant species favored by fire alongside native species further complicates the long-term composition and trajectory of ecosystems (Perry et al., 2014).

The ecosystems characterizing sheep and beef cattle farms in Aotearoa New Zealand are subject to the consequences of agricultural intensification, resulting in the loss and degradation of natural habitats (Zhang et al., 2020). This alteration in resource availability for native animals has led to habitat destruction and fragmentation, where once-connected habitats are now divided into smaller, isolated fragments surrounded by human-transformed land cover. The resulting landscape structure poses challenges for the movement and dispersal of organisms, restricting their ability to recolonize and occupy new habitat patches. Agroecosystems in these farms often harbor generalist species that adapt to fragmented habitats, tolerating human and livestock disturbances.

Nature-Based Solutions (NbS), which involve strategic tree planting and restoration, emerge as promising strategies to address carbon sequestration, climate change, and the biodiversity crises. Practices such as the restoration of native vegetation and the strategic implementation of tree planting can significantly contribute to enhancing habitat connectivity and providing corridors for movement across fragmented landscapes. Moreover, the deliberate placement of trees within these farms can serve multiple objectives, including carbon sequestration for climate change mitigation, promoting biodiversity by creating green corridors, enhancing the ecosystem functions of woody vegetation, and providing essential habitat for native species. Landscape connectivity, a crucial aspect for facilitating movement between locations, becomes a pivotal concept in conservation management. In the context of New Zealand's extensive sheep and beef farming, where remnant forest patches are common, the strategic planting of trees as part of NbS becomes imperative. Addressing knowledge gaps on how native species interact with fragmented landscapes through NbS is essential for enhancing connectivity and fostering effective conservation measures in these agroecosystems.

One of the main focuses of the 1BT Program was the use of marginal agricultural land for afforestation. This was in recognition that agriculture, including sheep and beef cattle farms, although playing a significant role in the economy, can also contribute to environmental degradation such as biodiversity loss, increased erosion, and increased greenhouse gas emissions (Te Uru Rākau, 2018a). Therefore, by incentivizing farmers to plant trees on their farms, the 1BT Program was designed as an alternative use of lower productivity areas on agricultural land, which would lessen the negative environmental impacts of farming practices. In addition, landowners can diversify their farming systems to include carbon farming through the ETS, where forest owners can earn carbon credits by maintaining their forest cover (Office of the Minister of Forestry, 2017). This would encourage improved land management, increase farmers' incomes, and contribute to climate change mitigation by promoting tree planting and discouraging deforestation (Ministry for Primary Industries, 2020c).

Although it may be beneficial for carbon and biodiversity, the combination of tree planting and emission trading schemes has the potential to enhance the risk of loss of existing woody vegetation on sheep and beef cattle farms, as it is threated to be replaced with newly planted forests (Norton et al., 2020). Due to the lower opportunity costs associated with sheep and beef cattle farmland, compared to other pastoral systems such as dairy farming, the majority of future afforestation by tree planting incentives, such as the 1BT Program, are anticipated to occur in these areas (West, et al., 2020). Given that woody vegetation in agricultural land is not yet fully taken into consideration as part of the carbon credit policy, and could potentially be replaced by new plantation areas, such incentives could potentially create challenges for the preservation of ecologically important vegetation cover (Norton et al., 2020). To achieve climate mitigation and biodiverse, multifunctional landscapes in agricultural areas, it is crucial to understand the role of the existing woody vegetation in sheep and beef cattle farms in the provision of carbon stocks and other ecosystem functions. This has implications for the potential of agricultural areas to bring positive benefits via schemes such as the 1BT Program.

The government has also suggested 2.8 million hectares of farmland out of 11.5 million hectares could be suitable for afforestation, which can be part of the permanent forest under the ETS to mitigate climate change (Nash & Shaw, 2022). In addition to the 1BT and ETS programs that target agricultural areas, there is a forthcoming policy on National Policy Statement for Indigenous Biodiversity (NPSIB) that contains objectives and policies to identify, protect, manage and restore indigenous biodiversity, that will map important native habitats, including those in farming areas (Ministry for the Environment, 2022b).

Questions remain regarding how we can minimize environmental impact—or even contribute to a more sustainable and multifunctional environment—while agricultural production is maintained in these areas. How can national tree-planting policies benefit sheep and beef cattle farmers in their efforts to increase their contribution to climate change mitigation and biodiversity conservation? How important are sheep and beef cattle farms for sequestering carbon, and can they also support biodiversity and other ecosystem functions? Can sheep and beef cattle farms become a valuable resource in our attempts to establish multifunctional landscapes? How can tree planting improve multifunctionality of the sheep and beef cattle landscapes?

This thesis explores the synergies between climate mitigation, biodiversity conservation, and ecosystem functions of woody vegetation on Aotearoa New Zealand sheep and beef cattle farms. It identifies instances where trade-offs exist between carbon storage, biodiversity, and other ecosystem functions, based on land management strategies associated with existing woody vegetation patches on farms. In addition, the research examines how future native woody plant restoration and revegetation on sheep and beef cattle farms can be improved to positively contribute to climate change mitigation and biodiversity preservation within agricultural landscapes. This analysis will assist decision-makers in identifying management practices available for achieving multiple benefits of large-scale tree planting policies in Aotearoa New Zealand and elsewhere. Threats from continuous degradation, such as erosion, conversion to monocultural plantations, and conversion to alternative land uses, are, in this context, major obstacles to maximizing the contributions of agricultural landscapes. Considering Aotearoa New Zealand has high native species endemicity, many unique ecosystems, and the primary agricultural industry is the production of livestock on exotic-dominated production pastures, it serves as an ideal case-study location for answering these research questions.

1.5 Thesis structure, goals, and significance

With regard to Aotearoa New Zealand's recent tree planting policies, the primary goals of this research were to: (1) evaluate how existing tree planting initiatives can contribute to biodiversity and carbon objectives, (2) examine how existing woody vegetation in sheep and beef cattle farms can contribute to climate change mitigation, and (3) quantify multifunctional landscape contributions under possible future land-use change scenarios where existing woody vegetation is protected and new trees are added. The potential of existing woody vegetation to contribute to a multifunctional landscape was quantified in three case-study sheep and beef cattle farms, particularly in relation to carbon storage and co-benefits for other ecosystem functions and biodiversity conservation. The data collection methodology used in this thesis was selected because it has previously been used in Aotearoa New Zealand to estimate the carbon stocks within different land uses; this is as part of the nation's obligation to report greenhouse gas emissions to the United Nations Framework Convention on Climate Change (UNFCCC). The specific thesis objectives are given below.

Chapter 2 investigates the gaps and opportunities associated with the One Billion Trees initiative (1BT) and describes the potential of the Program to achieve co-benefits for carbon and biodiversity. At the conclusion of the chapter, I offer recommendations for enhancing the current tree-planting Program to increase biodiversity and reduce carbon emissions. Several findings from this policy study directly influence how the land-cover change scenarios were designed to optimize the co-benefits of carbon and biodiversity (Chapter 4).

Chapter 3 estimates woody plant (including tree ferns) community structure, carbon stocks per hectare, and ecosystem functions. Multifunctionality was quantified and used to evaluate trade-offs in ecosystem functions among the different woody community types. This was achieved by: (1) mapping the composition of woody plant species across three farms using vegetation plots that were then classified into different plant community types, (2) comparing the characteristics of each plant community type in terms of relative abundance, diversity, and stem size, (3) estimating the mean and variation of carbon stocks per hectare among different on-farm community types and identifies plant community characteristics that contribute to a higher total carbon stock, and (4) estimating and comparing the relative potential multifunctionality of each plant community type. Field data on soil, biomass carbon stocks and vegetation inventories were used in this chapter. The study employed these measures to look at the patterns of variation among the various types of woody vegetation plant communities on the three case-study farms. The results of the carbon stock estimation from this analysis were used to calculate the carbon stocks used in the next chapter (Chapter 4).

Chapter 4 simulates land-cover change patterns in three agricultural landscapes under future native restoration and revegetation scenarios and to quantify changes in carbon storage, biodiversity, fragmentation, and connectivity, as well as observe trade-offs caused by predicted future land-cover change, under different native plant restoration and revegetation scenarios; based on the recommendations of Chapter 2and the results of Chapter 3This was achieved by: (1) developing land-cover change scenarios for the restoration of shrubland and exotic woodlots, as well as the revegetation of existing bare gullies with native trees, in addition to protecting the existing remnant forest and mixed-native woodlots, (2) running a series of Geographic Information System (GIS)-based model to assess land-cover change and its effects on habitat amount, carbon provision, biodiversity potential, landscape connectivity, and fragmentation, (3) evaluating trade-offs in various scenarios by comparing the changes of variables. This chapter's results illustrate co-benefits and trade-offs between key results variables, as well as how to analyze multifunctionality based on the diverse outcomes of each scenario.

Chapter 5 concludes the thesis with a synthetic discussion of the significance and implications of the results emerging from the previous three chapters. Some recommendations for possible directions of future research on woody vegetation in farms and its potential carbon, biodiversity, and multifunctionality co-benefits are also given.

Chapter 2 Large-scale tree planting initiatives as an opportunity to derive carbon and biodiversity co-benefits: a case study from Aotearoa New Zealand

Large-scale national and international tree planting initiatives are of increasing interest across many areas of the globe, especially as a tool for enhancing sustainability while tackling climate change (IPCC, 2019). This interest is reflected in international climate and sustainability policies, which encourage afforestation and reforestation activities as part of climate change mitigation and adaptation efforts (UNDP, 2019; UNFCCC, 2006, 2015). These are recognized among the most impactful kinds of ‘natural climate solutions’, a concept which refers to ‘conservation, restoration, and improved land management actions that increase carbon storage and/or avoid greenhouse gas emissions across global forests, wetlands, grasslands, and agricultural lands’ (Griscom et al., 2017). Tree planting objectives have also been pursued through international commitments such as the Bonn Challenge, backed by 48 nations, that aims to restore 350 million hectares of degraded and deforested land around the world by 2030 (IUCN, 2011). There are also three separate campaigns with the goal of planting one trillion trees; the World Economic Forum’s One Trillion Trees Program (World Economic Forum initiative, 2020), the Trillion Trees joint venture between BirdLife International, Wildlife Conservation Society, and the Worldwide Fund for Nature (www.trilliontrees.org) and the Plant-for-the-Planet Trillion Trees Campaign (www.plant-for-planet.org).

Such commitments have led to a range of large-scale national tree planting movements by countries around the world. For example, in 2014, Pakistan launched the Billion Tree Tsunami Program, which was achieved in 2017, adding 350 thousand hectares of trees to the country (Kamal et al., 2018) . Ethiopia’s Green Legacy Initiatives planted 350 million trees in a single day in July 2019 (UNEP, 2019). Planting trees has also become a core part of China’s efforts to tackle climate change, with over 4 million hectares planted each year (Xu, 2011). However, these efforts have also caused the displacement of native trees and decreased the area of native forests in China by 6.6 %, as a result of increasing tree cover and conversion of this land to single-species plantations for production (Hua et al., 2018; Zhai et al., 2017), with negative impacts on water balance (Ge et al., 2019; Zastrow, 2019b). Large-scale afforestation of fast-growing species has also been shown to threaten water systems, wildlife, and native habitats in Japan, the UK, and Brazil (Abreu et al., 2017; Bunce et al., 2014). Thus, while tree planting Programs have noble goals to mitigate the impacts of climate change, one consequence may be perverse outcomes for native forests and biodiversity.

The contested nature of forests within climate policy is increasingly evident in the academic literature. Bastin et al. (2019) declared that tree planting initiatives at a global scale could be one of the most effective strategies to combat climate change, largely through the rationale of rapid biomass accumulation. However, Veldman et al. (2019) were quick to critique the study for overestimating the carbon sequestration potential by nearly five times because they miscalculated soil carbon, did not subtract existing carbon vegetation from their estimates, and did not account for the warming effect of trees due to decreased albedo (Friedlingstein et al., 2019; Li et al., 2019; Mykleby et al., 2017), and for suggesting trees should be planted on grasslands and savannas, which would replace these diverse biotic communities with lower-diversity forests (Strömberg, 2011; Veldman et al., 2015). There are also increasing critiques of tree planting approaches for carbon sequestration because these approaches tend to prioritize creating tree plantations over native forests (Seddon et al., 2019), while neglecting the potential for biodiversity conservation (Lindenmayer et al. 2012), the carbon value of old-growth forests (Luyssaert et al., 2018), and the uncertainty over future carbon sink strength (Duffy et al., 2021; Hubau et al., 2020). A 'carbon-centric' approach to forestry is overtaking an appreciation of the diverse ecological and livelihood benefits that multifunctional forest landscapes can provide (Ojha et al., 2019). It is becoming clear that we must not only think about the number of trees being planted, but also what species and genetic lineages trees are being planted, where, and for what purpose? The survival rates of the trees planted must also be accounted for (e.g., Kodikara et al., 2017).

Concerns about the perverse outcomes of large-scale tree planting initiatives have started to emerge in the literature (e.g., Di Sacco et al., 2021; Fleischman et al., 2020; Holl & Brancalion, 2020). Here this study aims to contribute to this growing body of knowledge while also providing contextual insights around the incentives and down-stream impacts of a tree-planting initiative in a temperate forest system. This study draws upon Aotearoa New Zealand's One Billion Tree (1BT) Program as a case study to evaluate the potential of this initiative to contribute to the country's carbon sequestration target. This study also outlines how biodiversity might be negatively impacted by the 1BT Program in its current form and provide recommendations for how the 1BT strategy could be reimagined to achieve multiple wins for biodiversity and carbon in Aotearoa New Zealand. As this Program is still in an early phase, and continues to adapt as it develops, I hope the recommendations in this article provide important insights for 1BT and other large-scale tree-planting initiatives around the world going forward.

2.1 The One Billion Trees Program: an upgraded package of tree planting

In 2017, the Aotearoa New Zealand government launched its 1BT Program to increase current rates of tree planting across the country and reach at least one billion trees by 2028 (Te Uru Rākau, 2018a). The goal of this Program is to reduce the effects of climate change through increasing tree planting rates to deliver 'improved social, environmental, and economic outcomes for New Zealand, [and] play a significant role in moving towards a low emissions economy' (Te Uru Rākau, 2020). 1BT is strategically designed to optimize land productivity by increasing the number of trees planted in productive landscapes across the country and is managed by Te Uru Rākau (TUR), the recently established government body servicing Aotearoa New Zealand forestry.

1BT has been designed to be a catch-all replacement for a range of previous afforestation grants (Office of the Minister of Forestry, 2017). To achieve the target of one billion trees by 2028, the 1BT Program aims to plant 500 million exotic pine trees (*Pinus radiata D. Don*) through existing commercial forestry projects (Ministry for Primary Industries, 2020a), and an additional mix of 500 million exotic (mainly pine) and native trees as new forests on public and private land through Direct Grants and Partnership Funding to landowners. Direct Grants provide funding through four categories: native planting (NZ$4000 per hectare to establish new areas of native forest), native reversion (NZ$1000 per hectare to retire land and encourage natural regeneration of native forest), native mānuka/kānuka planting (NZ$1800 per hectare to plant blocks of the native pioneers *Leptospermum scoparium* (mānuka) or *Kunzea* spp. (kānuka) for erosion control, as a nurse crop for native forest, or as a native crop for pollinators and honey production), and exotic planting (NZ$1500 per hectare to plant a non-native single species tree plantation) (Ministry of Primary Industries, 2019) . Partnership Funding is aimed towards projects that can contribute towards the long-term success of tree planting in New Zealand and prioritizes projects focusing on improving forestry labour and workforce development, landowner knowledge about planting trees and forests as a sustainable land use, catchment-based or landscape-scale tree planting, science and innovation, or seedling and nursery production (Ministry of Primary Industries, 2019) . By 9 April 2021, 1BT had planted 258,686,000 trees, of which only 12.59 % were of native species that were directly funded by the Program (Ministry for Primary Industries, 2022b) .

2.2 The carbon potential of One Billion Trees

Aotearoa New Zealand's current pledge to the United Nations is to reduce greenhouse gas emissions by 30 % below 2005 levels by 2030 (UNFCCC, 2020) . The main policy mechanism to achieve this is the Emissions Trading Scheme (ETS) whereby companies can offset their emissions by purchasing carbon credits on the ETS market from eligible landowners who are planting trees (Ministry for the Environment, 2020b). One of the primary goals of the 1BT Program is to increase the number of trees planted in Aotearoa New Zealand and enhance the country's carbon sequestration efforts, with new trees expected to sequester an additional 384 million tons of carbon dioxide (CO_2) over 20 years. This was estimated from pine plantation sequestering approximately 600 tons CO_2 per hectare per year, and native forest sequestering up to 1000 CO_2 per hectare per year (Gibson, 2019; Te Uru Rākau, 2018b, 2020) (See *Corrigendum* in Appendix C). Carbon stock in New Zealand's native forest was estimated at around 1,697.3 tons CO_2 per hectare (Holdaway et al., 2014) and up to 1,185.43 tons CO_2 per hectare for pine plantation (Jayawickrama, 2001). In doing so, 1BT is expected to make a significant contribution to the ETS and the country's international climate commitment.

However, it is important to consider which tree species are incentivised by 1BT, their respective contributions to carbon sequestration, and the downstream impacts of this in terms of how tree planting intiatives in Aotearoa New Zealand are realised. Half of the trees 1BT aims to plant are part of planned commercial forestry efforts (Ministry for Primary Industries, 2020a; Office of the Minister of Forestry, 2017), and would have probably been planted under a business-as-usual scenario with or without 1BT. These trees are predominantly exotic species, with *P. radiata* making up 90 % of all New Zealand planted forests, typically planted as single-species plantation (NZFOA, 2019). With around half of this target allocated as re-planting on already established plantations, only approximately 50 % of the 384 million tons of sequestered carbon will be additional compared to a no-intervention scenario. We must also adjust these values to consider the number of trees lost per year due to changing land-use. For example, in 2016, the area of forest in New Zealand that was deforested was nearly equal to the number of new forests planted in the same year (Ministry for the Environment, 2018); 4.2 % of forest that was planted after 1989 has already been lost due to deforestation (Ministry for the Environment, 2018). When considering the country's climate contributions, it is important not to just count the number of trees in the ground, but account for net change in trees and additionality.

The relative carbon value of exotic species over native forest must also be considered. While fast-growing single-species plantations sequester carbon rapidly, their reduced tree longevity and regular harvesting means that, in the longer term, they are likely to sequester less carbon than native forest, which tends to be made up of longer-living slower-growing trees (Büntgen et al., 2019); much of the stored carbon in exotic plantations returns to the atmosphere when they are harvested through the removal of the trees, the release of carbon from the soil, and the decomposition of plantation waste and products (Lewis, et al., 2019). While both native and exotic trees may develop additional carbon benefits where young plants grow in the understorey (Forbes et al., 2015), these benefits will be removed through regular harvesting of plantation forests instead of allowing the understorey to thrive and develop into the complex forest structure typically associated with native forests. Berthrong et al. (2009) showed that conversion of non-forested lands to plantation forest can cause a 6.7 % decrease in soil carbon and other nutrients due to continuous harvest, including the decline of carbon in soil following afforestation in New Zealand (Scott et al., 2006).

Despite the carbon benefits of native forest the use of exotics tends to dominate afforestation and reforestation schemes in New Zealand. This is mostly due to two reasons. The first is that the historical focus of research on forestry means more is known about planting and growing species of commercial interest which has created path-dependency for exotic species such as *Pinus radiata* (Ministry for Primary Industries, 2020b; NZFOA, 2019). More research is needed to understand how we can best grow native trees and forests. The second reason reflects limitations of the ETS and how it operates within the New Zealand national carbon accounting system (Cameron, 2011; Leining et al., 2019; Ministry for the Environment, 2020b). This accounting system incentivises quick gains which tends towards planting commercial single-species plantations.

There is also a policy gap around old-growth native forests in Aotearoa New Zealand policy and similar policies around the world. Protection of existing forest with high carbon stocks has been identified as a priority (Canadell & Raupach, 2008; S. L. Lewis, Wheeler, Mitschard, et al., 2019; Luyssaert et al., 2018; Shukla et al., 2019). Native restoration of existing but degraded forests has been identified as the second priority as such an approach is considered relatively low risk and low cost compared to planting new trees. Yet, 1BT and previous forestry grants have been directed toward the establishment of new forests. Moreover, the ETS was designed as a policy to incentivize afforestation and disincentivize deforestation while regulating old-growth forest land with primarily exotic species but makes no provisions for protecting old-growth native forest (native forests that existed before 1 January 1990). Much of this forest belongs within the national conservation estate, but public funding for Department of Conservation is constrained, and protection of pre-1990 indigenous forest outside the conservation estate has few if any avenues for funding. A recent study by Goldstein et al. (2020) identified many of Aotearoa New Zealand’s old-growth native forests as being some of the world’s densest carbon stores, made up of ‘irrecoverable carbon’ that must be protected if we are to achieve our climate goals. Irrecoverable carbon is that which could not be recovered on timescales relevant for avoiding dangerous climate impact.

A longer-term commitment between landowners and the Aotearoa New Zealand government is needed to ensure that existing forests are not removed or replaced by new trees and will continue to sequester and store carbon (Funk et al., 2014). Enhancing sequestration by planting trees coupled with strong forest management is part of a global strategy for tackling climate change (Bauhus et al., 2010; Lindenmayer et al., 2012a). Protecting and restoring native ecosystems and existing habitats should be a top priority (IPCC 2019). It is more important to protect and restore existing forest than to plant new forest, and it is much easier and more cost-effective to do so (Moreno-Mateos et al., 2017).

2.3 The biodiversity potential of One Billion Trees

Well-designed tree planting initiatives can provide numerous environmental co-benefits while achieving their carbon goals, including improved land productivity, reduced erosion, improved water quality, improved soil quality, and the provision of important habitats for a range of native species (Soto-Navarro et al., 2020; Te Uru Rākau, 2020) . However, there have also been numerous examples where such initiatives can also deliver unintended negative consequences for biodiversity, including displacement of native trees, deforestation of native forests, and conversion of native ecosystems to single-species plantations (Ge et al., 2019; Hua et al., 2018; Zastrow, 2019a). Such approaches neglect the potential benefits of biodiversity conservation (Lindenmayer et al. 2012) while risking negative impacts on native species, habitats, soil, and water systems (Abreu et al., 2017; Bunce et al., 2014; Sloan et al., 2018) . Similar concerns have also been raised regarding 1BT in Aotearoa New Zealand, with exotic trees still tending to dominate the majority of planting efforts (Ministry for Primary Industries, 2020c), evidence of land conversion to plantation forests (Clifton, 2019; Hyndman, 2019), and inadequate consideration of impacts on biodiversity and native species (Church, 2019; Eder, 2019; Gibson, 2019; Salmond, 2019). Conversion of old growth forest to alternative land uses was undertaken at a rate of around 13 % per year of the total area deforested between 2008 and 2019 (Ministry for the Environment, 2018); 12,500 hectares of native regenerating forest, and 2,200 hectares of native old-growth forest have been converted to non-native cover, predominantly to exotic plantations (Allen et al., 2013a).

Native trees and forests have higher native biodiversity and ecosystem benefits in comparison to exotic plantations (Braun et al., 2017; Deconchat et al., 2009; Watson et al., 2018). Native forest does not only provide longer-term opportunities as a carbon sink (S. L. Lewis, Wheeler, Mitschard, et al., 2019), but it also provides a wide range of other ecosystem and biodiversity benefits (Brockerhoff et al., 2003; J. M. Hall et al., 2012; IPCC, 2019). Native patches that are located in a fragmented landscape can also provide additional conservation benefits by providing buffer zones at forest margins to improve connectivity between forest patches, allow wildlife movements and exchange of genetic material among fragmented sub-populations, sustain species that live in small forest remnants, and enable native species to recolonize remnants (Case et al., 2020a; Lamb, 2011). Therefore, on balance, structurally complex native forest ecosystems left to develop over the long term will return significant biodiversity and carbon co-benefits (Carswell et al., 2012). With planting Programs, however, it is challenging to create large tracts of native forests on pastoral soils (Forbes et al., 2020). The function of native organisms in rural landscapes clearly depends on the provision of certain thresholds of available resources, habitats, and associated intact food webs (Fahrig et al., 2011a). For example, Arroyo-Rodríguez et al. (2020) suggest that agricultural landscapes configured to support native biodiversity should contain at least 40 % forest cover, composed of roughly 10 % very large patches and 30 % smaller patches and semi-natural treed elements dispersed throughout based on their work in the Brazilian Atlantic forest. Thus, it is important to account for the ecosystem function and biodiversity potential of tree planting initiatives such as 1BT within a landscape ecology context (e.g., Landis, 2017) to maximize potential co-benefits.

It is also critical to consider the biodiversity impacts of tree planting initiatives on non-tree ecosystems. Concerns have been raised about new trees being planted over native scrubland, grasslands, and other naturally occurring ecosystems, degrading the distinct biodiversity value of these areas (Eder, 2019; Monks et al., 2019; Joseph W Veldman et al., 2015). Biosecurity issues, such as the spread of kauri dieback disease (Bradshaw et al., 2019) and myrtle rust (Toome-Heller et al., 2020) need to be better considered as does the unintentional spread of wilding *Pinus radiata* from mass-planting sites and into adjacent ecosystems (Hulme, 2020a).

Tree-planting initiatives must support the needs, values and priorities of landowners to be successful (Fleischman et al., 2020). A recent survey of rural decision makers in Aotearoa New Zealand reported that those farmers who were considering planting trees in the next two years had a strong preference for native species over exotics (MW-LR, 2020) . However, the survey also highlighted hurdles to planting these trees, including concerns over financial barriers, uncertainty around what tree to plant where, labor constraints, and negative perceptions about trees. (Maseyk et al. (2021) also identified cost, lack of knowledge, and lack of time and resourcing as potential barriers for managing and protecting native biodiversity in New Zealand, with landowners expressing a clear desire for greater support in overcoming these hurdles. Yet landowners are not eligible for any carbon or biodiversity grants to help them protect, restore, and maintain trees and forest that already exist on their land, leading some to turn to selective felling to cover the cost of protecting the rest of the forest (Gibson, 2020; Ministry for Primary Industries, 2002). 1BT must consider these hurdles and how to overcome them if they are to increase the number of trees planted on private land. Overcoming these hurdles would not contribute to the carbon goals of 1BT but realizing landowners' strong preference for planting native species would also deliver multiple benefits for carbon, biodiversity, and the farmers themselves.

2.4 Ten recommendations for One Billion Trees to achieve carbon and biodiversity goals

Tree planting is not a simple solution for climate change (Holl & Brancalion, 2020). Planting the right trees in the right place and securing its long-term survival is a fundamental consideration of large-scale reforestation Programs if they are to avoid perverse outcomes. Here, we make ten recommendations to improve the Aotearoa New Zealand One Billion Trees Program while achieving multiple wins for biodiversity and carbon together:

Recommendation 1. Diversify strategies - protect first, restore second, plant third.

While protection is not yet recognized as a climate strategy in Aotearoa New Zealand, stronger forest management (Bauhus et al., 2010; Lindenmayer et al., 2012b) and the importance of protecting existing forest (Lewis, et al., 2019; Norton et al., 2018) is an important part of achieving global climate goals (IPCC 2019). Both protection and restoration of degraded forests are relatively low cost and low risk solutions versus planting new trees, yet they are both currently under-considered in planning and policy. The nation's carbon accounting should not focus on tree planting alone; it should instead aim to diversify strategies and prioritize different approaches to better account for the relative value of protection, restoration, and planting.

Recommendation 2. Consider net change in trees – do not just count trees planted.

1BT reports the number of trees planted, but it is important to consider that at least half of these trees would have been planted under a business-as-usual scenario without 1BT funding (Ministry for Primary Industries, 2020a). Commercial forestry efforts will be regularly harvested (Ministry for the Environment, 2020b) and will not provide the long-term biodiversity and carbon co-benefits of structurally complex native forest systems (Carswell et al., 2012) . Trees are also lost every year due to forest degradation and land-use change (Ministry for the Environment, 2018) as well as non-trivial mortality rates and natural self-thinning (Forbes et al., 2020). We cannot assume the number of trees planted is equivalent to carbon and biodiversity benefits, we must also consider the additionality, structure, and longevity of these trees. One way to measure net change would be to quantify gains and losses using remote sensing of forested land cover (Liu et al., 2019).

Recommendation 3. Consider the co-benefits of carbon and biodiversity from the outset

The planting of native species should be further incentivized to pivot away from the path-dependency of exotic trees and achieve increased carbon and biodiversity co-benefits (e.g., Büntgen et al., 2019; Lewis, et al., 2019; Norton et al., 2020; Seddon et al., 2019). Native biodiversity and carbon sequestration should be monitored to track these co-benefits, which will provide justification for expanding such initiatives and prioritizing the planting of native trees over exotic species wherever possible.

Recommendation 4. Consider the broader landscape

Optimizing the carbon and biodiversity benefits of tree planting also requires important considerations of the broader landscape (Arroyo-Rodríguez et al., 2020; Case et al., 2020b; Landis, 2017). Co-benefits will need to be optimized on a case-by-case basis, depending on the existing landscape the trees may be planted in, the spatial requirements of tree patches, habitat connectivity to other tree patches, and the arrangement of trees. This means not only considering the types of trees planted (Recommendation 3) but also evaluating where these trees could be planted to optimize carbon and biodiversity benefits across the broader landscape.

Recommendation 5. Consider the carbon and biodiversity benefits of soil

Focus on trees as a carbon sequestration tool has overshadowed the carbon value of the soils beneath these trees as part of the climate solution. In decisions about which tree to plant where and why, it is also important to consider the impacts different tree species may have on the carbon and biodiversity potential of the soil in these areas (e.g., Waller et al., 2020).

Recommendation 6. Consider the importance of existing carbon stocks

There is a policy gap around old-growth forest, carbon, and the multiple ecosystem benefits these forests provide. While old-growth forests may add little to annual carbon sequestration efforts in comparison to new trees, their deforestation and/or degradation risks releasing the carbon they store back into the atmosphere. National climate policy needs to better account for these existing old-growth forests to ensure they do not degrade over time, protecting these important carbon stores and their multiple ecosystem benefits.

Recommendation 7. Consider potential impacts to non-tree ecosystems.

Tree planting initiatives must also consider the impacts of new plantings to biodiversity of non-tree ecosystems at potential planting sites. Replacing grasslands, savannas, wetlands and other non-tree ecosystems with trees overlooks the existing biodiversity value of these areas (Monks et al., 2019; Veldman et al., 2015) and can release the carbon already stored in these systems (Lewis, et al., 2019). Expanding tree planting means finding new areas to plant trees, but existing habitats and changes in land use should also be considered when selecting new sites.

Recommendation 8. Consider the longevity of the future forest

The 1BT Program aims to plant one billion trees across Aotearoa New Zealand by 2028. But what happens to these trees after the Program finishes? Carbon and biodiversity co-benefits will only be realized if these trees remain in the ground long into the future. Safeguards and ongoing support are required to ensure these trees are not removed post-1BT in response to shifting priorities in land-use or policy. Forests also need to be climate resilient to withstand the shocks and disruptions anticipated from invasive pests, weeds, and diseases, in addition to global heating. Better consideration of the genetic material to be planted is critical to ensure the longevity and adaptive capacity of the future forest (Bozzano et al., 2014; Fady et al., 2021).

Recommendation 9. Support landowners in planting and maintaining native trees

Landowners still suffer numerous technological and financial barriers to planting native trees. Better incentives and support for farmers to both plant and maintain native trees on their own land could help overcome these hurdles and secure carbon and biodiversity benefits into the future. Such an approach would include a significant and dedicated investment into research on enhancing native tree planting, resilience, and survivorship in different landscapes and under different climate scenarios.

Recommendation 10. Remember that climate goals cannot be achieved by planting trees alone.

While tree planting initiatives can provide multiple benefits for climate and biodiversity, and contribute towards our targets, climate goals cannot be achieved through planting trees alone (Holl & Brancalion, 2020; Seddon et al., 2020). Achieving national and international climate targets will also mean transitioning away from extractive industries and high-emissions economies. It is especially vital that tree planting is not used as an avoidance strategy for reducing emissions (McLaren & Jarvis, 2018) and that the socioeconomic dimensions of policy – such as equity, inclusivity and indigenous rights – are anticipated and addressed in policy design (Hall, 2019). The newly established Aotearoa New Zealand Climate Change Commission provides independent, expert advice to Government to help the country transition to a socially-just climate-resilient future and could offer an appropriate platform for a holistic approach (www.climatecommission.govt.nz).

2.5 Conclusion

Well-designed, evidence-informed large-scale tree plantings have great potential for delivering multiple benefits, if they are intentionally designed to do so. However, many large-scale tree planting schemes unintentionally incentivize single species tree plantations that can have negative and unintended consequences for native biodiversity. Here this study has demonstrated how these perverse outcomes have also occurred in a temperate forest case study in Aotearoa New Zealand. This study has provided ten recommendations that could be used to improve the design and development of the One Billion Trees Program to help Aotearoa achieve a future forest that jointly delivers benefits for carbon and biodiversity. Such opportunities need to be supported by evidence and enhanced by relevant climate policy. I hope these recommendations will inform climate thinking in Aotearoa New Zealand and globally, to assist in the achievement of enhanced benefits for science, nature, and society.

Chapter 3 Beyond carbon sequestration: Opportunities for multifunctionality of woody vegetation on New Zealand sheep and beef cattle farms

Two-thirds of Earth's global ice-free land surface, which amounts to approximately 196 million square kilometers has been converted to productive land uses, of which 45% is agricultural, leaving only 31% classified as natural ecosystems (Shukla et al., 2019). The conversion of natural ecosystems to agriculture and the continuous practice of intensive agriculture have many documented negative consequences, including the loss of biodiversity (Tscharntke et al., 2005) and alterations to the ecosystem functions (Bennett et al., 2009), such as the reduction of carbon sequestration from biomass and soils (Lal, 2004), increased soil erosion (Pimentel et al., 1995), and reduced habitat provision for native wildlife (Green et al., 2005).

With the growing popularity of global programs to reduce the impact of climate change through tree establishment, including natural regeneration, people are beginning to look for more opportunities to incorporate trees into a wide variety of land uses, including integrating trees in agricultural ecosystems. Countries such as New Zealand, Australia, and the United Kingdom are aiming for planting more trees on agricultural land (Australian Government, 2018; Ministry for Primary Industries, 2020; UK Government, 2021). Forest Landscape Restoration is a global initiative to bring 150 million hectares of deforested and degraded land into restoration by 2020, and 350 million hectares by 2030, that emerged in response to the Bonn Challenge, includes planting trees on agricultural land as one of the strategies (The Bonn Challenge, 2020). Private sector organisations, such as Nestle and Del Monte, have committed to support planting trees on and around farms as part of the One Trillion Trees initiative, the World Economic Forum’s efforts to accelerate Nature-based Solutions (NbS) (World Economic Forum initiative, 2020).The UNEA-5 resolution formally adopted the definition of NbS as ‘actions to protect, conserve, restore, sustainably use and manage natural or modified terrestrial, freshwater, coastal and marine ecosystems, which address social, economic and environmental challenges effectively and adaptively, while simultaneously providing human well-being, ecosystem services and resilience and biodiversity benefits’ (UNEP, 2022). The ‘Resolution on Nature-based Solutions for Supporting Sustainable Development’ also calls on UNEP to support the implementation of NbS, which safeguard the rights of communities and indigenous peoples (UNEP, 2022). Combining trees and agricultural systems on the same land area, could provide a valuable strategy to reconcile ecological and socio-economic objectives by treating agroecosystems as a multifunctional landscape (Reith et al., 2020). In such systems, native forest remnants, naturally regenerated forests, agroforests, mixed species plantations, and commercial monoculture plantations all co-exist, providing a wide variety of ecosystem functions, including those that contribute to biodiversity loss, climate mitigation and climate adaptation (Meli et al., 2019; Renting et al., 2009). Restoration of woody vegetation on agricultural lands could therefore provide valuable NbS for the enhanced provision of ecosystem functions while achieving multiple co-benefits (IUCN, 2020).

Although some research has focused on the benefits of woody vegetation, including in agricultural landscapes, for mitigating and adapting to climate change (e.g., Fischer et al., 2006; Verchot et al., 2007; Jose et al., 2019), there are many knowledge gaps. Research has so far largely been limited to adding, converting, and optimizing only a few woody features, such as woodlots and shelterbelts, and a few ecosystem functions, such as increasing carbon stocks; this narrow focus limits the potential for recommendations that will increase the heterogeneity of these landscapes (England et al., 2020; Lindenmayer et al., 2012a; Tscharntke et al., 2005). Research that quantifies the contributions of a diversity of existing vegetation is also lacking, potentially leading to the devaluing and conversion of certain plant community types, such as shrublands and native grasslands; such vegetation often supports unique and rare remaining native biodiversity (Lindenmayer et al., 2012). Further, quantifying the ecosystem functions associated with increasing woody vegetation that have negative impacts on farm productivity or native biodiversity is also important for understanding the consequences of incentivizing farm multifunctionality (Case et al., 2020b) and, thus, gaining balanced information for landscape planning and determining the necessary level of management intervention (England et al., 2020; Fahrig et al., 2011; Lindenmayer et al., 2012).

Woody vegetation on sheep and beef cattle farms in Aotearoa New Zealand offers a case study example of how various ecosystem functions interact in these agricultural landscapes. Increases in the area and intensity of agricultural production pose a major threat to Aotearoa New Zealand's unique native biodiversity and ecosystem services in New Zealand (Kirschbaum et al., 2012), potentially contributing to the loss of natural habitat area, native species diversity, and a net release of carbon into the atmosphere (Moller et al., 2010). Pastoral lands, including sheep and beef cattle farms, cover almost half of the country (Pannell et al. 2021) and are predominantly located on the lowland areas where the majority of native vegetation cover has been converted to other land uses (Dominati et al., 2019; Norton et al., 2020). Pastoral lands, including sheep and beef cattle farms, cover almost half of the country (Pannell et al., 2021) and are predominantly located in lowland areas where, nation-wide, the majority of native vegetation cover has been converted to other land uses (Dominati et al., 2019; Norton et al., 2020). Despite this, sheep and beef farms in New Zealand still contain a relatively high proportion of native woody vegetation at approximately 1,389,000 hectares, representing about 13.05% of the total sheep and beef farm area, or 17.1% of the total native woody vegetation cover in New Zealand, and about 4.62% of the total land area of the country (Pannell et al., 2021). This presents opportunities for farm managers to achieve high levels of agricultural production alongside native habitat conservation and other ecosystem services (Norton &

Miller, 2000). Many farmers in Aotearoa New Zealand are working to preserve and restore native and exotic woody vegetation as part of the agricultural landscape matrix (Pannell et al., 2021) alongside exotic species dominated woody features such as shelterbelts and timber woodlots (Parkyn et al., 2003). These different woody plant communities have both production and non-production benefits, and are becoming increasingly recommended as part of sheep and beef cattle farm planning (Dominati et al., 2019b; Easdale et al., 2021).

Despite increasing aboveground biomass becoming one of Aotearoa New Zealand's climate change mitigation strategy, there are still gaps in the understanding of the contribution of woody vegetation in agricultural landscapes as an important part of this mitigation. The total carbon stock held in woody vegetation on agricultural land is considered minor compared to the carbon stock per hectare (i.e., carbon stock density) of natural and exotic forest (Ministry for the Environment, 2020a). Considered to have a lower carbon value than other land cover types, puts woody vegetation on agricultural land at risk of being transformed to a higher-carbon-value forest for the purpose of mitigating climate change (Norton et al., 2020). Consequently, only a few estimates of carbon stock densities have been quantified for woody vegetation on Aotearoa New Zealand farms (e.g., Burrows et al., 2018; Welsch et al., 2016). Woody vegetation patches that are smaller than the threshold for the definition as 'forest' are not included in calculations for estimating Aotearoa New Zealand's carbon stocks (Burrows et al., 2018). The country's carbon credit or emissions trading scheme (ETS) also does not provide carbon credits for native remnants that do not meet the forest definition's eligibility criteria (Norton et al., 2020). This scheme only provides carbon credits for replanted stands of exotic trees and excludes replanted native stands that were established in areas that were covered by native forest prior to 1990. Currently, there is no policy to encourage farmers to retain existing woody vegetation on agricultural land (Welsch et al., 2014a). Furthermore, compared to exotic plantations, the carbon credits awarded for native trees are lower (Norton et al., 2020). Despite incentivization for planting native trees through Aotearoa New Zealand's national tree-planting program (One Billion Trees Program, Ministry for Primary Industries), which targeted planting into less productive land, a lower carbon credit rate was assigned for native species. T Although the incentivization in 1BT and ETS pay-outs was originally designed to encourage carbon capture as an ecosystem service, the implementation of both policies results in a lower economic return to landowners for planting native species compared to planting intensive exotic trees. Thus, in combination with the inclusion of existing planned exotic plantations into the scheme, a much higher number of exotic trees is being planted, despite the fact that the expansion of these plantations has been criticized for serving primarily socio-economic goals rather than biodiversity goals (Suryaningrum et al., 2021).

Without sufficient incentive for regenerating and conserving woody vegetation on farms, sheep and beef cattle farms with woody vegetation,, especially with native woody vegetation, will be economically valued at lower levels than other land uses such as intensive dairy farming and exotic plantation forestry (David Hall & McLachlan, 2022). The combination of tree planting incentives and the emission trading scheme may further increase the competitive value of commercial exotic plantations, which will drive further clearance and conversion of native vegetation, including on sheep and beef cattle farms (Norton et al., 2020; Walker et al., 2006). Hence, it is important to demonstrate the value of woody vegetation occurring on sheep and beef cattle farms and the multiple ecosystem functions it provides, such as carbon and native biodiversity, to decision-makers, such as landowners and land use planning authorities, so that management can be improved to optimize these functions alongside agricultural productivity.

Our study is the first to explore variation in carbon storage, species richness, and other ecosystem functions among different plant community types on three sheep and beef cattle farms. These case study farms were representative of this common Aotearoa New Zealand land use. This study addressed the following research questions: (1) How do woody plant (including tree ferns) community structure and carbon stocks per hectare differ among different on-farm community types, and what plant community characteristics contribute to a higher total carbon stock? (2) How does multifunctionality differ, and what functions trade off, among different community types? To address these questions, I: (1) mapped woody plant species composition across the three farms using vegetation plots that were then classified into different plant community types; (2) compared the characteristics of each of the plant community type in terms of relative abundance, diversity, and stem size; (3) estimated the mean and variation of carbon stocks per hectare for each plant community type; and (4) quantified the relative potential multifunctionality by summing the estimated ecosystem functions associated with each woody plant species and then aggregating these values for each vegetation type. This research illustrates how the detailed assessment of the ecosystem functions associated with woody species can be used to estimate the relative benefits of different woody plant communities on farms for farm multifunctionality.

3.1 Methods

3.1.1 Study area

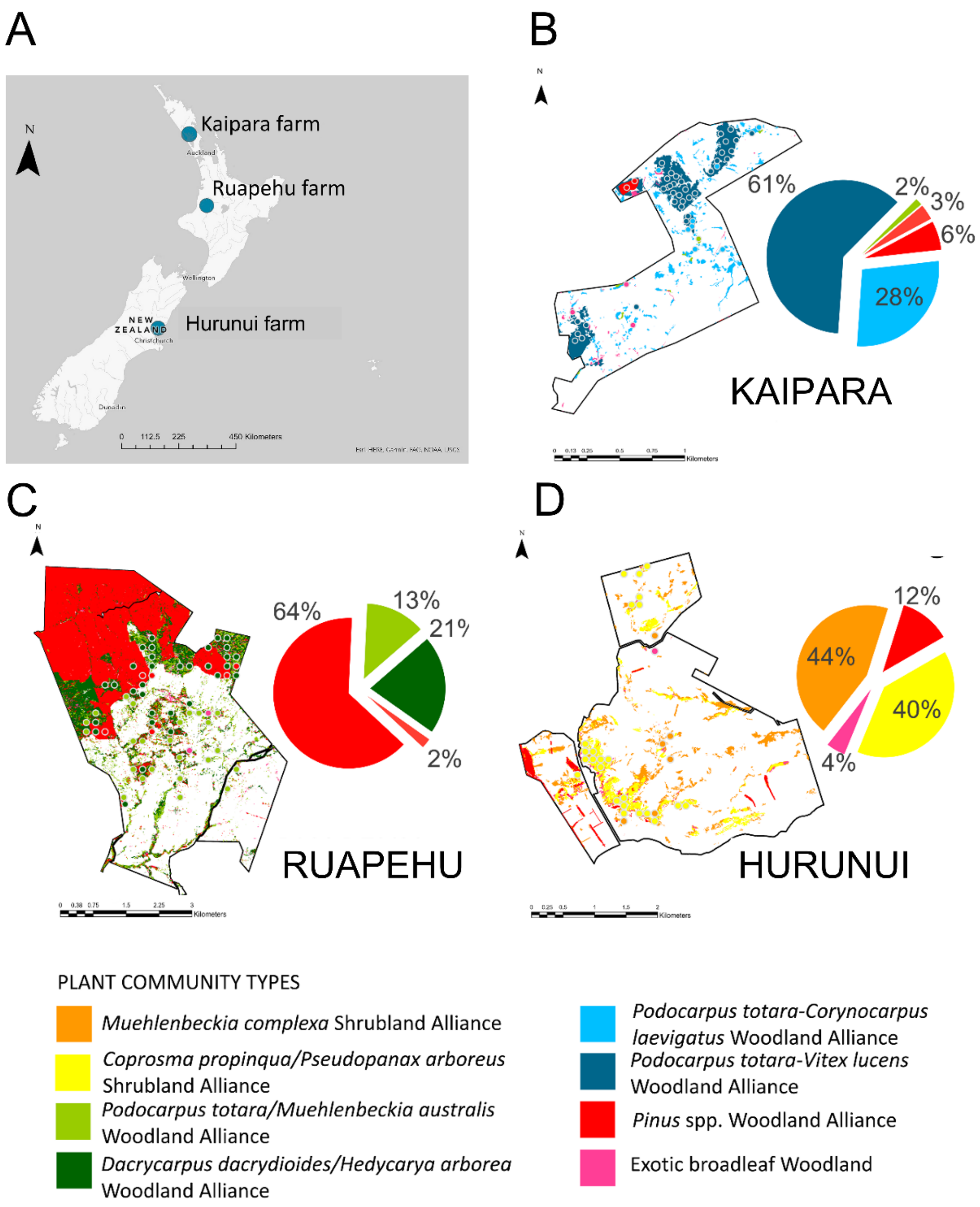
Our three case study sites were two farms in Aotearoa New Zealand’s North Island (the Kaipara and Ruapehu farms), and a third farm in the South Island (the Hurunui farm) (Figure 1). The three farms were sheep and beef cattle farms in the area dominated by beef cattle farming, followed by dairy farming, sheep-beef cattle farming, sheep farming, forestry, and annual crop (Statistics NZ, 2017). The vegetation of all three sheep and beef farms was typical of New

Zealand farming landscapes in that it was highly fragmented and consisted of remnants or regenerating cut-over patches of native forest, as well as commercial exotic plantations. These three sheep and beef cattle farms were predominantly managed as sheep and beef farms, with some areas managed as exotic monoculture plantations (Manaaki Whenua, 2019).

