The identification of unproductive hill country for planting carbon farmed vegetation and the economics of doing so

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i. Abstract

The Emissions Trading Scheme (ETS) is New Zealand's main legislative tool for ensuring the domestic and international greenhouse gas reduction commitments are met. Tradeable 'Carbon Credits' (NZUs) are awarded to forestry owners for the carbon storage capabilities of their forests. Afforested areas under 100 hectares in size are subject to carbon storage estimation through the use of default 'carbon look-up tables'. Recent mass land use change on New Zealand hill country from pastoral production to blanket planted exotic trees is divergent to government policy, which promotes the integration of native trees into the landscape. A pilot study tests whether 'carbon look-up tables' for Indigenous Forest are aligned with actual carbon sequestration of *Leptospermum Scoparium* (Manuka). This work aims to determine whether mass land use change is a consequence of the ETS in its current form. This is achieved through the economic modelling of exotic *Pinus radiata* (Pines) and Manuka on different slope classes of a typical New Zealand hill country farm. Allometric equations used on Manuka stands identified by random grid sampling showed that carbon sequestration of regenerating bush may be underestimated by over 82%. Economic modelling highlights the large earnings associated with blanket planted fast-growing Pines (peaking at $2,915/ha in Year 7) compared to the existing pastoral system ($242/ha/yr). Net Present Value (NPV) and sensitivity analysis results parallel findings at different inflation rates and market prices. Pines outperform natural regeneration of Manuka in every model from an economic perspective, although preferential income revenues like honey are not accounted for. Targeted plantings are not cost effective due to (a) increased fencing costs and (b) exclusion from the ETS. Recommendations made to align the impacts of the ETS with government policy include: (a) collecting carbon sequestration data for specific native species in different environments; (b) removing spatial constraints associated with the ETS' definition of a forest; and (c) assigning a monetary incentive to promote certain forest values; such as biodiversity. Silvopastoral systems where pastoral and forest species spatially coexist are a potential solution. Areas requiring research in a New Zealand hill country context are identified using a recently improved framework.

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1.1 Introduction

As a result of industrialisation and anthropogenic activity, increased atmospheric greenhouse gas levels including carbon dioxide are impacting the Earth's climate (IPCC, 2007). In a bid to combat climate change, New Zealand has adopted international legislation reinforced by stricter internal commitments. The Emissions Trading Scheme (ETS) was created through national legislation and is currently New Zealand's main tool for reducing net emissions (Evison, 2017).

New Zealand's ETS is unique in the sense that forestry is included within the accounting system (Adams & Turner, 2012; Kerr et al., 2012; West et al., 2020), a feature that European counterparts do not share (Diaz-Rainey & Tulloch, 2018). Effectively monetizing carbon emissions/sequestration, the ETS has created a 'carbon market' with forestry increasingly becoming a profitable enterprise, even on the steepest of land. Faster growing exotic species (primarily Pinus radiata) are preferred over native species due to increased carbon sequestration rates (Dodd et al., 2021; Leining et al., 2020). This has resulted in unparalleled land use change, with the returns of 'carbon farming' outweighing those of existing hill country sheep and beef systems.

In an effort to future proof their businesses in the face of strengthening freshwater quality legislation, hill country farmers have been planting woody vegetation to prevent erosion on erodible slopes (Fernandez & Daigneault, 2017; Mackay-Smith et al., 2021). Deeper roots increase the cohesion of a soil body, reducing the quantity of sediment and associated Phosphorus (P) lost in runoff (Marden et al., 2020; Spiekermann et al., 2021). Largely utilising natives for aesthetic and biodiversity purposes, this has resulted in targeted plantings coexisting with existing sheep and beef operations (Tozer et al., 2021).

The purpose of this literature review is to investigate the nuanced differences between government policy and current practice. The effect that climate change legislation has upon the landscape of New Zealand hill country is investigated. Whether the use of forestry as a tool for controlling pollutants from agricultural runoff/mass movements, has had a compounding effect is analysed. The degree of common ground between these widespread changes and resource management legislation is critiqued. During the process, areas which require further research are identified. Findings and research are kept within the scope of New Zealand's hill country.

1.2 Discussion

1.2.1 Climate Change and New Zealand's Commitments

Globally, atmospheric carbon dioxide concentrations have increased from 280ppm in preindustrial times to 379ppm in 2005 (IPCC, 2007). Attributed to the exponential increase in anthropogenic activity, increased greenhouse gas concentrations effectively trap infra-red radiation within Earth's atmosphere. The result is the gradual increase of Earth's surface temperature, with an average increase of 0.85 [0.65-1.06] °C measured from 1880 to 2012 (IPCC, 2014). Global impacts include increased occurrence of extreme climatic events (Hattermann et al., 2015; Lundquist et al., 2011), rising sea levels (Cazenave et al., 2014; Lundquist et al., 2011; Sommerkorn, 2008; Tebaldi et al., 2021), ocean acidification (Lundquist et al., 2011), loss of ecosystems (Hussein, 2009), food security (Asseng et al., 2019; Li et al., 2009) and the displacement of terrestrial organisms (Brown et al., 2017).

In a bid to mitigate the effects of climate change, New Zealand has signed international agreements, derived from meetings of the United Nations Framework Convention on Climate Change (UNFCCC). These agreements have the goal of stabilising atmospheric greenhouse gas concentrations at a level preventing the degradation of global ecosystems as a result of anthropogenic activity. To help ensure international obligations are met, national legislation has been passed, which includes emission reduction targets tougher than those of international documents.

1.2.1.1 United Nations Framework Convention on Climate Change

The UNFCCC, of which New Zealand is party too, is a forum where countries can collectively decide how to mitigate climate change. Adopted by New Zealand in 1992 (Bullock, 2012), it initiated processes to encourage the mass accounting of greenhouse gas emissions and the application of mitigations. However, it fails to quantify targets for emission reductions. Subsequent agreements have been made under the UNFCCC parent body which address the lack of legally binding emission targets. Most notable are the Kyoto Protocol and Paris Agreement. Only parties that ratify the individual contracts are bound by its contents.

New Zealand agreed to the Kyoto protocol in late 2002 (Adams & Turner, 2012; Bullock, 2012); which set mandatory reduction targets and reporting of emissions. It was agreed that emissions from the year 1990 would act as a baseline of which current emissions could be compared against. For the first reporting period (2008-2012) (Evison, 2017; Wang et al., 2022), New Zealand met its target of reducing emission levels below gross 1990 levels.

A second reporting period was agreed upon which ran from 2013-2020, commonly referred to as the Doha round (Wang et al., 2022). New Zealand signed in 2015, however chose to make its reduction commitment under the UNFCCC, instead of the Kyoto Protocol (Price & Duffin, 2013). This meant that the countries committed reduction target of 5% under 1990 levels was not legally binding; something it would have been under the Kyoto Protocol. As of April 2021, New Zealand was projected to have met these targets.

The Paris Agreement, with a commitment period of 2021 through to 2030, aims to limit global warming to less than 2°C (preferably <1.5°C) above pre-industrial levels (Fernandez & Daigneault, 2016; Fernandez & Daigneault, 2018; Leahy et al., 2020; Minasny et al., 2017). Unlike previous international agreements, it is signed by nearly all countries in the world, with reduction commitments from all major polluters. The participation of so many nations make it a landmark piece of international legislation, with all efforts going towards a common goal. A disparity in previous agreements is often cited as one reason why attempts were not successful at their intended scale (Grunewald & Martinez-Zarzoso, 2016; Holtsmark & Alfsen, 2004; Rosen, 2015). New Zealand's greenhouse gas reduction target under the Paris Agreement is 30% below 2005 levels; its first nationally determined contribution (NDC) (Fernandez & Daigneault, 2016).

1.2.1.2 National Policy

To meet the goals set by international law, New Zealand has implemented legislation to formally recognise and act upon the need to reduce its net greenhouse gas emissions. In order to reduce net emissions, gross emissions must decrease or sequestration of such gases in long term pools must increase. Legislation and government policy tackles both sides, with the result hoped to be a combination of efforts. The Climate Change Response Act has been a cornerstone piece of greenhouse gas legislation, detailing New Zealand's approach to dealing with meeting international greenhouse gas reduction targets. Recent amendments to the act have only solidified this status.

What is commonly referred to as the 'Zero Carbon Bill', is a 2019 amendment to the 'Climate Change Response Act 2002' (New Zealand Government, 2019). It puts a legal framework in place to help New Zealand meet its international commitments set out in the Paris Agreement. This includes the setting of national targets for the reduction of net greenhouse gas emissions; of which there are two. Firstly, all net emissions of greenhouse gases, excluding biogenic methane, should be zero by 2050. Secondly, biogenic methane emissions should be reduced by 24-47% below 2017 levels by 2050, with a 10% reduction by 2030 (Wang et al., 2022). The bill also formed the Climate Change Commission (Leining et al., 2020), of whose purpose is to monitor and give advice to the government on how to reach these targets (Climate Change Commission, 2021, 2022), as well as the ETS (Evison, 2017), New Zealand's tool for imposing an economic tax on emissions (New Zealand Government, 2019). The Climate Change Commissions latest advice to the government regarding the meeting of internal greenhouse gas targets include the creation of emission budgets, guidance on policy and strategy as well as recommendations on New Zealand's NDC to the Paris Agreement (Climate Change Commission, 2021).

The ETS is a government led initiative which effectively taxes industry for the greenhouse gases they emit, passing 'carbon credits' (NZUs) onto forestry owners, of whose trees sequester carbon (Evison, 2017; Kerr et al., 2012). Perhaps the largest tool for the government in meeting domestic and international targets (Rontard & Hernandez, 2022), the ETS places a price on emissions in every sector apart from Agriculture. Having to pay for their emissions in the form of carbon credits (1NZU = 1tCO_2-e) (Kerr et al., 2021; Manley & Maclaren, 2012), emitters are economically incentivised to reduce their emissions (Diaz-Rainey & Tulloch, 2018). In turn, NZUs are earned from the government by owners of greenhouse gas sequestering activities such as forestry. These credits are traded for a profit (Manley & Maclaren, 2012), creating a supply and demand dynamic between greenhouse gas emitters and those sequestering the harmful gases.

Due to increasing demand of a limited number of NZUs, the carbon price has risen to \$75/tCO_2-e (Jarden Securites Ltd, 2022), and is expected to increase further (Funk et al., 2014). This value is significant as it breaks the \$70/tCO_2-e threshold that has been modelled as the lowest point where mass land use change will occur (New Zealand Productivity Commission, 2018). A rising price increasingly creates a large incentive for 'carbon farmers' to blanket plant fast growing exotic tree species, namely Pinus radiata, with the expected income from the trading of NZUs far exceeding the potential earnings from other enterprises. The result is unprecedented mass land use change (West et al., 2020), with no current mitigation as the carbon price continues to make the blanket planting of fast-growing exotic forestry increasingly profitable.

1.2.2 Environmental Losses on Hill Country

Sixty percent of New Zealand's land mass consists of steep hill country (Derose et al., 1993). Runoff following precipitation events on hill country can transport environmental pollutants from slopes into catchments. Sediment and P are two pollutants which can contaminate aquatic environments; smothering natural ecosystems, inducing eutrophication and reducing the visual clarity of water bodies (Buda et al., 2009; Kleinman et al., 2011; Payen et al., 2020). This can have larger impacts downstream, where the rate of flooding events and other economic impacts; such as recreation, tourism and industry are also negatively affected (Dymond et al., 2010). Critical Source Areas (CSAs), where high concentrations of erodible sediment and P exist in fluvially active areas, still account for a large percentage of losses in hill country (Burkitt et al., 2017).

Erosion on hill country induced by overland flow, transports sediment, livestock dung and particulate P from slopes into catchments (Tozer et al., 2021), while Dissolved Reactive Phosphate (DRP) is lost as phosphate ions in soil solution (Ballantine et al., 2009; McDowell et al., 2001). Historically, sediment flows coincided with a reduction in soil stability after colonial deforestation (1840-1870) to provide grazing (MacLeod & Moller, 2006). This was followed by the subsequent intensification of hill country prompted by aerial topdressing of superphosphate (Burkitt et al., 2017). Removal of vegetative cover and associated roots along with increased soil P levels saw an increase in losses via both mechanisms (North et al., 2022). As a result, sediment and P losses to waterways have been high in recent decades. Recent advances in the application of remote environmental sensing and computer -

controlled topdressing has improved fertiliser use efficiency on hill country (White et al., 2017). In addition, other, best management practises such as riparian planting and wetland restoration have begun to have positive effects upon soil and nutrient conservation (Brown et al., 2019; Dymond et al., 2016; McDowell et al., 2021).

1.2.2.1 Sediment Loss

Since the arrival of Polynesian and European settlers in New Zealand, the country's erosion rates have increased considerably (Wilmshurst, 1997). Between the years of 1900 and the mid 1970's, the amount of land hosting some form of primary production enterprise increased from 35% to 60% (MacLeod & Moller, 2006), with the vast majority being cleared of native forests for ruminant grazing (Dymond et al., 2010; Wangui et al., 2021). On hill country, such deforestation has led to much higher erosion rates (Dymond et al., 2016; Page & Trustrum, 1997), largely due to the removal of root systems which stabilise the landscape (Wilmshurst, 1997). The soft brittle nature of New Zealand's underlying geology exacerbates this effect (Dodd et al., 2008; Page et al., 1994; Spiekermann et al., 2021; Tozer et al., 2021; Wilmshurst, 1997), with sediment being washed into catchments during precipitation events. North et al. (2022) estimate upto c.29.2 Mt y^{-1} of soil is lost via surface erosion every year.

The delivery of sediment to waterways in hill country catchments has negative effects on the health of streams and rivers. These include the suffocating of natural habitats and the reduction of light able to be utilised by phototrophs (Cournane et al., 2010; Dymond et al., 2010; Levine et al., 2021; McDowell et al., 2021). Other effects include increased risk of flooding due to deposition of sediment downstream. Sediment can also act as a transport medium for pathogens and nutrients (North et al., 2022), which can cause other detrimental effects to the health of waterways.

1.2.2.2 Phosphorus Loss

The addition of P to waterways can cause eutrophication (Ballantine et al., 2009; Buda et al., 2009; McDowell et al., 2001; Payen et al., 2020); the biological enrichment of aquatic environments (Kleinman et al., 2011). Algal growth in New Zealand waterways is generally constrained by P concentrations (Cournane et al., 2010), with nitrogen (N) levels sufficient in existing ecosystems. The effects of increased algal activity from P addition include reductions in dissolved oxygen and light levels within the water column, diminishing quantities of plant and animal biomass. An increase in turbidity also affects ecosystem dynamics, impacting activity of sight feeders such as koura (freshwater lobster) and shags.

Phosphorus loss in New Zealand agricultural systems occurs via two mechanisms, Particulate P and DRP (Cournane et al., 2010; McDowell et al., 2001). Both are instigated by runoff during precipitation events. P leaching is minimal due to specific adsorption by iron (Fe) and aluminium (Al) hydrous oxides on soil clay minerals.