The Kaipara Farm (152 ha; Figure 1 B) is located in the Kaipara region, close to Kaipara Harbor in the North Auckland Peninsula. The farm spans across several elevations, ranging from approximately 50 meters to 100 meters above sea level. The average daily temperature ranges from 7-8°C in the winter to 22–24 °C in the summer (Pearce et al., 2020). The farm experiences a mean annual rainfall of 1454 mm (Chappell, 2013). The region is known for its strong and gusty winds, with a high frequency of gusts exceeding 63 km/hr and 96 km/hr each year. The farm is mostly composed of high-productivity grassland, along with a small amount of shelterbelt, two small remaining native forest patches, and exotic woodland. However, the northern area of the farm is vulnerable to soil slip, while the southern area consists of gullies and is vulnerable to earthflow erosion. The Ruapehu farm (2,167 ha; Figure 1 C) is situated in the Manawatu-Wanganui region. The farm spans across several elevations, ranging from approximately 200 meters to 400 meters above sea level. The average annual rainfall for the area is 1522 mm, while the median winter average daily minimum temperature can range from 2–14 °C and the median summer average daily maximum temperature is 10–24 °C (Chappell, 2015). The Ruapehu farm mostly comprised of high-productivity grassland and a sizable area of exotic woodland that was close to some small patches of remaining old forest. The Canterbury farm (770 ha; Figure 1 D) is situated in the Canterbury region and consists primarily of high-productivity grassland, at an elevation of around 100 m above sea level, with a few tiny patches of shrubs and exotic shelterbelts. The region receives 618 mm of rain on average per year, and the median winter average daily minimum temperature can range from -2 to 11 °C and the median summer average daily maximum temperature can range from 4 to 22 °C (Macara, 2016).

3.1.2 Data collection

Ground truthing of aerial photographs was initially used to classify the woody vegetation occurring across each farm into five broad vegetation types, based on differences in species composition and structure: exotic pine forests, exotic shelterbelts, remnant native forests, regenerated native forest, and shrublands. This classification was used as a basis for a stratified random sampling scheme to quantify the aboveground and belowground carbon stock densities of the different tree and shrub communities present within woody vegetation patches on each farm. Within this study, tree ferns were also included due to their large size and functional role as sub-canopy trees within woody vegetation patches. Sampling plots were randomly positioned within each farm. At each sample location, woody vegetation was inventoried within square, 0.01-ha (10 x 10 m) survey plots (10 × 10 m), following a standard reconnaissance plot sampling method (Department of Conservation, 2012; Hurst & Allen, 2007); 145 plots in total were sampled across the three farms (Figure 1). We recorded the percent cover of trees and shrubs, including tree ferns (see Appendix 2 for complete name of species) on a six-point scale: <1%, 1-5%, 6-10%, 11-25%, 26-50%, 51-75%. For each plot, we collected the following data: (1) identification and measurement of the diameter and estimated height of trees and shrubs, identified to the lowest taxonomic level possible, for all stems with a diameter of at least 2.5 cm at 1.3 m height (DBH) and for discrete shrubs (multiple stems at 1.3 m height), (2) the number of individuals, (3) the length and width of coarse woody debris (CWD, i.e., the remnants of dead trees with a diameter and length greater than 10 cm), and (4) the width and height of discrete shrubs, which was subsequently used to calculate cuboid volume. Heights were estimated visually for all stems that belonged to the same species in a sample plot's four quadrants. Plant nomenclature followed the New Zealand Plant Names Database (Landcare Research, 2010). To measure soil carbon in the upper mineral soil, we collected an approximately 15 × 15 cm by 15 cm deep soil sample from each of the four quadrants of the plot.


A
N
Kaipara farm
Ruapehu farm
NEW ZEALAND
Hurunui farm
0 112.5 225 450 Kilometers
B
N
61%
2%
3%
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28%
KAIPARA
C
N
64%
13%
21%
2%
RUAPEHU
D
N
12%
44%
40%
4%
HURUNUI
PLANT COMMUNITY TYPES
Muehlenbeckia complexa Shrubland Alliance
Coprosma propinqua/Pseudopanax arboreus Shrubland Alliance
Podocarpus totara/Muehlenbeckia australis Woodland Alliance
Dacrycarpus dacrydioides/Hedycarya arborea Woodland Alliance
Podocarpus totara-Corynocarpus laevigatus Woodland Alliance
Podocarpus totara-Vitex lucens Woodland Alliance
Pinus spp. Woodland Alliance
Exotic broadleaf Woodland

Figure 1. A) Inset map shows the locations of three sheep and beef cattle research farms in Aotearoa New Zealand: Kaipara, Ruapehu, and Hurunui; and the three other maps show the distribution of plant communities across three farms on B) Kaipara, C) Ruapehu, and D) Hurunui. The colors of polygons and pie charts represent eight plant communities: two shrubland alliances (*Muehlenbeckia complexa* and *Coprosma propinqua*/*Pseudopanax arboreus* shrubland alliances), four native-dominated woodland alliances (*Podocarpus totara*/*Muehlenbeckia australis*, *Dacrycarpus dacrydioides*/*Hedycarya arborea*, *Podocarpus totara*-*Corynocarpus laevigatus*, and *Podocarpus totara*-*Vitex lucens* woodland alliances), and two exotic-dominated woodland alliances (*Pinus* spp. and Exotic broadleaf woodland alliances). Pie charts illustrate the proportions of plant community area relative to the total area of all plant communities in each farm.

3.1.3 Woody plant community classification

To classify plots by their species composition, I used the canopy cover data. I first categorized the plots manually into three, very different compositional types: (1) native-dominated plots that were classified within remnant or regenerating native forests or shrub patches (n = 131); (2) *Pinus* spp.-dominated plots that occurred in monocultural plots (within pine plantation areas) (n = 8); and (3) exotic broadleaf plots that either in the form of dispersed single trees or group of trees that were planted as monoculture stands of exotic species (*Erythrina ×sykesii* Barneby & Krukoff (n = 1) and *Salix* L. (n = 5).

For the native-dominated plots, an Importance Value Index (IVI) for each species was calculated by summing its cover-score midpoint recorded in each plot, multiplied by the depth of the height tier (the difference between the minimum and maximum values) (Norton & Leathwick, 1990). Once calculated, the plot level IVIs were summed by species per plot to give a single IVI per species per plot. I then generated a dendrogram for the native-dominated plots based on the importance values for each species plant species composition using the Two-Way Indicator Species Analysis (TWINSPAN; Kent, 2011) implemented in the community analysis software package PC-ORD (McCune et al., 2002). TWINSPAN is a hierarchical, divisive method of classifying plots based on their species composition (Kent, 2011). The default settings (McCune et al., 2002) were used for the maximum number of indicator species per division (five), the minimum group size (five plots), and for the maximum number of species in the output table (total number of species occurring in the plots). The pseudospecies cut levels (McCune et al. 2002) were set to 0, 0.15, 1.65, 31, and 194, based on the importance values for species in each community type.

Groups of compositionally similar plots resulting from the TWINSPAN were then assigned a name following the International Vegetation Classification (IVC) method (Grossman et al., 2001). These names were given as an "alliance", or a physiognomically uniform group of plant associations, sharing one or more dominant or diagnostic species, that occurred in the uppermost stratum of the vegetation (Grossman et al., 2001). Alliance names were a combination of the floristic name and the name of the structural community type. I determined the dominant species for inclusion in the floristic name by ranking the species in the alliances according to their relative cover and/or relative constancy. The slash symbol in the floristic name '/' is used to distinguish that the two dominant species occurred in different tiers, and the hyphen symbol '-' indicates where the two dominant species occurred in the same tier. The species that occurred in the uppermost stratum are listed first. I categorized alliances based on the dominant growth form; the term "woodland alliance" was used to describe the plant communities where trees were dominant over shrubs, and the term "shrubland alliance" was used to describe the plant communities that were dominated by shrubs (Grossman et al., 2001). Plots dominated by the exotic-*Pinus radiata* were classified as one alliance: *Pinus* spp. woodland alliance; the rest of exotic-dominated plots were assigned to one group: exotic broadleaf woodland.

A Principal Coordinates Analysis (PCoA) was performed on the plot-level dataset to visualize the dissimilarities in species composition among plant community types, based on an Ochiai dissimilarity matrix computed using abundance data (Roberts & Roberts, 2019). I then generated a spatial map to visualize the distribution of plant community types in each of the farms, where all woody vegetation patches were assigned plant community types based on the similarity of the sample plots.

3.1.4 Tree and shrub abundance and species richness estimation

The abundance of each tree and shrub species was estimated from the number of stems on each plot. Woody species diversity was calculated as relative species richness (number of species in each sample plot) using all woody and tree fern taxa that occurred within the 1-ha survey plot (10 × 10 m) (Magurran, 2004).

3.1.5 Carbon estimation

I calculated the total carbon stored in two carbon pools: aboveground and in the soil; and then estimated the carbon stored belowground from the calculated aboveground carbon stock. To quantify aboveground carbon stock, I estimated carbon stored on live stems, shrubs, and coarse woody debris using known allometric equations and other published information (Appendix Table 1). I estimated the live stem biomass for individual trees within each community type on each farm by using the collected tree stem diameter and height data. First, for the Aotearoa New Zealand native species and *Pinus* spp., I estimated tree heights for individuals that had been assigned an average height (i.e., where multiple individuals of the species were in the plot) using species-specific diameter-height equations. For species for which only one individual occurred in a subplot and species for which species-specific diameter-height equations were not available, the field estimated height was used. Second, I used the estimated stem heights, the stem diameter data from plots, to estimate the biomass of individual live stems a within plots. I used species-specific equation when available, and when the species-specific allometry was not available, I applied country-specific equations. Different equations and methodologies were used for estimating biomass from live stems (for native tree species, selected exotic tree species, native tree ferns) (Appendix Table 1, See equation 1-6). Wood-specific density to calculate live stem biomass was obtained from two online databases: 1) Aotearoa New Zealand species wood specific density and 2) global wood specific density dataset (Holdaway et al., 2014; Zanne et al., 2009). For species without species-specific wood specific density values (n = 48), I used the corresponding genus-level mean; when a genus-level mean was not available, a growth-form mean was used (Holdaway et al., 2014).

After estimating carbon stored on live stems, I estimate carbon stored on shrubs, using the measured volume of discrete shrubs (Appendix Table 1 equation 6). Discrete shrub density was obtained from Coomes et al. (2002) and when the shrub density of specific species was not available, I used the mean shrub density of all species.

To calculate biomass content from coarse woody debris, I applied two different equations for coarse woody debris from 1) trees and shrubs, and 2) tree ferns and palms (Appendix Table 1, see equation 8-11). For coarse woody debris of trees and shrubs, I first estimated the volume from standing and fallen logs. that was then calculated to estimate the biomass content of coarse woody debris. I calculated the biomass of coarse woody debris differently for logs classified into four decay classes: 1, 2, 3, and 4 (Coomes et al., 2002). A decay-stage modifier (DSM) was obtained from Holdaway et al. (2014). Fresh-wood specific density of fallen and standing logs was obtained from Coomes et al. (2002), and when the species-specific fresh-wood specific density was not available in Coomes et al. (2002), I applied wood specific density from Holdaway et al. (2016). The carbon fraction of live stems, discrete shrubs, and coarse woody debris was then estimated as 0.5 of biomass (Holdaway et al., 2014). Total carbon stored on live stems, discrete shrubs, and coarse woody debris were summarized as carbon from the aboveground biomass (AGB) pool. Then, I made the sum and estimation for each plot.

To estimate soil organic carbon (SOC), first I estimate the amount of total organic carbon (TOC) in the soil samples. Mineral soil samples taken from each plot subplot were aggregated and a 10 grams subsample of soil was combusted at 450 °C in a muffle oven for eight hours following the loss-on-ignition (LOI) method (Fourqurean et al., 2015). The total organic carbon content (TOC as a percent measure) quantified for each of these samples represented the percentage weight loss of the sample due to combustion (Equation 11; Appendix Table 1). The %TOC converted to a soil organic carbon (SOC) per hectare quantity, assuming that carbon concentrations were representative of the top 30 cm of mineral soil at each plot (Equation 12; Appendix Table 1); carbon estimates were made relative to a 30 cm depth in order to align with other soil carbon studies in Aotearoa New Zealand.-Soil bulk density was obtained from Sparling et al. (2000), from two representative classes in sheep and beef cattle farms; native forests (0.84 Mg m^{-3}) and *Pinus radiata* plantation (0.81 Mg m^{-3}). The conversion to soil carbon per-hectare densities required the incorporation of estimates of bulk density at the plot locations; for this purpose, the mean bulk density of native forests was used for the native-dominated woodland and shrubland plots and the mean bulk density of *P. radiata* plantation was applied to the exotic-dominated woodland plots (Sparling et al., 2000). When soil samples were not available for some plots (n =12), a mean soil carbon stock per hectare of plots of similar plant communities in the nearby habitat was applied.

I then estimated belowground biomass carbon (BGB), comprising biomass of live roots, as 25 % of aboveground carbon (Holdaway et al., 2017) in each plot. Finally, I summed the carbon densities calculated for the three carbon pools: aboveground carbon, soil carbon, and belowground carbon, as total carbon stock density (t C ha^{-1}) for each plant community on each farm.

3.1.6 Quantification of ecosystem functions

In addition to carbon, I estimated a range of metrics for nine ecosystem functions using data from each plot. The nine selected metrics indicate functions that support or represent both economic farm productivity and native biodiversity on sheep and beef cattle farms (McWilliam et al., 2017). These metrics were estimated as the number of individuals of specific woody species in each plot that are recognised in the published literature for: (a) timber provision, (b) stabilizing gullies, (c) stabilizing soil from erosion, (d) reducing soil surface erosion by wind (soil erosion caused by strong wind), (e) providing livestock shelter, (f) providing food for birds, and (g) enhancing native plant biodiversity. In addition, two indicators of ecosystem ‘disservice’ (i.e., that have a detrimental effect on farm production) were selected: (h) the number of individuals of introduced woody plant species that have become invasive weeds, and (i) a community weighted measure of mean flammability across all species in a plot.

Data on the characteristics of each species contributing to the above indicators were acquired from secondary databases (Hawke's Bay Regional Council et al., 2008; Northland Regional Council, 2005, 2020; Tane's Tree Trust, 2022; Te Mära Reo, 2020; Weedbusters, 2020), except for community weighted flammability. Plant relative flammability data that were acquired from shoot-level flammability measurements, using a standard methodology applied to samples collected on the three farms for this study and existing datasets (Wyse et al., 2016, Cubino et al., 2018, Cui et al., 2020). Community weighted flammability was estimated from individual relative flammability weighted using IVI following Cubino et al. (2018). For classifying number of introduced woody plant species that have become invasive weeds, we excluded species that are not perceived as weeds by farmers, e.g., willow (Salix spp.). All species-level data are given in Appendix 2. We then calculated average total carbon stock per hectare and the average scores for ecosystem functions for each plant community. The average scores was then normalized as Z-score.

3.1.7 Data Analysis

To address the first questions: how do woody plant (including tree ferns) community structure and carbon stocks per hectare differ among community types, and what plant community characteristics contribute to a higher total carbon stock, we conducted a Principal Component Analysis (PCA) on the plot-level dataset to determine the relationships among the stem size, abundance, tree and shrub richness, and the aboveground carbon stock densities for the different plant communities. To address the second question on how multifunctionality differs and what functions trade off among different plant community types, we first calculated the relative values of each ecosystem function, abundance, and richness for each community type. We obtained these values by dividing the total value of each variable across all plots by the number of plots for each community type. To visualize these values, we created three bar plots: one for abundance, one for richness, and one for the score of multifunctionality. For the bar plots of abundance and richness, we used the actual values, while for the bar plot of multifunctionality, we plotted the sum of Z-scores, with a mean of zero and standard deviation of one. In addition, we created a radar plot to visualize the Z-scores, rescaled from 0 to 10, of each ecosystem function for each plant community type. This allowed us to compare the relative multifunctionality of each plant community type on each farm. For the last two ecosystem functions (number of introduced woody plant species that have become invasive weeds and community weighted flammability), the score was taken as the reverse so that higher values represented the more desirable situation for farm productivity. The sum of the Z-scores for each plant community was used as an estimate of the relative total multifunctionality of each plant community. All analyses, except TWINSPAN, were conducted in R 4.0.3 (R Development Core Team, 2020).

3.2 Results

3.2.1 Woody plant community types and stand structural characteristics

Native-dominated plots across the three farms were classified into six, distinct, plant community types. Together with the two exotic-dominated woodland community types, this resulted in eight different plant community types across the three farms (Figure 2 A): two native-dominated shrubland alliances, four native-dominated woodland alliances, and two exotic-dominated woodland alliances (*Pinus* spp. woodland and Exotic broadleaf woodland). Across all farms, the native *Podocarpus totara*-*Vitex lucens* woodland alliance was the most common plant community in the plots sampled (*n* = 35; 100 % of plots on the Kaipara farm), followed by the *Dacrycarpus dacrydioides/Hedycarya arborea* woodland alliance (*n* = 32; 100 % of plots on the Ruapehu farm), and the *Coprosma propinqua*-*Pseudopanax arboreus* shrubland alliance (*n* = 24; 100 % of plots on the Hurunui farm), respectively. The Kaipara farm had the greatest number of different plant community types.

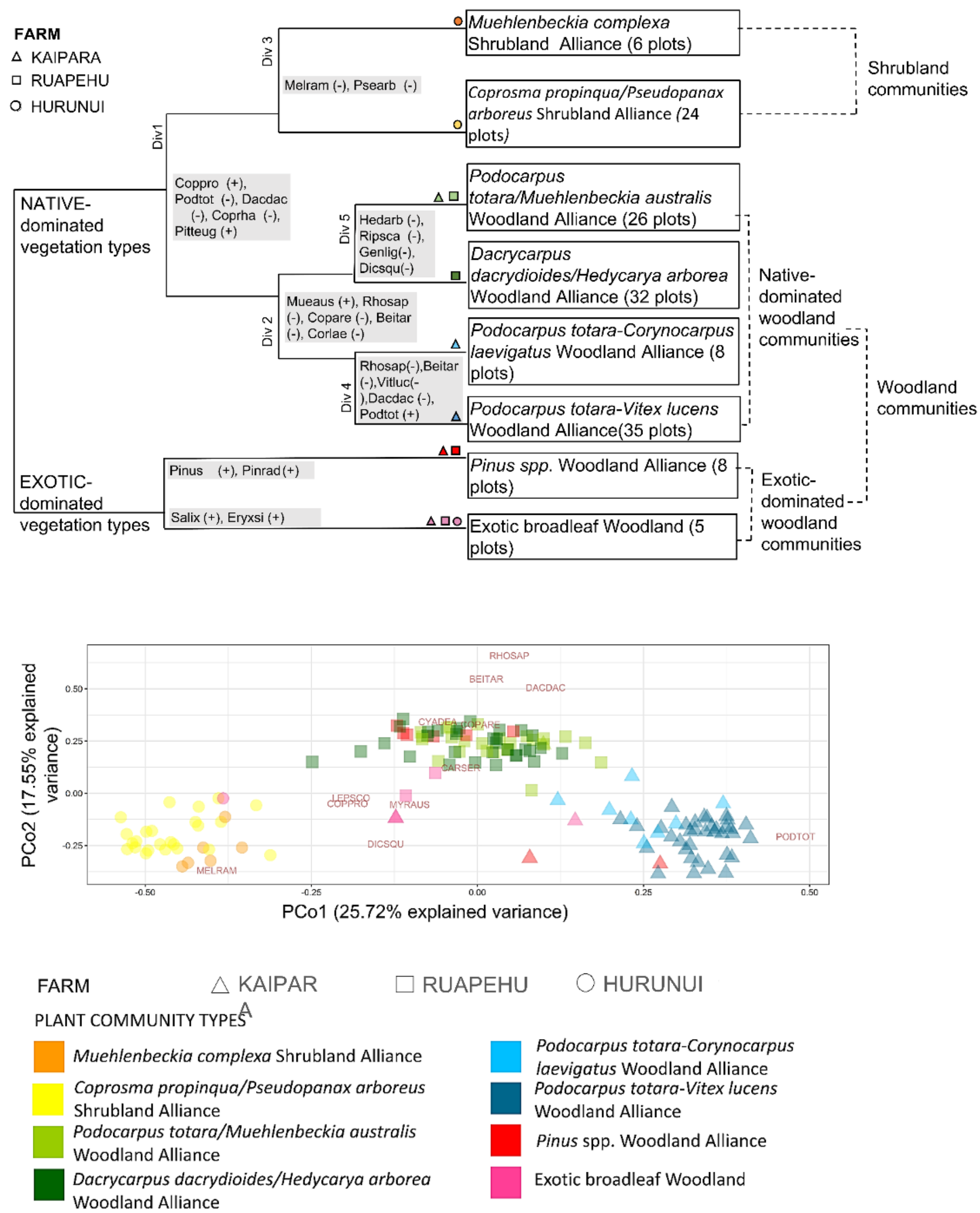
The ordination diagrams (Figure 2B) showed a clear separation in species composition among the three farms, and a clear distinction between the two structural woody vegetation types: woodland alliances and shrubland alliances. Only 12 species occurred in more than 10 stems per plot (Figure 2B). The ordination also showed that most plots of exotic-dominated woodland alliances in Kaipara and Ruapehu farms had species composition similar to the native-dominated woodland alliances and only very few exotic-dominated plots that had distinctive species composition (Figure 2B).

Average woody species abundance and richness were highly variable among the eight plant communities (Table 1). The average number of species ranged from one (*Pinus* spp. woodland alliance) to 5.60 ± 2.52 (*Podocarpus totara*-*Vitex lucens* woodland), and the average number of individual stems ranged from 4.13 ± 2.03 (*Pinus* spp. woodland alliance) to 29.68 ± 9.14 (*Podocarpus totara*-*Vitex lucens* woodland) (*Muehlenbeckia complexa* shrubland).

Smaller diameter stems under 25 cm dominated the sample plots in all three farms, with only a few larger diameter stems exceeding 25 cm or 50 cm (Table 1). The diameter of the stems ranged from 2.67 ± 4.19 cm (*Muehlenbeckia complexa* shrubland) to 47.77 ± 11.41 cm (*Pinus* spp. woodland alliance), and the height of the stems varied from 1.62 ± 2.55 cm (*Muehlenbeckia complexa* shrubland Alliance) to 20.28 ± 1.38 cm (*Pinus* spp. woodland alliance) (Table 1). Woodland alliances had larger and higher stems compared to shrubland alliances (Table 1). Exotic-dominated woodland alliances had the most stems with diameters greater than 25 cm (*n* = 36), while native-dominated woodland communities had the fewest (Appendix Figure 1). Similarly, stems longer than 10 cm were seen on woodland alliance plots, with exotic plots having a higher frequency (Appendix Figure 1; Appendix Figure 2). This general distribution was similar on the three farms, with only very few plots having stems with an average diameter larger than 50 cm and higher than 15 cm (Appendix Figure 1; Appendix Figure 2).

A

B



FARM
KAIPARA
RUAPEHU
HURUNUI
Div1
Div 2
Div 3
Div 4
Div 5
Melram (-), Psearb (-)
Coppro (+), Podtot (-), Dacdac (-), Coprha (-), Pitteug (+)
Hedarb (-), Ripsca (-), Genlig(-), Dicsqu(-)
Mueaus (+), Rhosap (-), Copare (-), Beitar (-), Corlae (-)
Rhosap(-),Beitar (-),Vitluc(-),Dacdac (-), Podtot (+)
Pinus (+), Pinrad (+)
Salix (+), Eryxsi (+)
NATIVE-dominated vegetation types
EXOTIC-dominated vegetation types
Muehlenbeckia complexa Shrubland Alliance (6 plots)
Coprosma propinqua/Pseudopanax arboreus Shrubland Alliance (24 plots)
Podocarpus totara/Muehlenbeckia australis Woodland Alliance (26 plots)
Dacrycarpus dacrydioides/Hedycarya arborea Woodland Alliance (32 plots)
Podocarpus totara-Corynocarpus laevigatus Woodland Alliance (8 plots)
Podocarpus totara-Vitex lucens Woodland Alliance(35 plots)
Pinus spp. Woodland Alliance (8 plots)
Exotic broadleaf Woodland (5 plots)
Shrubland communities
Native-dominated woodland communities
Woodland communities
Exotic-dominated woodland communities
RHOSAP
BEITAR
DACDAC
CYADEA
COPARE
CARSER
LEPSCO
COPPRO
MYRAUS
DICSQU
MELRAM
PODTOT
PCo2 (17.55% explained variance)
PCo1 (25.72% explained variance)
FARM
KAIPARA
RUAPEHU
HURUNUI
PLANT COMMUNITY TYPES
Muehlenbeckia complexa Shrubland Alliance
Coprosma propinqua/Pseudopanax arboreus Shrubland Alliance
Podocarpus totara/Muehlenbeckia australis Woodland Alliance
Dacrycarpus dacrydioides/Hedycarya arborea Woodland Alliance
Podocarpus totara-Corynocarpus laevigatus Woodland Alliance
Podocarpus totara-Vitex lucens Woodland Alliance
Pinus spp. Woodland Alliance
Exotic broadleaf Woodland

Figure 2. A) A dendrogram of six native-dominated and two exotic-dominated plant communities and its two structural groups: shrubland alliances and woodland alliances; across the Kaipara, Ruapehu, and Hurunui farms. The floristic name of plant communities represented the two species with the highest mean IVI across all strata (≥10) and/or the highest species occurrence in each plot, combined with the structural community's name. Species codes (in grey boxes) are the first three letters of the generic and species name for each pseudospecies (see Appendix Table 2 for full species names and life-forms); positive (+) and negative (-) signs indicate the preferences. The symbols indicate the location of community occurrence; and B) A two-dimensional scatterplot from a Principal Coordinate Analysis of species composition of on the Kaipara, Ruapehu, and Hurunui farms. Colors represent plant community type, each point represents a plot, and the distance between plots indicates their relative similarity. Red species labels on each graph indicate taxa represented by more than ten stems on each plot and represents the first three letters of the genus name and specific epithet (see Appendix Table 2 for full names).

Table 1 Summary (mean ± standard deviation) of stand characteristics of eight plant community types, in terms of size measurements for: live stem mean height (m) and mean diameter at breast height (DBH - cm), continuous shrub mean height (cm) and mean volume (cm^3), coarse woody debris mean diameter (the vertical measurement of coarse woody debris; cm) and mean length (the horizontal measurement of coarse woody debris/ length; cm), and mean live stem species richness and abundance.

Plant community types	Live stem size		Shrub size		Coarse woody debris size		Stem species richness	Stem Abundance
	Height (m)	DBH (cm)	Height (m)	Volume (cm^3)	Diameter (cm)	Length (cm)		
Muehlenbeckia complexa shrubland Alliance	1.62 ± 2.55	2.67 ± 4.19	0.9 ± 0.48	0.91 ± 0.72	N/A	0.02	2.67 ± 1.21	29.67 ± 9.14
Coprosma propinqua/Pseudopanax arboreus shrubland alliance	5.39 ± 1.48	9.2 ± 4.87	0.08 ± 0.38	0.14 ± 0.71	0.60 ± 0.26	10.92 ± 4.99	3.46 ± 1.72	17.33 ± 9.90
Podocarpus totara/Muehlenbeckia australis woodland alliance	10.9 ± 4.72	28.07 ± 23.65	N/A	N/A	0.39 ± 0.36	12.29 ± 12.92	2.58±1.65	6.00±4.71
Dacrycarpus dacrydioides/Hedycarya arborea woodland alliance	7.75 ± 2.24	19.04 ± 6.20	N/A	N/A	0.54 ± 0.28		4.75 ± 2.02	18.66 ± 11.83
Podocarpus totara-Corynocarpus laevigatus woodland alliance	8.48 ± 2.15	19.47 ± 8.07	N/A	N/A	0.1 ± 0.29	5.81 ± 16.44	2.25 ± 1.16	8.88±8.48
Podocarpus totara-Vitex lucens woodland alliance	10.06 ± 3.06	18.14 ± 11.67	N/A	N/A	0.47 ± 0.35	13.49 ± 13.77	5.60 ± 2.52	18.40 ± 9.84
Pinus spp. woodland alliance	20.28 ± 1.38	47.77 ± 11.41	N/A	N/A	0.67 ± 0.08	14.99 ± 2.83	1.00 ± 00	4.13 ± 2.03
Exotic broadleaf woodland	10.99 ± 7.22	40.65 ± 38.23	0.55 ± 1.34	4.39 ± 10.75	0.11 ± 0.27	1.71 ± 4.18	1.67 ± 1.03	5.50 ± 5.65

3.2.2 Carbon stock per hectare for each plant community type

Mean total carbon stock densities were highly variable across all eight plant community types; ranged from 19.83 ± 18.87t C ha^{-1} (*Muehlenbeckia complexa* shrubland alliance) to 318.62 ± 278.8 t C ha^{-1} (Exotic broadleaf woodland), with an average 175.58 + 99.93 t C ha^{-1} across all plant community types (see all results in Appendix Table 3; Appendix Table 4; Appendix Table 5). Total carbon stock densities from the three measured carbon pools (aboveground, belowground, and soil) were greater in the woodland alliances than in the shrubland alliances; the two exotic-dominated woodland alliances, the *Pinus* spp. woodland alliance and the exotic broadleaf woodland, had the highest total carbon stock per hectare (Figure 3; Appendix Table 3). Although a significant difference in carbon stock per hectare was observed between the two native shrubland alliances and the exotic-dominated woodland alliances, neither of these groups were significantly different from the native woodland alliances (Figure 3A).

The greatest carbon pool was aboveground biomass (71.62 %), followed by belowground biomass (17.90 %) and soil carbon (10.47 %) (Appendix Table 3). Soil carbon stocks showed the lowest variation among all plant communities (Appendix Table 3) compared to the other two carbon pools.

Aboveground biomass carbon was shown to be more closely associated with stem size than species richness and abundance (Figure 3B). In total, smaller diameter stems that appeared more frequently in the sample plots produced a lower total aboveground carbon stock than larger diameter stems that occurred less frequently (Figure 3C).

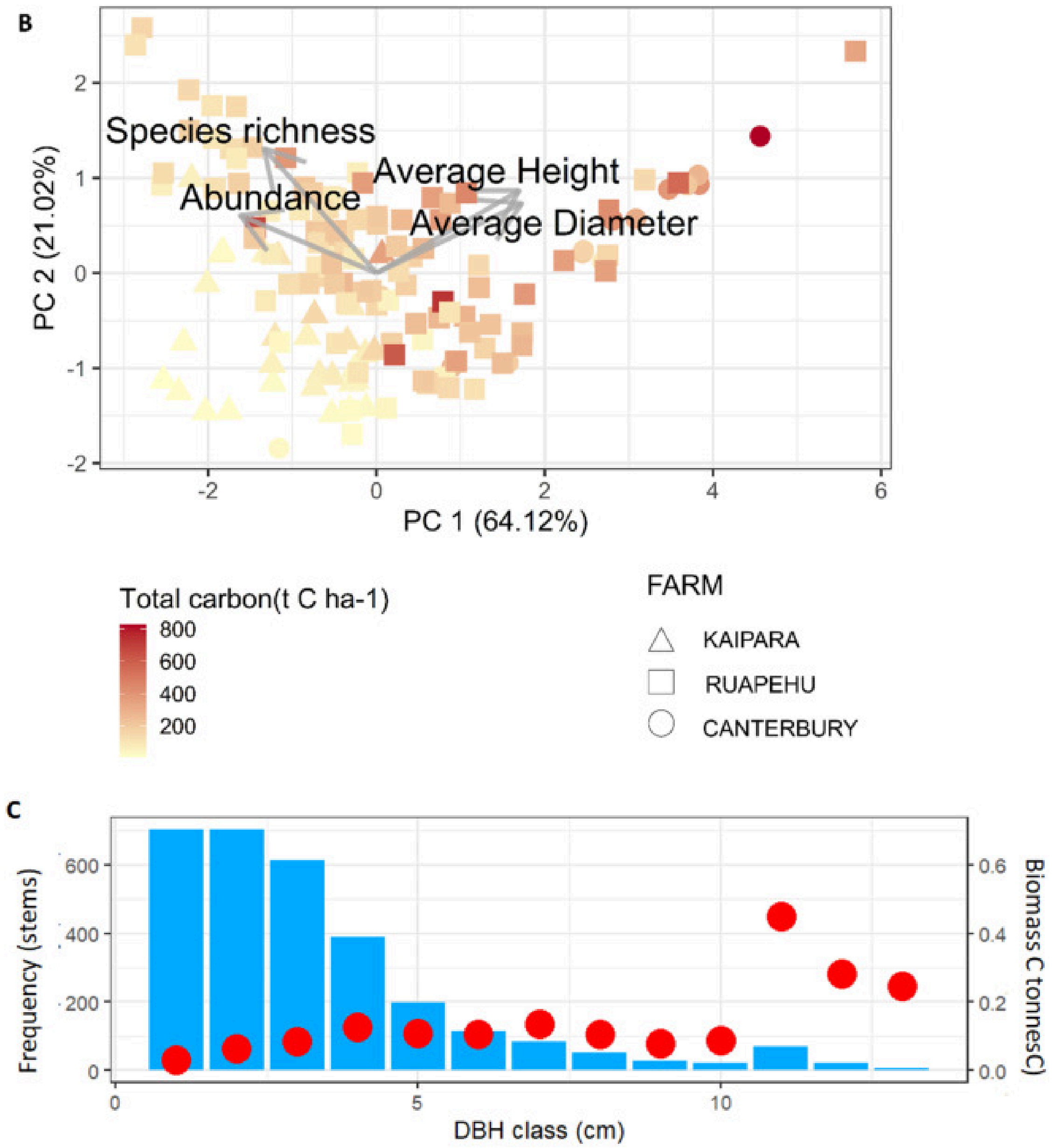


Figure 3. The average total carbon stock per hectare for each plant community type, and the strong relationship between total carbon stock per hectare and stem size: A) Boxplots of total carbon stock per hectare (t C ha−1). The numbers on A represent the mean value for each plot. The bottom and the top of each box represent the 25th and 75th percentiles, the thick band in the box represents the median, the whiskers represent 1.5 times the interquartile range, and the black dots represent outlier values that are bigger than the upper whisker and less than the lower whisker. The means sharing a letter are not significantly different (according to Tukey test); B) Principal Component Analysis (PCA) of the variables used to estimate the total carbon stock densities from all carbon pools on the Kaipara, Ruapehu, and Hurunui farms: stem diameter (cm), stem height (m), species richness, and abundance. Points on B show sample plots, variable loadings on the PC1 and PC2 are shown by blue and grey arrows: point shapes represent farms, and colour gradients represent mean aboveground carbon stock per hectare (t C ha−1); and C) Proportion of total biomass carbon stock contributed by each DBH class: the bar plot shows the

frequency of stems by DBH class, and the red dots shows the total carbon stock of all trees in all plots within each diameter class (1000 tonnes C).

3.2.3 The relative potential multifunctionality of plant community types

A relationship was observed between multifunctionality scores and plant diversity. Native-dominated plant communities showed higher multifunctionality and diversity scores compared to exotic-dominated plant communities (Figure 4). Muehlenbeckia complexa shrubland alliance that had the highest abundance had the highest multifunctionality score, while Pinus spp. woodland that had the lowest richness and abundance had the lowest multifunctionality score (Figure 4 A, B and C). In communities with low species diversity, the number of ecological functions inherent to each species had a greater influence on multifunctionality. For example, a low abundance, a low species richness and a low multifunctionality score were observed for Pinus spp. woodland, which mostly consisted of species with fewer ecosystem functions (Pinus spp. only had a function in timber provisioning; Appendix 2, Figure 4).

The sum of Z-score (Figure 4C) represents overall plant community multifunctionality scores, ranging from 20.78 (Pinus spp. woodland) to 54.95 (Coprosma propinqua/Pseudopanax arboreus woodland alliance). Plant communities with more individual stems and associated ecosystem functions generally scored higher on total multifunctionality. The multifunctionality scores of shrubland alliances and native-dominated woodland alliances were higher than exotic-dominated woodland alliances that were relatively low in stem (Figure 4). The exotic-dominated woodland alliances had a lower overall multifunctionality score compared to shrubland alliances and native-dominated woodland alliances due to a higher number of stems of introduced woody plant species that have become invasive weeds and a higher value of community flammability, which lowered the total area of multifunctionality.

A relationship was found between multifunctionality scores and plant diversity. Native-dominated plant communities showed higher multifunctionality and diversity scores compared to exotic-dominated plant communities (Figure 4). The highest multifunctionality scores were found in the *Podocarpus totara*-*Vitex lucens* woodland alliance (Figure 4B). In communities with low species diversity, the number of ecological functions inherent to each species had a greater influence on multifunctionality. For example, low species richness was found for *Pinus* spp. woodland, which consisted of species with low ecosystem function (timber provisioning; Figure 4G). In contrast, the low species richness of *Muehlenbeckia complexa* shrubland alliance had a higher multifunctionality score due to the presence of species with multiple ecosystem functions (e.g., *Melicytus ramiflorus* had three ecosystem functions, *Coprosma propinqua* and *Muehlenbeckia complexa* had two functions; Appendix

Table 2; Figure 4.D).

Figure 4. Bar plots and a radar graph present results for eight different plant communities: A) A bar plot illustrates the abundance stems per plot of eight plant communities; B) A bar plot shows the species richness stems per plot of eight plant communities; C) A bar plot displays the sum of Z-scores for nine ecosystem functions in eight plant communities; and D) Radar graphs depict the sum of Z-scores for 10 ecosystem functions of eight plant communities. The Z-scores of all ecosystem functions are plotted on the radar graphs, from center top to right top for eight plant community types, of TC (carbon sequestration), TI (timber provision), GS (gully stabilization), EC (soil erosion control), WE (wind erosion/soil surface erosion control), SH (shelter), BF (bird food), NS (enhancing native species), I_WD (fewer weeds), and I_FL (lower community flammability). All values on the radar graphs are relative, calculated as Z-scores with a mean of zero and a standard deviation of one, and rescaled from zero to 10. The color of the bar plots and the radar graph depict the plant community. The complete dataset can be found in Appendix Table 6 and Appendix Table 7.

3.3 Discussion

Our research on New Zealand sheep and beef cattle farms illustrates the necessity of incorporating native trees and shrub to achieve multifunctionality on farms that contain woody plant communities. In doing so, we have identified NbS actions that can help landowners enhance multifunctionality on their farms to deliver multiple co-benefits (as per IUCN, 2020; UNEP, 2022). We observed possible relationships among carbon stocks, woody plant species richness, and multiple ecosystem functions across three New Zealand sheep and beef cattle farms. Carbon stocks were strongly and consistently related to stem diameter and height, rather than abundance or woody plant richness; however, plant richness and the presence of key species were essential for high relative multifunctionality on farms. Combining multiple species with different ecosystem functions can result in a multifunctional land use that meets as many ecologically- and socially-beneficial outcomes as possible. This increased diversity and function is likely to increase farm resilience in the face of multiple land management challenges such as climate change. Thus, our findings have implications for achieving multiple benefits for the management of woody vegetation in these agroecosystems. While this study provides insights into the potential ecosystem service provision of native species on farms, the assessment of ecosystem services was binary rather quantitative, which may not capture the quantity or quality of ecosystem service provisions or weigh some services more highly than others. However, it provides a relative comparison of the multiple ecosystem functions provided by the woody vegetation in productive landscapes.

3.3.1 Carbon estimates in farms and natural ecosystems

Our research showed comparable carbon stock values to those of other studies conducted in Aotearoa New Zealand. The average total carbon stock per hectare in our study was 293.48 ± 200.01 t C ha^{-1} for the exotic-dominated woodland alliances, 192.84 ± 132.86 t C ha^{-1} for the native-dominated woodland alliances, and 64.31 ± 66.96 t C ha^{-1} for the shrubland alliances (Appendix Table 5). Compared to the study of aboveground carbon stock per hectare in natural ecosystems, the average aboveground biomass in native-dominated woodland alliances (145.20 ± 106.26 t C ha^{-1}) and in the exotic-dominated woodland alliances (202.87 ± 160.33 t C ha^{-1}) in our study (Appendix Table 5) were higher than the aboveground carbon stock per hectare in secondary forest (67.15 t C ha^{-1}; Holdaway et al., 2017), of the old growth forest (200.65 t C ha^{-1}; Holdaway et al., 2017), and native forest (169.1 ± 18.4 t C ha^{-1}; Coomes et al., 2002). The average aboveground carbon stock per hectare of the shrubland alliances (44.61±53.02 t C ha^{-1}; Appendix Table 5) was also higher compared to the aboveground carbon stock density in native shrubland community in the natural landscape (53.80 t C ha^{-1}; Coomes et al., 2002). The average aboveground carbon stocks per hectare of native and exotic-dominated woodland alliances in our study farms were higher compared to the national estimate carbon stock per hectare of woody vegetations on farms, that was between 13.05 - 60.57 t C ha^{-1} for grassland with woody biomass (Ministry for the Environment, 2020a).

Soil carbon contributed only a small percentage of the total carbon stock on these sheep and beef farms (14.99 ± 5.44 t SOC ha-1 under shrubland alliances and 19.19 ± 5.69 t SOC ha-1 under exotic-dominated woodland alliances and 20.39 ± 6.96 t SOC ha-1 under native-dominated woodland alliances). This result is consistent with the range of values presented in another study on soil carbon stocks under shelterbelts in agricultural landscapes in New Zealand (20.00 ± 12.1 t SOC ha^{-1}; Welsch et al., 2016). Grazing that commonly occurs within woody plant communities on sheep and beef cattle farms likely has an effect on soil carbon sequestration by increasing disturbance, which may result in lower sequestered soil carbon. Further comparative investigations on soil carbon stock in sheep and cattle farms are needed to understand the observed variation in soil carbon.

The link between aboveground carbon stocks and tree size has been well established (Lutz et al., 2018) . In our study, as expected, plots in the woodland alliances contained trees of the larger sizes, compared to plots in the shrubland alliances (Table 1); therefore, woodlands consistently contained more carbon per ha than shrublands (Figure 3). Among the woodland alliances, exotic trees were, on average, larger than native trees (Table 1). Our analysis showed that, for explaining variation in carbon stocks, tree size and abundance were more important than richness (Figure 3. D). These findings are similar to the results of a previous study

(Shirima et al., 2015) showing that tree diameter is an important predictor of aboveground biomass carbon stock in Tanzania. Similarly, Chave et al. (2005), estimated the carbon stock of the aboveground biomass on three continents (America, Asia and Oceania) and showed that trunk diameter and height to be important predictors of aboveground biomass.

Retaining large trees is crucial when tree size is the most important indicator of carbon storage, which is consistent with conclusions from prior research (Ali et al. 2019). In this study, however, the majority of remaining native trees were small-sized trees, whereas the majority of the large-sized trees were exotic tree species that were purposefully planted. Thus, it is important for policy makers and landowners to prioritise the retention of existing trees, especially large and tall trees and species that have the potential to reach large diameters. When implementing restoration, managers also should prioritise the establishment of these type of species, especially native species that can grow larger than exotic (Williams & Norton, 2012).

Managers should also aim to reduce the effects of disturbances, such as wind exposure, grazing, and drought, that slower growth and therefore reduce carbon sequestration (Carswell et al., 2009). The retention and restoration of native species, given the multifunctionality they bring, is a long-term land and biodiversity management strategy that should be promoted to farmers. For example, although the exotic trees on our study farms were very fast-growing species, e.g., P. *radiata*, growth rates can be nearly matched by some native species, e.g., P. *totara*, and maximum potential size and life span is by far exceeded by other native species, e.g., D. dacrydioides, when properly managed (Carswell et al., 2008; Kimberley et al., 2014).

3.3.2 Co-benefits of carbon and biodiversity: The multiple outcomes of managing for multifunctionality

Our study shows that native-dominated woodlands and shrublands provide greater multifunctionality than exotic-dominated woodlands (Figure 4; see Appendix Table 2 for ecosystem functions of each species). This is likely because some species contribute to multiple ecosystem functions simultaniously. For example, *Muehlenbeckia complexa* shrubland alliance had low woody species richness but had a relatively greater multifunctionality due to the presence of *Muehlenbeckia complexa* that has multiple functions: soil erosion and wind erosion reduction. In contrast, the exotic *Pinus* spp. plantations have low diversity and low multifunctionality because of the limited functions inherent to this species, i.e., timber provision (Figure 4). Interestingly, some native species on the three farms have potential ecosystem functions that are similar to exotic species. For example, the native *Podocarpus totara* that has multiple functions, including use a source of timber, could be a better alternative to exotic *P. radiata*, which is considered invasive (Bergin & Kimberley, 2014); or the native *Kunzea* spp that benefits birds and bees, as an alternative to the exotic *Populus* spp for shelterbelt plantings (Mackay-Smith et al., 2021). Conserving and restoring native species that provide multiple ecosystem functions increases both native biodiversity and the multifunctionality of the landscape. Therefore, policy incentives that preserve and restore multifunctionality, rather than only provision of carbon stocks are desirable in Aotearoa New Zealand farms to realise their potential for enhancing native biodiversity and its funtioning. This woud disincentivise the conversion of native ecosystems to intensive monocultures (e.g., *Pinus* spp. plantations), which have multiple social and ecological negative consequences (e.g. invasive species, post harvest, myrtle rust) (Hulme, 2020b; Marden et al., 2014; Toome-Heller et al., 2020). Landowners needs to carefully evaluate the tradeoffs between these two aspects—carbon stock and multifunctionality—to achieve win-win solutions for the economy and biodiversity. Careful and informed species selection, for both retaining and planting, will be vital to maximise the benefits of multifunctionality.

Our multifunctionality scores were designed to illustrate the set of functions that will help landowners on assessing the multifunctionality of different plant community types. Some plant communities received high scores on several ecological functions, whereas others received only minimal or medium scores. This knowledge will assist landowners in determining which plant communities are most suited for specific purpose. For example, managing *Pinus* spp. and exotic broadleaf woodland plots will increase the farm's carbon stock, while shrubland management will benefit biodiversity. However, it was necessary to find a balance between multiple ecosystem services to preserve the farm's long-term sustainability (Kremen et al., 2018); in other words, multifunctionality and sustainability should receive priority above maximal production. For example, although the *Pinus* spp. woodland alliance stored more carbon, it did so at the expense of other ecosystem services (gully stabilization, erosion management, and higher flammability) (Figure 4 H), making the plant community more vulnerable to recurring extreme weather and less beneficial to biodiversity. On the other side, having a larger shrubland community on the farm may reduce the farm's economic value, despite the benefits to birds and soil health. Although converting to a more sustainable system may reduce average yields, an ideal sustainable system should be both multifunctional and more adaptable to change while still being able to produce, as seen as example by plant community Podocarpus totara-Vitex lucens woodland alliance (Figure 4 A), where most functions were distributed relatively evenly, despite having few optimal functions.

When deciding what type of plant community to prioritize on land, our research shows that land managers should consider trade-offs. Our study should therefore be useful to land managers when choosing what type of combination is needed (Norton et al., 2020). If increasing carbon stocks on farms is the only management goal, then the land managers could retain or plant exotic-dominated plant community. However, if the land manager wants to achieve multiple objectives that can lead to a more sustainable farm system, then the land manager may seek a strategy of combining multiple plant community types depending on the functions needed. Alternatively, the land manager could prioritize plant communities that can achieve multiple objectives simultaneously. More importantly, although pasture can have a significant soil carbon stock of up to 110.4 tonnes of carbon per hectare (Wall et al., 2021), it is important to consider that converting patches of woody vegetation to a land cover without biomass cover, such as pasture, will result in the loss of the biomass carbon that is shown to be significant in our study. Therefore, such actions should be avoided, as these areas of woody vegetation retain significant carbon stocks per hectare not only from soil, but also biomass carbon. Farmers need greater support for the retention or enhancement of woody vegetation that can provide a pathway for farmers to receive carbon benefits and biodiversity conservation at a lower cost than new planting (Gilroy et al., 2014; Norton et al., 2020). Landowners frequently search for possibilities to conserve and/or restore existing woody vegetation in their farm planning. This involves selecting the appropriate plant community for the appropriate functions to be conserved. The common practice of preserving woody vegetation on steep slopes or along stream corridors (Maseyk et al., 2017), to minimize erosion on farms, thereby protecting soil resources and ensuring water quality as well as increasing carbon storage, is an example of a policy that needs to be encouraged and improved.

However, continuous grazing, which is typical in pastoral landscapes (Pannell et al., 2021), has likely affected the quality and quantity of native woody biomass in agricultural land (Norton et al., 2020). Fragmentation might also reduce the ability of remnants of native woody vegetation to maintain structure and species within the woodland vegetation patches (Norton & Miller, 2000). The common practice of selective logging would also threaten the sustainability of the structure because it has removed larger trees from the native-dominated woodland on agricultural landscapes. This is similar to other studies in other countries, for example in Brazil (Nair & Kumar, 2011) and in Colombia (Gilroy et al., 2014) where plant composition and structure of woodlands in agriculture landscape was lower compared to the natural ecosystems, resulted on a lower aboveground biomass. Tree species present in pastoral systems in Central America, without proper conservation, will continuously be degraded (Harvey et al., 2011). Although this has been observed overseas, this is also likely to happen in Aotearoa New Zealand (Norton & Miller, 2000b). Therefore, conserving the remaining trees and shrubs is an important action to retain the multifunctionality.

Our analysis measured the standing stocks of woody vegetation on sheep and beef cattle farms, which could serve as a baseline for meeting national requirements for monitoring increases in carbon sequestration, while also providing opportunities for additional multifunctionality through the presence of native biodiversity. On the other hand, the sheep and beef cattle farms with existing woody vegetation are situated on lowland areas that are suitable for other land uses such as agriculture and monoculture plantation forestry (Norton et al., 2020). It is important to note that regenerating native forests in rural New Zealand that often occurred on these sheep and beef cattle farms, are at risk of conversion to commercial exotic plantations due to the potential for generating greater carbon-credits through exotic forestry (Norton et al., 2020). Land use with lower economic value, such as conventional sheep and cattle farms, may also be impacted by the need for land for carbon farming (West, et al., 2020). Some New Zealand farms have been purchased with the intention of converting them to carbon forestry, or where exotic radiata pine is planted for the purpose of obtaining carbon credit from the emissions trading scheme without the intention of harvesting the trees (Orme & Orme, 2021). While the current policy does not specifically protect existing woody vegetation on farms, understanding of the importance of keeping and managing trees and shrubs in the agricultural environment, as well as preventing their conversion to reduced multifunctional land use, should be encouraged. For the maintenance or improvement of woody vegetation, farmers require greater assistance, which can take the shape of incentives or innovative best practices (Suryaningrum et al., 2021). However, developing an appropriate and acceptable policy, including incentives for farmers to diversify their farms for climate change adaptation and mitigation, is challenging and requires additional research and effort to tailor the policies to farmers' needs (Zonneveld et al., 2020).

3.4 Conclusions

Overall, we have shown that the carbon stock per hectare currently held by woody vegetation patches on sheep and beef cattle farms is significant, and that the multifunctionality varies widely, depending on the species composition within patches. Management practices that conserve and restore native woody vegetation on farms can provide ecosystem benefits, potentially higher carbon stocks relative to exotic plantations, and the opportunity to increase the economic value of the land. Future research should aim to quantify the quantity and quality of ecosystem service provisions to provide a more comprehensive understanding of the trade-offs and co-benefits associated with conserving and restoring woody vegetation on farms. We further advise that, in order to better achieve win-win outcomes of carbon and biodiversity conservation for agricultural systems from the existing woody vegetation, a comprehensive understanding of trade-off dynamics and the integrated implementation of strategies needs to be used to choose the most feasible combination of co-benefits and trade-offs between ecosystem functions. Based on current findings, we recommend the following practices: 1) recommendations to landowners to retain the existing trees and shrubs species to conserve their multifunctionality and carbon stock, with additional management techniques to maintain the quality of the existing woody vegetation and large trees; and 2) when conserving existing woody vegetation and/or if the landowners want to plant more trees, careful species selection and species composition are necessary to enhance multifunctionality. In contrast to non-collaborative ecological restoration initiatives that may not explicitly target biological diversity, these recommendations can help support landowners to identify the NbS actions that explicitly include conserving and/or planting woody vegetation on farms to protect the long-term ecological integrity of the area while achieving multiple co-benefits (IUCN, 2020). If woody vegetation on farms is not valued, it risks being converted to more ‘productive’ land use (e.g., intensive dairy farming, residential areas, pasture for livestock, pine plantations) (Allen et al., 2013).

Chapter 4 Simulating the potential impacts of farmland restoration and revegetation intervention scenarios on landscape-scale ecological multifunctionality

Detrimental losses of biodiversity and ecological function have been occurring in agricultural landscapes globally for many decades due to the ongoing impacts of land use and climate change (e.g. Tscharntke et al., 2022). The reintegration and restoration of woody vegetation cover in these landscapes has therefore become recognized as an important strategy for improving specific ecological outcomes and enhancing the overall multifunctionality of farms (Case et al., 2020c; Kuyah et al., 2016). In Aotearoa New Zealand, for example, ways to practically address the significant and ongoing threats to native species, water, and soils, especially across lowland agricultural areas, are urgently required (e.g. Ministry for the Environment & Stats NZ, 2019). Woody vegetation on farms provides various ecosystem functions and biodiversity conservation benefits, which can be beneficial for farms (Case et al., 2020c; Cunningham et al., 2015a), including for erosion control, water quality mitigation, enhanced carbon sequestration, improved animal welfare, and increased and better-quality wildlife habitat. However, how and where to intervene in agricultural landscapes, to most effectively revegetate and restore ecological functioning, is complex and requires a landscape design approach that is based on ecological principles (*sensu* Landis, 2017). Thus, the development of approaches to explore possible scenarios and trade-offs across different landscape contexts are critically needed.

Of particular interest is how the integration of woody vegetation into agricultural production landscapes could generate improved co-benefits, such as for both biodiversity conservation and carbon sequestration (Schoeneberger, 2009). As a result of climate change, there may be more uncertainty about agricultural output and dairy products, which may make landowners wish to diversify their investments by planting trees to receive carbon credit incentives, often known as "carbon farming" (e.g., Sharma et al., 2021), or – by planting forest patches that can also qualify for carbon credits in a variety of trading schemes (Cunningham et al., 2015b). Adding tree cover to agricultural landscapes can also promote biodiversity, increase agricultural productivity, reduce vulnerability to exotic species invasion, and increase ecological resilience to pressures such as climate change (Hooper et al., 2005). Despite the potential for achieving multiple benefits from tree plantings, there have been few studies that have investigated how to spatially target and assess revegetation efforts with a specific focus on simultaneously achieving multiple ecological objectives, such as the improvement of landscape connectivity or the expansion of native habitat availability, along with carbon sequestration. Indeed, research has primarily focused on how tree integration could enhance a single specific outcome, such as for increasing carbon storage or regulating runoff. However, focusing on one potential outcome, such as the use of monocultural forests for carbon storage, could have unintended negative consequences for other key values such as native biodiversity (Lindenmayer et al., 2012a), including possible reductions in species richness (Lautenbach et al., 2017) and increased introductions of invasive taxa to the area (Lindenmayer et al., 2012a).

Since the arrival of European settlers in Aotearoa New Zealand in the mid-1800s, just over half of country's land's original native forest area has been cleared for agricultural production purposes (MacLeod & Moller, 2006). In more recent decades, a significant amount of this agricultural land has undergone further conversion to intensive agricultural systems, such as dairy farming, following similar trends globally (MacLeod & Moller, 2006). These activities are mainly concentrated in lowland areas in the country and have caused significant biodiversity loss (Walker et al., 2006), increased greenhouse gas emissions (Kirschbaum et al., 2012), and increased fragmentation of native forest habitat (Moller et al., 2008). Over the past decade, a number of measures have been established by the government to try and improve environmental outcomes and the reintegration of native woody vegetation into these production landscapes has become a key strategy to achieve this (Case et al., 2023). The One Billion Trees (1BT) program, for instance, was established in 2018 to facilitate the goal of planting a billion trees by 2028 (Ministry for Primary Industries, 2020c); since the 1BT Program's commencement in 2018, there has been considerable interest in reforesting farmland as a way to reduce greenhouse gas emissions (West, et. al, 2020). More recently, the Aotearoa New Zealand Climate Change Commission, tasked to provide advice to the government regarding how to achieve carbon neutrality by 2050, has also suggested 2.8 million hectares of farmland out of 11.5 million hectares could be suitable for afforestation (Nash & Shaw, 2022). A forthcoming policy on the National Policy Statement for Indigenous Biodiversity (NPSIB), which contains objectives and policies to identify, protect, manage, and restore indigenous biodiversity, will map important native habitats, including those in farm areas, indicated an ongoing interest in native vegetation on farms (Ministry for the Environment, 2022b). Additionally, a new permanent forest category of the ETS would start allowing both exotic and indigenous forests to be registered in the ETS as of 2023 (Ministry for Primary Industries & Ministry for the Environment, 2022). Thus, there are multiple, critical motivating factors that are driving interest in woody revegetation across the country. However, it is not clear whether the trade-offs between carbon and biodiversity measures from this initiative are being recognized comprehensively as the results of restoration and revegetation.

Research is therefore urgently required to understand possible tradeoffs in ecological multifunctionality as a result of decisions made by landowners regarding revegetation, driven by some of the policies mentioned above. Different restoration and revegetation interventions might create a variety of landscape outcomes in terms of carbon, biodiversity, and connectivity in Aotearoa New Zealand agricultural landscapes (Maseyk et al., 2018). It is clear that some policies are currently leading to more exotic afforestation (Suryaningrum et al., 2021), and such policies have the potential to replace the existing native vegetation that is typical of Aotearoa New Zealand's agricultural landscape with exotic commodity plants (Norton et al., 2020). Although there is research on how tree planting such as 1BT will improve vegetation cover in the future, such as that conducted by West et al. (2020a), it was conducted exclusively on exotic plants. There has been no spatial assessment of the impacts of reforestation with native species, while taking into account the existing vegetation cover. This is important since agricultural land, notably sheep and cattle farms, has been shown to comprise a substantial amount of the country's remaining native vegetation, which consisted primarily of forest types that are typically underrepresented on public conservation land (Pannell et al., 2021).

Spatial modelling provides a means to realistically evaluate the structural and functional outcomes of tree planting in order to forecast how the tree planting scenario would impact the landscape configuration. To date, there have been few studies that have focused on the landscape's current spatial configuration and how it will affect the results of tree planting policy. Some models, such as Marxan (Watts et al., 2009), LUMASS (The Land Use Management Support System; Watts et al., 2009), and LUWES (Land Use Planning for Low Emission Development Strategy; Lawlor & Swan, 2014), have been developed to assist in the formulation of policies aimed at optimizing the many advantages derived from land use planning including those on carbon farming. Despite incorporating several purposes into spatial planning, the majority of these models were based on zonation, where the landscape areas were planned separately based on each function (e.g., production area, conservation area, residential area, etc.). Further models that can spatially anticipate the future landscapes following the intervention are still required. Some of the existing spatial planning models do not take into account the fact that other benefits may be enhanced or decreased as a result of trade-offs or synergies when one benefit is maximized by modifying the location and type of landscapes (Bodnaruk et al., 2017). Nevertheless, the lack of a comprehensive analysis of the numerous benefits of integrating trees into a landscape can hinder the achievement of a multifunctional and sustainable landscape.

This study presents a modeling framework that considers the impact of incorporating additional native vegetation cover, or restoring existing lower-quality vegetation types, on landscape woody vegetation patterns and ecological multifunctionality. Specifically, I present a new model for this purpose: SLIPSTReaM (Spatial Landscape Intervention and Pattern Scenario Testing for Revegetation and Multifunctionality). The SLIPSTReaM model was developed to enable the exploration of spatial revegetation and restoration intervention possibilities and outcomes in farm landscapes. Changes in patterns of vegetation configuration and composition generated by the model are evaluated in terms of impacts on landscape-scale functional indicators associated with biodiversity, carbon, and putative animal habitat availability and connectivity. Specifically, for three different Aotearoa New Zealand farm landscapes, I spatially explore and evaluate a range of possible intervention scenarios involving the conservation of existing high-biodiversity-value native vegetation, the restoration of existing, lower-priority woody vegetation to tall native forest, and the revegetation of bare gully areas with native tall forest.

The three main objectives of this study were to: (1) explore the behavior of the SLIPSTReaM model under different model parameter settings, (2) assess and understand how initial landscape values and spatial configuration affect model outcomes, and (3) quantify and compare outcomes and multifunctionality scores emerging from different model scenarios of different types of revegetation and restoration interventions in different landscapes. Ultimately the aim is to provide insights on the impacts of woody restoration and revegetation in farm landscapes and how this can be used to support spatial planning and design of multifunctional landscapes. Further, being able to assess the possibly complicated tradeoffs among ecological function indicators would provide useful information for reducing uncertainty when evaluating land management policy alternatives.

4.1 Methods

4.1.1 The SLIPSTReaM model

The purpose of the Spatial Landscape Intervention and Pattern Scenario Testing for Revegetation and Multifunctionality (SLIPSTReaM) model is to spatially simulate the addition of non-production, woody vegetation into a real agricultural landscape, under specific restoration and revegetation scenarios. Further, it quantifies the impact of these scenarios on five ecosystem indicators related to carbon storage, relative biodiversity potential, and the amounts, fragmentation and connectivity of habitat patches.

Overview

The model is a spatially explicit, stochastic, GIS-based simulation model, developed within the ArcGIS Pro 2.7.3 GIS software (ESRI, 2021) using ArcGIS API for Python 3.0.1 scripting in a Jupyter Notebook environment (Jupyter Team, 2015). Thus, the model makes direct use of the geoprocessing tools and functionality afforded by the ArcGIS Pro software via its ArcPy Python module to perform the modelling (ESRI, 2022).

Restoration, revegetation, and a combination of these are the main intervention actions implemented in the model to achieve woody vegetation pattern change in the landscape.

‘Restoration’ actions are implemented via the relabeling of existing woody vegetation patch types from those of a lesser stature or diversity (e.g., early successional shrubland) or exotic vegetation type (e.g., pine forest, gorse shrubland) to an assumed more biodiverse, mixed-native tall forest class. ‘Revegetation’ actions are implemented via the establishment of novel mixed-native tall vegetation patches on more marginally productive pasture areas occurring in relatively steep gully areas. Thus, the outcomes of particular restoration and/or revegetation actions in the model depend on: (i) the types and amounts of existing vegetation in a landscape, (ii) the prevalence of steep gully areas in a landscape, and (iii) the spatial distribution (landscape configuration) of the vegetation types and gullies. The output of a model iteration is therefore a consequence of how the restoration/revegetation actions implemented in a scenario change the spatial configuration, amount and composition of woody vegetation in the landscape compared to the current, baseline situation. The landscape ‘map’ of woody vegetation patches resulting from a model simulation run is summarized in terms of the changes in five ecosystem indicators (see ‘Monitored state variables’ section below) being tracked.

The spatial outcome from restoration and/or revegetation is generated instantaneously in each run of the model; in the real world, these outcomes would be achieved by a variety of mechanisms, such as via management of vegetation patches (e.g., clearing and in-planting), natural regeneration processes, assisted natural regeneration, and broad-acre revegetation on pasture soils (Case et al., 2023), although these mechanisms are not specified in the model. Thus, each model run implements an immediate change in the spatial distribution and composition of woody vegetation in the landscape based on the specified restoration or revegetation rules (or combinations of these). The model therefore has no temporal component and assumes that some relevant amount of time has passed in each iteration to achieve the desired, final restoration or revegetation outcome.

4.1.2 Model set-up and GIS datasets

Restoration and/or revegetation actions in the model are carried out on a selection of target areas of existing vegetation (in the case of restoration) or on bare gullies areas (for revegetation). Thus, the model requires three primary datasets to initialize the model: (i) a relatively fine-resolution, classified vegetation polygon layer, using vegetation type categories specific to each agricultural landscape (Figure 5A), (ii) a gully polygon layer (Figure 5B and C), and (iii) a GIS point dataset, spaced on a regular grid design, that represents the potential intervention locations available in the landscape (Figure 5D).

Grid-cell based representations of the vegetation and gully datasets are used to perform the modelling; existing vegetation and gully polygons are represented as a grid or ‘fishnet’ of 50

× 50-m (0.25-ha) square polygon cells (hereafter called ‘patches’). The purpose of this representation of the data is to ensure that the patches being targeted and selected for restoration/revegetation are of a standardized size, are of sufficiently fine resolution to represent most small vegetation components occurring in the landscape, and reasonably mimic the scale at which real-world restoration/revegetation actions might be carried out. The model requires that the vegetation and gully layers be combined into one input layer, which requires some geoprocessing to prepare the layers for modelling. Therefore, each 0.25-ha patch (grid cell) in this input layer contains information on the vegetation type occurring at each cell location (Figure 5A) and/or whether the cell is in a gully (Figure 5B); some of the cells will therefore be classified as ‘bare gully’ (do not contain woody vegetation but occurring in a gully) (Figure 5C) and would become targets for revegetation.

A GIS vector point dataset is required to specify the potential locations (intensity) at which restoration/revegetation might occur (see next section) in the landscape. A regular grid of points, generated in the GIS at a 100-m spacing across a landscape, is used for this purpose (Figure 5D).

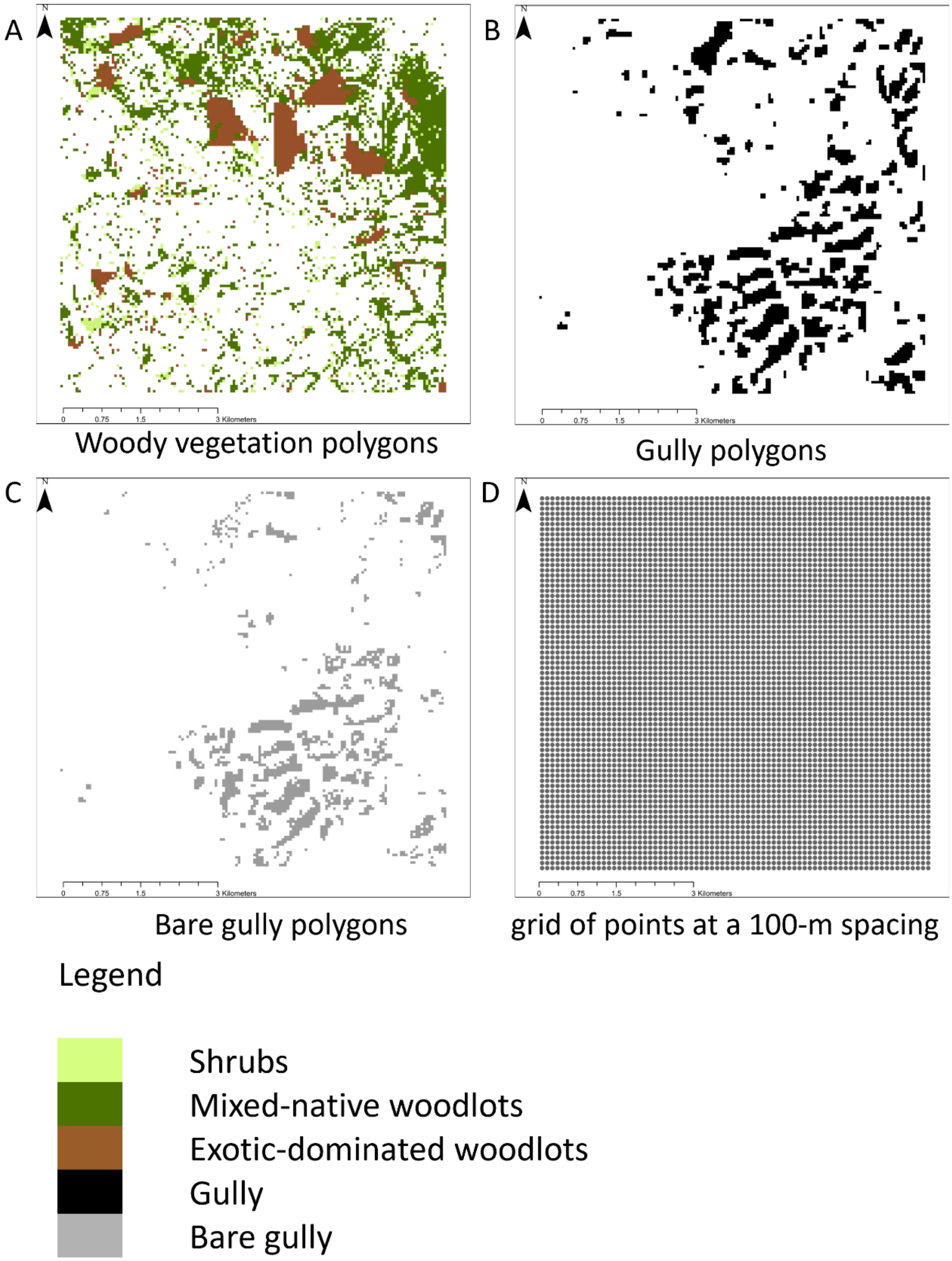


Figure 5 An example of scenario model input layers in the Kaipara landscape: A) woody vegetation layer with three main land cover classifications: shrubs, exotic-dominated woodlots, and native-dominated woodlots; B) gully polygons; C) bare gullies; and D) polygons points within 'fishnet' of 50 50-m (0.25-ha) square polygon cells created at 100-m spacing.

User-defined parameters

Restoration/revegetation rules

In reality, revegetation actions involve new tree plantings in habitats where natural regeneration and woody succession are unlikely to occur because of a loss of native seed supplies (through dispersal or the seed bank) and/or when the soil conditions have deteriorated considerably. Restoration, on the other hand, can involve either passive or active establishment of seedlings in existing vegetation areas where the aim is to restore the patch to a new vegetation type with enhanced ecological or biodiversity values (Case et al., 2023). The SLIPSTReaM model implements both revegetation and restoration actions via a ruleset parameter that dictates the woody vegetation types (in the 0.25-ha patches) that will be targeted for a restoration, revegetation, or combined intervention (Figure 6). Thus, these target ‘rule sets’ define the type of scenarios that will be simulated for a given set of model runs.

Intervention intensity

This parameter controls the overall number of locations in the model landscape at which restoration/revegetation will be potentially carried out. The 100-m resolution regular point grid input dataset is used to define the spatial domain of possible intervention locations. At each model iteration, a random sample of a user-specified intensity (e.g., 10 %) is selected from the available point locations, with the selected set become the seed locations at the landscape scale for restoration/revegetation activities (Figure 6).

Aggregation parameter

The aggregation parameter is used by the model in conjunction with the intervention intensity parameter. At each selected intervention location determined by the intensity parameter (above), a radial distance is specified by the user (e.g., 50, 100, 150 meters) that is used as a search distance for selecting target vegetation cells occurring in that local vicinity (Figure 6). Thus, for a given iteration of the model, the restoration/revegetation locations are chosen randomly in the landscape at a specified intensity and can comprise multiple 0.25-ha patches of a target vegetation that are found within a buffer radius of each location. The number of restoration/revegetation patches selected in a given iteration therefore depend on the interplay among the three parameters relative to the spatial configuration of woody vegetation and gully occurring in a given landscape.

Stochasticity and uncertainty

The selection of locations using a random selection process from the 100-m grid of possible intervention points, as described above, provides a mechanism to incorporate spatial, landscape-scale stochasticity for a given restoration/revegetation scenario across a set of iterations. At the patch scale, stochasticity is further incorporated by allowing for measurement uncertainties for two of the main patch-level quantities: carbon density and the potential biodiversity ‘score’.

This recognizes that these two patch-level quantities will probably vary according to some distribution around a set mean value for a given vegetation type. This is specified in the model by modifying the carbon stock density and biodiversity score value in a given patch by a percentage modifier, drawn from a normal distribution with a mean of zero and a specified standard deviation. Thus, the carbon and potential biodiversity score values in a patch have the potential to be increased or reduced by some percentage, determined by a draw from a normal distribution centered on zero for the former, and from a uniform distribution ranging from -1 to 1 for the latter.

The habitat-related model outcomes (total habitat area, percentage of small patches, and mean nearest-neighbor distance) are all emergent quantities determined by the baseline vegetation occurring in the landscape, in addition to the impact of restoration/revegetation activities. Thus, these quantities are ultimately stochastic because they are related to where and how much vegetation is being restored in a given model run.

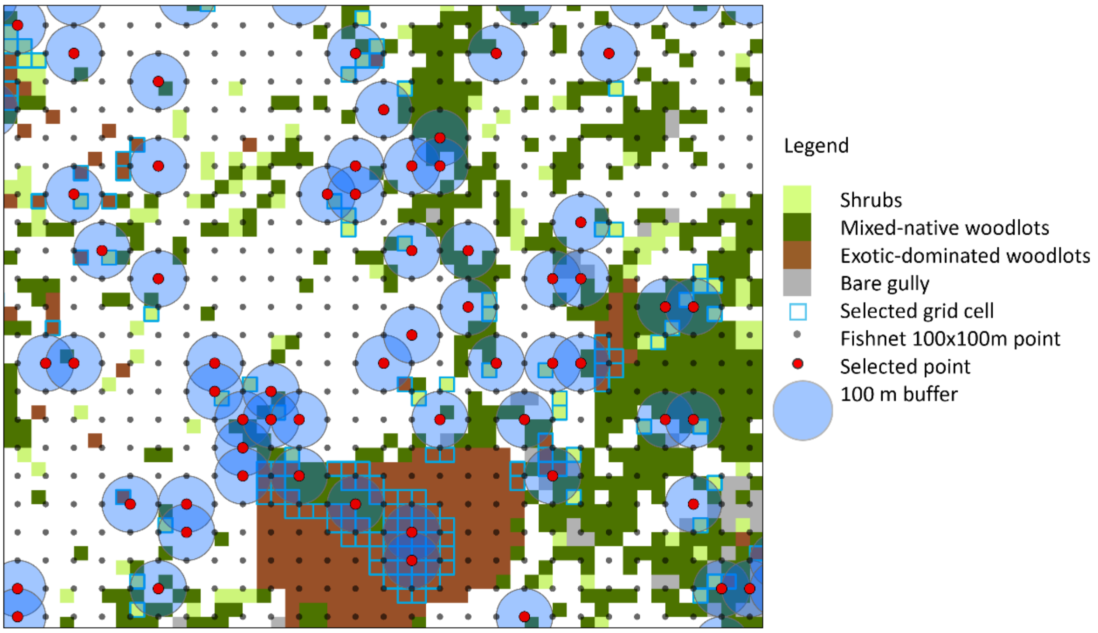


Figure 6 An illustration of how the model uses random points to select intervention locations in a landscape for, in this example, Scenario 4 (restoring mixed native tall forest in existing shrub and exotic-dominated patches). Here, the model is parameterized to use an intervention intensity parameter of 10 % (i.e., 10 % of the 100-m points are randomly selected each iteration as locations for restoration/revegetation) and an aggregation parameter of 100 meters. The red dots illustrate the intervention points selected by the model in this model iteration, while the blue circles illustrate how the aggregation parameter is implemented around those points as buffer zones. The model then selects target vegetation cells that intersect with these intervention locations (the blue-outlined grid cells). The selected intervention cells, in this case, are 'restored' by the model into a mixed-native tall forest and monitored model variables updated accordingly.

Monitored state variables

Five functional indicators are tracked for the model landscape in each iteration of the model: (i) total aboveground carbon stock held in woody vegetation, (ii) a landscape-scale mean biodiversity score, (iii) total habitat area, (iv) the proportion of small woody habitat patches in the landscape as a proportion of small habitat patches, and (v) the mean nearest-neighbor distance between habitat patches.

Carbon stock density

A mean aboveground carbon density (t C ha^{-1}) must be specified for each woody vegetation patch (0.25-ha polygon) in the input layer as part of the layer's attribute table. This value represents the mean carbon density for that vegetation type, as quantified either from literature values or other sources of information. If actual carbon density values per patch in a landscape is known, those could be used. Bare gullies areas receive a null carbon value. The total landscape carbon (tons) is quantified for each model iteration, incorporating uncertainty in carbon quantities in each patch as described above.

Potential biodiversity score

A biodiversity rank score is assigned to each 0.25-ha woody vegetation patch in the GIS input layer's attribute table. This is defined by the user based on an understanding of the relative biodiversity potential or condition associated with a given vegetation type. For example, for the modelling exercise presented here, a scoring system ranging from 1 (least biodiversity potential in bare gully zones) to 5 (high biodiversity potential in native tall forest types) was used. A landscape scale mean biodiversity score is generated at each model iteration, across all patches, incorporating uncertainties computed for patch-level biodiversity scores as described earlier.

The assignment of biodiversity scores for different land-cover classes takes into account various ecological factors, resulting in a spectrum of scores within each class. The assigned biodiversity scores reflect the richness and complexity of ecosystems within each land-cover class, considering factors such as the type of shrubs and the level of canopy coverage. While the 1BT policy prioritizes native species over exotic ones and tree systems over bare land, native species received a higher rank than exotics, and the tree system had a higher rank than bare land and shrubs. Higher scores indicate the presence of diverse native species and complex ecological interactions, while lower scores reflect areas with limited biodiversity due to factors such as lack of vegetation or dominance by non-native species.

1. Bare gully (Score: 1): Bare gully typically exhibits minimal biodiversity due to the absence of vegetation, offering limited habitat and resources for various organisms. This class receives a low biodiversity score of 1.
2. Exotic-dominated woodlots (Score: 2): Exotic patches introduce non-native plant species, offering a slight increase in biodiversity compared to bare land. However, the dominance of non-native species limits ecosystem complexity, leading to a moderate score of 2.
3. Shrubs (Score: 3): Shrubby areas provide better habitat and resources, fostering a variety of organisms. The biodiversity score for shrubby areas varies based on the type of shrubs present:
    a. Tall Native Shrubs (e.g., Kanuka) (Score: 3):** Tall native shrubs, such as Kanuka, offer significant habitat and resources, supporting a diverse range of species. This subcategory receives a high biodiversity score of 3.

b. Native Shrubs (e.g., Matagouri) (Score: 2):** Native shrubs like Matagouri, although beneficial, may not provide as much complexity or habitat as tall shrubs. Hence, this subcategory receives a moderate score of 2.
c. Exotic Shrubs / Gorse (Score: 2):** Shrubs comprising non-native species contribute to biodiversity but at a lower level due to the potential lack of ecological integration. This subcategory receives a moderate score of 2.

4. Mixed-native dominated woodlots (Score: 4 and 5): Mixed native patches represent a diverse range of native plant species, significantly enhancing biodiversity. Within this category, there exists a spectrum of scores based on the complexity of the ecosystem:
    a. Mixed Native - Old Growth Sparse (Score: 3): In areas characterized as mixed native old growth sparse, biodiversity is moderate due to a sparser canopy coverage, offering habitat for specific species. This subcategory receives a score of 3.
    b. Mixed Native - Old Growth Diffuse (Score: 4):Old growth areas with a more diffuse canopy coverage offer increased complexity, supporting a broader range of species. This subcategory receives a score of 4.
    c. Mixed Native - Old Growth Continuous (Score: 5): Old growth areas with a continuous canopy coverage represent the pinnacle of biodiversity within the mixed native patch class. These areas support a diverse array of flora and fauna, resulting in a high biodiversity score of 5.

Total habitat area

The model defines ‘habitat’ as patches meeting either a minimum biodiversity score (e.g., ≥ 4) or a minimum area threshold (e.g., ≥ 2.5-ha). Small patches of high biodiversity ranking can provide sources of critical resources for animals (e.g., birds) in the farm landscape; likewise, larger contiguous patches of woody vegetation of any type contain areas of core habitat and can provide areas for shelter and nesting. To generate putative habitat, the 0.25-ha cell-based vegetation map generated after restoration/revegetation actions in the landscape at each iteration is reconstituted by combining adjacent cells of a similar vegetation type into aggregate patches. These aggregated patches are assessed to determine if they meet mean biodiversity score and total size thresholds; patches that meet these thresholds are considered habitat and exported as a new layer.

Proportion of small habitat patches

As one way to monitor the effect of restoration and revegetation interventions, the model calculates the proportion of small habitat patches, defined as the number vegetation cells 0.5 ha or less in size, across a landscape area as a proportion of the total number of habitat patches.

Mean nearest neighbor patch distance

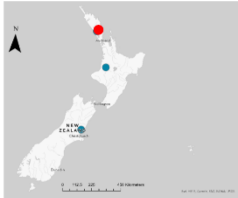
Habitat connectivity is quantified by the model as the distance among habitat patches or polygons large enough to support biodiversity (e.g., ≥ 2.5-ha) and/or having a higher species variety (e.g., biodiversity score ≥ 4). This variable aims to represent how restoration or replanting might improve connectivity in the landscape through activities that expand on the extent of existing patches, or adds new 'stepping stone' patches in bare areas, thereby lowering mean patch isolation (Cunningham et al., 2015). The model uses the 'Average Nearest Neighbor' tool in ArcGIS Pro to calculate the mean of the distances between the centroid of a focal woody patch and those of the nearest ten habitat patches.

4.1.3 Model simulations

Study landscapes

I applied the simulation model to three Aotearoa New Zealand pastoral agricultural landscapes (Figure 7): (1) Kaipara - a 5497-ha area east of the Kaipara Harbor in the North Island of Aotearoa New Zealand, straddling the Auckland and Northland administrative regions; (2) Ruapehu – 7078-ha area in the southwest of the North Island near Taumarunui; and (3) Hurunui – a 5661-ha landscape in a coastal region of Canterbury in Aotearoa New Zealand's South Island in the Hurunui administrative district. Each landscape encompassed one of the three sheep and beef cattle study areas described in Chapter 2. The vegetation of all three landscapes is typical of Aotearoa New Zealand pastoral farming landscapes: hilly areas dominated by pasture, intermixed with many small patches of woody vegetation and fewer patches of tall native forest. Land use in these three landscapes was predominantly sheep and beef cattle farms, with some areas managed for rotational crops, dairy farming, and exotic monoculture plantations (Manaaki Whenua, 2019). Despite the fact that the three landscapes are representative of a typical Aotearoa New Zealand sheep and beef cattle farm landscapes, each had distinct characteristics and spatial configuration in land use and land cover (Figure 7).

A KAIPARA



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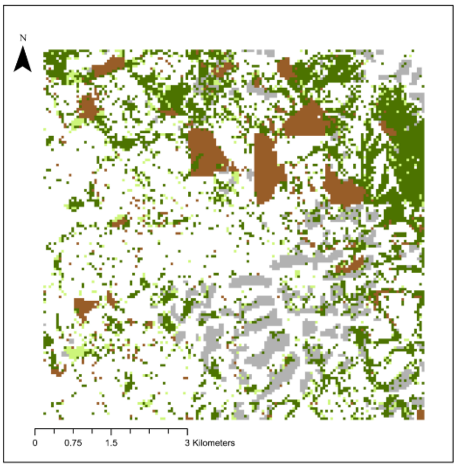
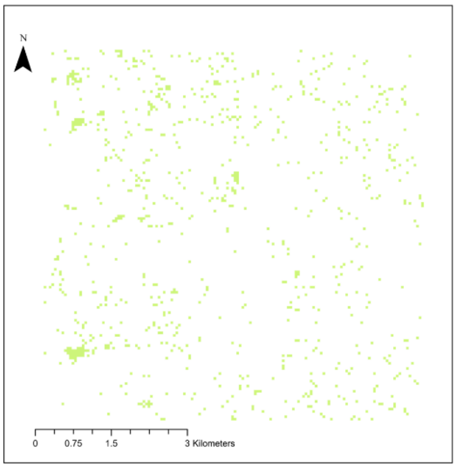


Land cover class	% Total area
Mixed-native woodlots	19.66
Shrubs	3.86
Exotic-dominated	6.95
Bare gullies	7.93
Total woody vegetation	30.47

A2



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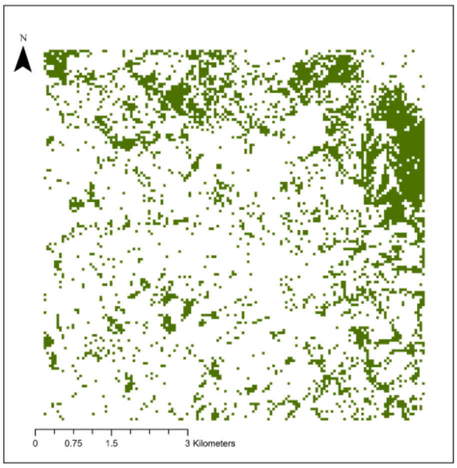

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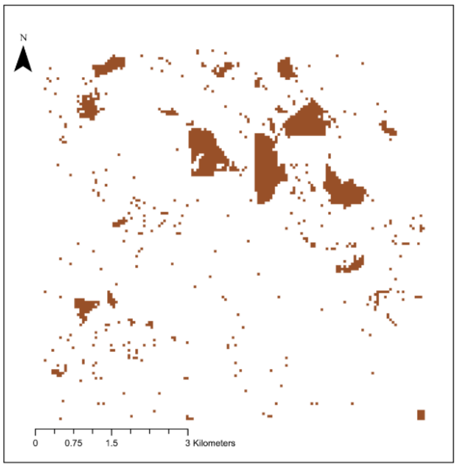

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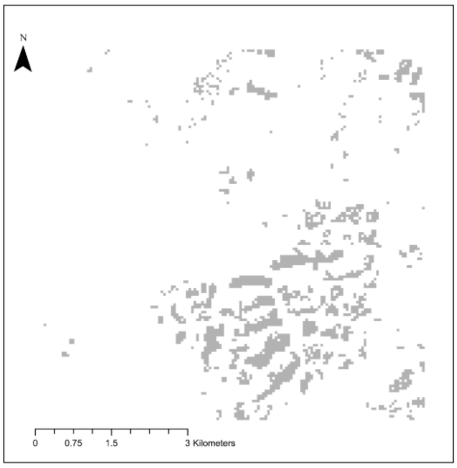
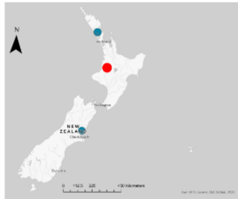


B RUAPEHU



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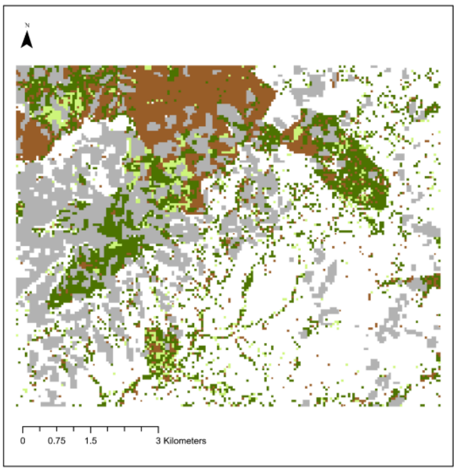
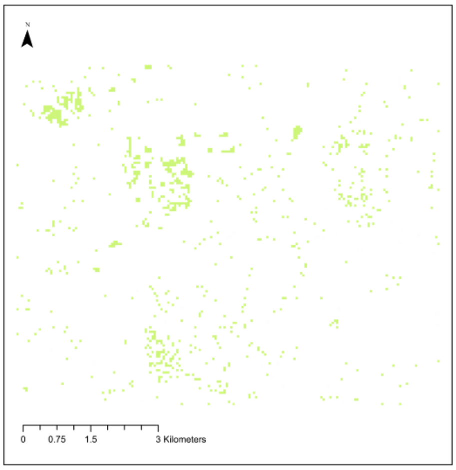


Land cover class	% Total area
Mixed-native woodlots	13.64
Shrubs	3.25
Exotic-dominated	11.36
Bare gullies	21.18
Total woody vegetation	28.25

B2



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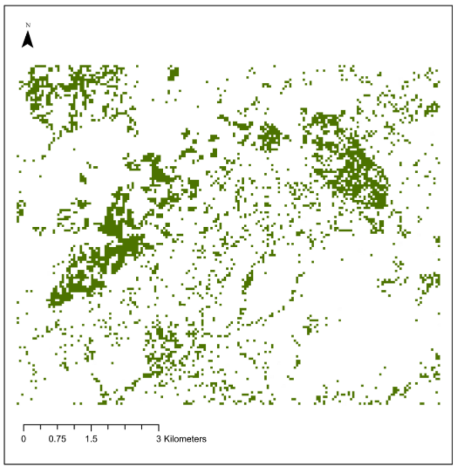

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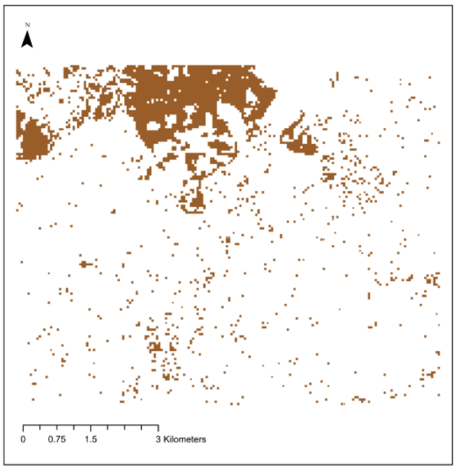

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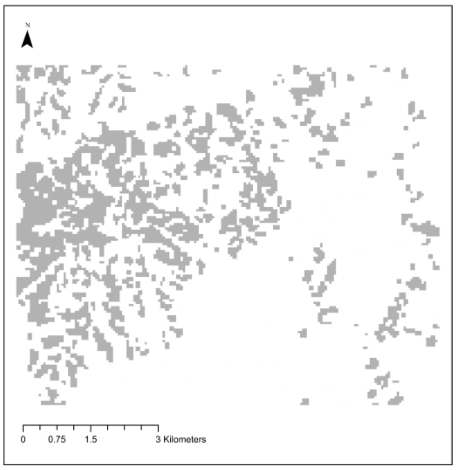
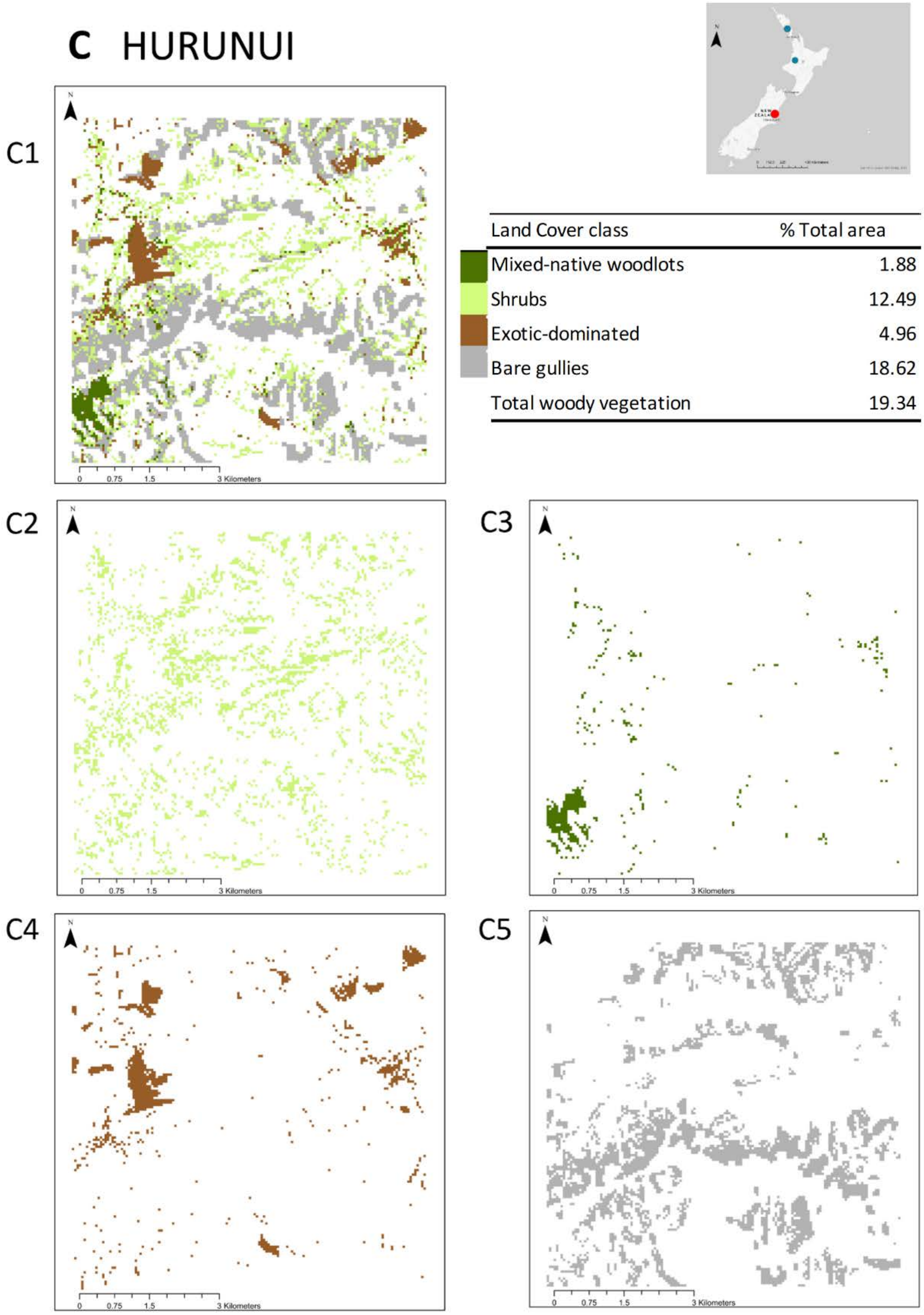




Land Cover class	% Total area
Mixed-native woodlots	1.88
Shrubs	12.49
Exotic-dominated	4.96
Bare gullies	18.62
Total woody vegetation	19.34

Figure 7 Maps showing the model input layers for the three landscapes: A) Kaipara, B) Ruapehu, and C) Hurunui. The location of each agricultural landscape in this study is represented by a red dot on the inset map. Each landscape map contains: 1) a complete set of woody vegetation and bare gully polygons, 2) shrub polygons, 3) native-dominated woodlot polygons, 4) exotic-dominated woodlot polygons, and 5) bare gully polygons.

Land cover input dataset for modelling

The primary input to the model was a high-resolution GIS polygon dataset of classified woody vegetation types and gully zones across the three farm landscapes. These datasets were generated previously as part of another research study (e.g., see Case et al., 2023; Zhang et al., 2021). In brief, a semi-automated, object-based classification procedure using eCognition image segmentation and classification software (https://geospatial.trimble.com/products-and-solutions/ecognition) was used to delineate vegetation types in high-resolution, color aerial images for the three study areas. Field surveys and visual inspection of the high-resolution imagery itself was used to identify training samples on the images for each vegetation class that was used in an object-based classification procedure (e.g., see Gupta & Bhadauria (2014). Object-based image analysis first creates objects or ‘segments’ (groups of adjacent pixels that are relatively homogeneous spectrally and which meet specific size/shape threshold criteria). Second, the segments are classified into vegetation types according to their mean spectral (color) properties, guided by the training samples. The eCognition software also enabled, as part of the classification procedure, the implementation of a set of spatial rules that could quantify relative canopy density among sets of vegetation type objects within a neighborhood zone. Gully on the three landscapes were automatically identified using a GIS analysis by one of the co-authors as portions of small river catchments with slope greater than 20 degrees and more than 20-hectare area. This step distinguished 22 original land cover classes: six native-dominated classes, six exotic-dominated classes, nine shrub classes, one bare-gully class, and one class of ‘other’ (road, water, building, etc.).