Particulate P loss is heavily associated with sediment loss (Ballantine et al., 2009). Fe and Al hydrous oxides specifically adsorb P (McDowell et al., 2001), which is subsequently occluded within soil aggregates by further weathering of clay minerals. P also makes up an integral component of organic matter complexes (McDowell et al., 2001) which hold soil aggregates together. Therefore, when sediment is lost to water induced erosion (Hart et al., 2004), accompanying P is lost as well (Kleinman et al., 2011; Levine et al., 2021; McDowell et al., 2001). A review by (Hart et al., 2004) states that particulate P is the main form of P loss from hill country, however, many of the reviewed results include streambank erosion, where particulate P losses are known to predominate.

Similarly to particulate P, DRP losses are initiated by overland flow (Hart et al., 2004). Phosphate ions in soil solution diffuse into runoff with a lower P concentration. Therefore, the magnitude of P loss in this form is highly dependent on soil factors, such as the P concentration and P-retention capacity of the soil. This occurs via the process of diffusion; compounds in soil solution moving from an area of high concentration (topsoil) to an area of low concentration (surface water). McDowell et al. (2001) state that DRP loss is the major form of P loss from higher intensity pasture based systems (Buda et al., 2009). However, manure lost to runoff proceeding a recent grazing event will contain both forms of P, with particulate P found in organic complexes. On hill country, the deposition of soluble P fertilisers into waterways via product placement or transport by surface runoff remains an issue.

1.2.3 Factors Affecting Phosphorus Loss

A number of factors determine the mode and quantity of P loss in hill country systems. Hillslope hydrology and topography (Burkitt et al., 2017), soil characteristics (McDowell et al., 2003) and the abundance of CSAs (Lucci et al., 2012) have the largest impact on losses. Therefore, the modification of one or more of these factors could potentially render a reduction in P loss to wider catchments.

1.2.3.1 Hydrology and Topography

Precipitation is the singular input of water on New Zealand hill country. Rainfall intensity has been linked to increased particulate P (Fraser et al., 1999) and DRP losses (Bayad et al., 2022) due to the larger generation of runoff and raindrop induced erosion. As explained by Buda et al. (2009), runoff is initiated by one of two processes. Infiltration excess occurs when the rate of precipitation exceeds the soils infiltration rate (McDowell et al., 2001). Runoff from saturation excess is possible when the soil is waterlogged (Lucci et al., 2012). As no more pore space are available, water is unable to be taken within the soil profile; leading to ponding. On hill country where steeper slopes are prevalent, infiltration excess is generally the mechanism for overland flow (Buda et al., 2009).

Slope has a major effect on the magnitude of sediment and P losses (Hart et al., 2004). The steeper the slope, the larger the kinetic energy (velocity) of runoff during precipitation events. The faster rainfall runs off the land, the less likely it is to infiltrate the soil profile, increasing the quantity of overland flow. Greater rates of infiltration excess runoff leads to higher amounts of sediment and particulate P loss (Burkitt et al., 2017). The removal of slope stabilising roots as a result of deforestation has also increased the incidence of mass movements (Page & Trustrum, 1997), particularly during storm events (Page et al., 1994). This transports sediment and associated nutrients from slopes directly to catchments.

1.2.3.2 Soil Characteristics

Higher erosion rates are found in unconsolidated fine soils. With a lower mass, particles require less energy be transported via water (Kleinman et al., 2011). They also tend to have higher P concentrations, due to the increase in relative particle surface area (Ballantine et al., 2009; Kleinman et al., 2011; McDowell et al., 2001). As more binding sites are present per unit surface area, losses of particulate P on fine soils will be greater than those of heavier nature, given equal soil mass and hydraulic activity. During small runoff events that can only transport finer sediment, this will be the main loss mechanism for Particulate P (Cournane et al., 2010).

Aggregate formation is the result of decomposed functional groups from organic matter binding with minerals in the soil. This process effectively ‘glues’ soil particles together, forming aggregates. It has been hypothesised that bound aggregates require more energy to be displaced, making them less erodible than unaggregated soil particles (Ballantine et al., 2009). However, the additional buoyancy of organic matter has been found to increase the transportability of soil aggregates; more so than their individual components (Cournane et al., 2010). Therefore, unconsolidated aggregates in pastoral soils are often associated with relatively high erosion rates and particulate P loss.

It is known that the amount of DRP lost during surface runoff events is positively correlated with the P status of the topsoil (Cournane et al., 2010; Gillingham & Gray, 2006; McDowell & Condron, 2004; McDowell et al., 2010; McDowell et al., 2003; Simmonds et al., 2017). The Olsen P test quantifies the amount of plant available phosphate ions a soil sample has on a per unit volume basis, giving primary producers information on the P status of their soils. This information is increasingly utilised by farmers/growers to better match plant demand with supply, mitigating potential P losses. As overland flow induces DRP losses via diffusion; an increase in the concentration of phosphate present in soil solution increases the DRP content in surface water that moves across the topsoil (McDowell et al., 2001).

Conversely, the amount of DRP lost in runoff is inversely proportional to a soil’s Anion Storage Capacity (ASC) (Burkitt et al., 2017). Previously referred to as the P-retention test, this metric ranks the ability of the soil to specifically adsorb P (Kleinman et al., 2011; McDowell & Condron, 2004). The higher the ASC, the greater the quantity of P able to be adsorbed and subsequently occluded by soil colloids. Weathered Fe and Al hydrous oxides represent adsorption points for plant available P in soil solution. Further weathering of primary minerals produces more hydrous oxides, which occlude previously adsorbed P. The stronger phosphate is held by the soil profile (as measured by ASC), the less likely it is to be lost via runoff. This process is particularly relevant in areas where Al enriched allophanic soils predominate (Burkitt et al., 2017).

1.2.3.3 Critical Source Areas

Losses of pollutants are only of concern if they make their way into the extended fluvial system. As runoff is the transport mechanism for sediment and P losses, hillslope hydrology dictates the quantity of pollutants lost to waterways (Ballantine et al., 2009; Kleinman et al., 2011). Without a means of transport, potential pollutants remain just that. Areas where high erodible sediment or P concentrations intersect with hydrologically active zones linked to wider catchments are CSAs (Ballantine et al., 2009; Buda et al., 2009; Burkitt et al., 2017; Kleinman et al., 2011; Lucci et al., 2012). Although easily identifiable and typically small in area, CSAs have been found to contribute a relatively large amount of environmental losses (Srinivasan & McDowell, 2007). With the uptake of new technologies and best management practises on flatter terrain with higher stock carrying capacities, focus has moved to diffuse losses (Howard-Williams et al., 2010; Kleinman et al., 2011). However, on typically low input sheep and beef hill country, these areas still account for a large percentage of sediment and P loss (Burkitt et al., 2017); exacerbated by varied environmental factors (Srinivasan & McDowell, 2007). Therefore, the targeting of CSAs for mitigation implementation such as forestry is increasing in uptake.

1.2.4 Forestry as an Environmental Loss Mitigation

Extended root systems from tree species have been shown to increase soil cohesion, holding the soil together (Spiekermann et al., 2021; Spiekermann et al., 2022). This leads to lower erosion rates, with subsequent runoff events transporting less sediment and P from the soil surface (Tozer et al., 2021). The likelihood of mass movements occurring on steeper slopes is also drastically reduced. As a secondary effect, the canopy of woody vegetation intercepts precipitation, reducing the force of impact it has upon the soil (Beardmore et al., 2019; Guevara-Escobar et al., 2000).