For the model simulations, I aggregated the original land cover classes into the four main classes relevant to this study on the selection of intervention class and presenting the results (see Appendix Table 8 for the explanation of land classes intervention groups): native-dominated woodlots (including old growth and mixed native), exotic-dominated woodlots, shrubs (including gorse, matagouri, and kanuka-manuka), and bare gullies (see an example in Figure 5 and Figure 7). Each land cover class obtained values for biodiversity potential and carbon stock that were extrapolated from plot level estimation (see Chapter 3 for the carbon stock estimation; see the variable values in Table 2). As input to the simulation model (see model description section), all existing woody vegetation and gully polygons in each landscape were intersected in the GIS with a 50×50m resolution polygon fishnet grid. This procedure resulted in all vegetation and gully areas being represented as 50-m resolution polygon ‘patch’ elements on which the model operates to carry out native restoration or revegetation interventions.

Table 2 Land cover classes and estimated carbon stock and biodiversity score for each Land cover class on the landscape. The total carbon stock was extrapolated from a plot-level average total carbon stock (Chapter 3). The original land cover map distinguishes 22 land cover classes (six native-dominated classes, six exotic-dominated classes, nine shrub classes, one bare-gully class, and one class of no data) that includes the subclasses based on the canopy cover (continuous, diffuse, and sparse). However, they were aggregated and refined into the five main classes relevant to this study on the selection of intervention class and presenting the results: native-dominated woodlots (including old growth and mixed native), exotic-dominated woodlots, shrubs (including gorse, matagouri, and kanuka-manuka), and bare gullies

Model Land cover	Original Land cover	Average total carbon stock (t C ha^{-1})	Biodiversity score
Mixed-native dominated woodlots	Old Growth (continuous)	211.96	5
	Old Growth (diffuse)	148.37	4
	Old Growth (sparse)	31.79	3
	Mixed Native (continuous)	164.43	5
	Mixed Native (diffuse)	115.10	4
	Mixed Native (sparse)	24.66	3
Shrubs	Kānuka (continuous)	73.39	3
	Kānuka (diffuse)	51.37	3
	Kānuka (sparse)	11.01	2
	Matagouri (continuous)	19.83	2
	Matagouri (diffuse)	13.88	2
	Matagouri (sparse)	2.97	2
	Gorse (continuous)	19.85	2
	Gorse (diffuse)	13.89	2
	Gorse (sparse)	2.98	2
Exotic-dominated woodlots	Deciduous (continuous)	364.99	2
	Deciduous (diffuse)	255.49	2
	Deciduous (sparse)	54.75	2
	Pine (continuous)	226.94	2
	Pine (diffuse)	158.86	2
	Pine (sparse)	34.04	2
Bare gullies	Bare gullies	0	1

Target patches for revegetation and restoration

The three farm landscapes were evaluated to determine the types of vegetation classes and areas that could be realistically targeted for restoration and revegetation interventions. Given the present tree-planting program aimed at marginal land (Ministry for Primary Industries, 2020c), I used shrubland patches as a proxy for 'marginal land', areas are typically not used for intensive agriculture activities and/or are not 'forest' (Ministry for Primary Industries, 2020c). Additionally, because some farmers were interested to know how much carbon could be gained by restoring exotic plantations or patches into native vegetation, I also targeted exotic cover types for woody vegetation restoration activity. Bare gullies were targeted for revegetation because erosion prone areas has been priority of the previous and current tree planting incentives (West et al., 2020). Native-dominated woodlots (i.e., old-growth and mixed-native vegetation) were set aside and excluded from target areas. Hence, the target land-cover classes are: 1) bare gullies; 2) shrubs, and 3) exotic-dominated forest patches (see Table 2 and Appendix Table 8 for description of each land cover class).

Model behavior testing

To explore the behavior of the model, and to address the first study question, I used several alternative (Alt) simulations to test the sensitivity of the monitored model outputs to different model parameter settings. Three ruleset scenarios were constructed that enabled an exploration of model parameter space: 1) revegetation of clustered bare gully target cells (Alt1), 2) restoration of scattered woody vegetation target cells (Alt2), and 3) a combination of revegetation and restoration of both larger and more dispersed target grid cell areas (Alt3). For this exercise, I ran the model for combinations of a range of settings for the intervention intensity parameter (5, 10, 15, 20, 25 %) and the aggregation parameter (50, 100, 150, 200, 250 m). Fifty iterations of the model were run for each of the three ruleset scenarios in each landscape. The different landscape indicator variables were monitored to determine the model responses to these parameter settings.

Modelling effects of different restoration and revegetation scenarios

To answer the second research question regarding assessing and understanding how initial landscape structure affects model outcomes, I constructed seven scenarios and measured the changes in habitat quantity, carbon stock, biodiversity potential, proportion of small patches, and the distances among habitat patches. The model featured seven scenarios (Table 3), three of which used a single intervention and four of which combined the interventions of restoration and/or revegetation. The three single intervention scenarios were Scenario 1 (which targeted bare gullies), Scenario 2 (which targeted shrubs), and Scenario 3 (which targeted exotic-dominated woodlots). The scenarios that combined interventions were Scenario 4, which combined the restoration of shrubs and exotic-dominated woodlots; Scenario 5, which combined the restoration of shrubs and the revegetation of bare gullies; Scenario 6, which combined the restoration of exotic-dominated woodlots and the revegetation of bare gullies; and Scenario 7, which combined the restoration of shrubs and exotic-dominated woodlots with the revegetation of gullies. Scenario 1 was a portrayal of a common practice in Aotearoa New Zealand agricultural landscapes that involved planting trees on steep slopes or erosion prone areas (Marden, 2012), while Scenario 2 represented the current interest of landowners to plant native trees throughout Aotearoa New Zealand (Kimberley et al., 2021), Scenario 3 represented the current initiatives for planting exotic species for carbon farming throughout Aotearoa New Zealand (West, et.al., 2020b), and Scenarios 5,6, and 7 were the combination of Scenario 1, 2, and 3. These three activities were also promoted as part of environmental farm plans around the country (Maseyk et al., 2019). The seven scenario models were run on a 100 m aggregation parameter and a 10% aggregation distance. Fifty iterations of the model were run for each of the seven scenario models in each landscape.

After running the scenario model, I estimated the baseline and each scenario's mean total carbon stock, median biodiversity potential, mean habitat distance, mean proportion of small habitat patches, and mean habitat amount for the landscape. To examine the results of each scenario, the outcomes were then compared to the baseline (the observed current landscape) to assess their relative effect (Figure 8).

Table 3 Seven scenario models describing target land cover class for each scenario

Scenario	Target land cover
Scenario 1 Bare gully revegetation	Bare gullies
Scenario 2 Shrub restoration	Shrubs
Scenario 3 Exotic-dominated woodlot restoration	Exotic-dominated woodlots
Scenario 4 Shrubs and exotic-dominated woodlot restoration	Shrubs and exotic-dominated woodlots
Scenario 5 Shrub restoration and bare gully revegetation	Shrubs and bare gullies
Scenario 6 Exotic-dominated woodlot restoration and bare gully revegetation	Exotic-dominated woodlots and bare gullies
Scenario 7 Shrub and exotic-dominated woodlot restoration and bare gully revegetation	Shrubs, exotic-dominated woodlots, and bare gullies

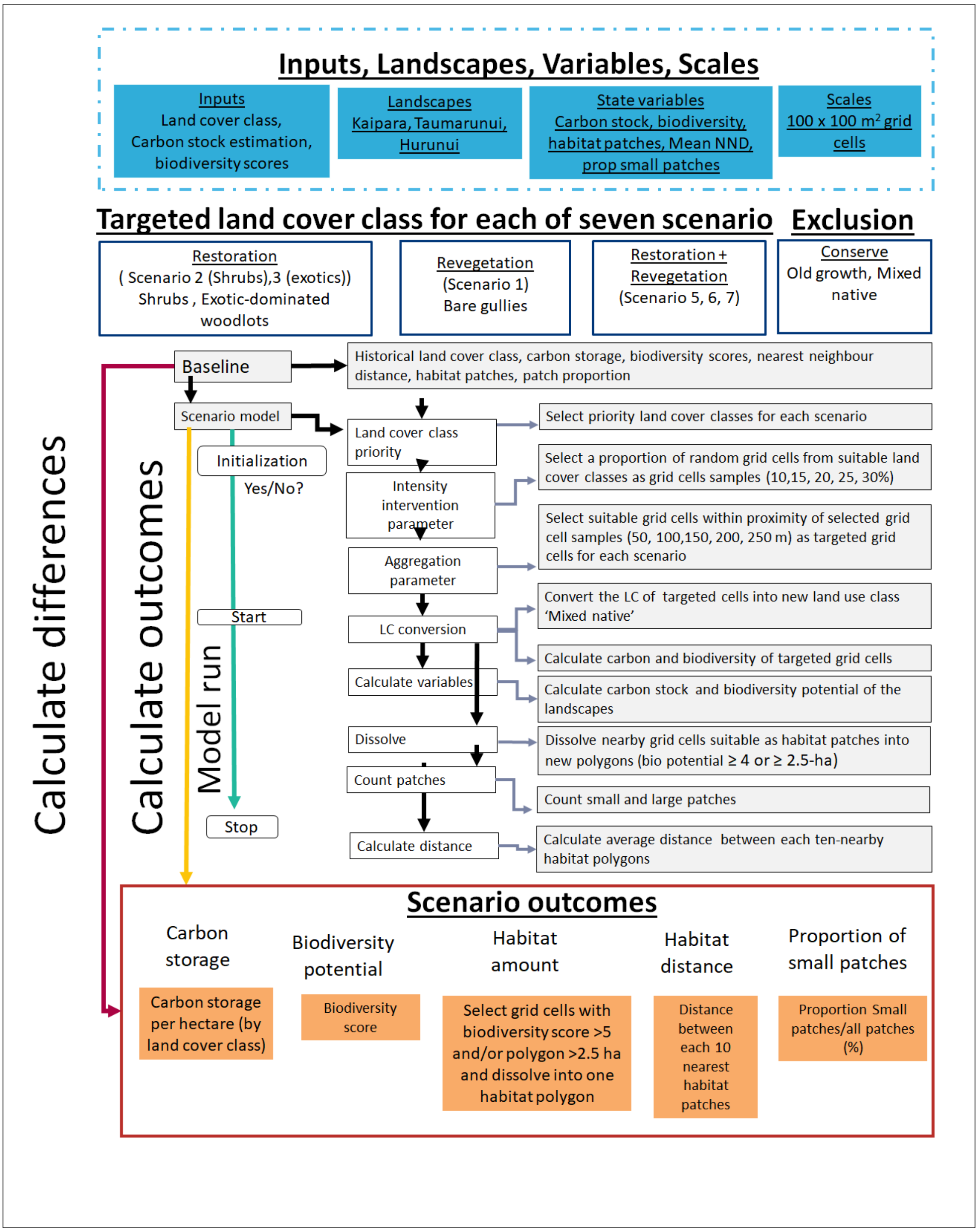


Figure 8. A flowchart illustrating the process used to generate five different outcomes: carbon stock, biodiversity potential, proportion of small patches, habitat distance, and habitat amount, from the seven woody vegetation restoration and bare gully revegetation scenario models; and to observe the differences between outcomes and the baseline values.

Modelling differences in outcomes and multifunctionality among farm landscapes

To address the third research question regarding evaluating multifunctionality tradeoffs among different model scenarios related to different types of revegetation and restoration interventions, I created A Principal Component Analysis (PCA) and radar graphs. The PCA was created from the outcomes of seven scenarios in the three landscapes to observe variations between baseline and scenario results in three landscapes. The radar graphs were created using the *Z*-score of each scenario's outcomes and was calculated as

$$Z = \frac{(x-\mu)}{\sigma},$$

where *x* represents the score difference from the baseline, μ is the mean, and σ is the standard deviation. The radar graphs were used to compare each scenario's multifunctionality score in each landscape. On the radar graph, the *Z*-score was inverted for the proportion of small patches and mean nearest neighbor distance (NND) to better reflect the predicted outcome. The multifunctionality score for each scenario in comparison to other scenarios in each landscape was calculated and displayed as the colored area of the radar graph.

ArcGIS Pro version 2.7.3 (ESRI, 2021) was used to conduct spatial analysis and quantify the results of the patterns of land-cover change under potential restoration and revegetation scenarios. Following the spatial analysis, all statistical analyses were carried out in R statistical software (R Development Core Team, 2020). ImageJ (Abràmoff et al., 2004) was used to measure the proportion of colored area of the radar graphs.

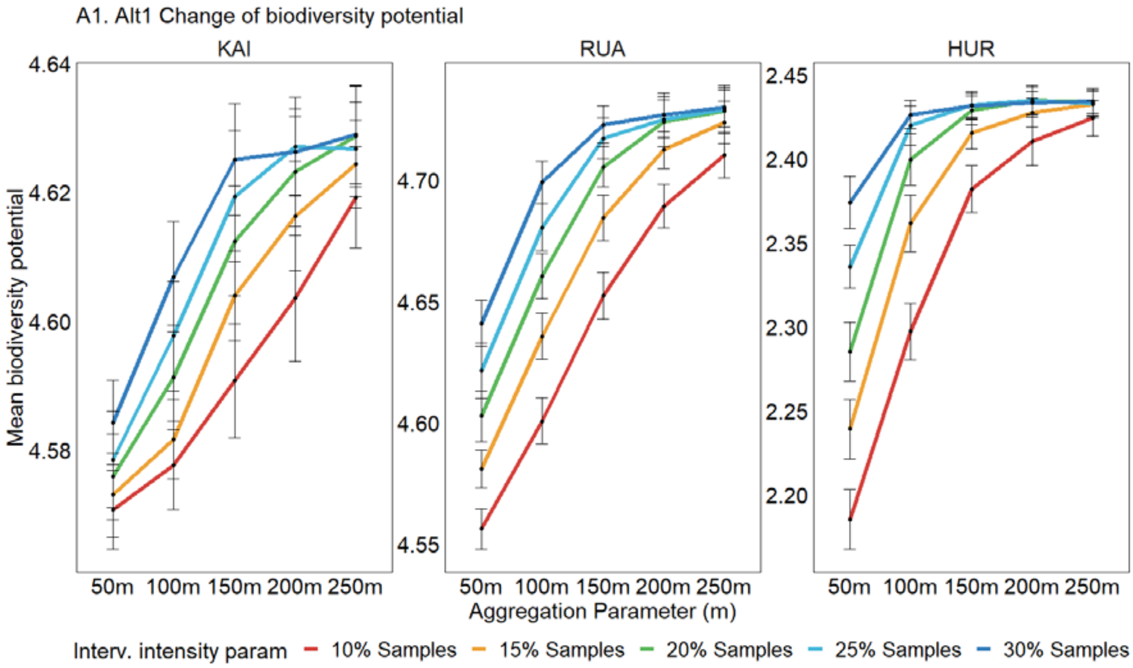
4.2 Results

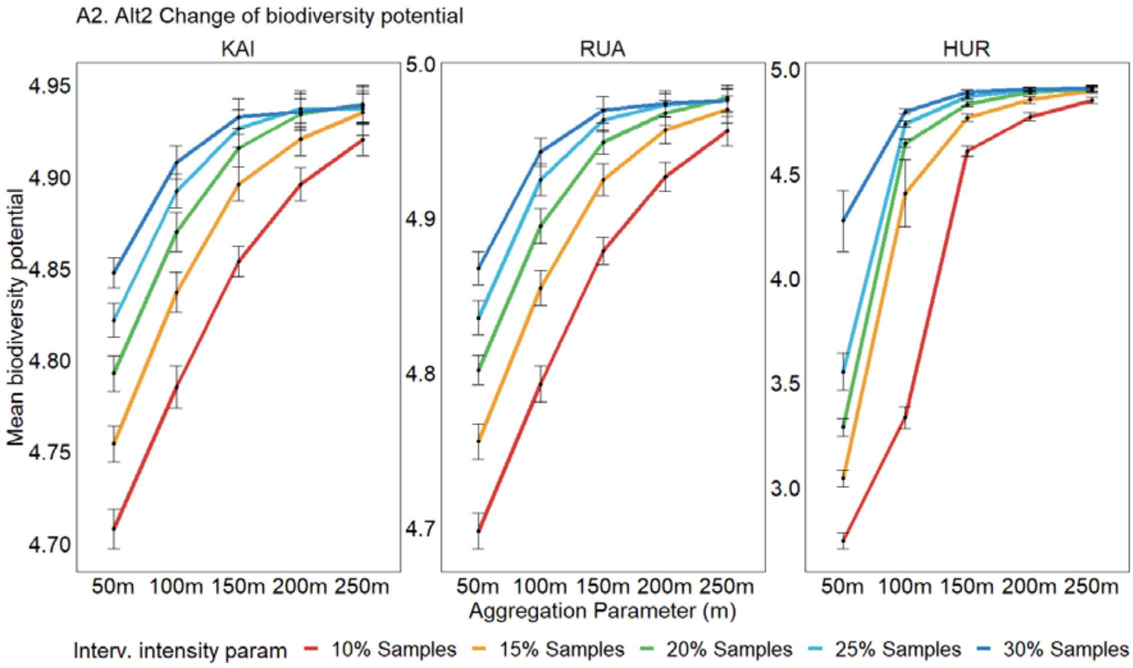
4.2.1 Model behavior

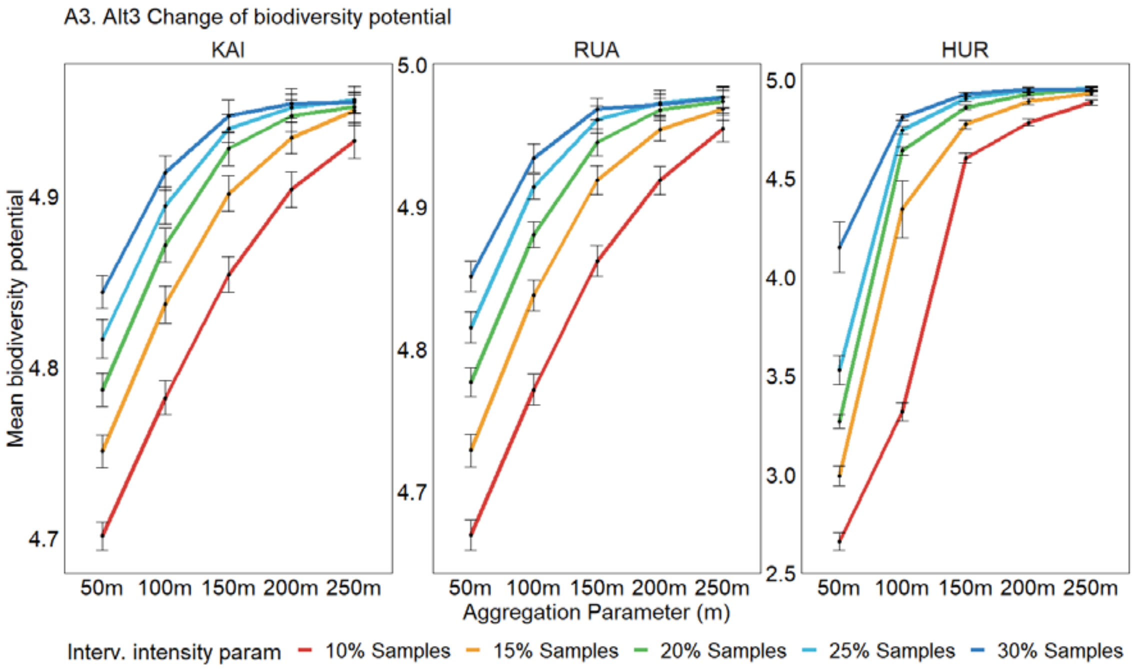
There was considerable variation in revegetation and restoration intervention outcomes as a result of applying different combinations of settings for the three model parameters (intervention intensity parameter, aggregation parameter, and intervention ruleset). The variety of changes could be divided into two categories: changes with a relatively consistent pattern of outcomes and changes with fluctuating patterns of outcomes. Variation in outcomes for two ecological state variables, namely biodiversity potential and habitat distance, was provided below as an illustration of the model's behavior based on the different combination of settings.

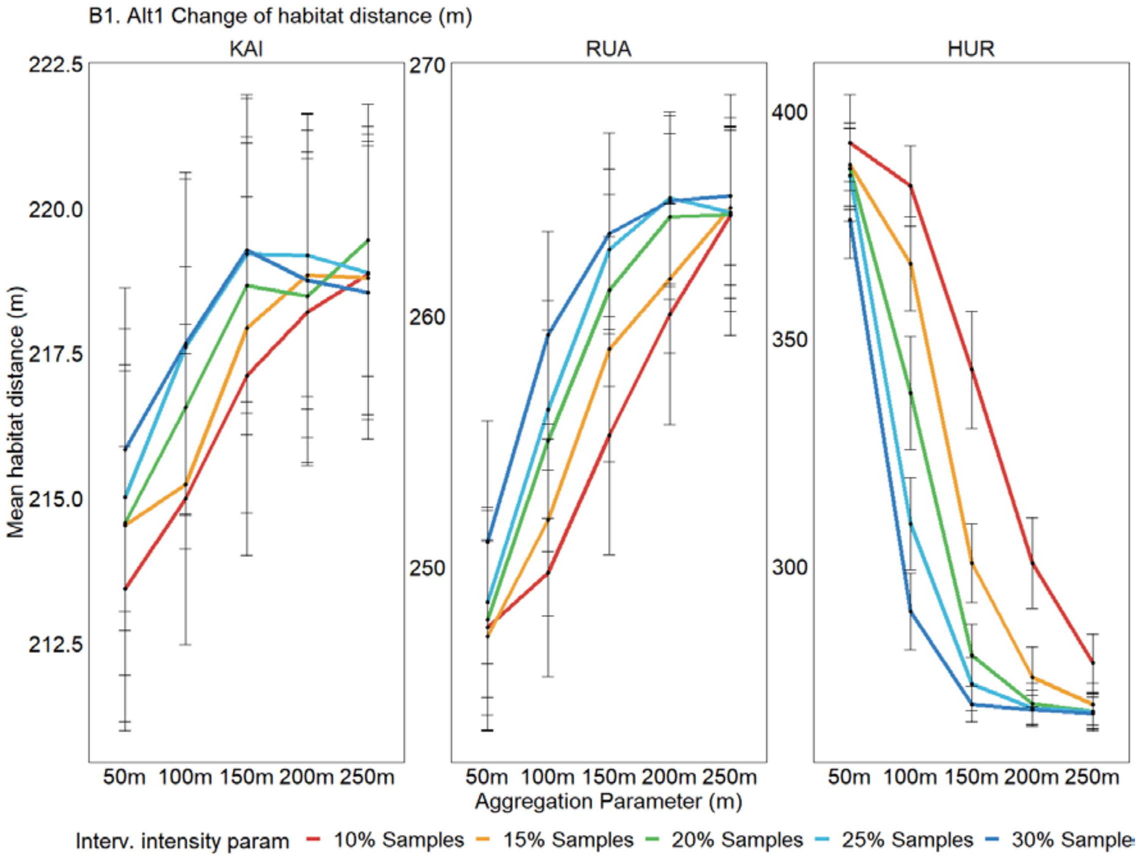
Increases in the intervention intensity and aggregation parameters resulted in a relatively similar pattern of non-linear increases in biodiversity potential outcomes in all three landscapes (Figure 9A). In response to changing the aggregation parameter, the slopes of the change in the carbon stock outcomes, were steeper between 50 and 150 m aggregate parameter and started to plateau after 150 m, with negligible change beyond 200 m aggregate parameter. Greater changes were observed for the three ruleset scenarios across the three landscapes following the increases in the aggregate parameter as compared to changes in the intensity intervention parameter. This pattern of change was consistent across the three ruleset scenarios across all landscapes, the changes of intensity intervention parameter resulted in the greatest change between 10 % and 15 %, a moderate increase between 15 % and 25 %, and a smaller change between 25 % and 30 %. Beyond a combination of the 150-m aggregate parameter and the 30 % intensity intervention parameter, the changes of parameters had minimal effects on biodiversity potential. This consistent pattern of change in biodiversity potential was also observed in the pattern of change in habitat amount (see Appendix Figure 3 for other outcomes).

In the contrary to the results that demonstrated a relatively similar pattern of gradual change, the changes in habitat distance, carbon stock, and proportion of small patches were more variable as the model setting parameters were changed across the three ruleset scenarios and across the three landscapes (see an example of variation of change in habitat distance in Figure 9 B, and Appendix Figure 3 for other variables). Although there were some results that showed a relatively consistent pattern where the habitat distance gradually decreased with the increasing setting parameters, e.g., in all three ruleset scenarios in the Hurunui landscape and in the Alt 2 ruleset scenario in the Kaipara landscape, the rest of the outcomes in other rule set scenarios in the three landscapes were more variable. For example, although the results of the Alt 1 ruleset scenario in the Kaipara and Ruapehu landscapes showed a pattern of gradually increasing outcomes, there were some exceptions of fluctuating outcomes in the Kaipara landscape. The Alt 3 ruleset scenario in the Kaipara and Ruapehu landscapes showed the greatest variation in habitat distance with varying parameter settings compared to all other ruleset scenarios across the three landscapes.

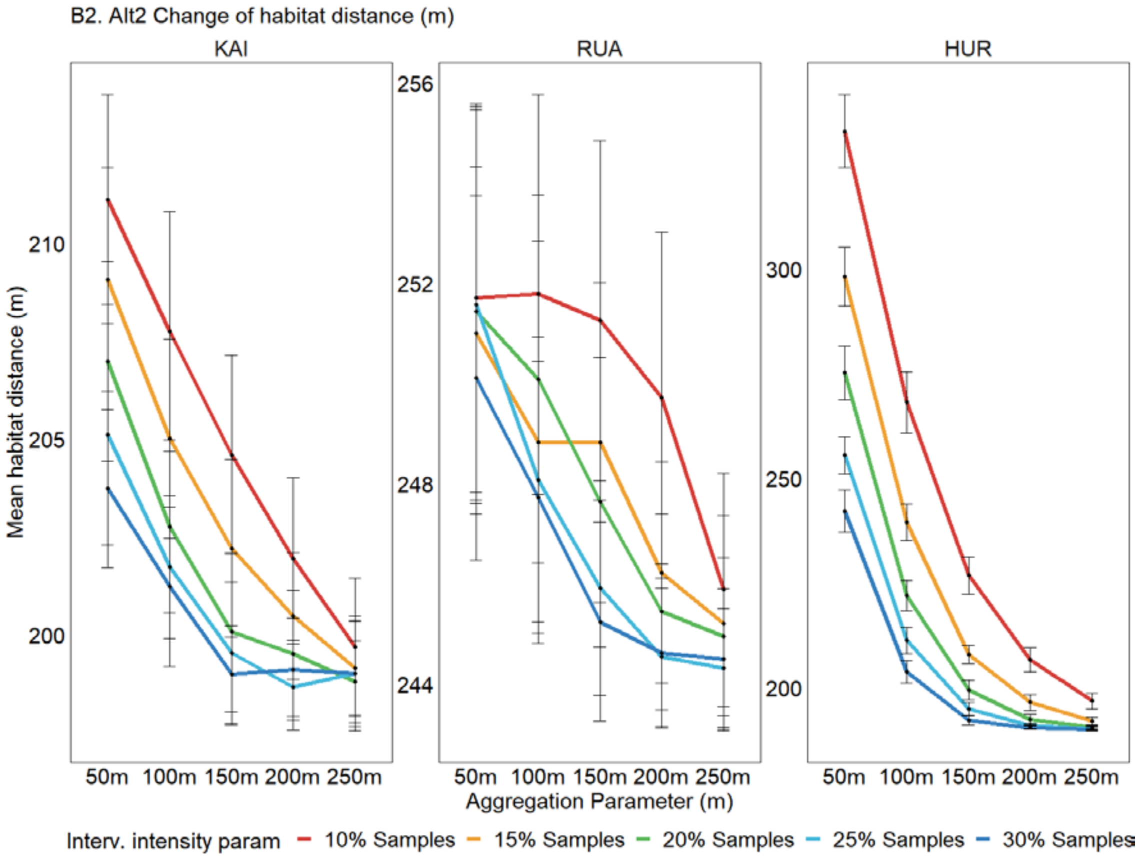



A2. Alt2 Change of biodiversity potential
KAI
RUA
HUR
Mean biodiversity potential
4.95
4.90
4.85
4.80
4.75
4.70
5.0
4.9
4.8
4.7
5.0
4.5
4.0
3.5
3.0
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples


A3. Alt3 Change of biodiversity potential
KAI
RUA
HUR
Mean biodiversity potential
4.9
4.8
4.7
5.0
4.9
4.8
4.7
5.0
4.5
4.0
3.5
3.0
2.5
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples



B1. Alt1 Change of habitat distance (m)
KAI
RUA
HUR
Mean habitat distance (m)
222.5
220.0
217.5
215.0
212.5
270
260
250
400
350
300
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Sample



B2. Alt2 Change of habitat distance (m)
KAI
RUA
HUR
Mean habitat distance (m)
210
205
200
256
252
248
244
300
250
200
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples

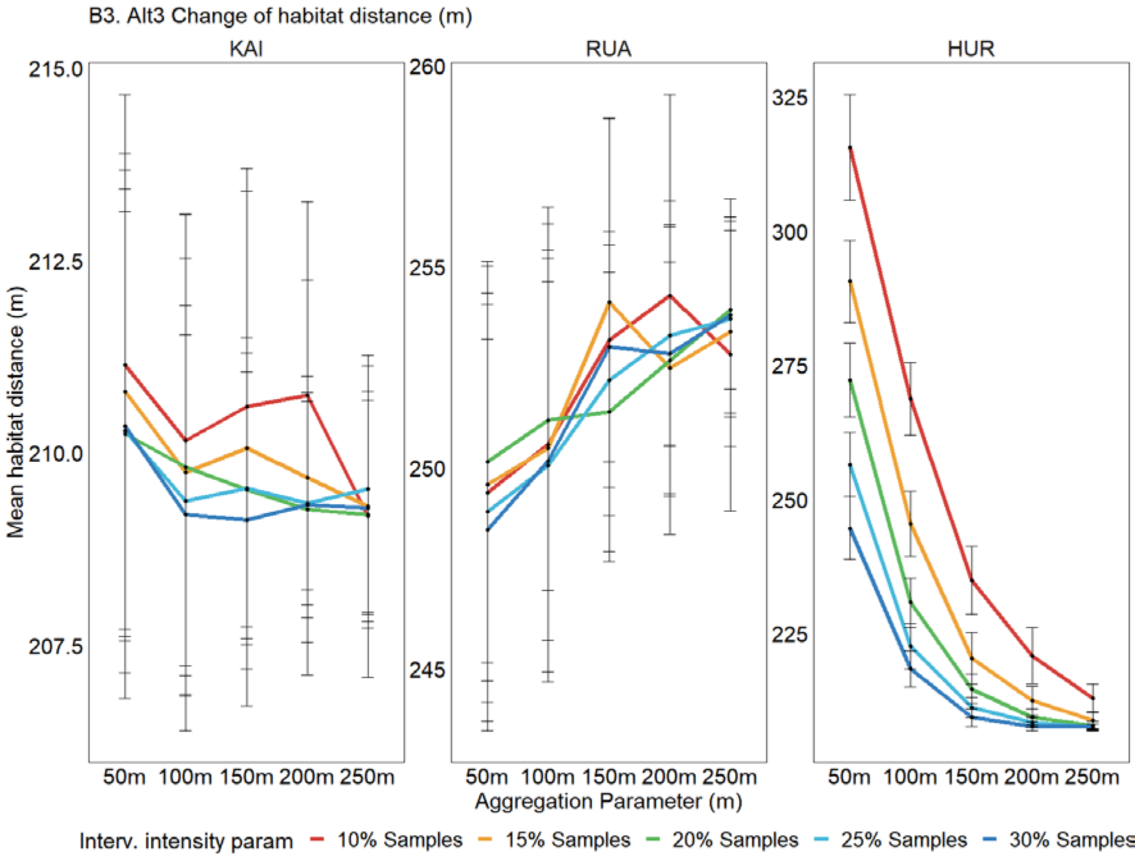


Figure 9 An illustration of how varying the aggregation parameter (from 50 m to 250 m) and intervention intensity parameter (from 10 % to 30 %) can affect the results of A) the median total biodiversity potential and B) the mean habitat distance (m) . for the three alternative scenarios 1) revegetation of clustered bare gully cells (Alt1), 2) restoration of scattered woody vegetation cells (Alt2), and 3) a combination of Alt 1 and Alt 2 (Alt3); across the three landscapes Kaipara, Ruapehu, and Hurunui. Note that the y-axes are on different scales for the different landscapes.

4.2.2 Baseline conditions for scenario simulations

The three landscapes had different spatial configurations and initial values for the variables depending on their existing states or baselines (see Figure 7A for infographic of each landscape and Appendix Table 9A for baseline values of each variable). Compared to the other two landscapes, the Kaipara landscape had the smallest area, with woody vegetation making up 30 % of the area and bare gullies making up 7.9 % of the total area (Figure 7). Small patches of shrubs were evenly distributed throughout the landscape, while exotic-dominated woodlots are divided into large patches in the northern part of the landscape and some smaller patches throughout the landscape areas. The medium- and large-sized patches of bare gullies were concentrated on the southeastern part of the landscape. Over 19 % of the entire area, or 64 % of the woody vegetation areas, were composed of mixed native vegetation, which was dispersed over the landscape with particularly large patches concentrated in the north-eastern area. The habitat area of the area was 1121 hectares, or 20.39 % of the total area, with a 215.45 m average distance between habitat patches. The woody vegetation area covered 58.85 % of the area. The estimated mean carbon stock and biodiversity potential were 265.55 t C ha^{-1} and 4.79, respectively.

Figure 7B showed that 21.18 % of the Ruapehu landscape was made up of many large, continuous, medium- and large-sized polygons of bare gullies. The woody vegetation area accounted for 28.25 % of the total landscape area, with mixed native woodlots (13.64 % of the total area) dispersed throughout the landscape, with large patches in the northeast and north-western areas, and exotic-dominated woodlands (11.36 % of the total area) dominating. These woodlots were mostly concentrated in sizable, continuous patches in the southern portion of the landscape. A small portion of shrubs (3.25 % of the total area) were scattered across the landscape. The estimated mean biodiversity potential of the woody vegetation was 4.12, and the mean carbon stock was the highest among the three landscapes, 363.49 t C ha^{-1}. The landscape featured 1257.99 hectares of habitat patches, or 17.77 % of the total area, and the distance between habitat patches is around 248.20 m. The proportion of small patches was 62.26 %.

The Hurunui landscape consisted of a sizable continuous polygon of bare gullies (18.62 % of the total area) that were scattered across the landscape (Figure 7C), similar to the Ruapehu landscape. A very small portion of mixed-native and exotic-dominated woodlots (1.88% and 4.96 %, respectively) and small patches of shrubs (12.49 %) made up the woody vegetation, which covered 19.34 % of the overall area. The proportion of small patches was 68.99 %. With only 184.50 hectares of habitat patches (3.26 % of the total area), the Hurunui landscape featured the smallest habitat amount, the lowest biodiversity potential (2.81), the smallest average carbon stock (102.20 t C ha^{-1}), and the greatest distance between habitat patches (449.52 m) across the three landscapes.

4.2.3 Variation in monitored model outcomes among relative to baseline conditions

There was considerable variation in revegetation and restoration intervention outcomes relative to baseline levels among the seven different intervention scenarios, and among landscape areas, in terms of the five ecological state variables being monitored in the model (Figure 10 A to E;

see results in Appendix Table 9 A and Appendix Table 9; see the example of spatial changes on Appendix Figure 4; Appendix Figure 5; and Appendix Figure 6).

Changes in carbon stock

Across all scenarios and landscapes, positive gains in carbon were generated for the majority of the seven model intervention scenarios; gains in carbon were highest in the Ruapehu and Hurunui landscapes, more than twice the increase observed in the Kaipara landscape (Figure 10 A). Among scenarios, the greatest gains in carbon stock, relative to the baseline conditions, were generated at all sites for Scenario 5, followed by Scenarios, 7, 6 and 1 (Figure 10 A). Scenario 3, where current exotic-dominated plantations were replanted with native vegetation, resulted in losses of carbon compared to baseline. For scenarios where gains in carbon were observed, gains ranged from a low of about 15 t C ha^{-1} (Scenario 2, Kaipara) up to a high of about 130 t C ha^{-1} (Scenario 5, Ruapehu), compared to baseline amounts.

Changes in potential biodiversity

All seven model intervention scenarios generated positive gains in biodiversity potential across all scenarios and landscapes, with results varying between scenarios in each landscape (Figure 10 B).Increases in biodiversity potential were higher in the Ruapehu, compared to the increase observed in the Kaipara and Hurunui landscapes. The combination of bare gully revegetation and shrubs and exotic-dominated plantation restoration in Scenario 7 generated the largest increase in biodiversity potential relative to baseline values across all scenarios in all three landscapes. Biodiversity potential decreased from a minimum of 0.003 (Scenario 4) to a maximum of -0.21 (Scenario 1) in the Kaipara landscape; 0.48 (Scenario 1) to a maximum of 0.67 (Scenario 4) in the Ruapehu landscape; and -0.51 (Scenario 3) to a maximum of 0.52 (Scenario 4) in the Hurunui landscape.

Changes in mean habitat distance

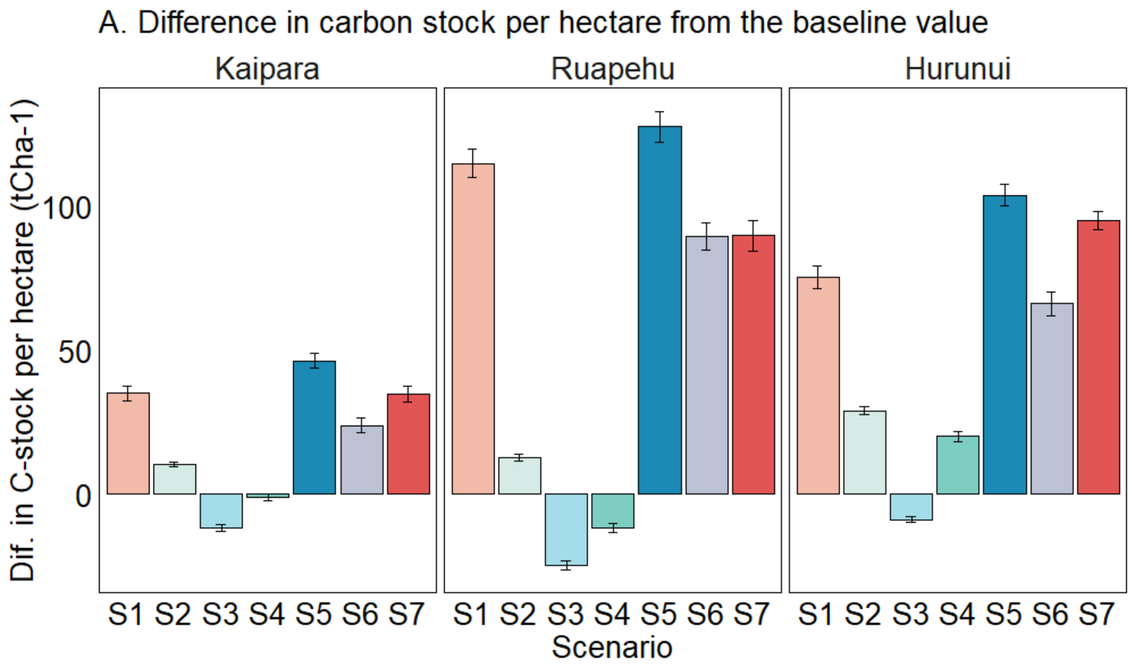
There was variation in the reduced habitat distance across all scenarios and landscapes; the change of habitat distance in the Hurunui landscape was much greater than in the other two landscapes (Figure 10C). Reductions in habitat distance in the Hurunui landscape (-180.87 m) was more than 23.49 times greater than the highest reduction in the Kaipara landscape (-7.7 m) and 41.11 times than the highest reduction in the Ruapehu landscape (-4.4 m). There were only Scenario 2 and 5 decreased the habitat distance in the Ruapehu landscape (Figure 10C). When compared to the baseline distances, the decreases in habitat distance across all landscapes ranged from -0.57 m (Scenario 1) in the Kaipara landscape to -181.39 m (Scenario 4) in the Hurunui landscape.

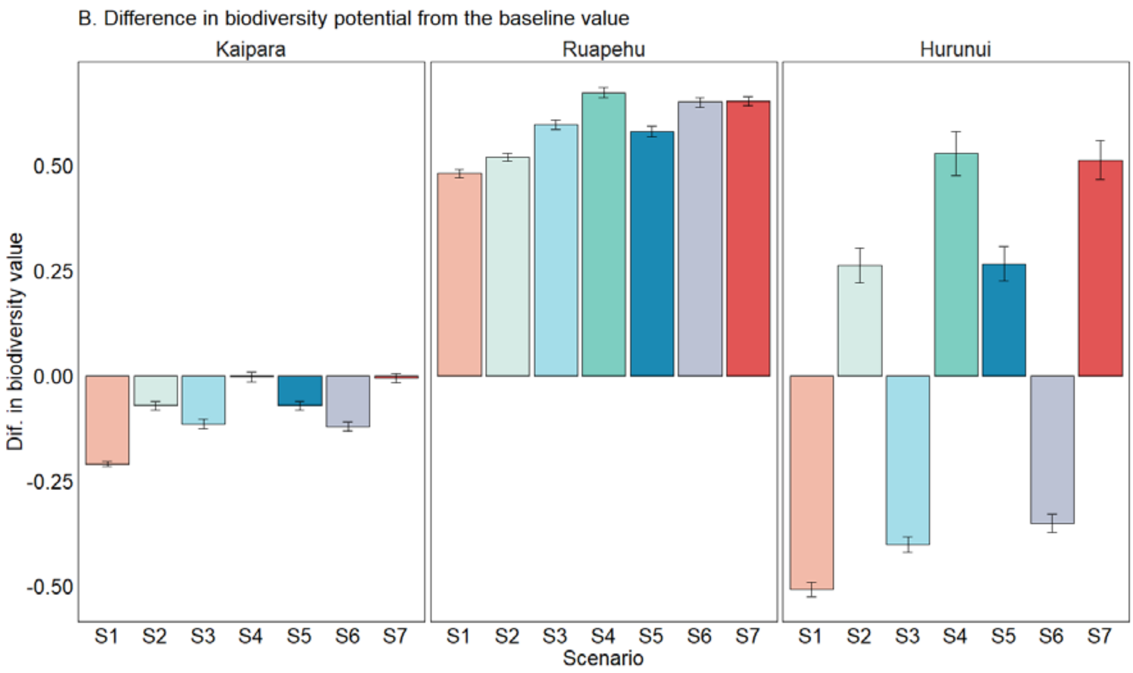
Changes in the proportion of small patches

Across all scenarios and landscapes, the changes on proportion of small habitat patches varied amongst scenarios in each landscape; the change of proportion of small patches in the Hurunui landscape was much greater than in the other two landscapes (Figure 10D). Amongst the scenarios that generated the highest reduction, the reduced proportion of small patches in the Hurunui landscape (-17.38 %) was more than nine times greater than in the Kaipara landscape (-1.93 %), and approximately five times greater than in the Ruapehu landscape (-3.48 %). Scenarios 1, 5, and 6 lowered the proportion of small habitat patches in all landscapes, whereas Scenario 7 only reduced the proportion of small habitat patches in the Ruapehu and Hurunui landscapes. In all three landscapes, Scenario 1 was the scenario with the highest reduction of the proportion of small habitat patches relative to the baseline. Across all three landscapes, for scenarios in which a decline in the proportion of small habitat patches was observed, the reduction varied from -0.37 % (Scenario 5, Kaipara) to -17.38 % (Scenario 1, Hurunui), relative to baseline values.

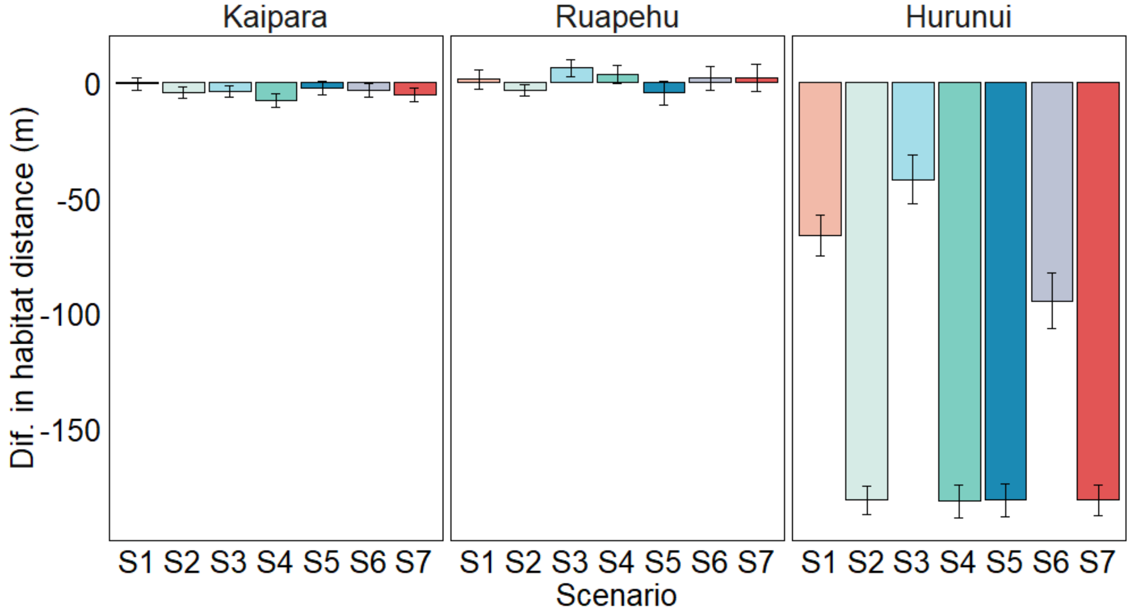
Changes in the habitat amount

All seven model intervention scenarios resulted in increased habitat amounts across all scenarios and landscapes. The greatest increases were in the Ruapehu and Hurunui landscapes, which were more than double the increase observed in the Kaipara landscape (Figure 10E). Scenario 7 in the Kaipara and Hurunui landscapes and Scenario 6 in the Ruapehu landscape generated the greatest increases in habitat amount relative to the baseline conditions. In all three landscapes, increases varied from approximately 47.83 hectare (Scenario 3, Hurunui) to approximately 1057.43 hectare (Scenario 7, Hurunui) relative to baseline values.

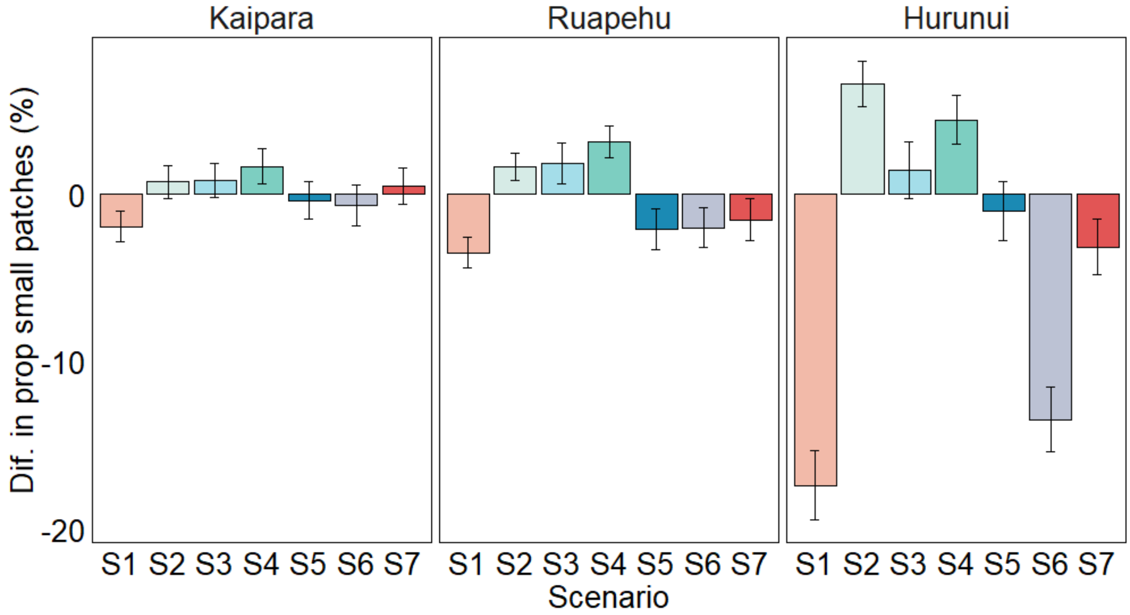

A. Difference in carbon stock per hectare from the baseline value
Kaipara
Ruapehu
Hurunui
Dif. in C-stock per hectare (tCha-1)
100
50
0
S1 S2 S3 S4 S5 S6 S7
S1 S2 S3 S4 S5 S6 S7
S1 S2 S3 S4 S5 S6 S7
Scenario


B. Difference in biodiversity potential from the baseline value
Kaipara
Ruapehu
Hurunui
Dif. in biodiversity value
0.50
0.25
0.00
-0.25
-0.50
S1 S2 S3 S4 S5 S6 S7
S1 S2 S3 S4 S5 S6 S7
S1 S2 S3 S4 S5 S6 S7
Scenario

C. Difference in habitat distance (m) from the baseline value



D. Difference in % of small patches from the baseline value



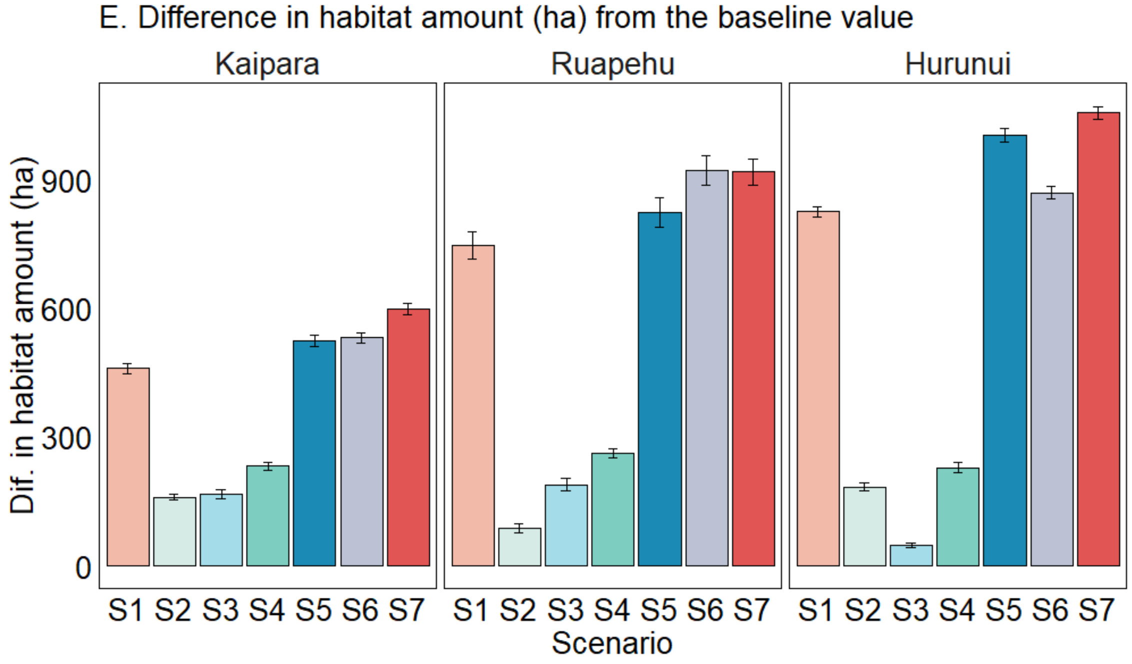
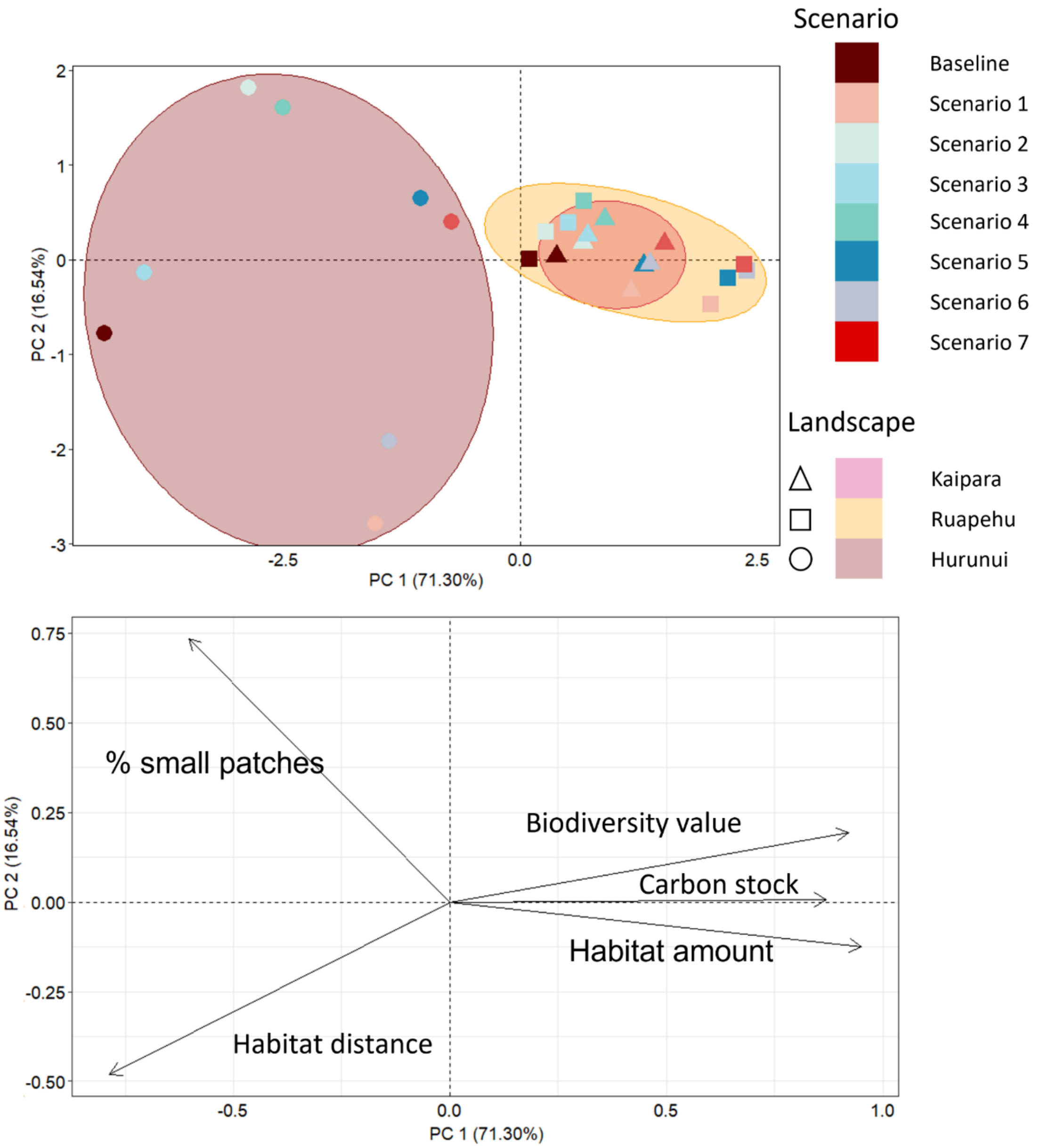


Figure 10. Bar plots shows changes in outcomes relative to the baseline in A) carbon stock ($tCha^{-1}$), B) biodiversity score, C) percentage of small patches (%), D) distance between habitat patches (m), and E) habitat amount (ha); between baseline and the results of each scenario in three landscapes: A) Kaipara, B) Ruapehu, and C) Hurunui, for each scenario : S1) Scenario 1 (revegetating bare gullies), S2) Scenario 2 (restoring shrubs), S3) Scenario 3 (restoring exotic-dominated woodlots), S4) Scenario 4 (a combination of restoring shrubs and exotic-dominated woodlots), S5) Scenario 5 (a combination of restoring shrubs and revegetating bare gullies), S6) Scenario 6 (a combination of restoring exotic-dominated woodlots and revegetating bare gullies), and S7) Scenario 7 (a combination of woody vegetation and bare gully revegetation).

4.2.4 Differences in outcomes and multifunctionality among sites

The seven different intervention scenarios across the three landscapes showed clear differences in outcomes. The first two components of the PCA explained 71.30 % and 16.54 %, or 87.84 %, accumulatively, of the variance in outcomes (Figure 11). The first principal component (PC) was primarily driven by increased habitat quantity, carbon stock, and biodiversity potential, whereas the second PC was characterized by a greater proportion of small habitat patches and increasing habitat distance. Both Kaipara and Ruapehu landscape scenarios formed a distinct group, characterized by an increase in carbon stock, biodiversity potential, and habitat amount. The scenarios from the Hurunui landscape formed a distinct group that could be separated from the scenarios from the other two landscapes due to their greater variation in characteristics (Figure 11).

Of the three landscapes, the outcomes from the seven scenarios for the Kaipara and Ruapehu landscapes were more similar to the baseline and to each other than they were compared to the Hurunui landscape (Figure 11). In comparison to the baseline, Scenarios 2, 3, and 4 in Kaipara and Ruapehu showed the smallest differences, whereas in Hurunui, only Scenario 3 produced outcomes that were very close to the baseline (Figure 11). In comparison to the baseline, some scenarios showed improved outcomes for almost all of the variables. For example, Scenarios 1,5, 6, and 7 in the Kaipara and Ruapehu landscapes generated increasing carbon stocks, biodiversity potential, and habitat amounts while also reducing the proportion of small patches and the distance between habitat patches. Some scenarios; however, generated increases in certain variables but decreases in others, for example Scenario 3 in the Kaipara and Hurunui landscapes that generated an increase in biodiversity potential but a decrease in carbon stock (Figure 11).

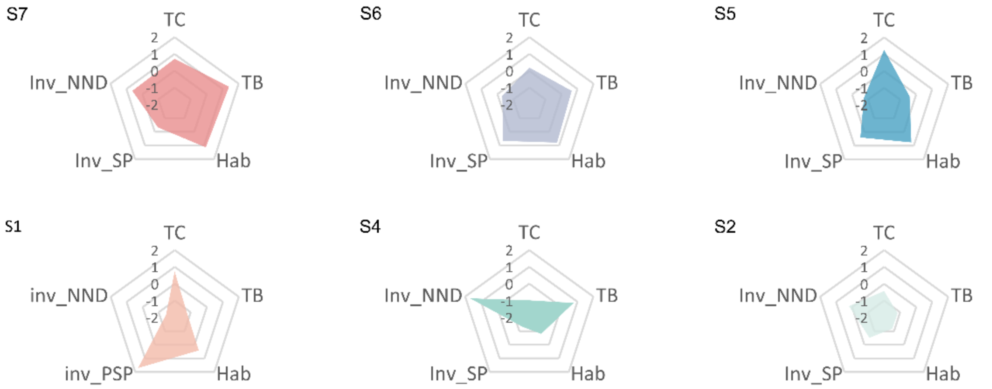


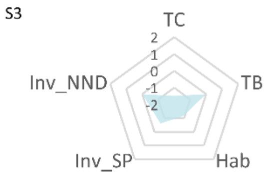
Scenario
Baseline
Scenario 1
Scenario 2
Scenario 3
Scenario 4
Scenario 5
Scenario 6
Scenario 7
Landscape
Kaipara
Ruapehu
Hurunui
PC 1 (71.30%)
PC 2 (16.54%)
% small patches
Biodiversity value
Carbon stock
Habitat amount
Habitat distance

Figure 11 PCA biplot showing scenario centroids and 95 % confidence intervals for the model outcome variables from the baseline and the seven scenario models computed for three landscapes: Kaipara, Ruapehu, and Hurunui. The colors of the points represent the baseline and seven scenario models: S1) Scenario 1 (revegetating bare gullies), S2) Scenario 2 (restoring shrubs), S3) Scenario 3 (restoring exotic-dominated woodlots), S4) Scenario 4 (a combination of restoring shrubs and exotic-dominated woodlots), S5) Scenario 5 (a combination of restoring shrubs and revegetating bare gullies), S6) Scenario 6 (a combination of restoring exotic-dominated woodlots and revegetating bare gullies), and S7) Scenario 7 (a combination of woody vegetation and bare gullies revegetation). The shape of the points and the ellipse colors represent the different landscapes. The PCA explains 71.30 % of the variance along the first axis and 16.54 % along the second axis. Arrows represent the vectors of four outcome variables: carbon stock density, biodiversity value, distance between habitat patches, proportion of small patches, and habitat amount.

The radar graphs showed marked variation in multifunctionality scores among the seven scenarios in each landscape, with Scenarios 5, 6, and 7 that combined restoration and revegetation intervention having the highest rank in all three landscapes (Figure 12 A, B, and C; see all values in Appendix Table 9 B, and C). Scenario 5 generated the highest multifunctionality score in the Ruapehu landscape, whereas Scenario 7 had the highest multifunctionality score in the Kaipara and Hurunui landscapes. In contrast, Scenario 3, in which exotic-dominated woodlots were replaced with mixed-native patches, generated the least multifunctionality score among all scenario of the three landscapes. The multifunctionality scores ranged between 10.30 to 46.63 % in the Kaipara landscapes, 7.16 % to 54.95 % in the Ruapehu landscapes, and 4.68 % to 52.72 % in the Hurunui landscape.

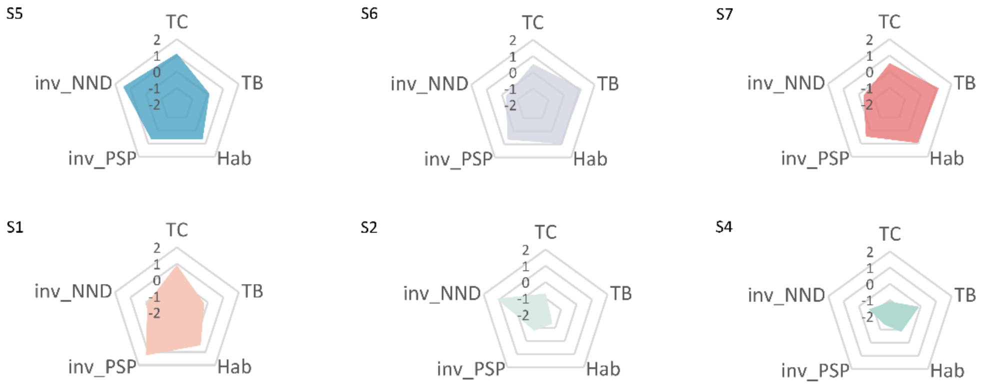
A. KAIPARA

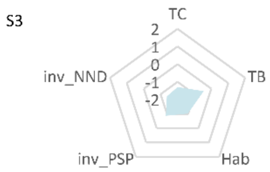




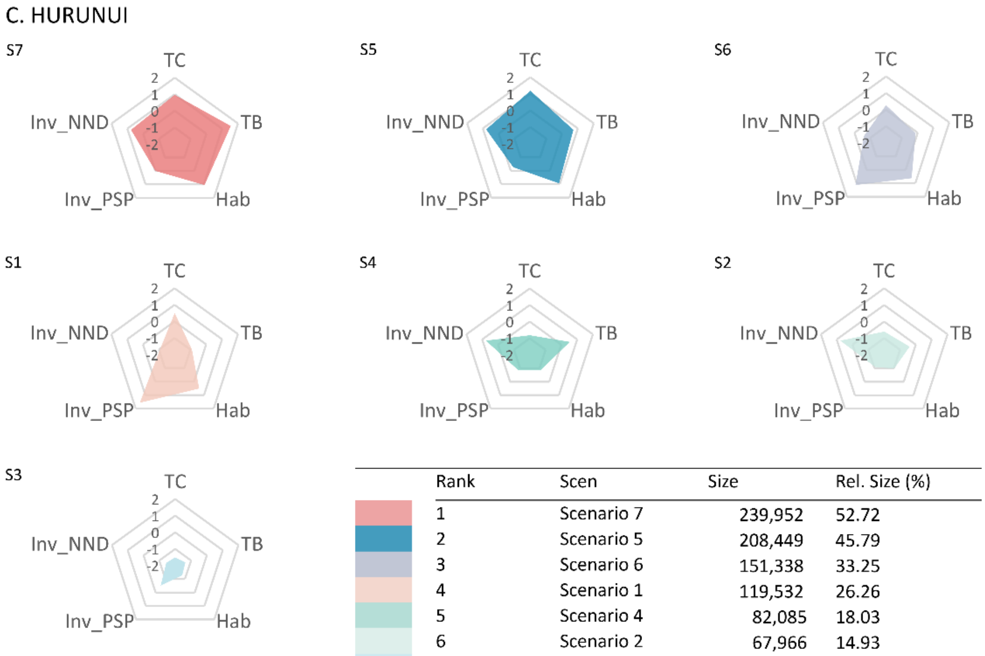
	Rank	Scen	Size	Rel. Size (%)
	1	Scenario 7	219,511	46.63
	2	Scenario 6	172,287	36.60
	3	Scenario 5	140,418	29.83
	4	Scenario 1	99,860	21.21
	5	Scenario 4	81,534	17.32
	6	Scenario 2	61,423	13.05
	7	Scenario 3	48,501	10.30

B. RUAPEHU





	Rank	Scen	Size	Rel.Size(%)
	1	Scenario 5	236,873	54.95
	2	Scenario 6	195,043	45.25
	3	Scenario 7	188,316	43.69
	4	Scenario 1	171,155	39.71
	5	Scenario 2	57,733	13.39
	6	Scenario 4	41,255	9.57
	7	Scenario 3	30,886	7.16



	Rank	Scen	Size	Rel. Size (%)
	1	Scenario 7	239,952	52.72
	2	Scenario 5	208,449	45.79
	3	Scenario 6	151,338	33.25
	4	Scenario 1	119,532	26.26
	5	Scenario 4	82,085	18.03
	6	Scenario 2	67,966	14.93
	7	Scenario 3	21,293	4.68

Figure 12 Radar graphs show Z-scores on 1) carbon stock,2) biodiversity score, 3) fraction of small patches, and 4 distance between habitat patches; between baseline and each scenario in three landscapes: A) Kaipara, B) Ruapehu, and C) Hurunui, for each scenario : S1) Scenario 1 (revegetating bare gullies), S2) Scenario 2 (restoring shrubs), S3) Scenario 3 (restoring exotic-dominated woodlots), S4) Scenario 4 (a combination of restoring shrubs and exotic-dominated woodlots), S5) Scenario 5 (a combination of restoring shrubs and revegetating bare gullies), S6) Scenario 6 (a combination of restoring exotic-dominated woodlots and revegetating bare gullies), and S7) Scenario 7 (a combination of woody vegetation and bare gullies revegetation). The value indicated the Z-score of the scenario outcomes for mean carbon stock (TC), mean biodiversity potential (TB), mean total habitat amount (Hab), mean reduction on proportion of small patches (Inv SP), and mean reduction on distance between habitat patches (Inv NND). The table ranked the percentage of colored area on the radar graph, with a lower rank indicating a more favorable result.

4.3 Discussion

This study presents a new, spatially-explicit simulation model for exploring how different woody vegetation restoration and revegetation interventions can generate considerable variation in a range of ecologically-important indicators and overall multifunctionality in Aotearoa New Zealand farm landscapes. The results show that actions such as woody vegetation restoration and revegetation of bare gullies, which can be motivated by government initiatives like the One Billion Trees program, offer a potentially important opportunity to deliver carbon and biodiversity benefits on agricultural lands. This study selected areas where land-cover change is anticipated to occur as a result of Aotearoa New Zealand's climate change and biodiversity policies. A spatially-explicit model that takes into account the objectives of actions to mitigate climate change and enhance biodiversity conservation will aid in the identification of synergies between these two goals. This study showed that the effects of woody vegetation restoration and gully revegetation depend on both the composition and configuration of the landscape, in addition to the intervention design parameters.

4.3.1 Model parameters and their relevance to the decision-making process

The SLIPSTReaM model results showed that increasing parameter values will create different patterns of landscape change. For some outcomes, such as biodiversity potential and habitat amount, increasing the parameters would have a generally consistent effect on the outcomes, with the aggregation parameter having a larger effect. For certain outcomes with fluctuating outcomes, such as habitat distance and proportion of small patches, the outcomes fluctuated when the parameter settings were modified.

The area, size, number and spatial arrangement of existing habitat patches that were affected by the scenario interventions were all likely to have affected the observed variation in habitat distance as a spatially-measured model outcome by interacting with changes in the model parameter settings. However, the Hurunui landscape was an exception to this by generating a relatively consistent value even when the outcomes were spatially measured. This was likely due to the high availability of all target classes in the Hurunui landscape, i.e., woody vegetation and bare gullies were relatively evenly distributed across the landscape (Figure *7* B). This ensured that an adequate number of grid cells were available for all rulesets. In contrast, the Kaipara landscape had a smaller proportion of woody vegetation and bare gullies that could be chosen for woody vegetation restoration and/or bare gully revegetation and were also distributed over different sides of the landscape (see the infographic of the Kaipara landscape in Figure 7 A), thus, resulting in more variable probabilities of outcomes.

Due to the limited availability of target class grid cells, the variation in the outcomes also occurred in the non-spatially measured outcomes, such as the carbon stock results of the restoring shrubs and exotic-dominated woodlots in the Kaipara landscape (Appendix Figure 3). The fluctuation in carbon stock outcomes in the Kaipara landscape was likely due to the spatial distribution and proportion of the target classes, as well as variation in the different initial carbon stocks of those classes. The small proportion of the two different target classes — shrubs and exotic-dominated woodlots (3.86 % and 6.95 % of total woody vegetation on the Kaipara landscape, respectively; see Figure 7A) — and their spatial distribution, resulted in an almost equal probability for both to be randomly selected as intervention targets as the intervention intensity parameter was increased. However, due to larger sized polygons of exotic-dominated woodlots in some areas of the landscape, increasing the aggregation parameter likely resulted in a higher probability of recruiting more exotic-dominated woodlots than small patches of shrubs. Because the initial carbon stock values of the exotic-woodlots was higher, increasing the aggregation parameter would primarily reduce the carbon stock outcomes, unless the target locations were distributed in an area where there were no exotic-dominated patches, such as in the southern part of the Kaipara landscape (see the Kaipara landscape in Figure 7A). Thus, although restoring woody vegetation in Ruapehu showed a similar decrease in carbon stock following increasing model parameters, there was no fluctuation in Ruapehu due to the greater proportion of exotic-dominated woodlots compared to shrubs (11.36 % and 3.25 %, respectively; see Figure 7B). Based on this type of observation, this suggests that landowners should carefully observe the composition and configuration the target classes in the landscape prior to any intervention to determine how the intervention will likely affect the outcomes.

The size and slope of differences in outcomes relative to the baseline across variation in the intervention intensity and aggregation parameters can be used to estimate the proportion of target areas that would achieve desired outcomes prior to a landscape-scale intervention. In this study, it was feasible to achieve a greater change in outcomes relative to the baseline by targeting 10 to 20 % of the landscape, and up to 150 m the surrounding area of each target location. Under the assumption that this simulation was effective, the model allowed the estimation of the proportion of areas that could be restored or revegetated, while still achieving increased outcomes for the three landscapes. A lesser aim would lead to results that gradually improved as opposed to a greater target, which would achieve higher results but would limit those results to a specific level. The intensity and aggregation parameters could only be increased until the maximum number of target cells available on the landscape were selected. This was illustrated by the negligible change in outcomes under a combination of 200 m for the aggregation parameter and 30 % for the intensity intervention parameter.

4.3.2 Impacts of different scenarios on different landscapes: the importance of understanding the baseline landscape context

This study demonstrated that a baseline is a crucial starting point for measuring and predicting the impact of restoration and revegetation interventions. This has previously been shown by the observation where a baseline is required to define an appropriate comparison framework when establishing the objectives for conservation efforts and evaluating their effectiveness after implementation (Bull et al., 2014). For example, in a carbon farming project, although the carbon stock generated by the intervention was important, the difference between the outcomes and the baseline was more significant for measuring the impact of the interventions. To achieve the goals of woody vegetation restoration and gully revegetation, it would be necessary to comprehend the principle that carbon farming activities should store more carbon than the initial stock (Nunes et al., 2019) and that biodiversity offset initiatives should result in gains that are equal to or higher than those lost (Kujala et al., 2022).

Understanding the value of a baseline is also necessary to understand the cost and therefore to reduce the negative consequences of carbon farming, which involves replacing existing woody vegetation with new trees in an effort to increase carbon sequestration (Lindenmayer et al., 2012). As an illustration, in the combination of the 1BT and ETS programs in Aotearoa New Zealand, it was often believed that growing pine forests on marginal land in agricultural landscapes would create a very high carbon stock. However, this was based on the premise that the baseline carbon stock was negligible due to the extremely low carbon stock value attributed to marginal land in Aotearoa New Zealand (i.e., 13.05 - 60.57 t C ha^{-1} for grassland with woody biomass (Ministry for the Environment, 2020a)), in comparison to the predicted carbon stock generated by the pine plantation (e.g., Pine plantation in Kaipara region accounted for an average 715 t C ha^{-1} for 25 years up to 1 313 t C ha^{-1} for 50 years plantation (Parliamentary Counsel Office, 2008). In reality, these marginal areas frequently had significant existing carbon stocks, even when the woody vegetation was dominated by shrubs, as demonstrated in Chapter 3. Therefore, the actual carbon gain from this approach is only the difference carbon stock between the newly planted vegetation and the woody vegetation that was replaced. In addition, because carbon farming was often created in an area that was not considered a 'forest,' it may overlook the current biodiversity and other ecosystem functions supplied by other habitats, such as native shrubs and wetland, that already existed on the landscape.

Examining the spatial configuration of habitat areas before any interventions or planting trees is significant because the spatial structure of the habitat network and the landscape matrix into which the network is incorporated have a significant impact on predicting the future effects of habitat quality, such as connectivity and the proportion of small patches across a variety of scenarios and landscapes. Scenarios involving the restoration of exotic-dominated woodlots and revegetation of bare gullies reduced the number of small patches in all three landscapes significantly more than scenarios involving the restoration of shrubs. Bare gully and exotic-dominated woodlots were made up of the originally medium to large polygons that were arranged in clusters within certain areas within the landscapes (Figure 7). Thus, these configurations caused the creation of larger woody vegetation patches when being revegetated. In contrast, scenario models with shrubs, which had smaller sized polygons and were distributed more sparsely across the landscape increased the proportion of small patches. Understanding the initial spatial arrangement of the woody vegetation will interact with the intervention to alter the current configuration is also essential to avoid the unintended consequences of tree planting. For example, recent research has shown afforestation was ineffective in increasing connectivity in Denmark because most new planting was concentrated in a few areas (Madsen, 2002).

Regardless of the magnitude of the shift or the benefits each scenario provided in terms of accessible resources and land, what matters most in future is that landowners are having more options and able to participate in efforts to improve the environmental quality of the landscapes. Improving the quality of existing woody vegetation, such as those in the Kaipara environment, could be an option for farmers who wish to contribute to carbon and biodiversity projects, but are unable to expand the amount of vegetation, such as in the Hurunui landscape. Understanding how much change will occur beyond the initial state is important because landowners need to make strategic decisions on how to contribute when the land values vary and different tree planting measures will require different resources (Hall & McLachlan, 2022). Additionally, landscapes like Kaipara and Hurunui, which currently have a high amount of woody plant covers, would nevertheless need to preserve the quality of the present woody vegetation. Therefore, even though restoring woody vegetation in this type of landscape would not result in a substantial difference in carbon emissions, it is still important because removing the existing woody vegetation could result in a considerable carbon loss.

4.3.3 Comparison among scenarios, the significance of choosing the appropriate scenarios for each landscape, and recommendations

For a landowner, the findings of the various scenarios indicated how prioritization can be used to investigate multiple scenarios for tree planting that strategically target locations to maintain and add new trees to optimize desired ecosystem services and generate greater total benefits. While the particular objectives of tree planting may differ among locations, planners and decision makers should systematically analyze a range of situations, objectives, limitations, and stakeholder and societal preferences to get an understanding of the broad spectrum of potential solutions (Nyelele & Kroll, 2021). Tradeoffs between biodiversity and carbon stock enhancement on tree restoration and revegetation scenarios are likely to occur when only one objective is prioritized. This conclusion is consistent with earlier research (e.g., Choi et al., 2022; Reside et al., 2017). Choosing which scenarios to implement requires careful consideration. Any tree planting management plan should include assessment and evaluation of the objectives and comparative advantages of different tree planting initiatives, such as those illustrated in Figure 12.

Moreover, previous research has shown that the frame of reference that is used to evaluate the impact of an intervention determines whether the intervention will succeed or fail (Bull et al., 2014). For example, in this study, despite the fact that some scenarios did not yield a larger carbon stock than the baseline — and hence are not useful for carbon farming—such as Scenario 3 which restored exotic-dominated woodlots (Figure 10 A and B) — such scenarios increased the biodiversity potential of the landscapes. On the other hand, some scenarios that produced significant increases in carbon stocks, such as Scenario 1 which revegetated bare gully (Figure 10), may not improve habitat quality for native biodiversity. Thus, if the objective of revegetation or restoration was only to increase carbon stock, Scenario 1 would be deemed a success, while Scenario 3 would be deemed a feasible scenario for biodiversity enhancement. However, both scenarios would be deemed unsuccessful if carbon and biodiversity co-benefits were the goal. Therefore, it is essential, when designing and administering a multipurpose project, such as 1BT, to develop baseline measurements that encompass the desired outcomes of multiple objectives, so that it can be determined how the objectives were actually achieved following the intervention, and how to assess the efficacy of the interventions based on the multiple outcomes. This is significant because, frequently, not all objectives within a set of multiple objectives can be met entirely.

This study had implemented an integrated method of evaluation by applying spatially explicit assessment, which should be used to achieve multiple objectives, particularly when the climate change mitigation strategy also aims to support biodiversity conservation objectives, such as reducing fragmentation and establishing a landscape with greater connectivity (Muoz-Rojas et al., 2015). This study indicated that agricultural landscapes may contribute to ecological functions, such as Scenario 7 in the Kaipara and Hurunui landscapes, where restoring woody vegetation and revegetating bare gullies could increase carbon stock, habitat quantity, and habitat patch distance. While other studies (Arroyo-Rodríguez et al., 2020) have suggested that agricultural landscapes designed to support native biodiversity should contain at least 40% forest cover, comprised of roughly 10% very large patches, 30% smaller patches, and dispersed semi-natural treed elements; the intervention in three landscapes in this study have managed to increase the habitat amount of the three study landscapes. For example, the habitat amount of scenarios with the highest multifunctionality scores had increased to 23.70, 30,76, and 33.97% of the total area in the Kaipara, Ruapehu, and Hurunui landscapes; respectively. However, the amount of habitat in two of the three landscapes exceeded 20% prior to the intervention (20% and 24.11%, respectively, in the Kaipara and Hurunui landscapes). Therefore, while not being able to add larger habitat patches as much as in the Hurunui landscape, the inclusion of tiny patches of native trees ("tree islands") in the Ruapehu landscapes could nevertheless contribute to increasing biodiversity and facilitating landscape regeneration (Montoya-Sáncheza et al., 2022). Adding small natural features, such as small woody vegetation or native patches as midfield islets, to a relatively homogeneous ecosystem, such as pasture, can also increase environmental heterogeneity (Deák et al., 2021; Benayas et al., 2008; Benayas & Bullock, 2012). Therefore, additional research is required to determine whether existing initiatives such as the 1BT could increase landscape permeability, which implies facilitated environmental flows to increase plant and animal dispersal, by reducing non-vegetation areas, restoring key areas, and by revegetating stepping stones to reduce the distance between habitats.

This study has contributed a new method for spatially assessing the potential gains in multifunctionality resulting from native tree protection and tree planting in agricultural landscapes. Globally, there is a growing number of studies on the use of spatial methodologies for the strategic targeting of tree planting initiatives e.g. (Ausseil & Dymond, 2010; Munro et al., 2009; Rallings et al., 2019; Renwick et al., 2014; West, 2020a). However, little research has looked at how spatial targeting will affect the outcomes of afforestation, quantitatively and spatially. Compared to zonation-based models such as LUWES (Lawlor & Swan, 2014) and LUMASS (Herzig & Rutledge, 2013) that aims to maximize the outcomes of separated planning units (e.g., production zone, conservation zone, plantation zone), the modelling method used in this study highlighted the need for viewing a landscape as an unified planning unit rather than developing a strategy by dividing landscapes for production and conservation purposes. In addition, while previous planning models utilized a spatial approach, they did not modify the landscape's configuration to project how tree planting may alter the landscape, as what we have projected through this model. This is the first model in Aotearoa New Zealand to provide a future projection for landscape configuration following restoration and revegetation interventions using native species. The use of spatial projections of potential woody vegetation change could promote discussions between the public, landowners, and policymakers on prospective scenarios for woody vegetation conservation, establishment, and management at a landscape scale (Sherren et al., 2011).

4.4 Conclusion

This novel spatial analysis of the potential for woody vegetation restoration and revegetation in agricultural landscapes to improve multiple ecological functions provides useful information for landowners and decision makers around the world engaging with tree planting programs and related incentives. Only by carrying out such an analysis, can we assess the co-benefits and/ or trade-offs of tree planting that may result in more multifunctional landscapes. This research has shown that understanding the existing carbon stock and biodiversity potential of a landscape, as well as appropriate knowledge of the spatial configuration, are the most important factors in determining the outcomes of restoration and revegetation scenarios. This is an important step in identifying opportunities for achieving carbon benefits, and generating carbon credits, in a sustainable way that also benefits biodiversity.

In addition to planning for carbon and biodiversity co-benefits, improving spatially-related ecological outcomes, such as increasing habitat amount, reducing the distance between habitat patches, and lowering the proportion of small patches in the landscape, will require strategic spatial planning, taking into account the predicted spatial expansion of woody vegetation patches after the intervention. More process-based modelling research will be necessary to confirm this conclusion. Biodiversity conservation and climate change mitigation Programs should be based on spatial planning and modeled predictions of landscape carbon and biodiversity; this modelling should be underpinned by ecological and natural history knowledge of native species.

The conversion of native grasslands to native woody vegetation in New Zealand brings forth an understudied but potentially consequential threat to biodiversity. These tussock grasslands, shaped by over a century of livestock grazing, have recently seen efforts to retire sections for the protection of indigenous biodiversity. However, the replacement of short tussock grassland with native woody plants introduces uncertainties and risks (Rose & Frampton, 2007). Covering around 1 million hectares, these grasslands are vital to New Zealand's montane-subalpine landscapes. As native woody species and tall tussocks attempt to reclaim their space, the potential loss of seed sources across large areas emphasizes the need for thorough research to understand and address the yet-to-be-quantified but potentially substantial losses in biodiversity tied to this ecological transformation. The recent surge in tree planting efforts has also introduced additional threats to native grasslands, further complicating the ecological dynamics of these vital landscapes. While these endeavors may aim to enhance biodiversity or address climate concerns, the unintended consequences on existing ecosystems, particularly native grasslands, cannot be overlooked. The shift from native grassland to tree-dominated areas poses potential challenges, creating an intricate interplay of factors that might impact indigenous biodiversity. This underscores the importance of carefully assessing and mitigating the potential risks associated with these tree planting initiatives to ensure a balanced and sustainable approach to conservation.

Chapter 5 General discussion

Globally, tree planting continues to be one of the most popular climate change mitigation options. Goals for afforestation have already been developed using a target-based approach. For example, in response to UN Sustainable Development Goal 15, the UN Strategic Plan for Forests 2017–2030 (United Nations, 2017) aims to increase global forest area as a percentage of all land area by 3 %. This goal is being implemented through the Bonn Challenge and the Trillion Tree Campaign, which aims to globally restore 350 Mha of degraded and deforested land by 2030 (Brown, 2020). The European Union has committed to ambitious habitat restoration goals including planting three billion trees by 2030 and restoring a minimum of 10 % of woody vegetation cover on agricultural land (European Comission, 2020). Other countries have pledged to plant more trees. For example, the United States has proposed to plant more than one billion trees by 2030 (U.S. Department of Agriculture, 2022) and Vietnam will plant one billion trees by 2025 (Socialist Republic of Vietnam Prime Minister, 2021). However, in addition to emphasizing the need to preserve forests and utilize natural regeneration whenever possible, the majority of these campaigns have primarily promoted tree planting as the primary mechanism for achieving their goals (Martin et al., 2021). Understanding that large-scale tree plantings will need to be properly planned (Holl & Brancalion, 2020), the results from this thesis bring a deeper understanding of the multifunctionality trade-offs and the potential biodiversity and ecosystem function gains and losses associated with these ongoing worldwide initiatives to plant more trees with Aotearoa New Zealand as the case study.

The second chapter of this thesis used the 1BT Program in Aotearoa New Zealand as a case study to evaluate how a large-scale tree planting program can help Aotearoa New Zealand meet its carbon sequestration goal. This chapter focuses on understanding the gaps and opportunities of large-scale tree planting that can benefit carbon and biodiversity, but also identifies potential negative impacts tree planting can have when not planned strategically. We showed that, in 2020, the 1BT Program may have a negative impact on biodiversity, and offered recommendations for how the 1BT strategy might be redesigned to achieve multiple wins for biodiversity and carbon in Aotearoa New Zealand. We suggested ten recommendations for how these initiatives could be adapted to avoid perverse outcomes for native species, while jointly achieving carbon and biodiversity goals: (1) Diversify strategies—protect first, restore second, plant third; (2) Consider net change in trees—do not just count trees planted; (3) Consider the co-benefits of carbon and biodiversity from the outset; (4) Consider the broader landscape; (5) Consider the carbon and biodiversity benefits of soil; (6) Consider the importance of existing carbon stocks; (7) Consider potential impacts to non-tree ecosystems; (8) Consider the longevity of the future forest; (9) Support landowners in planting and maintaining native trees; (10) Remember that climate goals cannot be achieved by planting trees alone.

The 1BT Program stopped taking proposals in 2022, and as of 30 June 2022, 374,902,000 trees had been planted, only 9.02 % of which were native species directly financed by the program (Ministry for Primary Industries, 2022). With the approved 25,815 hectares of planned land, the initiative will run until 2028 (Ministry for Primary Industries, 2022a). It is anticipated that native plants will comprise 46.34 percent of the intended planted area (Ministry for Primary Industries, 2022a); indicating that the initiative currently prioritizes planting native species, which is pertinent to two of our recommendations (recommendation 1 and recommendation 9) that emphasize the significance of native plantings over exotics. Currently, the efficacy of the program was measured by the quantity of seedlings planted (Ministry for Primary Industries, 2022b), rather than the number of trees that established (recommendation 2). In the absence of a formal monitoring and evaluation strategy regarding levels of establishment after planting, and based on the current report, 1BT has assumed that all dispersed seed successfully increases forest cover. The current states of the initiative have not yet taken into account monitoring the co-benefits of carbon and biodiversity from the outset, the benefits of soil for carbon and biodiversity, the significance of current carbon stocks, or potential effects on non-tree ecosystems (recommendation 3 to 7). However, the government has agreed with the Climate Change Commission's recommendations to provide estimates for carbon stocks that are not currently listed in the Aotearoa New Zealand greenhouse gas inventory (Ministry for the Environment, 2022a) that can contribute on developing a database of the existing carbon stock (recommendation 6). These estimates will be useful for the future improvement of the ETS and will acknowledge the significance of the carbon stock from land cover that was not considered forest. In addition, beginning in 2023, a new permanent forest category of the ETS would allow both exotic and native forests to be registered in the ETS, which would potentially encourage landowners' motivation to plant native species (Ministry for Primary Industries & Ministry for the Environment, 2022) (recommendation 9). Even though the proposal period for 1BT Program has ended, it will still engage with various landowners to develop additional strategies that will help plant “the right trees, in the right place, for the right purpose” (Te Uru Rākau, 2018a). The recommendations from this study, therefore, can continue to provide valuable insights to help shift planting strategies away from commercial exotic plantation, and towards planting native biodiversity to achieve both climate change mitigation and biodiversity conservation together. Additionally, more research is required to monitor how well 1BT Program is implemented between its approval and 2028, determine if the 1BT Program is able to achieve its goal of planting one billion trees, and evaluate how the program will be able to achieve its many objectives on climate change, biodiversity, social issues, and economic sectors. More work is also required to determine the future climate priorities of Aotearoa New Zealand after the 1BT Program is over. Furthermore, Aotearoa New Zealand will need to make more effort to combat climate change, and not just by planting trees (recommendation 10).

The second chapter of this thesis demonstrated the importance of preserving existing woody vegetation, especially native woody patches, to achieve multifunctionality on sheep and beef cattle farms in Aotearoa New Zealand. The results highlight the complex connections between the carbon stocks, diversity of woody vegetation species, and other ecosystem functions, on three, case study sheep and beef cattle farms. Instead of abundance or woody plant richness, stem diameter and height strongly and consistently positively correlated with carbon stocks; yet plant richness and the presence of key species were critical for high relative multifunctionality on farms. This result shows how multiple species with various ecosystem functions can be combined to create a multifunctional landscape that achieves the greatest number of ecologically and socially positive outcomes. Farm resilience in the face of many land management difficulties, such as climate change effects and product price volatility, is projected to rise as a result of this increased diversity and function. To manage woody vegetation in these sheep and beef cattle farms and agricultural landscapes in a way that benefits farm production, climate change mitigation and biodiversity, it is important to consider the implications of our findings. This study provides recommendations for avoiding the unintended impacts of extensive tree planting initiatives. In contrast to other recent studies that developed recommendations for large-scale tree planting initiatives, our research emphasized the importance of recognizing existing trees and habitat as a necessary first step in establishing effective land management to increase carbon storage, while also expanding the multifunctionality of agricultural land. Following this recommendation, landowners will be able to prevent the most negative change that occurs as a result of extensive tree plantings, that is, the disappearance of natural habitat when land is converted to exotic monoculture plantation.

This study generated new estimates for carbon stocks on Aotearoa New Zealand sheep and beef cattle farms. The estimated average total carbon stock per hectare for the three broad vegetation cover types in this study were 293.48 ± 200.01 t C ha^{-1} for the exotic-dominated woodland alliances, 192.84 ± 132.86 t C ha^{-1} for the native-dominated woodland alliances, and 64.31 ± 66.96 t C ha^{-1} for the shrubland alliances (Chapter 3). These results are comparable to some other similar studies, such as the estimated biomass carbon stock from shelterbelts in agricultural areas in Aotearoa New Zealand (237 ± 77 t C ha^{-1}; Czerepowicz et al., 2012) and the estimate of carbon stocks on farms in other countries, e.g., the average carbon storage in biomass and top soil in coffee agroforestry in Costa Rica was 93 ± 29 t C ha^{-1} (Häger, 2012). However, the estimates were lower than in other studies in other country, for example, the estimated aboveground biomass carbon stock of mixed plantings in agricultural land in Burkina Faso was between 256.8 t C ha^{-1} and 587.9 t C ha^{-1} (Dimobe et al., 2018). Climate, geography, management approach, and the selection of non-forest trees are a few of the variables that contribute to the variances in carbon stocks among different regions (Canadell & Raupach, 2008). Thus, this research contributes to a growing understanding of the role of woody vegetation in farms, which is crucial knowledge required to support valuation of existing woody vegetation on agricultural land. This is essential for currently growing efforts in Aotearoa New Zealand to conserve and establish more native trees within farms.

Our findings in Chapter 3 showed how multifunctionality differed across different land cover types. This is beneficial to landowners when making decisions that will result in different combinations of land cover (Chapter 3); for example, multifunctionality scores for mixed-native-dominated woodland communities showed the highest multifunctionality compared to the shrubland plant communities and exotic-dominated woodland plant communities (Chapter 3). However, our results also show that co-benefits and trade-offs between ecosystem functions are common, and thus, while it can be difficult to maximize all functions where many aspects (i.e., financial, environment, etc.) have conflicting effects on the component functions of a multifunctionality measure (Manning et al., 2018), it is feasible to choose the most suitable mix of trade-offs and co-benefits among various alternatives (Chapter 3).

Findings presented in Chapter 3 are particularly valuable for Aotearoa New Zealand as the nation continues on its present path toward carbon neutrality under the Paris Agreement and the National Policy Statement for Indigenous Biodiversity (NPSIB) (Ministry for the Environment, 2022b). Aotearoa New Zealand's government has released a new domestic goal to decrease greenhouse gas emissions, with the exception of biogenic methane, to zero by 2050 (New Zealand Government, 2019). Forestry-related activities will continue to be one of the primary methods for responding to climate change under this new commitment. Understanding the need for appropriate methodologies to estimate carbon stocks from forestry and other land uses is important; the government recently agreed with the recommendations made by the Climate Change Commission regarding the development of appropriate methods to account for changes in carbon stored in above-ground biomass, as well as the development of methods for tracking emissions and removals by sources and sinks (Ministry for the Environment, 2022a). This includes approaches that have not yet contributed to the existing national inventory, for example, accounting for carbon in small patches of trees or in regenerating vegetation outside of forest protection areas (Ministry for the Environment, 2022a), such as the plant communities described in this study (Chapter 3).

The quantification of carbon stock density for several plant communities of woody vegetation on farms (Chapter 3) can contribute to Aotearoa New Zealand's greenhouse gas inventory database, especially for carbon pools that have not yet been measured, such as shrubs, woody vegetation, and shelterbelts on farms that do not meet the criteria for 'forest'. With the upcoming new permanent forest category of the ETS, which would allow both exotic and native forests to be registered in the ETS beginning in 2023 (Ministry for Primary Industries & Ministry for the Environment, 2022), the estimated carbon stock from native-dominated plant community in Chapter 3 can contribute to the estimation of carbon stock of native species. Globally, the findings in Chapter 3 contribute to the current general value for estimating carbon emissions in the Land Use and Land Use Change and Forestry (LULUCF) sector (Intergovernmental Panel on Climate Change, 2014). In addition, the results in Chapter 3 provide insight into the varieties of native and non-native plant communities and the variety of ecosystem functions that can contribute to a database of biodiversity and ecosystem functions in agricultural landscapes. The results of Chapter 3 also demonstrate the multifunctionality of native vegetation on farms that can encourage farmers to integrate more trees on farms. The multifunctionality database can also contribute important insights for implementing recommendations from the National Policy Statement for Indigenous Biodiversity (NPSIB), such as identifying Significant National Areas (SNA) targeted towards native vegetation in agricultural areas (Ministry for the Environment, 2022b).