The planting of woody vegetation on historic sheep and beef hill country production systems has been largely utilised as an environmental mitigation practice (Bergin & Kimberley, 2014; Douglas et al., 2007; Lambie et al., 2021; Mackay-Smith et al., 2021; Marden et al., 2020; Tozer et al., 2021), with plantings being concentrated on identified CSAs. On New Zealand hill country, diffuse P and sediment loss is largely synonymous with slope (McDowell, 2006) due to the increased gravitational potential of rain droplets and the brittle nature of sedimentary parent material (Page et al., 1994; Spiekermann et al., 2021;

Tozer et al., 2021). This has largely resulted in plantings being targeted on steeper land, with permanent native species being utilised to increase biodiversity (Bergin & Kimberley, 2014; Tozer et al., 2021; Wangui et al., 2021). Space planting of exotic species such as poplars have also been incorporated on less steep areas where grazing occurs yet erosion is still a prevalent mechanism of nutrient loss (Douglas et al., 2013; Mackay-Smith et al., 2021; Thompson & Luckman, 1993; Tozer et al., 2021; Wangui et al., 2021).

As the targeted planting of native species are often sporadic and not uniform in nature, they are often not covered by the ETS. Spatial limitations as to what can be deemed a forest capable of earning NZUs mean smaller plantations are ineligible. Four broad rules dictate what can be considered a forest; be at least a hectare in size, have a canopy closure of 30%, have the potential to reach five metres in height and have an average width of at least 30 metres (Ministry for Primary Industries, 2017). These rules exclude the likes of shelter belts and targeted plantings of native species from the scheme, despite still sequestering carbon from the atmosphere. If farmers were compensated for these areas, on land identified as unsuitable for pastoral farming, an argument could be made that the ETS supports resource management policy (Leining et al., 2020; Norton et al., 2020).

1.2.5 Current Land Use Change and Implications of Blanket Planted Exotic Forestry

As carbon farming is a relatively new concept having only recently become financially feasible due to increases in the carbon price (Funk et al., 2014), there is negligible research that relates to its practicality or applications. The allure of exotic species over native species is in their fast-growing nature (Ministry for Primary Industries, 2017). As the ETS reimburses landowners for the carbon sequestration capability of their land, exotic species are comparatively more profitable than their native counterparts (Dodd et al., 2021). This means that the majority of plantings, including those across whole hill country stations, are exotic Pinus radiata; even though this species reaches maturity and perishes long before natives.

Adams and Turner (2012) model increasing areas of long rotation (Tee et al., 2014) untended forestry with an increasing carbon price (Fernandez & Daigneault, 2016). On hill country, where the management and subsequent harvesting of such trees have proven both financially unfeasible and physically impractical (Manley & Maclaren, 2012), blanket planted pine trees are left to topple. They eventually make their way into extended freshwater catchments, impacting the environment, with widespread effects often seen downstream. Rural communities are also affected on a social and economic level, due to the immense change in land use (Adams & Turner, 2012).

1.2.5.1 Environmental Impacts

Matured exotic forestry left unmanaged on hill country eventually makes its way into the extended fluvial system (Lambie et al., 2021). This negatively affects the use of waterways for recreational activities, with inaccessibility becoming an issue due to the scale of debris. As a result, the social value of waterways with steep catchments predominantly in blanket planted exotic forestry decreases. Future work could investigate whether this also has further implications on freshwater ecosystems (Aristi et al., 2017), including inhibiting fish passage.

Lundquist et al. (2011) claim that forestry's inclusion in the ETS has decreased biodiversity loss. Protection of existing indigenous forestry does help protect biodiversity, however the author argues that this is due to a reduction in net deforestation (Adams & Turner, 2012). What the particular study fails to take into account is the species composition of any newly afforested land (Nguyen & Nghiem, 2016; Norton et al., 2020). Monoculture Pines made up 87,000 ha of plantings in forests over 40ha in size during 2020 (Ministry for Primary Industries, 2021d), the very opposite of diversity (Leining et al., 2020; Norton et al., 2020; Wangui et al., 2021). The amount of land in mixed indigenous forest continues to decrease (Ministry for the Environment & Stats NZ, 2021), despite planting efforts through the likes of the one billion trees programme.

Soil acidification occurs under exotic forestry (Alfredsson et al., 1998; Giddens et al., 1997; Parfitt et al., 1997; Yeates et al., 2000). This is due to plant uptake processes, plant removal and atmospheric decomposition (Alfredsson et al., 1998); effectively retiring the land from future pastoral farming as a result. Only a large uneconomic quantity of agricultural lime can reverse this effect. Unpruned and unthinned, blanket planted pine plantations also pose a large fire risk (Kim & Langpap, 2015; Leining et al., 2020; Manley & Maclaren, 2012). A lower initial planting density (Ning & Sun, 2017) coupled with long or non-existent rotations (Adams & Turner, 2012), compared to that of thinned production forestry, contributes to this effect. Often, fire breaks are not even considered, as planters are not necessarily experienced in forestry. Both soil acidification and fire have destructive qualities upon the land.

Plantations of Pinus radiata can facilitate natural regeneration of native species below the canopy (Forbes et al., 2019; Lambie et al., 2021). However, this requires a commitment not to harvest, which has been shown to be the cause of a plethora of issues. Reaching maturity, Pinus radiata trees undertake a physiological change, allowing conditions more suitable for native species to persist. The available space within the understory increases. The distance and seed dispersal mechanism of existing native vegetation is also an important factor regarding the rate of native species regeneration (Bergin & Kimberley, 2014; Forbes et al., 2019; Funk & Kerr, 2007). Experience has also shown that active management is required to keep invasive competing species from invading. Management of light levels is also essential for the growth of taller canopy trees, as compared to understory species (Forbes et al., 2016). Longer term studies are required to see whether this practice is a practical and cost-effective method of native forestry restoration.

1.2.5.2 Social and Economic Impacts

Rural community degradation is a result of the minimal labour requirement for carbon farmed blanket planted pine trees on hill country (Adams & Turner, 2012; Miller & Buys, 2014), with people moving from rural communities in search of employment elsewhere (Paul et al., 2013; West et al., 2020). As a result, whole farms are being brought out by carbon farmers. This in turn has had a large impact on New Zealand's ability to produce food and fibre. Kerr et al. (2012) suggest that carbon farming will substitute hill country sheep and beef production systems, with higher intensity dairy land least sensitive to changes in the carbon price. However, this modelling was undertaken at a much lower carbon price, raising the question if such systems are under threat (Adams & Turner, 2012). There is limited work comparing the financial viability of these 'carbon farming' systems over the existing farming operations they occupy (Funk et al., 2014; Tozer et al., 2021).

On a broad national scale, Adams and Turner (2012) estimate that 1.8 million hectares of marginal land could be more profitable in exotic forestry at a \$20/tCO_2 carbon price. A modelling study by Wangui et al. (2021) proposes a carbon price of over \$49.90/tCO_2 - \$91.24/tCO_2 could break even with existing farm systems for native forestry. Undertaking economic modelling upon a case study farm could provide useful insights as to the large profitability gap between fast growing exotic forestry and existing sheep and beef operations amid rising carbon prices (Dodd et al., 2021; Wangui et al., 2021). There is also limited work comparing the differences in profitability between Pinus radiata and alternative native vegetation enterprises (Pizzirani et al., 2019). Such a study could also take differing planting regimes, enterprise mixes and species composition into account.