Understanding the importance of establishing a more robust strategy to address the increasing popularity of extensive tree planting, the analysis of native restoration and revegetation in Chapter 1 of this thesis can contribute to illustrate how different scenarios that were projected on different landscapes will produce different results. This was done by running a set of revegetation scenarios based on recommendations from Chapter 1: (1) Diversify strategies—protect first, restore second, plant third; (2) Consider net change in trees—do not just count trees planted; (3) Consider the co-benefits of carbon and biodiversity from the outset; (4) Consider the broader landscape; (5) Consider the carbon and biodiversity benefits of soil; and (6) Consider the importance of existing carbon stocks. Despite the difference targets among the scenario, substantial variation among landscapes in the results was due to the original carbon stock and biodiversity potential of each landscape (Chapter 1), and the original spatial configurations. To maximize carbon, biodiversity, and multifunctionality, landowners will need to account for the distinctive characteristics of the given landscape. This study demonstrates that restoring shrubs and exotic-dominated woodlots and/or revegetating bare gullies — without removing existing native-dominated patches — would result in changes in carbon stocks, biodiversity, connectivity, and fragmentation; these could be greater or less than the baseline. The multifunctionality assessment in Chapter 4 demonstrated the importance of considering both the baseline and a combination of co-benefits and trade-offs among all results from different scenarios prior to selecting the best option with the greatest potential to increase landscape multifunctionality. The multifunctionality assessment will be beneficial to guide landowners in their decision-making process. The 1BT and ETS Programs require landowners to produce a thorough spatial map of the proposed area as part of the application process, which includes the estimation of the planned carbon stock, but not the estimation of existing carbon stocks, if the area is not considered ‘forest land’. In addition, both the 1BT and ETS Programs do not include the assessment of biodiversity or ecosystem functions that will be needed to determine if the program meets its numerous objectives. In order to achieve multiple benefits associated with woody vegetation on farms, landowners shall be required to assess not only their carbon stocks, but also other ecosystem functions during the strategic decision-making process. However, not all Aotearoa New Zealand farms have detailed information on their existing carbon and biodiversity and a great deal of effort is needed to provide such information. Therefore, additional research that contributes to the development of an open access database of ecosystem functions provided by woody vegetation on farms, best practice guidance and further scenario simulations on the integration of strategies to achieve various objectives in agricultural landscapes, should be a target for future research efforts.

The results of Chapter 4 support previous work showing that a spatially-targeted approach is needed to maximize potential greenhouse gas reductions from revegetation due to the importance of the local landscape context (Brown, 2020). Integrative methods of evaluation, such as spatially explicit approaches to land-system assessment, should be used to achieve multiple objectives, especially when the climate change mitigation strategy also aims to support biodiversity conservation objectives such as reducing fragmentation and establishing a higher connectivity landscape (Muñoz-Rojas et al., 2015). In relation to 1BT, the findings of Chapter 4 illustrated various future scenarios of native restoration and revegetation, and how it will be advantageous for calculating whether each scenario will be beneficial for increasing carbon stocks and other ecosystem functions. To assist the government and landowners in evaluating the best approaches to attain multiple benefits from native tree planting, it will be necessary to conduct more research and develop best practices for more spatially explicit planning and monitoring. Moreover, the implementation of any strategy that would result in land-cover change on privately owned rural land in Aotearoa New Zealand has depended heavily on the decisions of individual landowners. Given that the strategy requires the cooperation of a large number of landowners, further efforts and research are required to engage various stakeholders in order to achieve the objective of a multifunctional landscape. In this context, a spatially explicit plan that combines different interests within a multifunctional landscape, such as the one presented in Chapter 4, will be valuable for fostering communication among landowners and between landowners and policy makers.

According to a global study (Zomer et al. 2009), approximately one billion hectares of agricultural land, or roughly half of all farmed area on Earth, was estimated to have more than 10 % tree cover. The area managed by integrating trees into agricultural systems is believed to have increased as a result of the development of farmer-managed natural regeneration (Reij et al. 2009). Globally, Nair (2012) estimates that 1.6 billion hectares of agricultural land might be managed in ways that incorporate trees in the near future. As a result, there will be more opportunities to accumulate carbon and support numerous other ecological processes globally by retaining and adding woody vegetation on agricultural land. However, there are still knowledge gaps regarding the nature and functions of woody vegetation in agricultural settings, as well as how to manage the ecosystem functions it provides. Therefore, there are needs to increase research on the carbon storage and multifunctionality of trees in diverse land cover, and to develop fresh and innovative solutions to develop a multifunctional agricultural landscape that integrates trees (Nair & Garrity, 2012). In addition, it is essential that the study of the multifunctionality of trees in agricultural landscapes be properly translated into policy and practice for landowners to realize the benefits of such research.

Land use change and natural disturbances have an ongoing impact on woody vegetation in agricultural settings. More study is required to identify solutions that will encourage landowners to retain existing woody vegetation and to plant new trees, particularly native species, outside of forest areas and in agricultural landscapes. This is especially important for countries with extensive agricultural areas that still retain high woody vegetation cover, such parts of Aotearoa New Zealand. Although Aotearoa New Zealand’s plans to increase its forest cover are strongly aligned with climate change targets (Ministry for the Environment, 2022c), the main challenge remains to avoid achieving these targets at the expense of native biodiversity in agricultural landscapes. More research should be conducted on the impact of land-use change, including tree planting initiatives to enhance the existing native woody vegetation in agricultural landscapes. Efforts should also be made to investigate new financing options for agricultural mitigation and adaptation strategies, to encourage the use of trees in climate change strategy, given the increased international attention on climate change (David Hall & McLachlan, 2022). Such investments would greatly contribute to the advancement of multiple goals, including the improvement of food security, the improvement of rural livelihoods, and the enhancement of climate change mitigation and adaptation.

Climate change is intensifying the effects of global processes such as increasing reliance on intensive agricultural production, rising food consumption and ongoing biodiversity loss. Responding positively to these challenges requires investment and innovation in agricultural practices. Such approaches should be particularly targeted toward land management methods that provide co-benefits for production alongside preserving and restoring biodiversity and its associated ecosystem functions. This thesis identified significant carbon sequestration as well as multiple ecosystem functions that can enhance the multifunctionality of Aotearoa New Zealand's sheep and beef cattle farms and greenhouse gas mitigation. Through simulations of the protection and development of mixed-native woody vegetation, we demonstrated a strategy that could produce multiple benefits. With this approach, sheep and beef cattle farms can potentially contribute to climate change mitigation and biodiversity conservation. This research has provided important insights that can be used to help achieve multiple benefits across Aotearoa, while also providing broadly relevant for agricultural landscapes and tree planting initiatives around the world.

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Appendices

Appendix A *Supplementary Tables*

Appendix Table 1 Equations used for estimating 1) live tree height, 2a.) volume of standing dead logs,2b.) volume of fallen dead logs; and biomass carbon stock densities for 3a.) native and exotic live stems, 3b.) exotic *Pinus* spp. live stems, 3c.) exotic *Erythrina sykesii*, 3.d.) exotic *Salix* spp., 3.e.) native tree ferns and palms, 4) discrete shrubs, 5a) biomass of dead tree, and 5b) biomass of dead tree ferns and palms. Where *D* was the diameter at breast height (cm), H was the height (m); *A* was the normalized elevation (elevation (m above sea level (a.s.l.)/100) of the plot scaled to be similar in range to the other predictors, a, b, c, and d were model parameters, ε_H was the coefficient of the height-diameter model, *BLS* was live stems biomass (kg), *W* was wood specific density, C_{Shrub} was the biomass carbon of discrete shrubs (kg), S was shrub density. Dm was the diameter of the dead log, L was the length of the log, r1 and r2 were two radii measures of the log, C_{CWD_T} was the biomass carbon of the coarse woody debris of trees (kg), C_{CWD_TFP} was the biomass carbon of the coarse woody debris of tree ferns and palms (kg), *V* was the volume of the standing log (V_{SL}) or the fallen log (V_{FL}), *FW* was fresh-weight density, and *DSM* was the decay-stage modifier, DW_{oven} was the oven-dry weight (g) and IW_{400} was the weight after ignition (g), TOC was the total organic carbon concentration (%), *BD* was the soil bulk density (Mg m^{-3}), and *h* was the soil depth (*h* = 30 cm)

	Carbon pools	Components	Variable	Applied to	Equation	Citation
1	Aboveground biomass	Live stems	Height	Native species trees, and *Pinus radiata* D.Don	$Ln\ (H - 1.35) = ln(a) + ln(1 - bA) + ln\left(1 - exp\left(CD^d\right)\right) + \varepsilon_H$	Beets et al., 2012; Holdaway et al., 2014
2			Biomass	70 species of native trees and shrubs (*n* = 2,443) and exotic species that were not dominant (*n* = 14)	$B_{LS} = 0.905 \times W \times 0.000483(D^2H)^{0.978} + 0.00175\ D^{1.75}$	Holdaway et al., 2014

3			Biomass	Biomass carbon of exotic *Pinus* spp. and *P. radiata* (n = 40)	$Ln(B_{LS}) = -9069 + 1.2273\ lnD + 0.1411\ (lnD)^2 - 0.0078\ lnh + 0.00840\ (lnH)^2$	Moore, 2010
4			Biomass	Biomass carbon of exotic *E.* ×*sykesii* (n = 68)	$Ln\ B_{LS} = -1.4595 + 2.0618\ lnD$	Keith et al., 2009
5			Biomass	Biomass carbon of exotic *Salix* spp. (n = 21)	$B_{LS} = Exp(-2.2094 + 2.3867\ lnD)$	Jenkins et al., 2004
6			Biomass	Biomass carbon of native tree ferns (n = 459)	$C_{TFP} = 0.0027\ (D^2H)^{1.19}$	Beets et al., 2012
7		Shrubs	Biomass	Biomass carbon of discrete shrubs (n = 189)	$C_{Shrub} = 0.5\ \times S \times cuboid\ volume$	Coomes et al., 2002
8		Coarse woody debris	Volume	Volume of standing logs V_S (n = 167)	$V_{SL} = \pi\ \times \left(\frac{Dm}{2}\right)^2 \times\ L$	Beets et al., 2012; Holdaway et al., 2014
9			Volume	Volume of fallen logs V_F (n = 279)	$V_{FL} = \frac{\pi L}{3}[(r_1^2 + r_2^2) + (r_1 \times\ r_2)]$	Holdaway et al., 2014
10			Biomass	Biomass carbon of coarse woody debris of trees	$C_{CWD\_T} = 0.5\ \times \sum V\ \times FW \times\ DSM$	Holdaway et al., 2014
11			Biomass	Biomass carbon of coarse woody debris of tree ferns and palms	$C_{CWD\_TFP} = 0.0027 \times DSM \times (D^2L)^{1.19}$	Holdaway et al., 2014
12	Soil carbon		Proportion of organic carbon	Total organic carbon	$TOC\ (\%) = \left[\frac{DW_{oven} - IW_{400C}}{DW_{oven}}\right] \times 100/7.2$	Welsch, et al., 2019
13			Soil carbon	Soil organic carbon	$SOC = \%TOC\ \times\ BD\ \times\ h$	Welsch, et al., 2019

Appendix Table 2 Summary data of species and genus identified on the plant inventory on the three study farms and it's frequency of stem occurrence on the four quadrants within 10 × 10 m sample plots, shows species/taxa codes on inventory, local name, Māori name, suitability as timber source (TI), preference for soil erosion control (EC), preference for gully stabilization (GS), preference for soil surface erosion caused by wind (WE), preference as foodsource for birds (BF), preference as shelter for animals (SH), and the frequency of stems (Fr). "Y" symbol signifies the species that was planted to provide the certain ecosystem functions, and "*" symbol in the frequency column indicates that the species was present on the plot but the stem was too small (2.5 cm) to be counted on the stem inventory

	Code	Preferred species name	Local name	Maori name	TI	EC	GS	WE	BF	SH	NS	WE	Fr
1	AGAAUS	*Agathis australis* (D.Don) Lindl. ex Loudon (1829)	Cowrie kauri, kauri pine	Kaore (sapling), kauri, ware							Y	Y	4
2	ALEEXC	*Alectryon excelsus* Gaertn. (1788)	New Zealand ash, titoki	tapitapi, titoki, titongi, tokitoki, tongitongi, topitopi	Y	Y					Y	Y	37
3	ARISER	*Aristotelia serrata* (J.R.Forst. & G.Forst.) W.R.B.Oliv. (1921)	Wineberry, makomako	Mako, makomako		Y	Y				Y	Y	35
4	ASPSCA	*Asparagus scandens* Thunb. (1794)	Climbing asparagus	-									3
5	BEITAR	*Beilschmiedia tarairi* (A.Cunn.) Benth. & Hook.f. ex Kirk (1889)	Taraire	Taraire	Y					Y	Y	Y	88
6	BEITAW	*Beilschmiedia tawa* (A.Cunn.) Benth. & Hook.f. ex Kirk (1889)	Tawa, tawaroa	Tawa, tawa rautangi							Y	Y	48
7	BRAREP	*Brachyglottis repanda* J.R.Forst. & G.Forst. (1775)	Bushman's friend, rangiora	Kōuaha, pukapuka, pukariao, puke-rangiora, rangiora, raurākau, raurēkau, whārangi, whārangi-tawhito		Y	Y	Y			Y	Y	18
8	BUDDAV	*Buddleja davidii Franch.*	Buddleia, butterfly bush, summer lilac	-									*
9	CALTUG	*Calystegia tuguriorum* (G.Forst.) R.Br. ex Hook.f. (1854)	Climbing convolvulus, New Zealand bindweed	Pauwhiwhi, pawhiwhi, rarotawake							Y	Y	11
10	CARAUS	*Carmichaelia australis* R.Br. (1825)	North Island broom	Tainoka, tawao;maukoro, makaka		Y					Y	Y	39

11	CARSER	*Carpodetus serratus* J.R.Forst. & G.Forst. (1776)	Marble leaf, Motorbike tree	Piripiriwhata, punawata, putaputawata, putawata□;				Y	Y	69
12	CLEFOE	*Clematis foetida* Raoul (1846)	-	-				Y	Y	6
13	CLEFOR	*Clematis forsteri* J.F.Gmel. (1791)	Small white clematis	Pikiarero, pōānanga, pōhue, pōhuehue, pōpōhue, puatataua, puataua, puatautaua, puawānanga, puawhānanga				Y	Y	7
14	CLEPAN	*Clematis paniculata* J.F.Gmel. (1791)	White clematis	Pikiarero, pōhue, pōpokonui-a-hura, pūānanga, puapua, puatataua, puataua, puatauataua, puawānanga, puawhānanga				Y	Y	8
15	CLEVIT	*Clematis vitalba* L.	Old man's beard, traveller's joy	-						3
16	COPARB	*Coprosma arborea* Kirk (1877) [1878]	Mamangi, tree coprosma	Māmāngi			Y	Y	Y	8
17	COPARE	*Coprosma areolata* Cheeseman (1885) [1886]	Thin-leaved coprosma	Aruhe			Y	Y	Y	80
18	COPGRA	*Coprosma autumnalis* Colenso (1887)	kanono, manono	Kākawariki, kanono, kānonono, kapukiore, karamū-kueo, kawariki, kueo (fruit), manono, pāpāuma, patutiketike, raurākau,raurēkau, tapatapauma, toherāoa			Y	Y	Y	29
19	COPPRO	*Coprosma propinqua* A.Cunn. (1839)	Mingimingi	Miki, Mingi, Mingimingi			Y	Y	Y	112
20	COPPXR	*Coprosma propinqua x robusta*	-	-			Y	Y	Y	7
21	COPRHA	*Coprosma rhamnoides* A.Cunn. (1839)	-	-			Y	Y	Y	77
22	COPRIG	*Coprosma rigida* Cheeseman (1886) [1887]	-				Y	Y	Y	15
23	COPROB	*Coprosma robusta* Raoul (1844)	Glossy karamu	Kākaramū, kākarangū, karamū, kāramuramu, karangū	Y	Y	Y	Y	Y	37

24	COPROT	*Coprosma rotundifolia* A.Cunn. (1839)	Round-leaved coprosma	-					Y	Y	Y	10
25	COPSPA	*Coprosma spathulata* A.Cunn. (1839)	-	-					Y	Y	Y	16
26	COPVIR	*Coprosma virescens* Petrie (1878) [1879]	-	-					Y	Y	Y	*
27	CORARB	*Coriaria arborea* Linds. (1868)	tree tutu	Pūhou, tāweku, tūpākihi, tutu		Y				Y	Y	7
28	CORAUS	*Cordyline australis* (G.Forst.) Endl. (1833)	Cabbage tree, giant dracena, grass palm, palm lily, sago palm, ti, ti kouka	Kāuka, kiokio, kōuka, tī, tī awe, ti kōuka, tī para, tī pua, tī rākau, whanake			Y	Y	Y	Y	Y	29
29	CORLAE	*Corynocarpus laevigatus* J.R.Forst. & G.Forst. (1776)	Karaka, karaka nut	Karaka, kōpī			Y	Y	Y	Y	Y	42
30	COTCOC	*Cotoneaster coriaceus* Franch. (1890)	-	-		Y	Y	Y				3
31	CUPMAC	*Cupressus macrocarpa* Hartw. (1847)	Macrocarpa, Monterey cypress	-				Y				*
32	CYADEA	*Cyathea dealbata* (G.Forst.) Sw. (1801)	Ponga, punga, silver fern	Kaponga, kātote, ponga, punga						Y	Y	136
33	CYAMED	*Cyathea medullaris* (G.Forst.) Sw. (1801)	Black mamaku; black tree fern, mamaku	Katātā, kōrau, mamaku, pītau						Y	Y	24
34	CYASMI	*Cyathea smithii* Hook.f. (1854)	Ponga, Smith's tree fern,soft tree fern	Kātote, neineikura, whē						Y	Y	18
35	CYSTOP	*Cystopteris speciosa* Bernh. (1805)	-	-						Y	Y	*
36	DACCUP	*Dacrydium cupressinum* Sol. ex G.Forst. (1786)	-	-						Y	Y	*
37	DACDAC	*Dacrycarpus dacrydioides* (A.Rich.) de Laub. (1969)	Kahikatea, white pine	Kahika, kahikatea, kaikatea, katea, kōaka, koroī	Y	Y		Y	Y	Y	Y	248
38	DICFIB	*Dicksonia fibrosa* Colenso (1844)	Golden tree fern, whekī-ponga	Kuranui-pākā, kurīpākā, pūnui, tūkirunga, wekī, whekī, whekī-kōhunga						Y	Y	15
39	DICSQU	*Dicksonia squarrosa* (G.Forst.) Sw. (1801)	Harsh tree fern, rough tree fern	Atewheki, pakue, pēhiakura, tiotirawa, tūākura, tūōkura, uruuruwhenua, whekī						Y	Y	325

40	DISTOU	*Discaria toumatou* Raoul (1844)	Matagouri, wild Irishman	Tūmatakuri, tūmatakuru, tūturi						Y	Y	97
41	DODVIS	*Dodonaea viscosa* Jacq. (1760)	Akeake, sticky hop-bush	ake, ake rautangi, akeake		Y	Y	Y		Y	Y	9
42	DYSSPE	*Dysoxylum spectabile* (G.Forst.) Hook.f. (1864)	Kohekohe, New Zealand's mahogany	Kohe, kohekohe, kohepi (flowers), kohepu (flowers), koheriki, māota (flowers)	Y				Y	Y	Y	31
43	ELADEN	*Elaeocarpus dentatus* (J.R.Forst. & G.Forst.) Vahl (1794)	-	Hangehange, hīnau, pōkākā, whīnau	Y					Y	Y	6
44	ELAHO O	*Elaeocarpus hookerianus* Raoul (1846)	-	Mahimahi, pōkākā, puka, whīnau	Y					Y	Y	12
45	ERYXSY	*Erythrina ×sykesii* Barneby & Krukoff (1974)	Coral tree	-					Y			14
46	FREBAN	*Freycinetia banksii* A.Cunn. (1837)	Kiekie	Kiekie						Y	Y	*
47	FUCEXC	*Fuchsia excorticata* (J.R.Forst. & G.Forst.) L.f. (1781)	Fuchsia, tree fuchsia	Hōnā (fruit), kōhutuhutu, kōnini (fruit), kōtukutuku, māti (fruit), tākawa (fruit)		Y		Y		Y	Y	35
48	FUCPER	*Fuchsia perscandens* Cockayne & Allan (1926) [1927]	-	-					Y	Y	Y	*
49	GENLIG	*Geniostoma ligustrifolium* A.Cunn. (1839)	-	-						Y	Y	60
50	GRILIT	*Griselinia littoralis* (Raoul) Raoul (1846)	Broadleaf	Huariki (fruit), kāpuka, māihīhi, pāpāuma, paraparauma, tapatapauma				Y		Y	Y	14
51	GRILUC	*Griselinia lucida* (J.R.Forst. & G.Forst.) G.Forst. (1786)	Puka	Akakōpuka, akapuka, puka, pukatea						Y	Y	*
52	HEDAR B	*Hedycarya arborea* J.R.Forst. & G.Forst. (1776)	Pigeonwood, porokaiwhiri	Kaiwhiri, kaiwhiria, kōporokaiwhiri, pōporokaiwhiri, pōporokaiwhiria, porokaiwhiri,					Y	Y	Y	70

				porokaiwhiria, poroporokaiwhiria									
53	HEDHEL	*Hedera helix* L.	English ivy, ivy	-									*
54	HOHANG	*Hoheria angustifolia* Raoul (1844)	Mountain lacebark, narrow-leaved houhere	Houhi, houhi-puruhi, puruhi		Y		Y			Y	Y	16
55	HOHSEX	*Hoheria sexstylosa* Colenso (1884) [1885]	Graceful lacebark, lacebark	Houhere, houhiongaonga		Y		Y			Y	Y	27
56	ILEMIC	*Ileostylus micranthus* (Hook.f.) Tiegh. (1894)	Mistletoe, small-flowered mistletoe	Pikirangi, pirinoa, pirirangi, pirita							Y	Y	*
57	KNIEXC	*Knightia excelsa* R.Br. (1810)	New Zealand honeysuckle. rewarewa	Rewarewa	Y			Y		Y	Y	Y	63
58	KUNROB	*Kunzea robusta* de Lange & Toelken (2014)	-	-	Y		Y		Y	Y	Y		30
59	LEPSCO	Leptospermum scoparium J.R.Forst. & G.Forst. (1776)	Red tea tree, tea tree, Mānuka	Kahikātoa, kātoa, mānuka, pata, rauiri, rauwiri		Y	Y	Y			Y	Y	196
60	LEUFAS	*Leucopogon fasciculatus* (G.Forst.) A.Rich. (1832)	Mingimingi, tall mingimingi	Hukihukiraho, kaikaiatua, mānuka-rauriki, mikimiki, mingi, mingimingi, ngohungohu, tūmingi							Y	Y	*
61	LEYFOR	*Leycesteria formosa* Wall. (1824)	himalaya honeysuckle	-									*
62	LIGLUC	*Ligustrum lucidum* W.T.Aiton	Broadleaf privet, tree privet	-									*
63	LIGSIN	*Ligustrum sinense* Lour.	Chinese privet, small-leaf privet	-									*
64	MELALP	*Melicytus aff. alpinus* (Blondin) (nom. inv.)	Porcupine shrub	-						Y	Y	Y	*
65	MELMIC	*Melicytus micranthus* (Hook.f.) Hook.f. (1852)	Swamp mahoe	Manakura						Y	Y	Y	*
66	MELRAM	*Melicytus ramiflorus* J.R.Forst. & G.Forst. (1776)	Māhoe, whiteywood	Hinahina, inaina, inihina, māhoe, moeahu		Y	Y			Y	Y	Y	184
67	MELSIM	*Melicope simplex* A.Cunn. (1839)	Poataniwha	Poataniwha, tātaka						Y	Y	Y	*

68	METCOL	*Metrosideros colensoi* Hook.f. (1852)	-	-		Y		Y	Y	29
69	METDIF	*Metrosideros diffusa* (G.Forst.) Sm. (1797)	White rata	Rātā		Y		Y	Y	25
70	METFUL	*Metrosideros fulgens* Sol. ex Gaertn. (1788)	Akakura, akatawhitawhi, Scarlet rata, vine rata	Aka, akakura, akatawhitawhi, akatawhiwhi, amaru, kahika, kahikahika, rata; ratapiki		Y		Y	Y	*
71	METPER	*Metrosideros perforata* (J.R.Forst. & G.Forst.) A.Rich. (1832)	Clinging rata, small white rata, Akatea	Aka, akatea, akatorotoro, koro, torotoro, whakapiopio		Y		Y	Y	47
72	METROB	*Metrosideros robusta* A.Cunn. (1839)	Northern rata, rata	Rātā	Y	Y		Y	Y	*
73	MUEAUS	*Muehlenbeckia australis* (G.Forst.) Meisn. (1841)	Large-leaved muehlenbeckia, pōhuehue	Pōhuehue, puka				Y	Y	75
74	MUEAXI	*Muehlenbeckia axillaris* (Hook.f.) Endl. (1848)	-	-				Y	Y	*
75	MUECOM	*Muehlenbeckia complexa* (A.Cunn.) Meisn. (1841)	Scrub pohuehue, small-leaved pohuehue, wire vine	Pōhue, pōhuehue, pōpōhue, tororaro, waekāhu	Y			Y	Y	16
76	MYOLAE	*Myoporum laetum* G.Forst. (1786)	Ngaio	Ngaio	Y	Y		Y	Y	*
77	MYRAUS	*Myrsine australis* (A.Rich.) Allan (1947)	Māpou, red mapou, red matipo	Māpau, māpou, mataira, matipou, takapou, tāpau, tīpau		Y	Y	Y	Y	85
78	NESLAN	*Nestegis lanceolata* (Hook.f.) L.A.S.Johnson (1958)	white maire	Maire, maire raunui, maire rauriki				Y	Y	22
79	OLEAVI	*Olearia avicenniifolia* (Raoul) Hook.f. (1864)	mountain akeake	Akeake		Y		Y	Y	*
80	OLEPAN	*Olearia paniculata* (J.R.Forst. & G.Forst.) Druce (1917)	Akiraho, golden akeake	Akepiro, akiraho		Y		Y	Y	51
81	OLERAN	*Olearia rani* (A.Cunn.) Druce (1917)	Heketara	Akewharangi, heketara, ngungu, taraheke, tātaraheke, wharangi-piro		Y		Y	Y	*
82	PARCAP	*Parsonsia capsularis* (G.Forst.) DC. (1844)	Akakaikiore, New Zealand jasmine, small flowered jasmine	Akakaikiore, akakiore, kaikū, kaikūkū, kaiwhiria, tōtoroene, tōtorowene				Y	Y	*

83	PARHET	*Parsonsia heterophylla* A.Cunn.	Akakaikiore, New Zealand jasmine	Akakaikiore, akakiore, kaihua, kaikū, kaiwhiria, poapoa, tautauā, tawhiwhi, tūtae-kererū						Y	Y	45
84	PASTET	*Passiflora tetrandra* Banks ex DC. (1828)	New Zealand passion flower, New Zealand passionfruit, Kohia	Aka, akakaikū, akakaikūkū, akakōhia, akakūkū, akatororaro, kāhia, kaimanu, kohe, kohia, kōhia, kūpapa, pōhue, pōpōhue						Y	Y	*
85	PENCOR	*Pennantia corymbosa* J.R.Forst. & G.Forst. (1776)	-	Ahikōmau, hine-kaikōmako, kahikōmako, kaikōmako						Y	Y	51
86	PHOTEN	*Phormium tenax* J.R.Forst. & G.Forst. (1776)	Flax, harakeke, lowland flax, New Zealand flax, swamp flax	Harakeke, harareke, kōrari		Y	Y			Y	Y	*
87	PHYOCT	*Phytolacca octandra* L.	Dyeberry, inkweed, red ink plant	-								*
88	PHYTRI	*Phyllocladus trichomanoides* G.Benn ex D.Don (1832)	Celery pine, tanekaha	Ahotea, nīko, tānekaha, tanekaha, tāwaiwai, toatoa	Y			Y		Y	Y	*
89	PINRAD	*Pinus radiata* D.Don	Monterey pine, radiata pine	-	Y							41
90	PIPEXC	*Piper excelsum* G.Forst. (1786)	-	-						Y	Y	*
91	PITEUG	*Pittosporum eugenioides* A.Cunn. (1840)	Lemonwood, tarata	Kīhihi, tarata		Y	Y			Y	Y	*
92	PITTEN	*Pittosporum tenuifolium* Sol. ex Gaertn. (1788)	Black matipo, kohukohu	Kaikaro, kōhūhū, kohukohu, koihu, kōwhiwhi, māpauriki, pōhiri, pōwhiri, rautāwhiri, tāwhiri, tawhiwhi			Y			Y	Y	33
93	PODTOT	*Podocarpus totara* D.Don (1832)	Tōtara	Amoka, tōtara	Y	Y	Y			Y	Y	302
94	PRUTAX	*Prumnopitys taxifolia* (Sol. ex D.Don) de Laub. (1978)	Black pine, black pine, matai	Kāī, kākāī, māī, mataī						Y	Y	*
95	PSEARB	*Pseudopanax arboreus* (L.f.) K.Koch (1859)	Five-finger, whauwhaupaku	Houhou, parapara, puahou, tauparapara, whau, whaupaku, whauwhau, whauwhaupaku		Y	Y		Y	Y	Y	77

96	PSEAXI	*Pseudowintera axillaris* (J.R.Forst. & G.Forst.) Dandy (1933)	Horopito, lowland horopito, lowland pepper tree;	Puhikawa; horopito				Y	Y	*
97	PSECOL	*Pseudowintera colorata* (Raoul) Dandy (1933)	Alpine pepper tree, horopito, mountain horopitor, pepper tree	Ramarama, red horopito, ōramarama				Y	Y	*
98	PSECRA	*Pseudopanax crassifolius* (Sol. ex A.Cunn.) K.Koch (1859)	Horoeka, lancewood	Hoheka, horoeka, koeka, kokoeka, ohoeka		Y	Y	Y	Y	63
99	RAUANO	*Raukaua anomalus* (Hook.) A.D.Mitch., Frodin & Heads (1997)	-	-				Y	Y	*
100	RHASOL	*Rhabdothamnus solandri* A.Cunn.	New Zealand gloxinia	Kaikaiatua, mātā, mātātā, taurepo, waiūatua				Y	Y	*
101	RHOSAP	*Rhopalostylis sapida* H.Wendl. & Drude (1878)	Feather duster palm, nīkau, nikau palm	Nīkau				Y	Y	177
102	RIPSCA	*Ripogonum scandens* J.R.Forst. & G.Forst. (1776)	Kareao, supplejack	Akapirita, kakareao, kakarewao, kareao, karewao, kekereao, pirita, taiore				Y	Y	41
103	ROSRUB	*Rosa rubiginosa* L.	Apple-scented rose, eglantine, sweet brier	Mihinare						*
104	RUBCIS	*Rubus cissoides* A.Cunn. (1839)	Bush lawyer	Taraheke, taramoa, tātaraheke, tātarāmoa				Y	Y	*
105	RUBFRU	*Rubus fruticosus* L.	Blackberry	-						*
106	RUBSCH	*Rubus schmidelioides* A.Cunn. (1839)	Bush lawyer, white-leaved lawyer, tātarāmoa	Tātarāmoa				Y	Y	*
107	RUBSQU	*Rubus squarrosus* Fritsch (1886)	Leafless lawyer, yellow-prickled lawyer					Y	Y	
108	SALALB	*Salix alba* L.	Golden willow, silver willow, white willow	-	Y					*
109	SALCIN	*Salix cinerea* L. (1753)	Grey willow	-	Y					*
110	SALXFR	*Salix ×fragilis* L.	Crack willow	-	Y					*
111	SAMNIG	*Sambucus nigra* L.	Elderberry, black elder, elder	-						*

112	SCAGEN	*Scandia geniculata* (G.Forst.) J.W.Dawson (1967)	-	-						Y	Y	*
113	SCHDIG	*Schefflera digitata* J.R.Forst. & G.Forst. (1776)	Seven-finger	Kohi, Kotētē, Patate, Patatē, Patē, Patētē		Y	Y			Y	Y	*
114	SENGLS	*Senecio glastifolius* L.f. (1782)	Holly-leaved senecio	-								*
115	SOPMIC	*Sophora microphylla* Aiton (1789)	Kōwhai, small-leaved kowhai, Weeping kowhai	Kōwhai	Y	Y	Y		Y	Y	Y	*
116	STRHET	*Streblus heterophyllus* (Blume) Corner (1962)	Milk tree, small-leaved milk tree, Turepo	Ewekuri, tāwari, tūrepo						Y	Y	*
117	URTFER	*Urtica ferox* G.Forst. (1786)	Ongaonga, tree nettle	Ongaonga, taraonga, taraongaonga						Y	Y	5
118	VERSAL	*Veronica salicifolia* G.Forst. (1786)	Koromiko	Kōkōmuka, kōkoromiko, kōkoromuka, korohiko, korokio, koromiko, koromuka						Y	Y	1
119	VERSTR	*Veronica stricta* Banks & Sol. ex Benth. (1846)	Koromiko	Kōkōmuka, Kōkoromiko,kōkoromuka, korohiko, korokio, koromiko, koromuka						Y	Y	1
120	VITLUC	*Vitex lucens* Kirk (1897)	New Zealand oak (English), pūriri (English)	Kauere, pūriri	Y				Y	Y	Y	40
121	WEIRAC	*Weinmannia racemosa* L.f. (1781)	Kāmahi	Kāmahi, tawhero, tōwai	Y	Y	Y			Y	Y	18
122	BRACH Y	*Brachyscome species* (Cass.) (1816)	-	-						Y	Y	1
123	CLEMA T	*Clematis species* L. (1753)	Clematis	Akakaikū, akakaikūkū, aka-kōpū-kererū, akakūkū, hokokūkū, pōānanga, pōhue, pōtaetae, puatataua, puatororaro, upokonui-a-ura						Y	Y	14
124	COPROS	*Coprosma* J.R.Forst. & G.Forst. (1775)	Coprosma, looking-glass plant, mirror plant	-				Y	Y	Y	112	
125	RUBUS	*Rubus* L. (1753)	Blackberry, bramble, brier, dewberry, raspberry	-						Y	Y	1

126	PINUS	*Pinus* L. (1753)	Pine	-			Y	14
127	PRUNUS	*Prunus* L. (1753)	Cherry, peach, plum;	-			Y	7
128	SALIX	*Salix* L. (1753)	Willow	-	Y		Y	11

Appendix Table 3 Mean and standard deviation (Mean ± Standard deviation) of carbon stocks per hectare of eight plant community types (t C ha-[1])

Carbon pools	Components	*Muehlenbeckia complexa* shrubland alliance	*Coprosma propinqua/Pseudopanax arboreus* shrubland alliance	*Podocarpus totara/Muehlenbeckia australis* woodland alliance	*Dacrycarpus dacrydioides/Hedycarya arborea* woodland alliance	*Podocarpus totara-Corynocarpus laevigatus* woodland alliance	*Podocarpus totara-Vitex lucens* woodland alliance	*Pinus* spp. woodland alliance	Exotic broadleaf woodland
Above-ground	Live stem	6.88 ± 16.13	49.83 ± 55.57	194.06 ± 141.78	133.76 ± 67.43	92.46 ± 64.92	120.18 ± 100.56	187.31 ± 92.74	244.43 ± 226.53
	Coarse woody debris	0	1.78 ± 2.91	3.11 ± 4.81	5.64 ± 10.01	4.26 ± 12.06	2.81 ± 5.59	7.4 ± 5.21	0.02 ± 0.06
	Shrub	2.28 ± 1.89	0.2 ± 1.00	0	0	0	0	0	2.31 ± 5.66
	Total aboveground	9.16 ± 15.13	51.82 ± 55.31	197.16 ± 141.99	139.4 ± 66.53	96.72 ± 66.36	122.99 ± 101.40	194.71 ± 91.49	246.76 ± 223.6
Below-ground	Belowground	2.29 ± 3.78	12.95 ± 13.83	49.29 ± 35.50	34.85 ± 16.63	24.18 ± 16.59	30.75 ± 25.35	48.68 ± 22.87	61.69 ± 55.9
Soil	Soil carbon	16.77 ± 2.78	14.55 ± 5.89	17.49 ± 3.67	19.00 ± 6.62	18.30 ± 4.72	24.29 ± 4.72	18.82 ± 4.98	19.69 ± 6.99
Total Carbon		28.22 ± 18.63	79.33 ± 72.20	263.94 ± 178.79	193.25 ± 81.77	139.21 ± 86.86	178.02 ± 126.71	262.21 ± 112.17	328.14 ± 276.87

Appendix Table 4 Mean, standard deviation (Mean ± Standard deviation), and sum of carbon stocks per hectare of three sheep and beef farms and all plots (t C ha^{-1})

Carbon pools	Components	Kaipara		Ruapehu		Hurunui		Average Plot	
		Mean±SD	Sum	Mean±SD	Sum	Mean±SD	Sum	Mean±SD	Sum

Aboveground	Live stem	128.12 ± 117.5	6277.68	163.66 ± 110.91	10638.18	50.26 ± 72.3	1558.08	127.41 ± 114.16	18473.93
	CWD	3.43 ± 7.01	167.93	4.38 ± 7.8	284.98	1.39 ± 2.66	42.94	3.42 ± 6.8	495.85
	Shrub	0	0	0.21 ± 1.72	13.86	0.6 ± 1.43	18.58	0.22 ± 1.34	32.45
	Total aboveground	131.54 ± 117.86	6445.61	168.26 ± 110.23	10937.02	52.25 ± 71.85	1619.61	131.05 ± 114.32	19002.23
Belowground	BGB	32.89 ± 29.47	1611.4	42.07 ± 27.56	2734.26	13.06 ± 17.96	404.9	32.76 ± 28.58	4750.56
Soil	Soil carbon	23.03 ± 7.36	1128.67	18.22 ± 5.60	1184.22	15.00 ± 5.35	465.13	19.16 ± 6.87	2778.01
Total carbon		187.46 ± 147.46	9185.67	228.55± 137.44	14,855.49	80.31 ± 91.45	2,489.64	174.44 ± 143.53	26,530.81

Appendix Table 5 Summary of mean and standard deviation (Mean ± Standard deviation) of carbon stocks per hectare of three structural plant community types (t C ha^{-1}): Shrubland alliances, Native-dominated woodland alliances, and Exotic-dominated woodland alliances

Carbon pools	Shrubland alliances	Native-dominated woodland alliances	Exotic-dominated woodland alliances
Total aboveground carbon stock	43.29 ± 52.60	145.20 ± 106.26	217.02 ± 156.37
Total soil carbon stock	14.99 ± 5.44	20.39 ± 6.96	19.19 ± 5.69
Total belowgroundcarbon stock	10.82 ± 13.15	36.30 ± 26.56	54.25 ± 39.09
Total Carbon stock	69.11 ± 68.02	201.89 ± 132.63	290.46 ± 193.40

Appendix Table 6 Summary of % radar area and mean total carbon stock (t C ha^{-1}), and mean scores of other ecosystem functions: timber provision, gully stabilization, wine erosion reduction, wind erosion reduction, animal shelter, source of bird foods, increasing native biodiversity, less invasive weeds, and less community flammability; of eight plant community types

Row Labels	% radar area	Mean of total Carbon Stock	Mean of timber provision	Mean of gully stabilisator	Mean of soil erosion reduction	Mean of wind erosion reduction	Mean of animal shelter	Mean of source of food for birds	Mean of increasing native biodiversity	Mean of inverted Invasive weeds	Mean of inverted Community flammability
Corposma propinqua/Pseudopanax	40.69	79.33	0	8.79	11.21	11.96	0.92	6.67	17.29	17.29	2.56

arboreus Shrubland Alliance											
Dacrycarpus dacrydioides/*Hedycarya arborea* Woodland Alliance	27.77	193.25	3	1.72	4.91	4.56	2.84	2.75	18.44	18.66	1.85
Exotic broadleaf Woodland	10.19	328.14	0	2	0	0.17	2	2.33	0.33	2	1.14
Muehlenbeckia complexa Shrubland Alliance	27.14	28.22	2.83	0.67	7	3.5	0	10.17	29.5	29.5	2.72
Pinus spp. Woodland Alliance	11.4	262.21	4.13	0	0	0	0	0	0	4.13	1.27
Podocarpus totara/*Muehlenbeckia australis* Woodland Alliance	20.54	263.94	4	0.12	4.42	4.42	3.46	1.61	5.96	5.96	1.28
Podocarpus totara-Corynocarpus laevigatus Woodland Alliance	22.71	139.21	4.38	2.63	7.13	7.88	5.13	1.62	8.87	8.88	1.35
Podocarpus totara-Vitex lucens Woodland Alliance	40.14	178.02	7.97	0.71	5.69	7.26	2.08	9.37	18.34	18.37	2.16

Appendix Table 7 Summary Z-score of eight plant community types on percentage of radar area, mean total carbon stock (t C ha-1) and mean scores of other ecosystem functions: timber provision, gully stabilization, wine erosion reduction, wind erosion reduction, animal shelter, source of bird foods, increasing native biodiversity, less invasive weeds, and less community flammability

Row Labels	% radar area	Mean of total Carbon Stock	Mean of timber provision	Mean of gully stabilisator	Mean of soil erosion reduction	Mean of wind erosion reduction	Mean of animal shelter	Mean of source of food for birds	Mean of increasing native biodiversity	Mean of inverted Invasive weeds	Mean of inverted Community flammability

Corposma propinqua/*Pseudopanax arboreus* Shrubland Alliance	1.32	-1.04	-1.28	2.34	1.64	1.74	-0.64	0.61	0.48	0.45	1.22
Dacrycarpus dacrydioides/*Hedycarya arborea* Woodland Alliance	0.23	0.09	-0.11	-0.12	-0.03	-0.1	0.45	-0.4	0.59	0.59	0.09
Exotic broadleaf Woodland	-1.3	1.43	-1.28	-0.03	-1.35	-1.19	-0.031	-0.51	-1.17	-1.18	-1.04
Muehlenbeckia complexa Shrubland Alliance	0.18	-1.55	-0.18	-0.49	0.52	-0.36	-1.16	1.51	1.67	1.75	1.48
Pinus spp. Woodland Alliance	-1.19	0.78	0.32	-0.72	-1.35	-1.24	-1.16	-1.11	-1.19	-0.95	-0.83
Podocarpus totara/*Muehlenbeckia australis* Woodland Alliance	-0.4	0.79	0.28	-0.68	-0.17	-0.13	0.8	-0.7	-0.62	-0.76	-0.82
Podocarpus totara-Corynocarpus laevigatus Woodland Alliance	-0.21	-0.45	0.42	0.19	0.56	0.72	1.74	-0.7	-0.34	-0.45	-0.7
Podocarpus totara-Vitex lucens Woodland Alliance	1.32	-0.07	1.82	-0.483	0.17	0.57	0.02	1.31	0.58	0.56	0.59

Appendix Table 8 Description of intervention classes based on original main classes of land-cover classification based on expert judgement’s ground truthing

Intervention class	**Description**	**Class**	**Species**

Mixed-native	Forest patches that have always been present and contain old-growth canopy trees (e.g. podocarp, tawa, beech, etc.), although they have been modified by logging (cut-over) and/or degraded by animals	Old-growth	Tawa (Beilschmiedia tawa (A.Cunn.) Benth. et Hook.f. ex Kirk), Podocarp (Podocarpaceae eg. Podocarpus spp., Dacrycarpus dacrydioides (A.Rich.) de Laub., New Zealand beech (Nothofagus antartica (G.Forst.)Oerst.), Fucospora spp.)
	A widespread type of woody vegetation with a combination of native species often dominated by totara or kahikatea characterized by having established on sites that were previously pasture or plantation forest.	Mixed-native	totara (P. totara) or kahikatea (D. dacrydioides)
Shrubs	Regenerated patches dominated by some tall-shrub/seral tree species such as kānuka, mahoe (Melicytus J.R.Forst. et G.Forst.), lemonwood (Pittosporum eugenioides A.Cunn.), five-finger (Pseudopanax colensoi (Hook.f.) Philipson), and relatively younger than the mixed-native patches	Kanuka-manuka	kānuka (Kunzea spp. de Lange et Toelken), manuka (Leptospermum scoparium J.R.Forst. et G.Forst.)
Shrubs	Early regenerated shrubland dominated by matagouri and other shrubs species	Matagouri	Matagouri (Discaria toumatou Raoul)
Exotic-dominated woodlots	Plantation forests or pine, eucalyptus, other conifers, etc.	Pine	Pine (Pinus sp. (L.)), Eucalyptus sp. (L'Her.), Douglas Fir (Abies grandis (Douglas ex D.Don) Lindl.))
Exotic-dominated woodlots	Planted patches of willow, sycamore, oak, etc., including shelterbelts	Deciduous	Willow ((Salix sp. (L.)), Erythrina xsykesii (Barneby & Krukoff.), oak (Quercus sp. (L.)), sycamore (Acer sp. (L.))
Shrubs	Vast area of scrubland dominated by exotic gorse	Gorse	Gorse (U. eurapaeus)
	Total area of woody vegetation		

Appendix Table 9 A) Summary of baseline and results (Mean ± Standard Deviation) of the mean total vegetation area (m^2), mean total carbon density (kt C ha^{-1}), mean biodiversity potential, and mean Nearest Neighbor Distance (NND) (m) on three landscapes: Kaipara, Ruapehu, and Hurunui, for baseline and after seven scenarios intervention: Scenario 1 (restore bare gullies), Scenario 2 (revegetating shrubs), Scenario 3 (Revegetating exotic-dominated woodlots) (restoring shrubs and exotic-dominated woodlots and revegetating bare gullies), Scenario 4(restoring shrubs and exotic-dominated woodlots), Scenario 5 (restoring shrubs and revegetating bare gullies), Scenario 6 (restoring exotic woodlots and revegetating bare gullies), and Scenario 7 (restoring shrubs and exotic-dominated woodlots and revegetating bare gullies) on intervention intensity parameter 10 % and aggregation parameter 100m , B) Summary for the differences from baseline for all scenarios models on intervention intensity parameter 10 % and aggregation parameter 100m; and C) Summary of the Z-score of t the results, relative to other scenarios in each landscape (Mean ± Standard Deviation) of the mean total vegetation area (m^2), mean total carbon density (kt C ha^{-1}), mean biodiversity potential, and mean Nearest Neighbour Distance (NND) (m) on three landscapes: Kaipara, Ruapehu, and Hurunui, for baseline and after seven scenarios intervention: Scenario 1 (restore bare gullies), Scenario 2 (revegetating shrubs), Scenario 3 (Revegetating exotic-dominated woodlots) (restoring shrubs and exotic-dominated woodlots and revegetating bare gullies), Scenario 4(restoring shrubs and exotic-dominated woodlots), Scenario 5 (restoring shrubs and revegetating bare gullies), Scenario 6 (restoring exotic woodlots and revegetating bare gullies), and Scenario 7 (restoring shrubs and exotic-dominated woodlots and revegetating bare gullies)