1.2.5.3 Agreement with Resource Management Policy

The fast-growing nature of exotic Pines, and their relative ability to sequester carbon is not up for dispute (Ministry for Primary Industries, 2017). The ETS is achieving its goals in guiding New Zealand to meet its environmental mandates set by the ratified Paris Agreement (Leining et al., 2020), by encouraging the planting of fast growing trees with a higher financial incentive. However, the current effect of the ETS seems to totally contradict the purpose and goals of statutory legislation and other government initiatives; the Resource Management Act 1991 and the One Billion Trees programme respectively. The formers purpose is to "…manage the use, development, and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic, and cultural well-being…" (New Zealand Government, 1991). The latter has a goal that reads "we want to see trees integrated into the landscape to complement and diversify our existing land uses, rather than see large-scale land conversion to forestry" (Ministry for Primary Industries, 2021a).

Current practice blatantly challenges this philosophy, suggesting that the current carbon pricing mechanism utilised by the ETS is not effective at mitigating long term industrial greenhouse gas emissions (Evison, 2017). Leining et al. (2020) propose that the ETS incentivises the sequestering of greenhouse gases with fast growing exotic tree species over the implementation of mitigations to lower our gross greenhouse gas footprint (Rontard & Hernandez, 2022). Norton et al. (2020) directly attribute recent mass conversion of pastoral land to forestry to legislative and peripheral policy. These studies highlight the disproportionate incentivization of exotic forestry over native species, bringing the proposed working life of the scheme into question.

1.3 Summary

Mass land use change to exotic forestry in New Zealand hill country is attributed to internal climate change legislation. More specifically, the ETS has created a largely unregulated 'carbon market' favouring fast growing exotic species with short turn around periods between planting and maturity. This is particularly poignant given many areas currently being planted, are done so without the intent of maintenance or harvesting. Left to topple, fallen trees lead to the congestion of waterways, affecting freshwater ecosystems and recreational access. Acidification of soils, increased fire risk, loss of biodiversity and disruption to rural communities are also consequences of blanket planted exotic forestry. One positive consequence of this situation could be the 'nursing' effect of mature Pines on native vegetation regeneration.

In contrast, existing hill country farmers looking to future proof their businesses from strengthened freshwater legislation have largely done so with the targeted planting of native species on land deemed most at-risk. However, due to the slow growing nature of natives, these areas are retired from the farms productive area; earning minimal amounts under the ETS. In addition, spatial constraints imposed by the ETS often means that targeted native plantings are not eligible under the scheme.

One major knowledge gap, as emphasized by this review of literature is the lack of work comparing the economics of novel 'carbon farming' systems with traditional sheep and beef enterprises on New Zealand hill country. A case study hill country farm will be used to model different planting regimes of exotic and native forestry within an existing farming operation, allowing for the quantification of system economics. It is hypothesised that a substantial economic disparity exists between the planting of exotic species, native species and existing pastoral systems. It is thought that this leads to mass land use change, negatively impacting biodiversity and rural communities.

This work aims to quantify the difference between blanket planted Pines and targeted native plantings/existing sheep and beef operations in an economical sense. The objective of this is to outline the vast economic differences between exotic and native species and highlight the failings of the ETS. Specifically, it should highlight the need for further work surrounding the growth rates of native tree species.

2 Materials and Methods

To investigate the knowledge gap identified in the literature review, a modelling-based study upon an existing hill country sheep and beef farm is deemed most suitable (Wangui et al., 2021). In this case, simulation modelling is used to evaluate the implications of greenhouse gas policy (Rizojeva-Silava et al., 2018). Modelling a typical hill country farm with different planting regimes of woody vegetation for 'carbon farming' purposes would provide actionable results, with economic benefits able to be compared with that of the existing farm system (Antle & Capalbo, 2001). It would also likely highlight the initial carbon stock present on sheep and beef farmland, in the form of native flora (Pannell et al., 2021). This is the next best alternative to physically establishing a trial on farmland; a laborious, expensive and time-consuming task.

2.1 Species genus

Two species are modelled on a case study farm. Manuka (*Leptospermum scoparium*) as a proxy for naturally regenerating native bush and exotic Pines (*Pinus radiata*). These two species are two commonly used tree species within hill country sheep and beef production systems. Manuka being commonly utilised for erosion control (Boffa Miskell Ltd, 2017; Hedley et al., 2013; Marden et al., 2020; Stephens et al., 2005) and honey production (Boffa Miskell Ltd, 2017; Lambie et al., 2021; Nickless et al., 2017) whilst Pinus radiata is planted for production forestry (Heaphy et al., 2014), shelter belts (Knowles, 1991; Price, 1993) and increasingly, offsetting farm emissions (Dodd et al., 2021) and earning NZUs (Norton et al., 2020; West et al., 2020). Both species have solid literature basis within these contexts. Where many studies have investigated the carbon sequestration capabilities of pine trees in a range of environments, few studies have documented similar growth rates of naturally regenerated bush, let alone Manuka specifically (Hedley et al., 2013). A pilot study is developed to collect such data to flow into case study modelling and act as a comparison to default carbon storage tables used by MPI (Dodd et al., 2021; Ministry for Primary Industries, 2017).

2.2 Manuka Carbon Sequestration

The aim of this pilot study is to provide an insight as to the carbon sequestration rates of naturally regenerated Manuka. This data will then be incorporated into the economic modelling of a case study farm. Being a pilot study, the author was time limited. The extent of the pilot lacked that of a full scale multi-faceted study, and results should be appropriately interpreted. Only one locality was used for data collection.

An expedition was organised to collect Above Ground Biomass (AGB) data (Beets et al., 2014) of naturally regenerated Manuka at a property in South Taranaki. Nukuhau Carbon Limited operate a 151 hectare carbon farming business at 1326 Okahutiria Rd, Waverley (Figure 1). Previously a pastoral farm, the property was planted primarily in Pines and eucalyptus c.2009/2010 Neil Walker (2022), providing an ideal environment for the natural regeneration of native bush (Lambie et al., 2021). Being a primary coloniser, Manuka is typically the first species to establish within gaps in the canopy where light levels are sufficient (Beets et al., 2011; Boffa Miskell Ltd, 2017). In this case, natural regeneration meant Manuka established almost immediately with the removal of ruminant grazing. Therefore, an establishment year of 2010 is used for the purposes of the study, with plants being approximately 12 years old at the time of measurement. This is done as Manuka establishment on pastoral hill country is usually achieved by the retirement of land (Burns et al., 2011; Morales et al., 2016), possibly aided by the spreading of seed capsules (Douglas et al., 2007). Being unpalatable to stock, seedlings are left relatively undisturbed by grazing ruminants (Boffa Miskell Ltd, 2017). Planting individual Manuka trees sourced from nurseries is usually reserved for higher input systems such as dedicated honey production, where a high concentration of Manuka compared to other species needs to be maintained. Without financial assistance (from the likes of national or regional afforestation support schemes) nor likely return on investment (e.g. honey or production forestry), sourcing seedlings from nurseries can be an expensive task, especially with volatile carbon prices. Collecting AGB data of naturally regenerated Manuka would benefit the study as Manuka seedlings transplanted from a nursery environment may display different growth characteristics. Manuka planted for honey is also planted at lower densities of 1,100-1,600 stems per hectare compared with that recommended for erosion control (>10,000 stems/ha) (Boffa Miskell Ltd, 2017), due to differences in tree allometry.

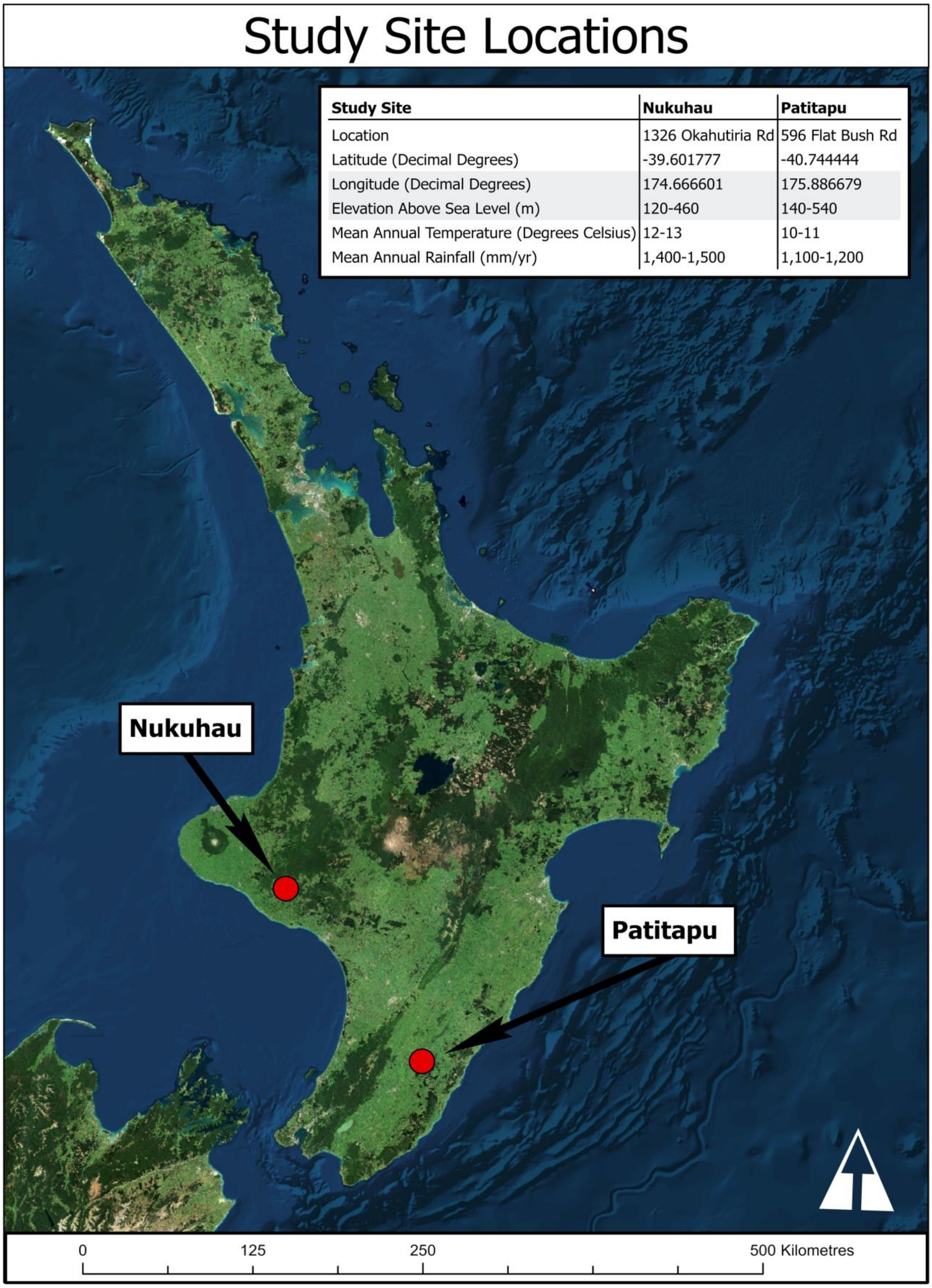

Figure 1: Property names and locations of study sites. Climate data sourced from (NIWA, 2012a, 2012b).

2.2.1 Random Grid Sampling

A shapefile of the property boundary was downloaded from the Taranaki Regional Council's open access data portal. This allowed for existing areas of indigenous bush (pre-2008) to be identified before the establishment of the carbon farming operation. Imagery dated 05/09/07 is used in Google Earth to manually identify these areas as well as existing infrastructure and waterways. With these areas removed from the analysis, a sampling grid of five metres by five metres is overlayed to the area using ArcGIS Pro v2.5.0 (Esri, Redlands, CA). A random number was assigned to each grid section, using a random number generator. The lowest 33 random numbers and their associated grid squares were selected, representing the randomly selected samples. These are shown in Figure 2. A corresponding centroid point file generated with the same ArcGIS Pro process (located in the centre of every grid square) where then selected (Gordon & Pont, 2015; Ucar et al., 2016) and transferred to a handheld GPS. As Manuka grows largely under the canopy of exotic species, it could not be established from the satellite imagery whether sample sites did in fact contain Manuka. As the presence of Manuka was uncertain, it was expected that some of the 33 sample sites would not be measurable.

2.2.2 Field Measurements

In terms of data collection, a method developed by Beets et al. (2014) is used to measure the AGB of the Manuka and subsequently, total carbon sequestered. As per Figure 3, a five metre by five metre square is measured with an industrial 30 metre measuring tape; the GPS centroid point located in the centre of each plot. A Juno T41 GPS (Trimble, Sunnyvale, CA) device was used to locate each point in the field. Plot orientation is aligned with magnetic north on each occasion.