Table A. Summary of baseline and results

Landscape	Scenario	Mean total habitat (ha)	Mean total carbon stock (t C ha^{-1}))	Mean total biodiversity potential	Mean prop small patches (%)	Mean mean NND (m)
Kaipara	Baseline	1121.98	265.55	4.79	58.85	215.45
	Scenario 1	1582.52 ± 11.05	300.48 ± 2.46	4.93 ± 0.02	56.92 ± 0.93	214.98±2.51
	Scenario 2	1281.89 ± 7.19	275.96 ± 0.71	4.95 ± 0.02	59.59 ± 1.00	211.18±2.59
	Scenario 3	1289.55 ±10.28	253.77 ± 1.20	5.18 ± 0.04	59.69 ± 1.03	211.68±2.62
	Scenario 4	1354.52 ± 10.03	264.42 ± 1.29	5.34 ± 0.03	60.53 ± 1.05	207.75±3.06
	Scenario 5	1647.34 ± 13.42	311.86 ± 2.70	5.10 ± 0.03	58.49 ± 1.12	213.31±3.11

	Scenario 6	1654.81 ± 12.2	289.36 ± 2.68	5.31 ± 0.04	58.17 ± 1.21	212.34±3.04
	Scenario 7	1722.19 ± 12.9	300.25 ± 2.76	5.47 ± 0.03	59.34 ± 1.10	210.16±2.93
Ruapehu	Baseline	1707.03	363.49	4.12	62.26	248.20
	Scenario 1	2005.16 ± 31.8	478.25 ± 4.9	5.00 ± 0.04	58.78 ± 0.92	249.76 ± 4.1
	Scenario 2	1346.13 ± 9.88	376.14 ± 1.01	4.31 ± 0.02	63.89 ± 0.81	245.01 ± 2.47
	Scenario 3	1447.22 ± 14.57	338.79 ± 1.65	4.89 ± 0.04	64.09 ± 1.2	254.61 ± 3.82
	Scenario 4	1521.34 ± 10.93	351.81 ± 1.47	5.06 ± 0.04	65.39 ± 0.93	251.78 ± 3.99
	Scenario 5	2083.28 ± 34.81	490.97 ± 5.37	5.16 ± 0.05	60.18 ± 1.23	243.80 ± 5.04
	Scenario 6	2181.53 ± 33.91	452.81 ± 4.77	5.66 ± 0.05	60.27 ± 1.22	250.36 ± 5.17
	Scenario 7	2177.27 ± 30.87	453.18 ± 5.32	5.66 ± 0.05	60.73 ± 1.24	250.58 ± 5.91
Hurunui	Baseline	284.50	102.20	2.81	68.99	449.52
	Scenario 1	1111.64 ± 11.64	177.44 ± 3.86	3.35 ± 0.04	51.62 ± 2.06	383.57 ± 8.83
	Scenario 2	467.9 ± 9.11	131.16 ± 1.25	3.55 ± 0.03	75.58 ± 1.38	268.6 ± 6.3
	Scenario 3	332.33 ± 6.04	93.25 ± 0.97	3.18 ± 0.03	70.44 ± 1.71	407.58 ± 10.55
	Scenario 4	513.79 ± 11.8	122.18 ± 1.77	3.91 ± 0.04	73.44 ± 1.43	268.12 ± 7.23
	Scenario 5	1291.6 ± 13.47	206.27 ± 3.86	3.99 ± 0.05	68.07 ± 1.54	266.65 ± 7.46
	Scenario 6	1154.71 ± 14.58	168.21 ± 4.23	3.65 ± 0.05	55.54 ± 1.93	355.06 ± 12.16
	Scenario 7	1341.93 ± 15.11	197.18 ± 3.16	4.31 ± 0.06	65.85 ± 1.65	268.64 ± 6.75

Table B. Summary for the differences from baseline

		Mean					SD				
Farm	Scenario	TC	TB	PSP	NND	Hab	TC	TB	PSP	NND	Hab
Hurunui	Scenario1	75.24	0.54	-17.38	-65.95	827.14	3.86	0.04	2.06	8.83	11.64
Hurunui	Scenario2	28.96	0.74	6.58	-180.91	183.4	1.25	0.03	1.38	6.3	9.11
Hurunui	Scenario3	-8.95	0.37	1.44	-41.94	47.83	0.97	0.03	1.71	10.55	6.04
Hurunui	Scenario4	19.98	1.11	4.44	-181.4	229.29	1.77	0.04	1.43	7.23	11.8
Hurunui	Scenario5	103.76	1.18	-1.03	-180.77	1,005.93	3.88	0.06	1.77	7.11	15.39

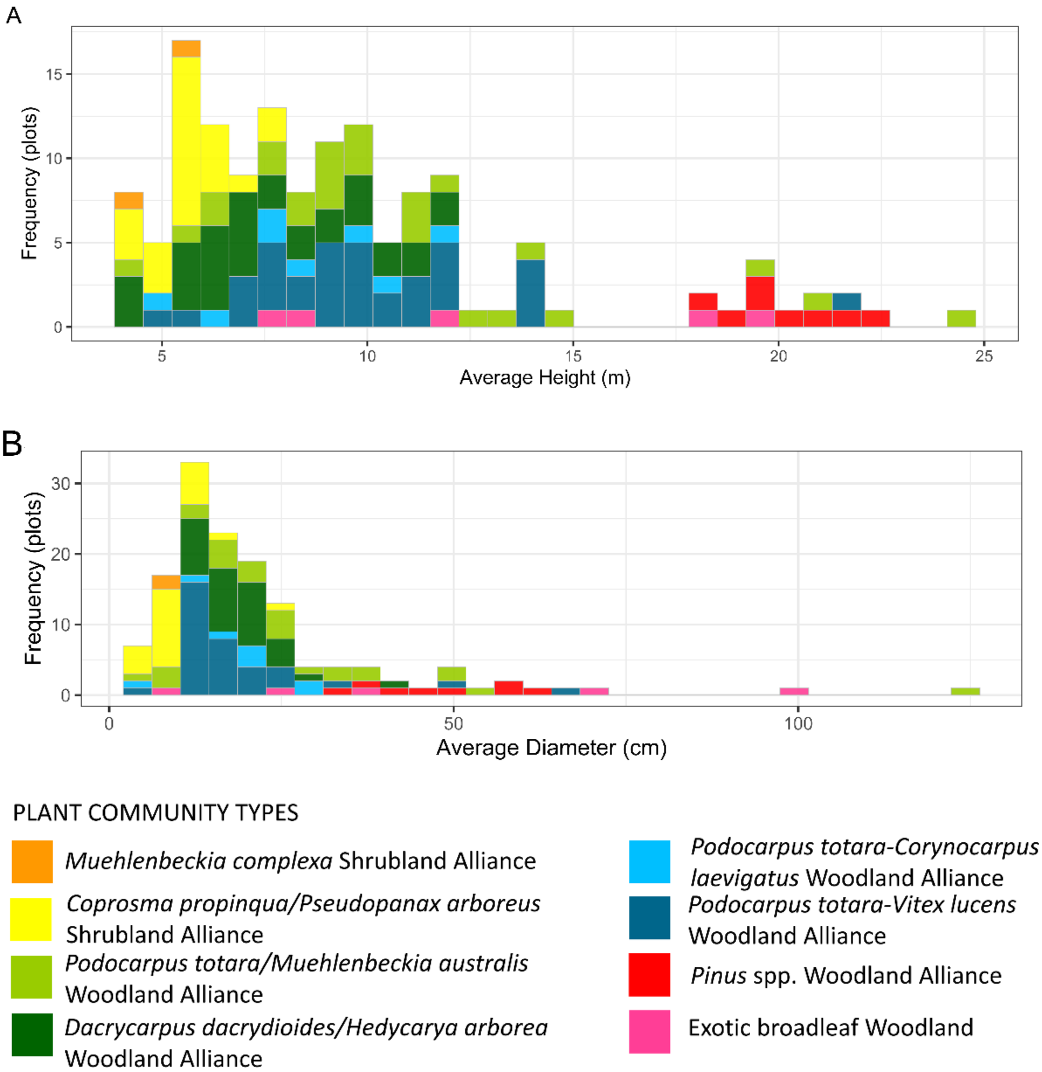
Hurunui	Scenario6	66.01	0.85	-13.46	-94.46	870.21	4.23	0.05	1.93	12.16	14.58
Hurunui	Scenario7	94.98	1.5	-3.15	-180.87	1,057.43	3.16	0.06	1.65	6.75	15.11
Kaipara	Scenario1	34.92	0.15	-1.93	-0.47	460.54	2.46	0.02	0.93	2.51	11.05
Kaipara	Scenario2	10.4	0.16	0.73	-4.27	159.91	0.71	0.02	1	2.59	7.19
Kaipara	Scenario3	-11.78	0.39	0.83	-3.77	167.58	1.2	0.04	1.03	2.62	10.28
Kaipara	Scenario4	-1.14	0.55	1.67	-7.7	232.54	1.29	0.03	1.05	3.06	10.03
Kaipara	Scenario5	46.3	0.31	-0.37	-2.14	525.37	2.7	0.03	1.12	3.11	13.42
Kaipara	Scenario6	23.8	0.52	-0.69	-3.11	532.84	2.68	0.04	1.21	3.04	12.2
Kaipara	Scenario7	34.69	0.68	0.49	-5.29	600.21	2.76	0.03	1.1	2.93	12.9
Ruapehu	Scenario1	114.76	0.88	-3.48	1.56	747.17	4.9	0.04	0.92	4.1	31.8
Ruapehu	Scenario2	12.65	0.19	1.63	-3.19	88.14	1.01	0.02	0.81	2.47	9.88
Ruapehu	Scenario3	-24.7	0.77	1.83	6.41	189.23	1.65	0.04	1.2	3.82	14.57
Ruapehu	Scenario4	-11.68	0.94	3.13	3.58	263.35	1.47	0.04	0.93	3.99	10.93
Ruapehu	Scenario5	127.48	1.04	-2.08	-4.4	825.29	5.37	0.05	1.23	5.04	34.81
Ruapehu	Scenario6	89.32	1.54	-1.99	2.16	923.54	4.77	0.05	1.22	5.17	33.91
Ruapehu	Scenario7	89.69	1.54	-1.53	2.38	919.28	5.32	0.05	1.24	5.91	30.87

Table C. Summary of the Z-score

Landscape	Scenario	Mean of Total C change	Mean of Mean Biodiversity change	Mean of Small patches proportion change	Mean NND change	Mean habitat amount change
Kaipara	Scenario 1	0.72	-1.23	-1.71	1.45	0.41
	Scenario 2	-0.43	-1.16	0.53	-0.19	-1.18
	Scenario 3	-1.48	-0.01	0.61	0.02	-1.14
	Scenario 4	-0.98	0.77	1.31	-1.68	-0.79
	Scenario 5	1.26	-0.41	-0.4	0.73	0.75
	Scenario 6	0.20	0.63	-0.67	0.31	0.79
	Scenario 7	0.71	1.40	0.32	-0.64	1.15

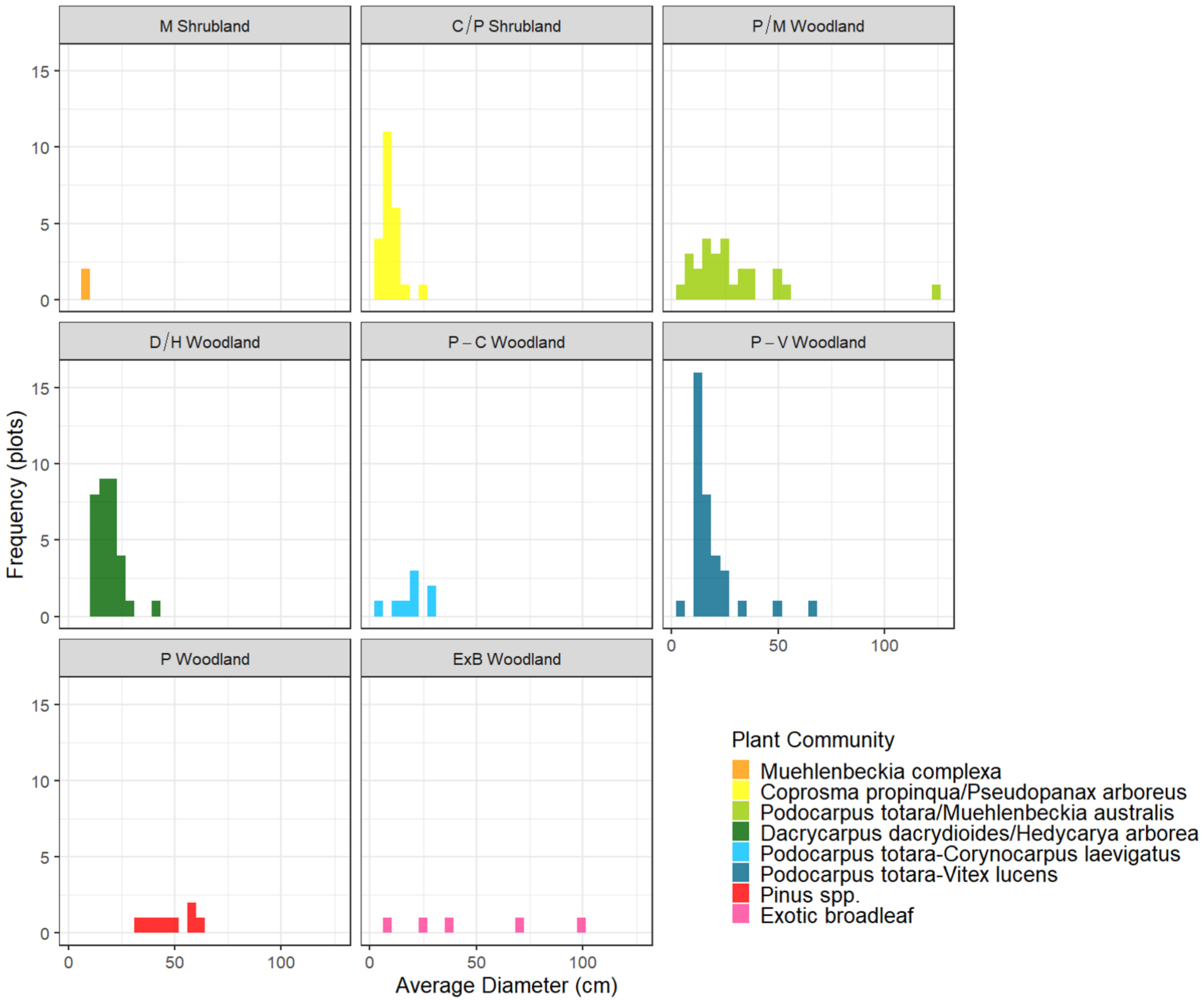
Ruapehu	Scenario 1	0.92	-0.23	-1.25	0.09	0.49
	Scenario 2	-0.7	-1.71	0.79	-1.16	-1.29
	Scenario 3	-1.29	-0.45	0.87	1.37	-1.02
	Scenario 4	-1.09	-0.10	1.39	0.63	-0.82
	Scenario 5	1.12	0.12	-0.69	-1.48	0.71
	Scenario 6	0.52	1.18	-0.65	0.25	0.97
	Scenario 7	0.52	1.19	-0.47	0.31	0.96
Hurunui	Scenario 1	0.50	-0.92	-1.57	1.06	0.52
	Scenario 2	-0.61	-0.39	1.09	-0.77	-0.97
	Scenario 3	-1.51	-1.35	0.52	1.44	-1.29
	Scenario 4	-0.82	0.53	0.85	-0.78	-0.87
	Scenario 5	1.19	0.72	0.25	-0.8	0.94
	Scenario 6	0.28	-0.13	-1.14	0.61	0.62
	Scenario 7	0.97	1.54	0.01	-0.77	1.05

Appendix B *Supplementary Figures*

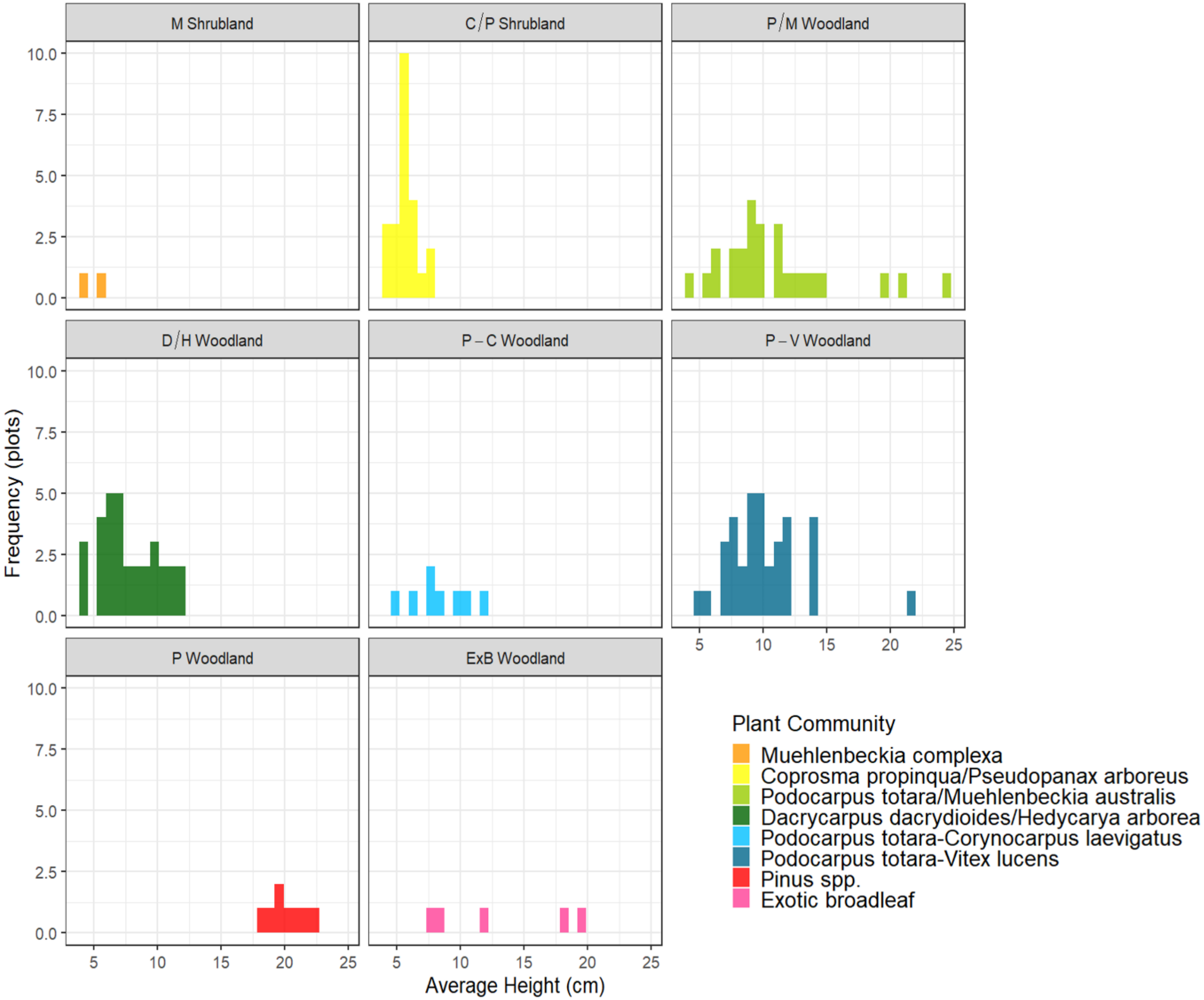


Appendix Figure 1 Cumulative histogram of Diameter at breast height (DBH) (cm) and height (m) of eight plant community types: A) Distribution of average DBH of all plots in three sheep and beef farms, B) Distribution of average height of all plots in three sheep and beef farms

A

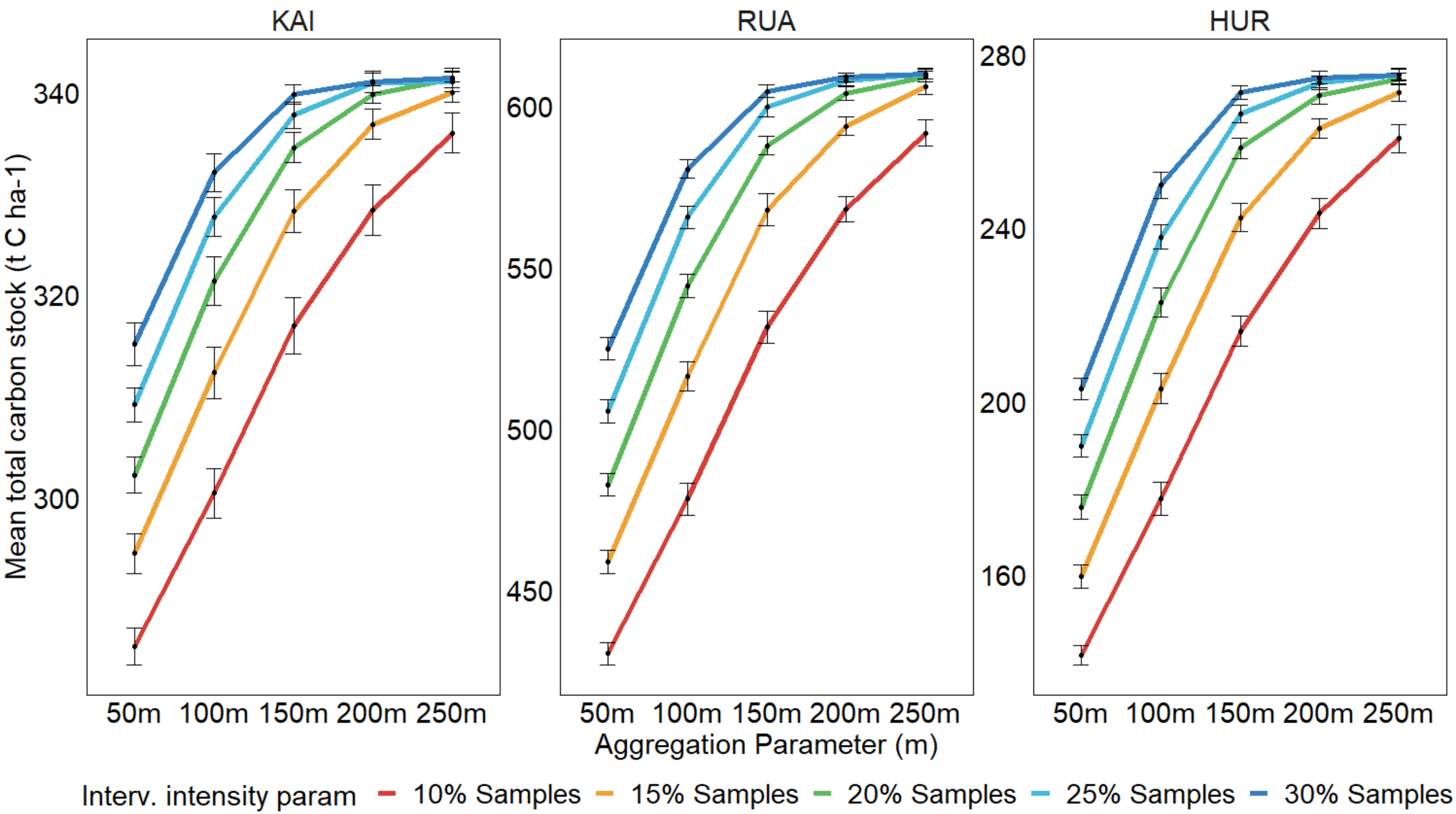

M Shrubland
C/P Shrubland
P/M Woodland
D/H Woodland
P – C Woodland
P – V Woodland
P Woodland
ExB Woodland
Frequency (plots)
Average Diameter (cm)
0
5
10
15
0
50
100
Plant Community
Muehlenbeckia complexa
Coprosma propinqua/Pseudopanax arboreus
Podocarpus totara/Muehlenbeckia australis
Dacrycarpus dacrydioides/Hedycarya arborea
Podocarpus totara-Corynocarpus laevigatus
Podocarpus totara-Vitex lucens
Pinus spp.
Exotic broadleaf

B

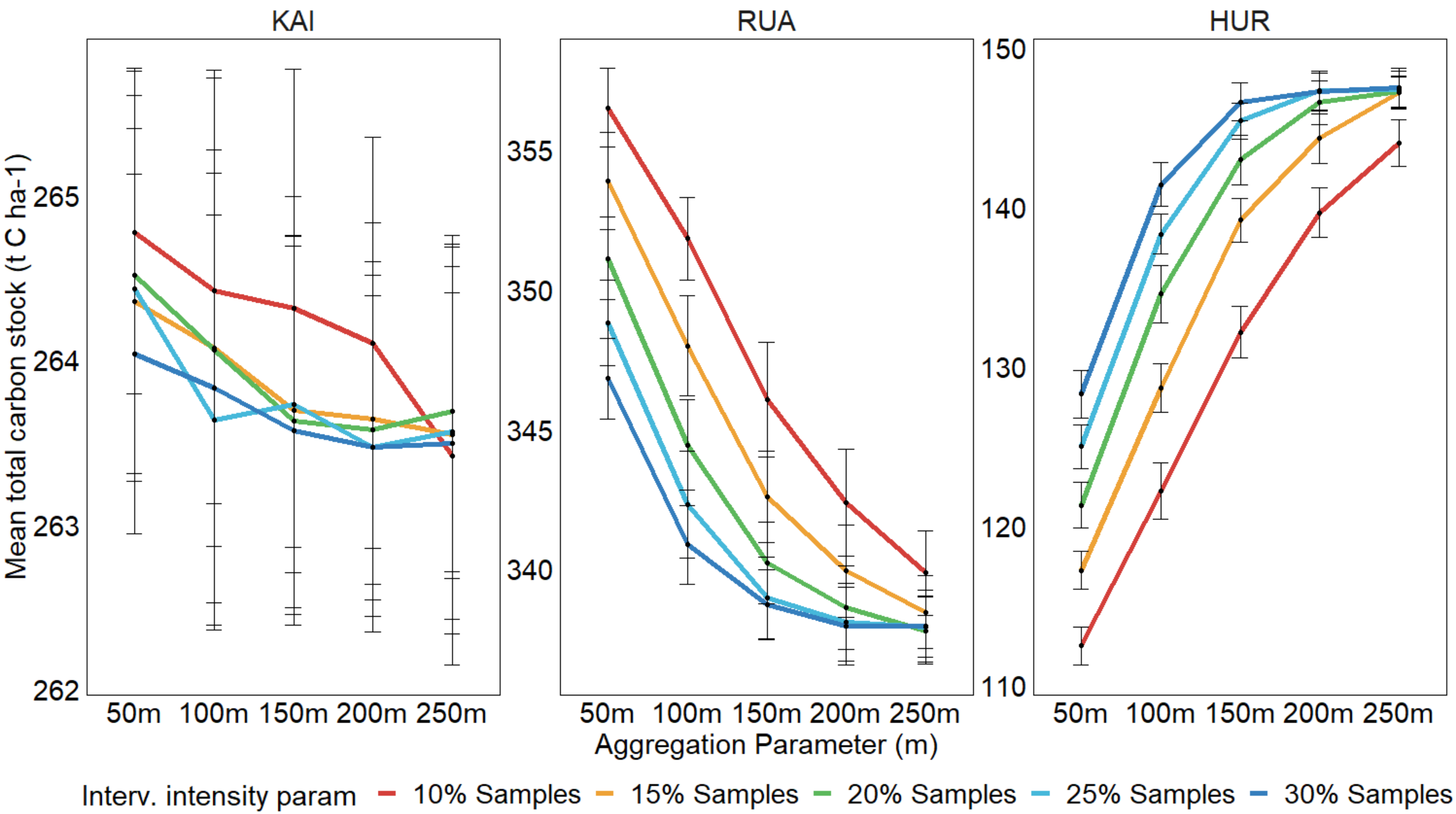

M Shrubland
C/P Shrubland
P/M Woodland
D/H Woodland
P – C Woodland
P – V Woodland
P Woodland
ExB Woodland
Frequency (plots)
Average Height (cm)
10.0
7.5
5.0
2.5
0.0
5
10
15
20
25
Plant Community
Muehlenbeckia complexa
Coprosma propinqua/Pseudopanax arboreus
Podocarpus totara/Muehlenbeckia australis
Dacrycarpus dacrydioides/Hedycarya arborea
Podocarpus totara-Corynocarpus laevigatus
Podocarpus totara-Vitex lucens
Pinus spp.
Exotic broadleaf

Appendix Figure 2 Frequency histograms of frequency of Mean Diameter at breast height (DBH) (cm) and Mean height (m) of all plots of eight plant community types: A) Histogram of frequency average DBH of all plots in three sheep and beef farms by plant community types, B) Histogram of frequency average height of all plots in three sheep and beef farms by plant community types

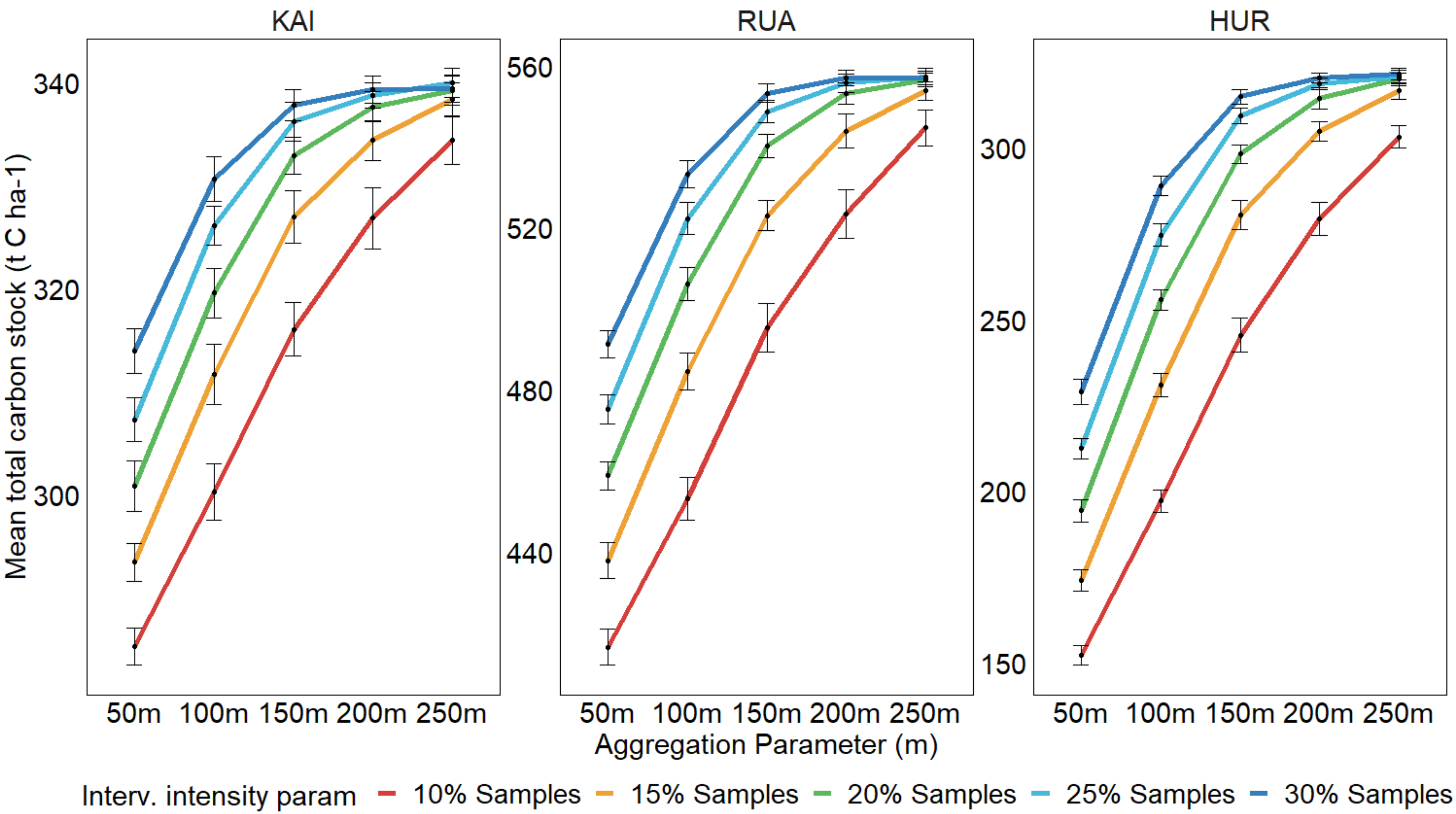
A1. Alt 1 Change of Carbon Stock per hectare

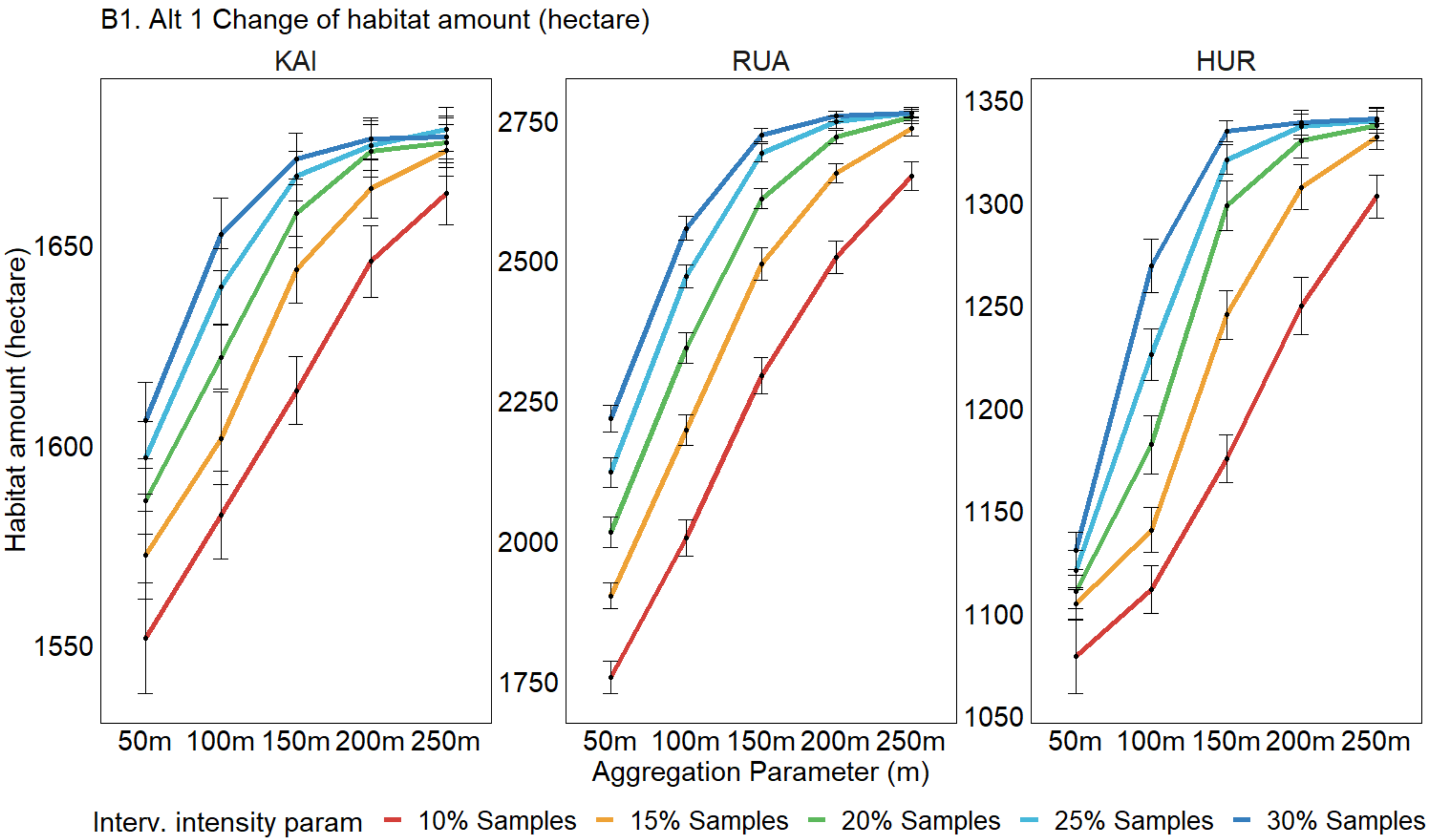


A2. Alt2 Change of Carbon Stock per hectare



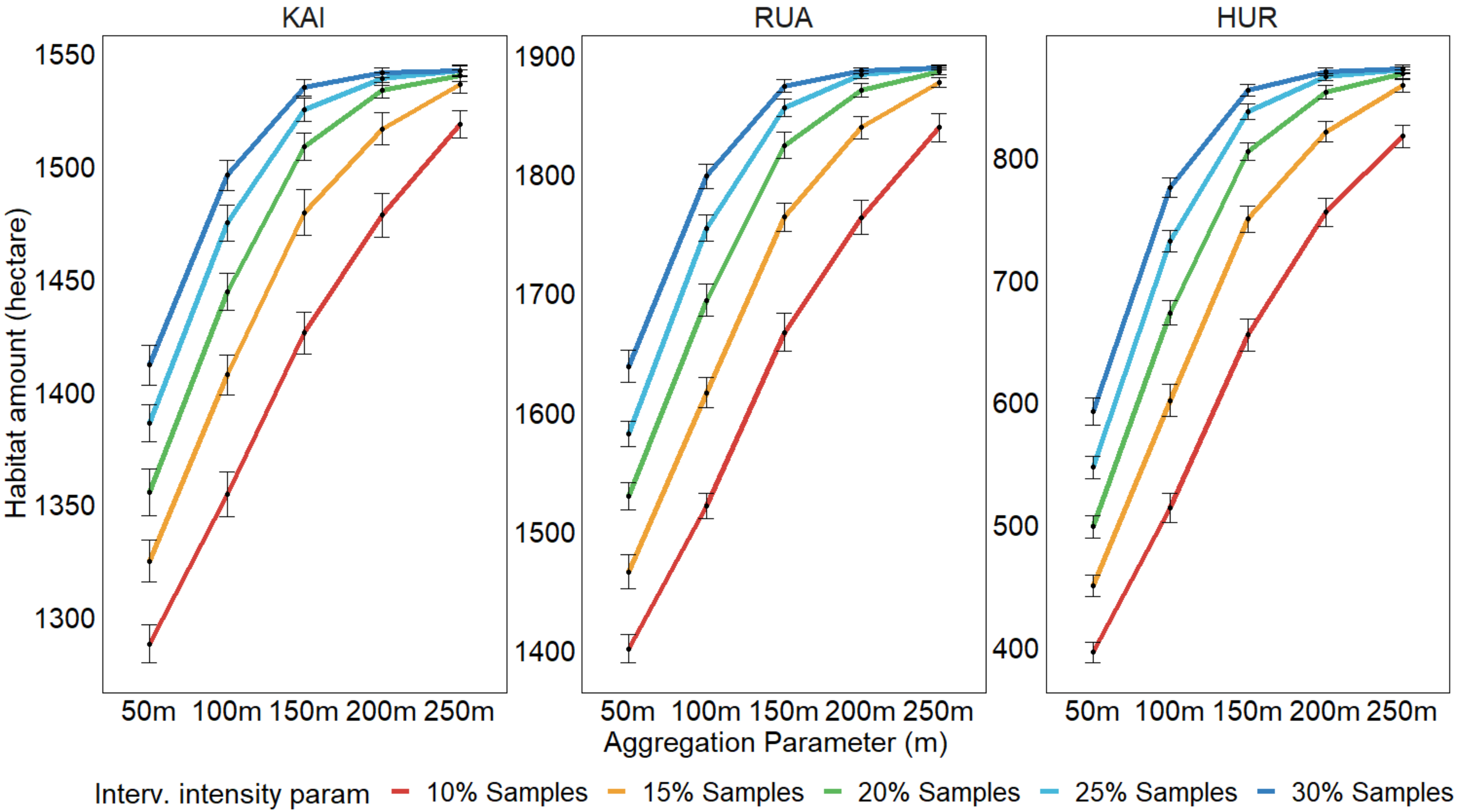
A3. Alt3 Change of Carbon Stock per hectare

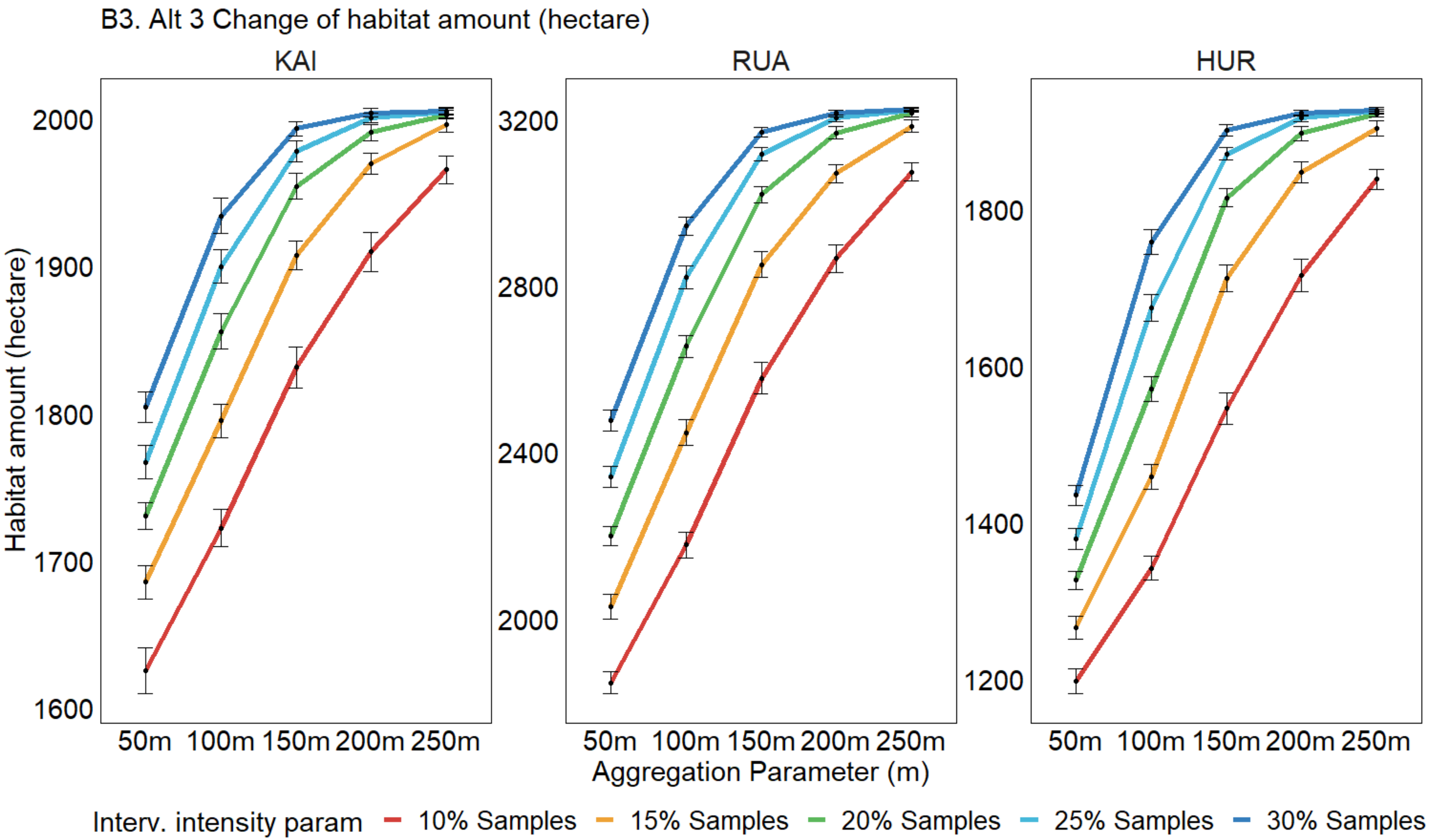


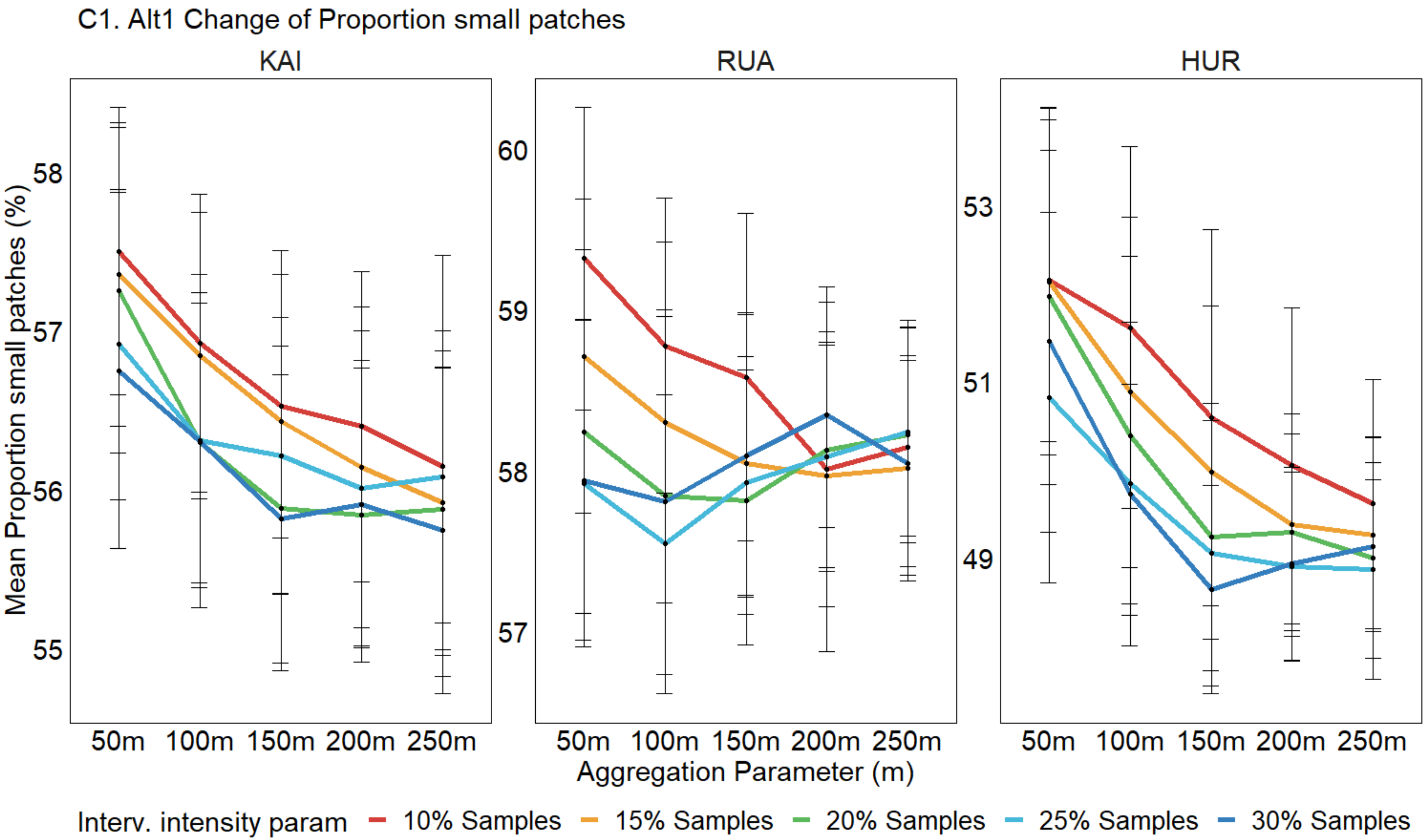


B1. Alt 1 Change of habitat amount (hectare)
KAI
RUA
HUR
Habitat amount (hectare)
1550
1600
1650
1750
2000
2250
2500
2750
1050
1100
1150
1200
1250
1300
1350
50m
100m
150m
200m
250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples