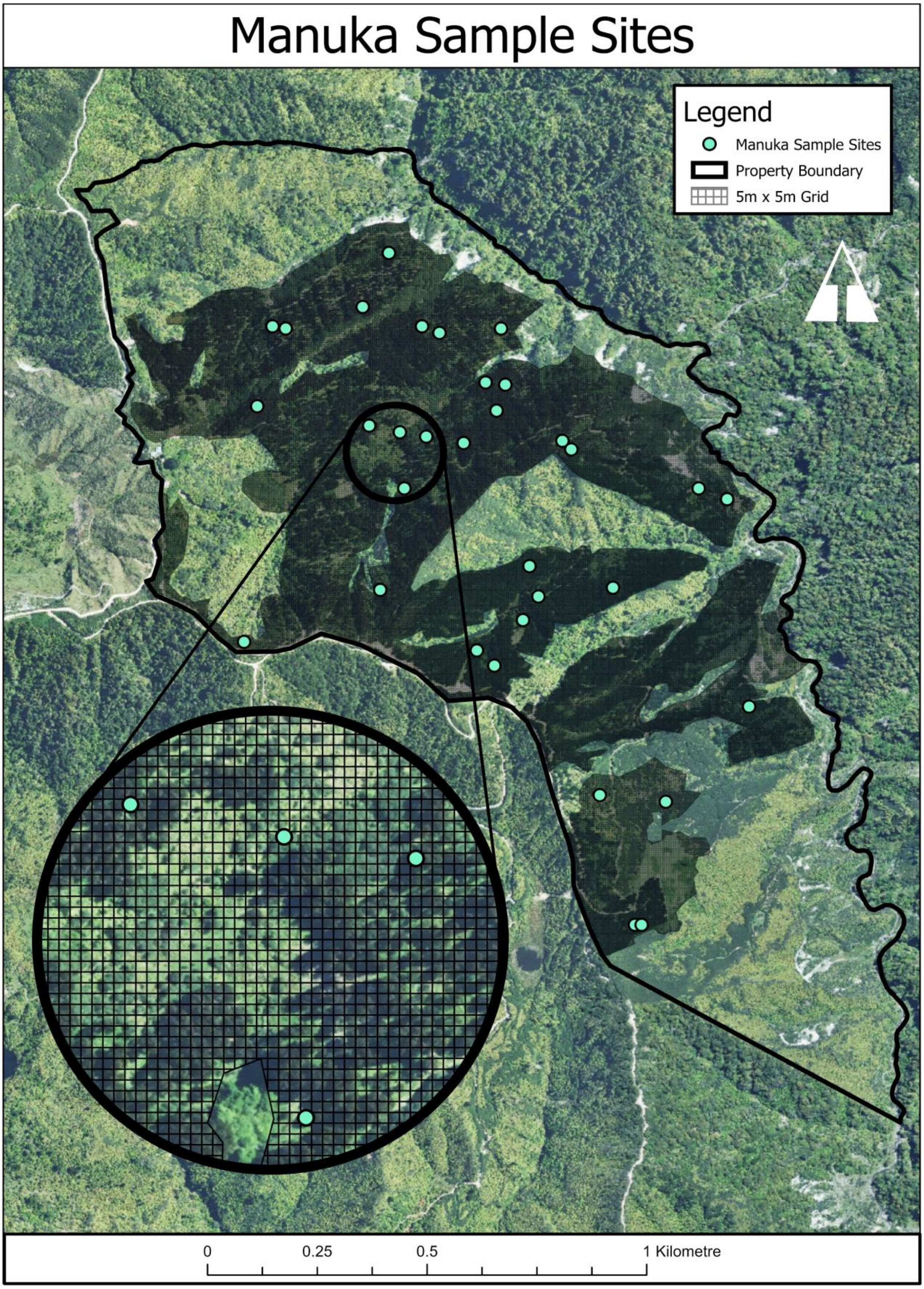

Figure 2: Map of Nukuhau detailing predetermined random Manuka Sample Sites from overlayed five metre by five metre grid.

Figure 3: An example of the five metre by five metre plots used to quantify Manuka AGB.

Within each plot, every Manuka tree over one meter in height is measured. Those under this height are deemed to be insignificant. Two measurements are taken; basal diameter and height. Basal diameter is measured 100 mm above the lowest point where the trunk intercepts the soil (Beets et al., 2014). This is done as basal diameter has shown to provide better estimates of stem volume of smaller shrubs compared to DBH (Scott et al., 2000). Where the trunk partitioned into two or more stems below 100 mm, each was measured separately. The cumulative diameter is then used. For stems with a diameter above 25 mm it was easiest to measure the circumference with measuring tape (Figure 4), with diameter calculated after field measurement. A calliper accurately measured diameter of smaller stems (Figure 4). It is assumed that stems are circular.

The height of the tallest stem on each tree is also measured. This was achieved with a Trupulse 200L laser rangefinder (Laser Technology, Inc, Centennial, CO). Measurement accuracy was limited as the device could only take measurements to the nearest metre. This could introduce some variation to the study. However, when height data is utilised in allometric equations, the accuracy of prediction has been shown to increase (Sullivan et al., 2018).

Figure 4: Measuring tape was used to measure the circumference of stems > 25 mm in diameter (left). A calliper was used to measure the diameter of stems < 25 mm in diameter (right). Measurements are taken 100 mm above the ground.

Under the current ETS, carbon stocks of forestry of 100 hectares or more is measured per the Field Measurement Approach (FMA) (Ministry for Primary Industries, 2018). This is done so using the Diameter at Breast Height (DBH) method. This measures the trunk diameter at 1.4 m above the ground. Smaller trees (such as Manuka) over 300 mm in height that have a DBH under 25mm, are subject to basal diameter measurements (Ministry for Primary Industries, 2018). For simplicities sake, the basal diameter method has been used to estimate carbon stocks of all Manuka trees, regardless of height and DBH within plots. This aligns with the methodology employed by Beets et al. (2014) and the inclusion of younger stems allows for a more complete picture of carbon sequestration to be calculated (Searle & Chen, 2017).

2.2.3 Biomass Calculations

Beets et al. (2014) devised a non-intrusive AGB measurement methodology. A SAS V2.0 GLIMMIX (SAS Institute, Cary, NC) procedure to calculate species specific coefficients to ensure the accuracy of using Calculation 1 in predicting AGB of different arboreal shrubs. This process assumed a gamma distribution, and the model was built by fitting *DryWeight* to the log function of *(Basal Area (BA) x Height)*. The result is a coefficient of 234+- 29 for *Leptospermum scoparium*, which is utilised as $a_{species}$ in Calculation 1. *Dry Weight* is the oven dry weight of the plant in kg. *BA* is the Basal Area (m^2) of the stems and is calculated using either circumference or diameter data measured 100mm above the ground. *Height* (m) is the height of the tallest stem. This calculation is used to estimate AGB of Manuka plots. Other potentially significant inputs of carbon to the system, including leaf litter and soil carbon fluxes are not measured (Lambie & Dando, 2019).

$$DryWeight = a_{species}(BA \times \ Height) \tag{1}$$

The *Dry Weight* of all Manuka trees within the plot are combined and subsequently scaled to an area of one hectare. Carbon content is assumed to be 50% of total dry matter (Beets et al., 2014; Dodd et al., 2021), however recent studies have shown that this value can differ slightly between species (Beets & Garrett, 2018; New Zealand Institute of Forestry Inc., 2005). From this, the total quantity of carbon sequestered per unit area is extrapolated. Results are subject to statistical analysis to provide meaningful context to the data. A national DEM with a spatial resolution of eight metres is used to derive slope and aspect data.

Economic returns for carbon sequestration through the NZ ETS are based upon the mass of carbon dioxide subsequently stored within woody biomass. The molar mass of carbon (12.01 $gmol^{-1}$) and carbon dioxide (44.01 $gmol^{-1}$) are used to convert between the mass of carbon and carbon dioxide. This assumes that 100% of sequestered carbon is transferred into woody biomass. Measured carbon dioxide sequestered is compared to that of twelve-year-old 'Indigenous Forest' in MPI's look-up tables. Carbon sequestration is then estimated for years 0-50 by scaling the lookup tables by the comparison factor. Therefore, it is assumed that Manuka exhibits similar growth characteristics as 'Indigenous Forest' over time. More work is required to understand the growth stages of regenerating natural bush, however carbon sequestration MPI's estimates are based on primary colonisers such as Manuka (Ministry for Primary Industries, 2017). Therefore, results should be comparable.

2.3 Economic Modelling of Carbon Farming Enterprise

Patitapu is a hill country station located on the boundary of Horizon's Regional Council and the Greater Wellington Regional Council in the Alfredton area (Figures 1 and 5). The farm supports sheep and beef totalling 17,500 stock units (SU) over the property for mainly breeding purposes (Mckenzie, 2015), the property is classed as a 'North Island Hard Hill Country' farm (Beef and Lamb New Zealand, 2021). However, it must be noted that Patitapu outperforms the average farm of the same class on a production level (Beef and Lamb New Zealand Economic Service, 2022). This is attributed to the property's shareholding managers, Doug and Jo McKenzie and their philosophy to run Patitapu 'as a high performing farming business' as compared to a medium/low performance station (Mckenzie, 2015).

Patitapu has been a trial site as part of a Primary Growth Partnership named 'Pioneering to Precision'; a collaboration between the Ministry of Primary Industries, Ravensdown Ltd and Massey University. The project aims to increase efficient use of fertiliser on hill country, subsequently increasing the productivity of the land. The centrepiece of the project is a hyperspectral camera; an AisaFENIX (Specim, Spectral Imaging Ltd, Oulu, Finland). Spectral reflectance is used to predict existing soil fertility and identify productive areas on pastoral sites. Hyperspectral bands between 380-2500nm are used to do so (Pullanagari et al., 2018; Pullanagari et al., 2016). As a result, high quality remotely sensed data exists for the property. Associated LiDAR also produces a high-quality Digital Elevation Model (DEM). It is the availability of these products coupled with the large variability in slope and existing vegetative cover that led to Patitapu being chosen as a suitable case study farm for the modelling of woody vegetation (Wangui et al., 2021).