B2. Alt 2 Change of habitat amount (hectare)

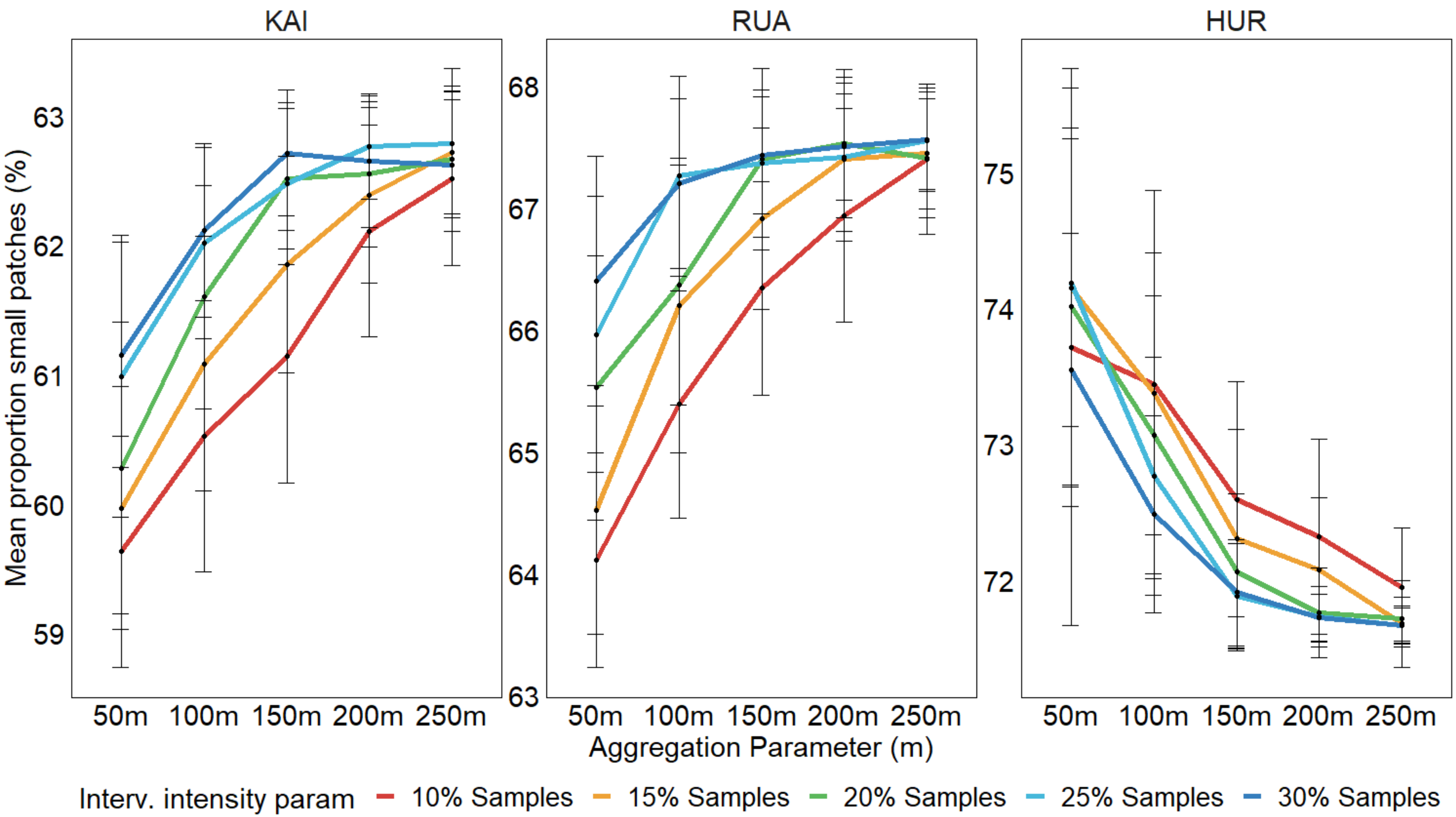


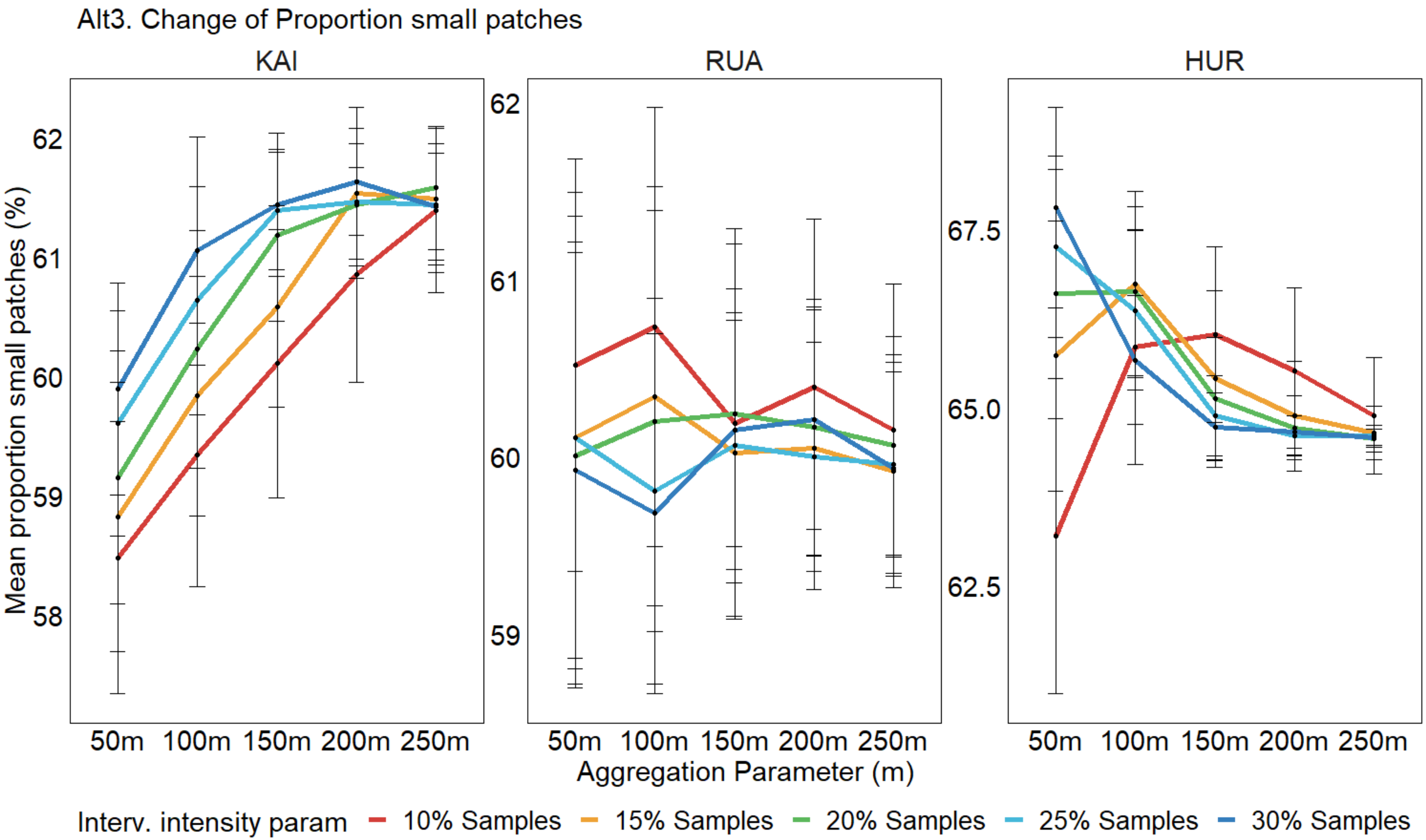

B3. Alt 3 Change of habitat amount (hectare)
KAI
RUA
HUR
Habitat amount (hectare)
2000
1900
1800
1700
1600
3200
2800
2400
2000
1800
1600
1400
1200
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples



C1. Alt1 Change of Proportion small patches
KAI
RUA
HUR
Mean Proportion small patches (%)
58
57
56
55
60
59
58
57
53
51
49
50m 100m 150m 200m 250m
50m 100m 150m 200m 250m
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples

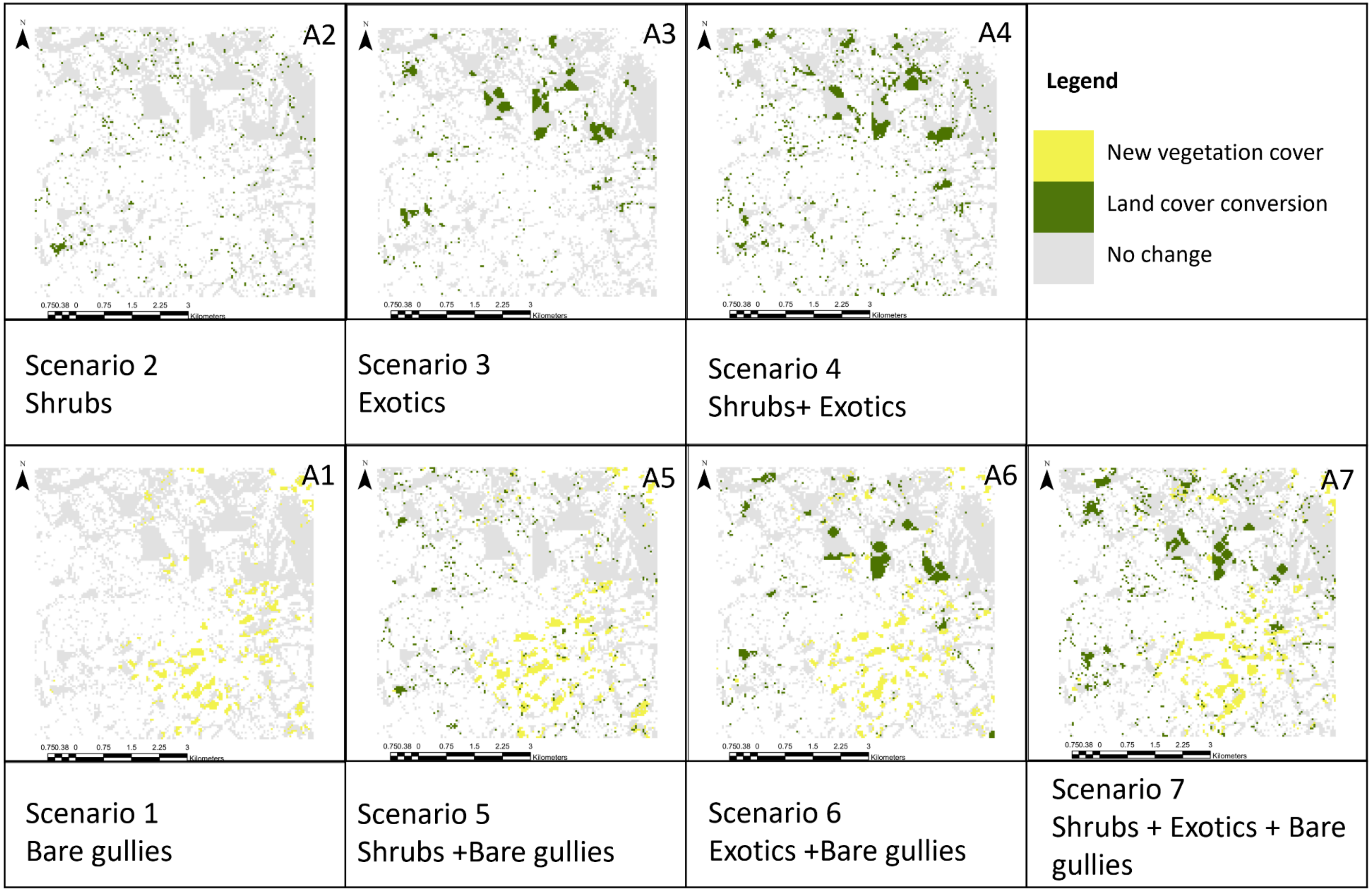
C2.Alt2 Change of Proportion small patches



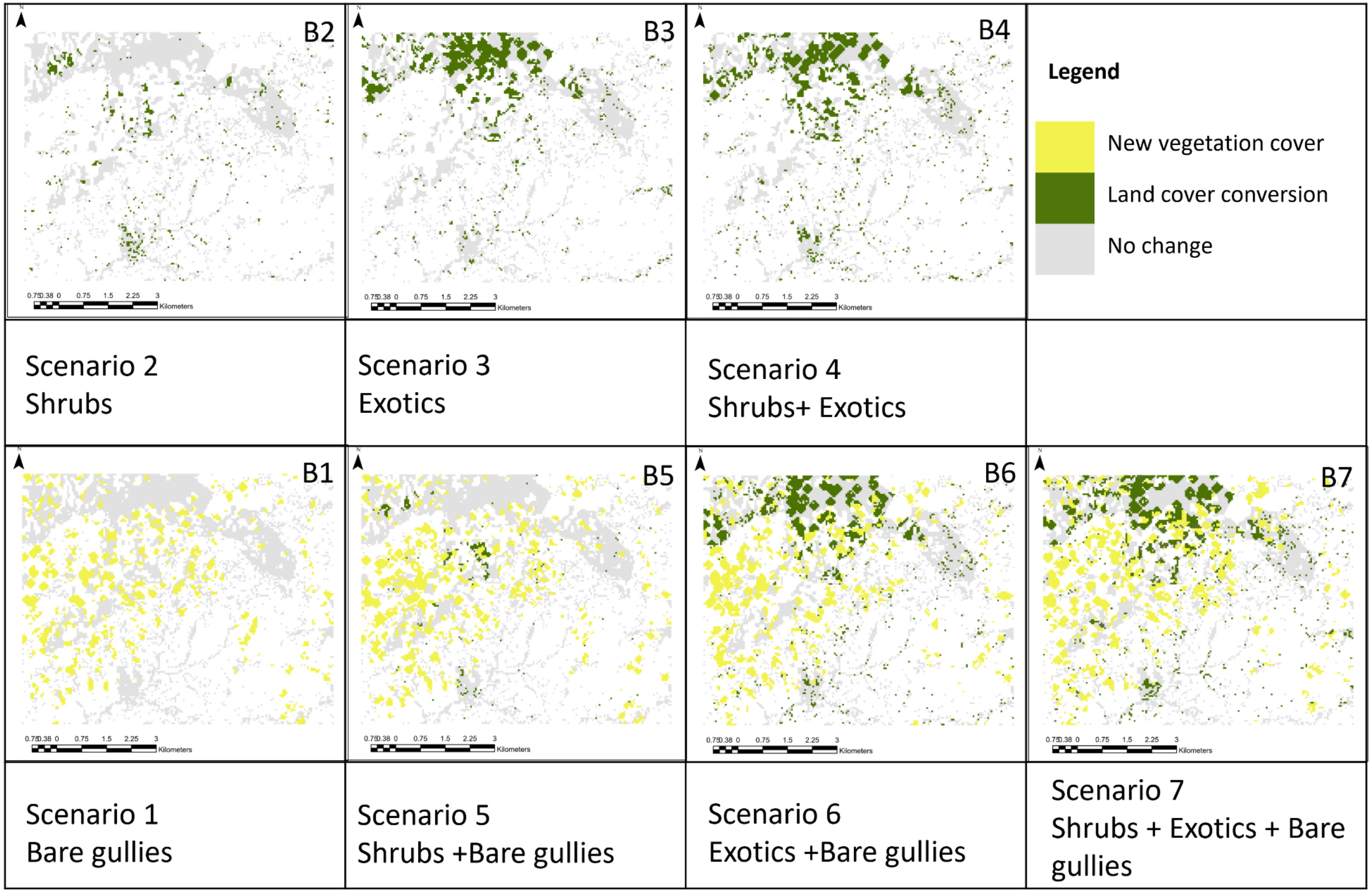

Alt3. Change of Proportion small patches
KAI
RUA
HUR
Mean proportion small patches (%)
58
59
60
61
62
62
61
60
59
67.5
65.0
62.5
50m 100m 150m 200m 250m
50m 100m 150m 200m 250m
50m 100m 150m 200m 250m
Aggregation Parameter (m)
Interv. intensity param
10% Samples
15% Samples
20% Samples
25% Samples
30% Samples

Appendix Figure 3. Changes of outcomes A) Carbon Stock ($tCha^{-1}$), B) Habitat Amount (ha), and C) Proportion of small patches (%) following the change of intervention intensity parameter (10, 15, 20, 25, 30 %) and aggregation parameter (50,, 100, 150, 200, 250, 300 m) of three model rulesets : 1) Alt 1 (revegetating continuous-large size bare gully polygons), 2) Alt 2 (restoring disperse small and large size woody vegetation polygons), and 3) Alt 3 (a combination of Alt 1 and Alt2)

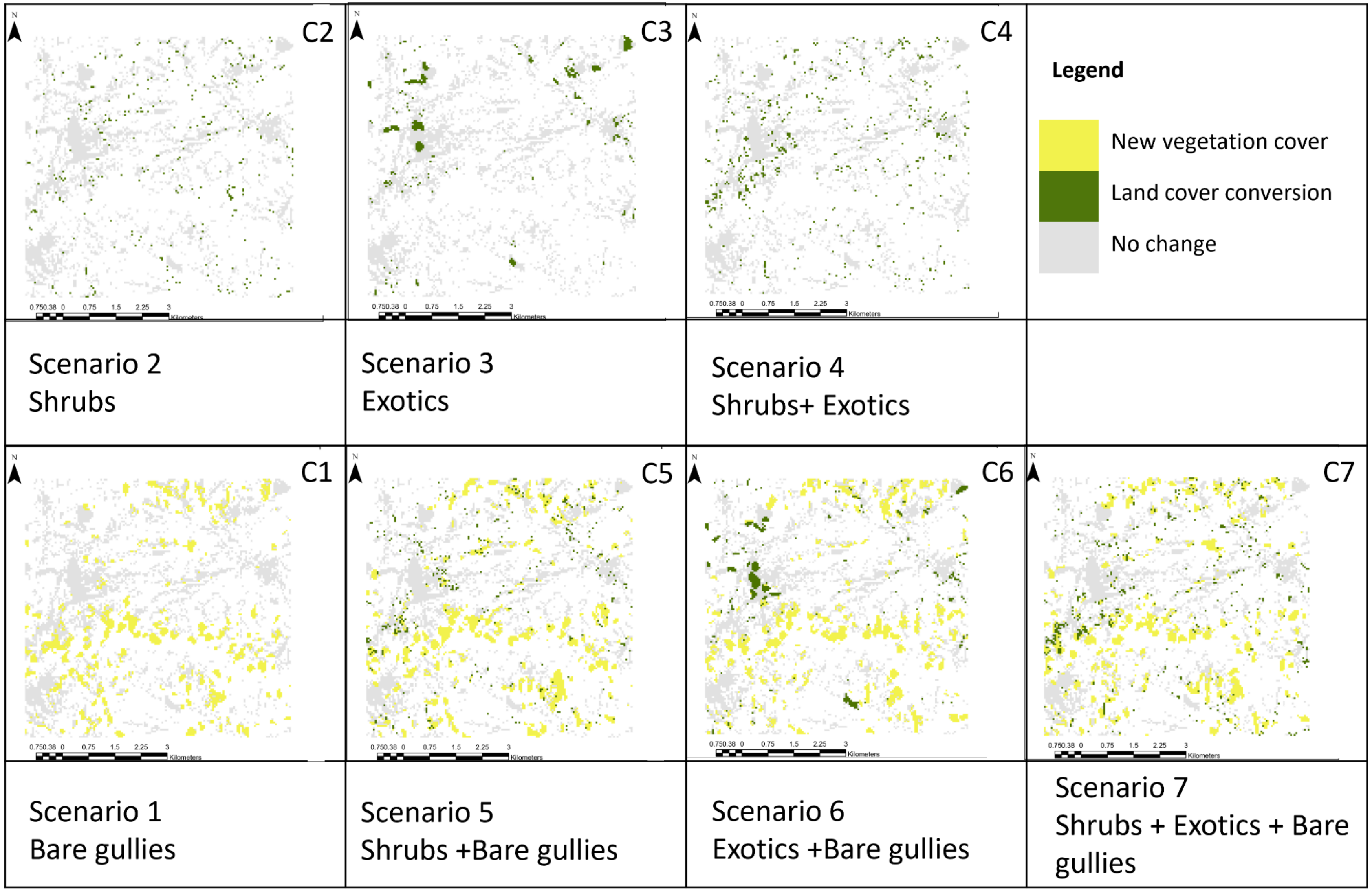
A KAIPARA Vegetation cover



B RUAPEHU Vegetation cover

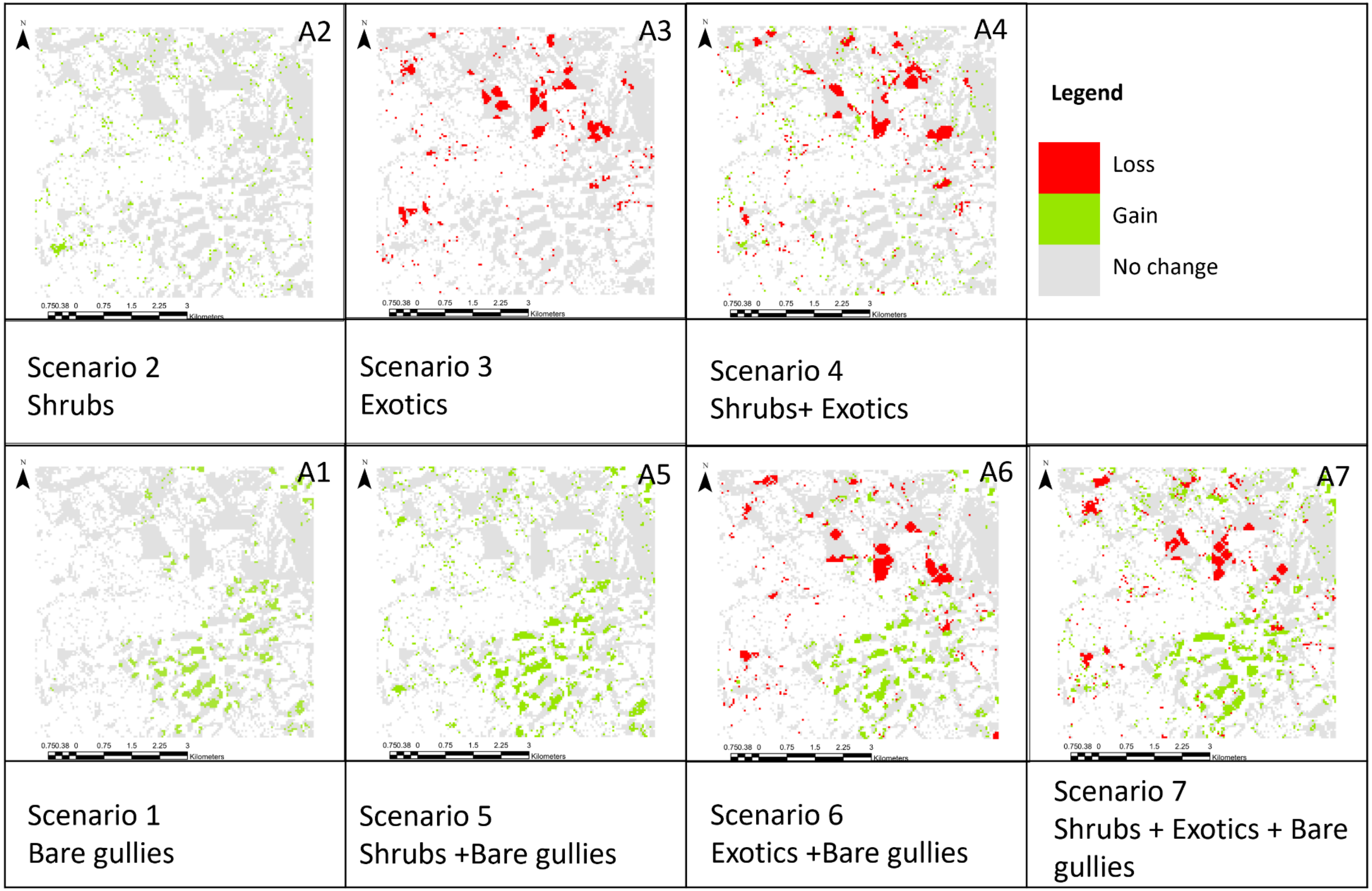


C HURUNUI Vegetation cover

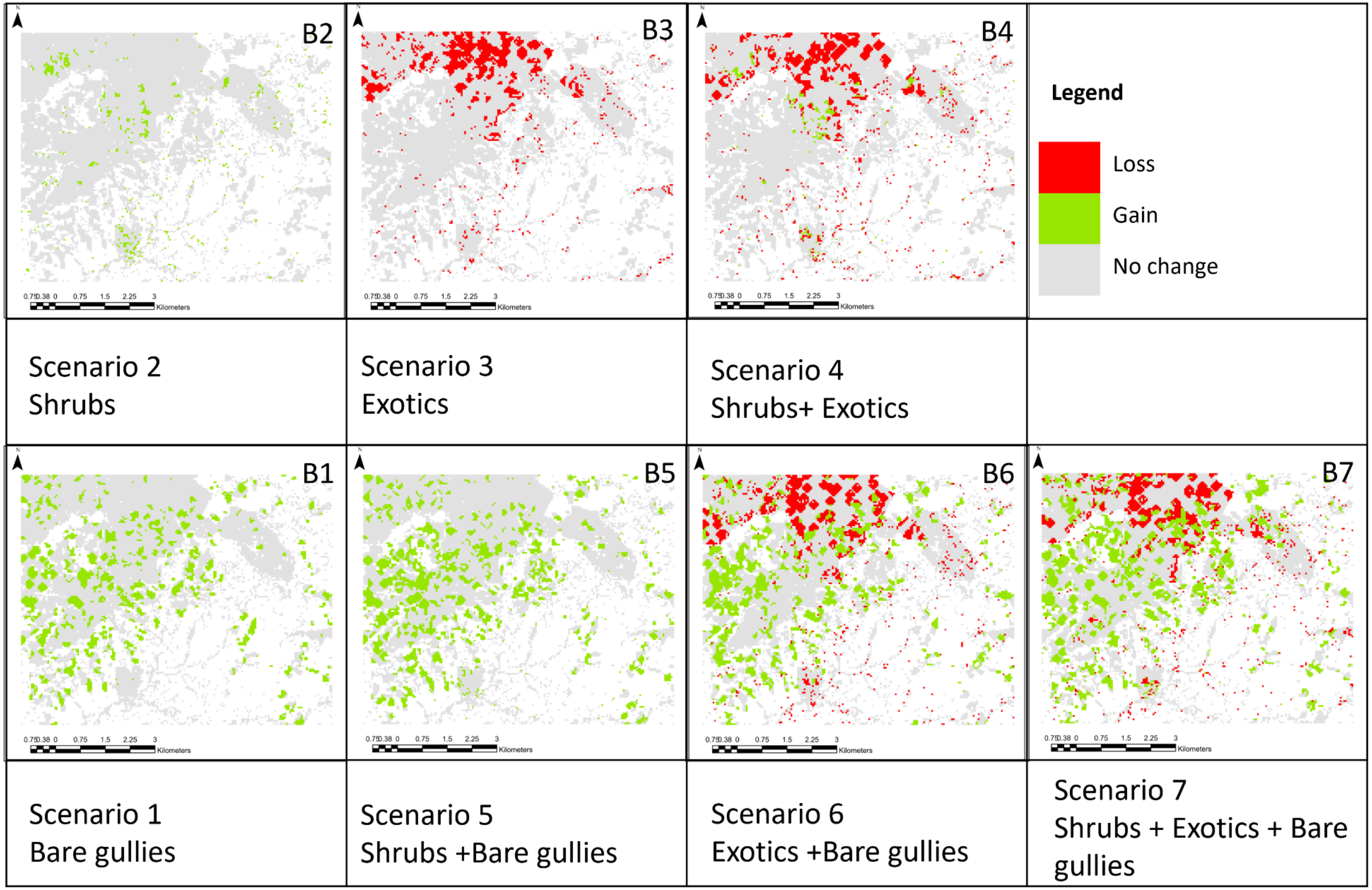


Appendix Figure 4 Three group of maps illustrating the distribution of land cover change and additional vegetation cover in three different landscapes: Kaipara (A), Ruapehu (B), and Hurunui (C); on the seven scenario models: 1) Scenario 1 (revegetating bare gullies), 2) Scenario 2 (restoring shrubs), 3) Scenario 3 (restoring exotic-dominated woodlots), 4) Scenario 4 (restoring shrubs and restoring exotic-dominated woodlots)), Scenario 5 (restoring shrubs and revegetating bare gullies), 6) Scenario 6 (revegetating bare gullies and restoring exotic-dominated woodlots), and 7) Scenario 7 (a combination of restoring shrubs and exotic dominated woodlots and revegetating bare gullies).

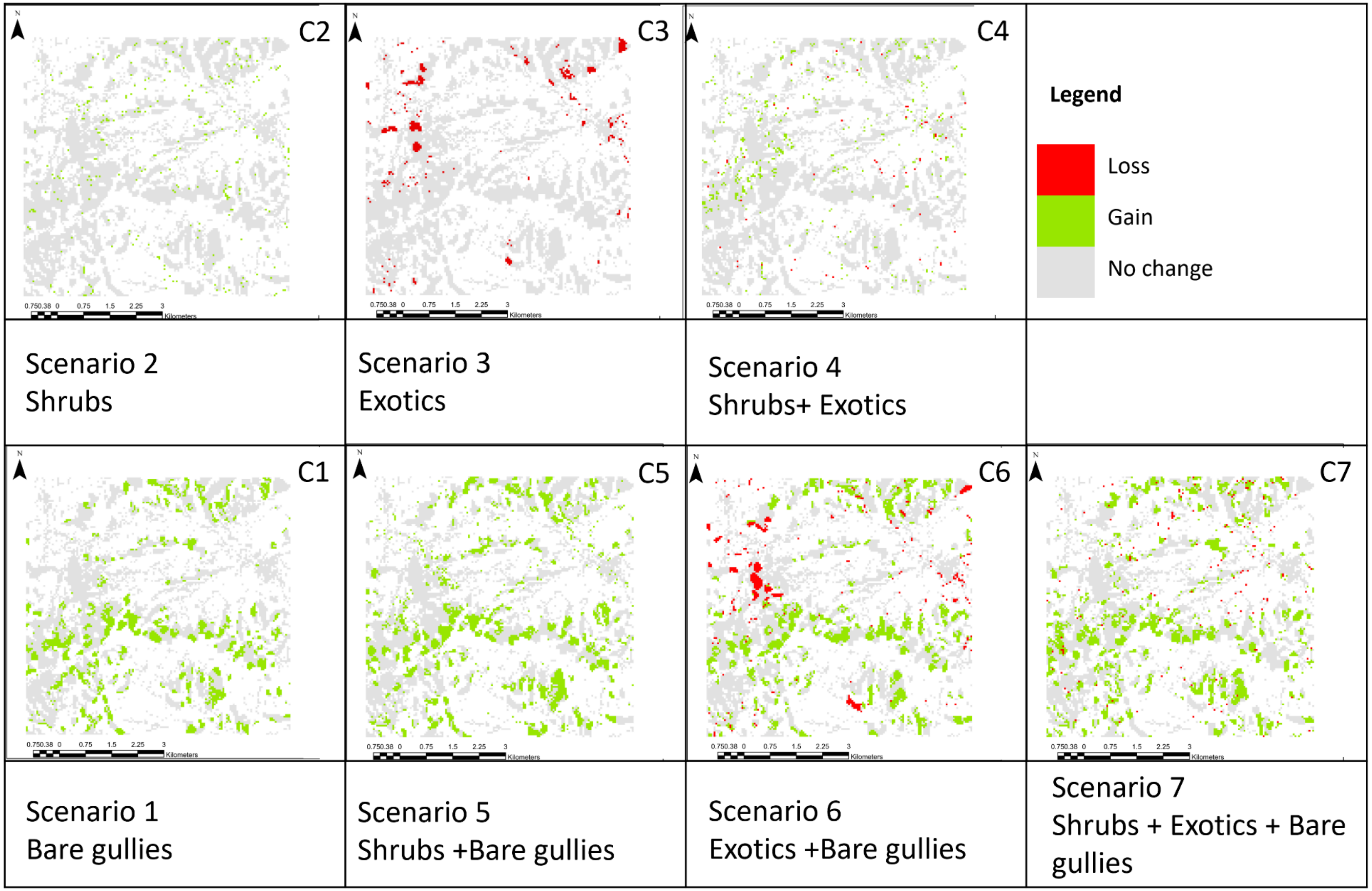
A KAIPARA Carbon stock



B RUAPEHU Carbon stock

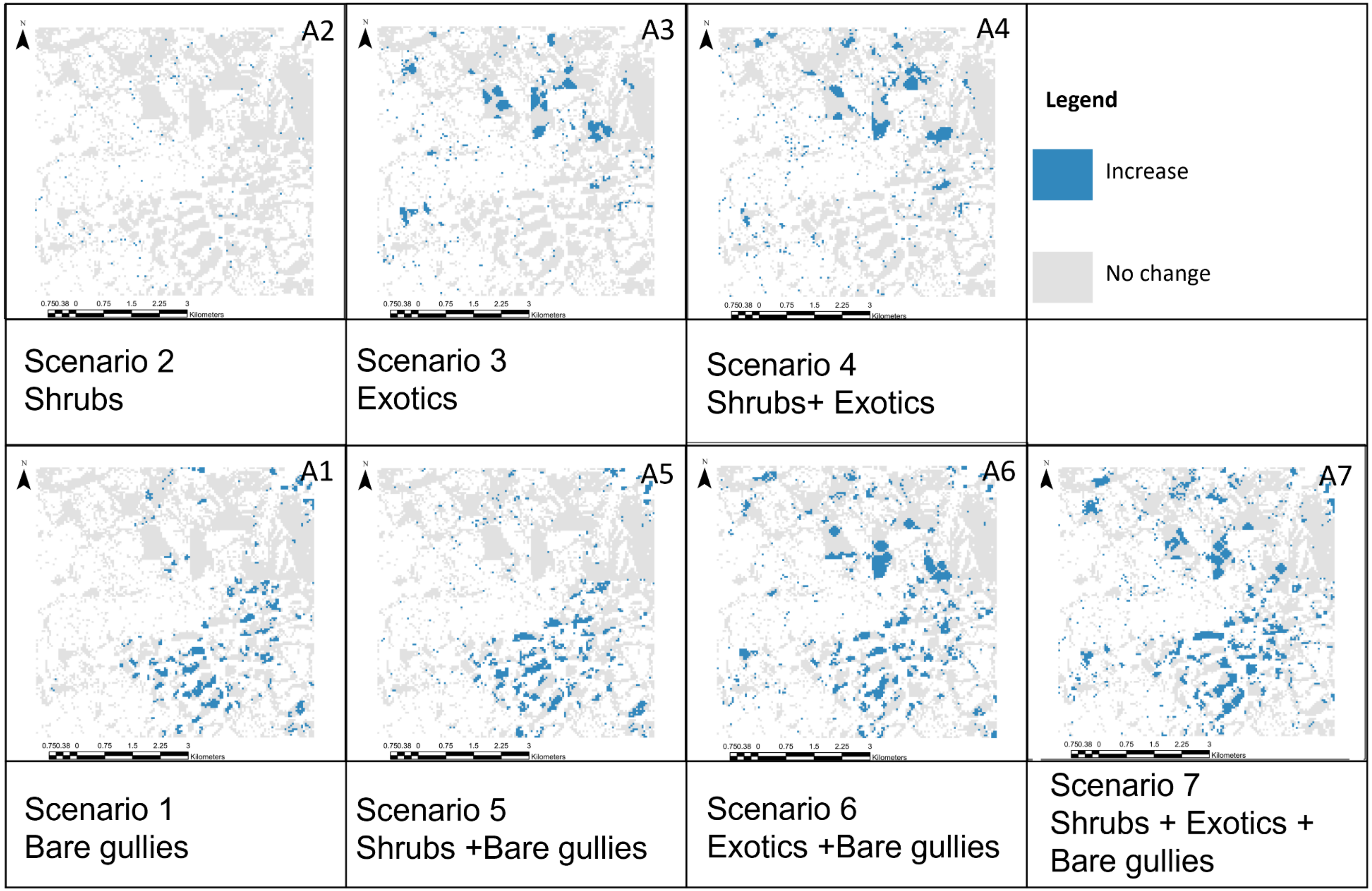


C HURUNUI Carbon stock

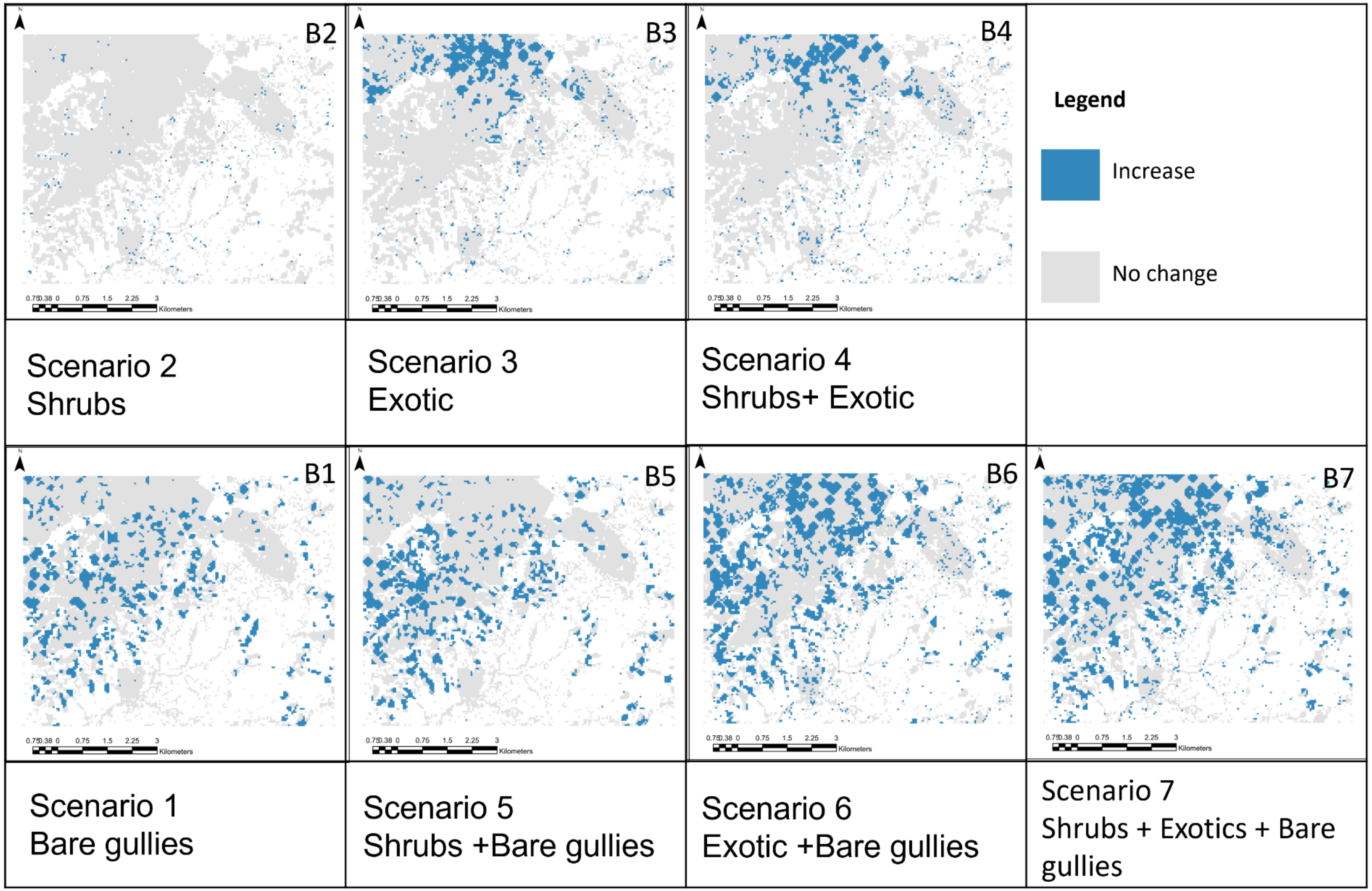


Appendix Figure 5 Three groups of maps that show how the carbon stock changes in three different landscapes: Kaipara (A), Ruapehu (B), and Hurunui (C); on the seven scenario models: 1) Scenario 1 (revegetating bare gullies), 2) Scenario 2 (restoring shrubs), 3) Scenario 3 (restoring exotic-dominated woodlots), 4) Scenario 4 (restoring shrubs and restoring exotic-dominated woodlots)), Scenario 5 (restoring shrubs and revegetating bare gullies), 6) Scenario 6 (revegetating bare gullies and restoring exotic-dominated woodlots), and 7) Scenario 7 (a combination of restoring shrubs and exotic dominated woodlots and revegetating bare gullies). The changes in carbon stock relative to the baseline are shown as gains (above the baseline carbon stock) that were represented by green color or losses (below the baseline) that were represented by red color. The areas that did not experience any change are colored in grey. These maps provide an illustration of the model that was run using a 10 % intervention intensity parameter and a 100 m aggregation parameter.

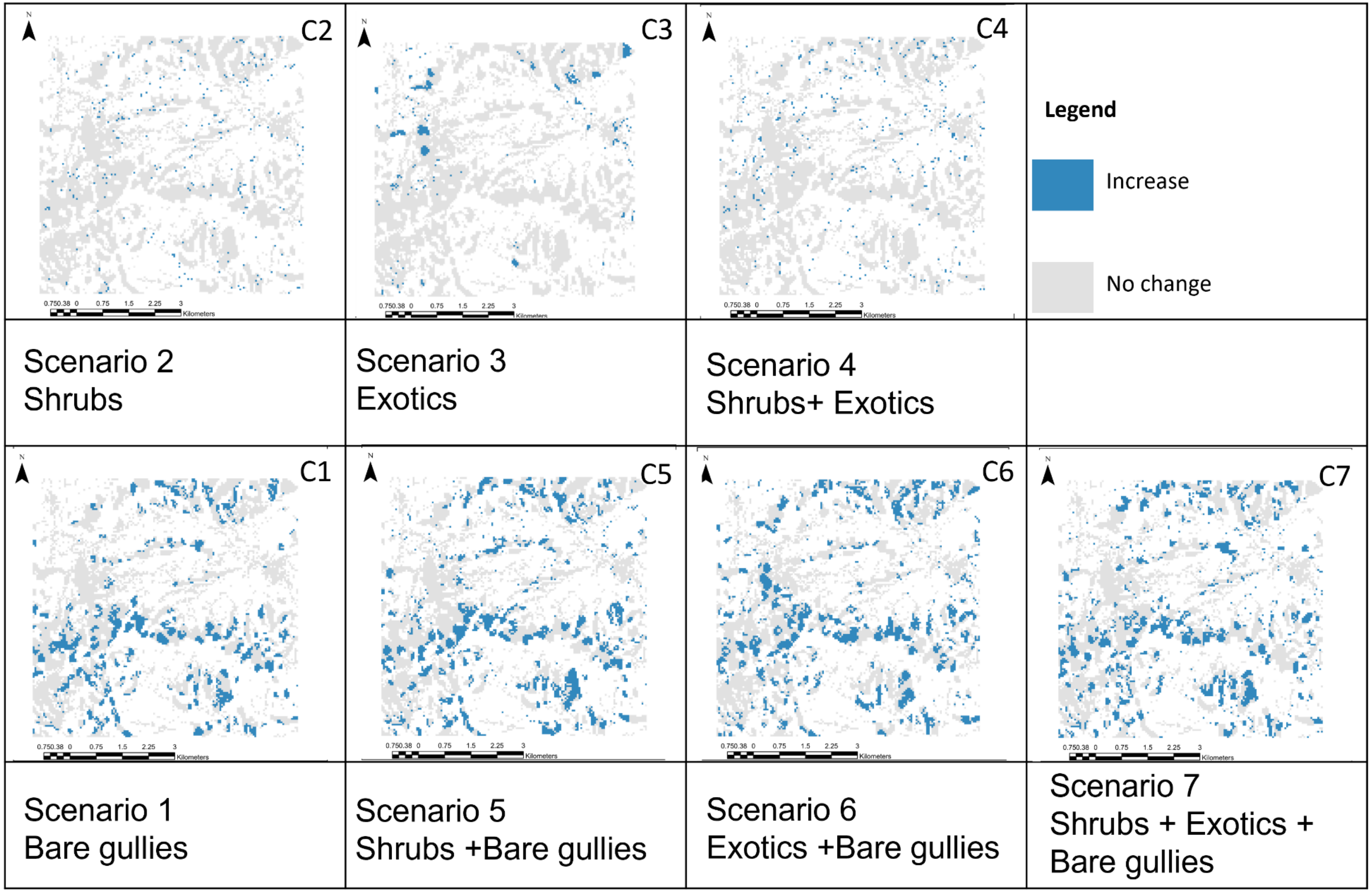
A KAIPARA Biodiversity value



B RUAPEHU Biodiversity value



C HURUNUI Biodiversity value



Appendix Figure 6 Three groups of maps that show how the change of mean biodiversity potential of each grid cell in three different landscapes: Kaipara (A), Ruapehu (B), and Hurunui (C); on the seven scenario models: 1) Scenario 1 (revegetating bare gullies), 2) Scenario 2 (restoring shrubs), 3) Scenario 3 (restoring exotic-dominated woodlots), 4) Scenario 4 (restoring shrubs and restoring exotic-dominated woodlots)), Scenario 5 (restoring shrubs and revegetating bare gullies), 6) Scenario 6 (revegetating bare gullies and restoring exotic-dominated woodlots), and 7) Scenario 7 (a combination of restoring shrubs and exotic dominated woodlots and revegetating bare gullies). Changes in the biodiversity potential of intervention cells relative to their initial biodiversity potential are depicted in blue. The areas that did not experience any change are colored in grey. These maps provide an illustration of the model that was run using a 10 % intervention intensity parameter and a 100 m aggregation parameter.

Appendix C

Corrigendum: Calculation and Clarification of CO2 Sequestration Figures

The calculation and sources behind the figure of 384 million tonnes of CO_2 expected to be sequestered by new trees on the 1BT scheme; that was estimated from pine plantation sequestering approximately 600 tons CO_2 per hectare per year, and native forest sequestering up to 1000 CO_2 per hectare per year. This information was based on the Cabinet Paper that was cited on the paper:
Te Uru Rākau. (2018b). One billion trees fund report on policy and design recommendations. https://www.mpi.govt.nz/dmsdocument/32908-3-Appendix1-Report-on-Policy-and-Design-Recommendations-OIA

The carbon sequestration rates mentioned in the provided information are as follows:

1. Native Forest/ Regenerating Indigenous Species:
   Average sequestration: Approximately 160 tonnes of carbon dioxide per hectare over 20 years. Long-term sequestration: Indigenous species can sequester carbon up to 1000 tonnes per hectare if natural conditions and management facilitate succession to tall forest species.
   Source: Estimated carbon stock of pre-1990 natural forest, specifically tall forest, as reported in the national greenhouse gas inventory.
2. Pine Plantation/ Plantation Forests:
   Short-term sequestration: Plantation forests, both exotic and indigenous species, have faster rates reaching approximately 600 tonnes per hectare over 20 years.
   Comparison: This rate is roughly equivalent to the long-term average carbon stock of rotational radiata pine.
   Source: Sequestration over 20 years for both exotic and indigenous species is derived from the LUCAS look-up tables used for our national inventory and target reporting.