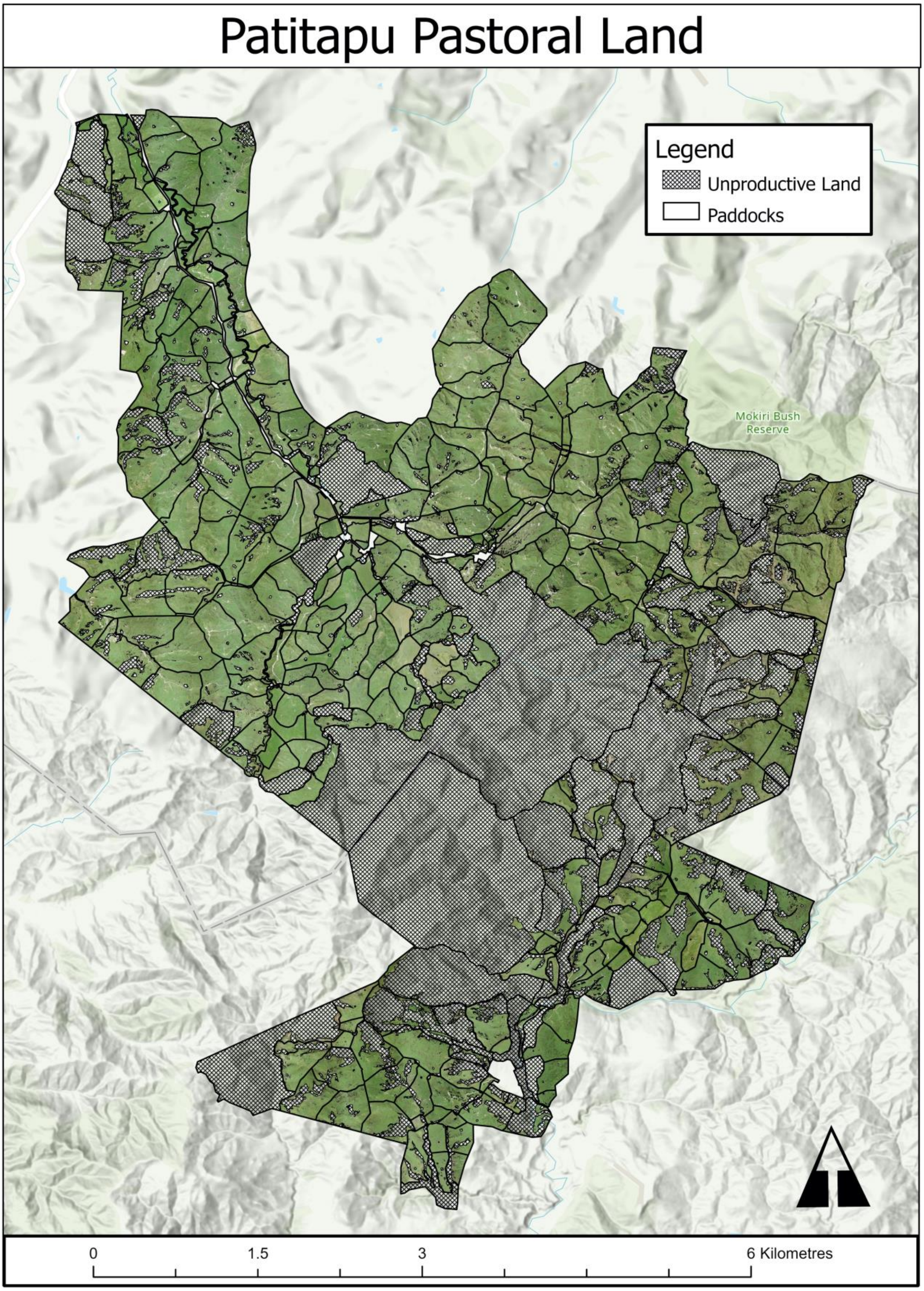

Figure 5: Map of Patitapu with unproductive areas removed. 'Unproductive land' is used in the context of the study; land which is not pastoral.

2.3.1 Identification of Land Suitable for a Hypothetical Carbon Farming Enterprise

The first step was to identify land suitable for 'carbon farming' on Patitapu, a variable that differs between models. From an economic point of view, the most unproductive of areas for ruminant production are the most suitable for the planting of woody vegetation. In an agricultural context, pasture production is taken as a proxy for the productivity of the land (Murray et al., 2007; Pullanagari et al., 2015) which in turn is largely affected by soil type, elevation, slope and aspect in New Zealand Hill country (Pullanagari et al., 2018; Pullanagari et al., 2016; White et al., 2017). Examples of work that has been used to identify these low pasture producing areas based upon these variables include (West & Turner, 2014), with slope being found to have the main influence on pasture production and (Lopez et al., 2003; PA Handford and Associates Ltd & AgFirst, 2011). It has been found that on sedimentary hill country, areas of low pasture productivity are commonly associated with heightened nutrient and sediment loss (Donovan & Monaghan, 2021; Spiekermann et al., 2021), which is to be expected on steeper slopes (McDowell, 2006). For the purposes of the study, slope will be the singular variable determining suitable areas for vegetation modelling.

An RGB image with a spatial resolution of 43 cm was used to manually identify unproductive farm areas in ArcGIS Pro (Figure 5). This included waterways (rivers and dams), existing forestry (native and exotic) and farm infrastructure. These areas were subsequently removed from the analysis. A farm boundary shape file was used to clip imagery and subsequent raster data to the extent of the farm. This methodology was used as supervised classifications without access to infra-red bands provided poor results (Martinez-Montoya et al., 2010; Panigrahy et al., 2009). For the purposes of the study, existing forestry is considered to be pre-1990 forestry land ineligible for earning NZUs under the ETS.

A one metre DEM provided from LiDAR with a sensing altitude of 660 m (Pullanagari et al., 2018), clipped to the property boundary with unproductive areas removed was used to produce a slope map using ArcGIS Pro. Values are then reclassed to slope classes as detailed by Lynn et al. (2009) as part of the Land Resource Inventory (LRI) , which provides the backbone to the Land Use Capability (LUC) scheme. These seven slope classes specified in Table 1 provide the basis for modelling areas suitable for vegetation under a 'carbon farming' enterprise. These slope classes were chosen as they provide more categories than the three broad slope classes (low, medium, steep) utilised by many (Wangui et al., 2021). Hill country farms are often characterised by the variability of such factors. A higher quantity of slope classes better captures this variability.

Table 1: LRI slope classes used in modelling of Patitapu, adapted from (Lynn et al., 2009).

Slope Class	Slope Angle (degrees)	Description	Typical Examples
A	0-3°	Flat to gently undulating	Flats, terraces
B	4-7°	Undulating	Terraces, fans
C	8-15°	Rolling	Downlands, fans
D	16-20°	Strongly rolling	Downlands, hill country
E	21-25°	Moderately steep	Hill country
F	26-35°	Steep	Hill country and steeplands
G	>35°	Very steep	Steeplands, cliffs

Table 2 illustrates the models used to identify land suitable for carbon farmed vegetation. A number of key spatial limitations exist to define a forest under the ETS. To be eligible to earn NZUs under the ETS, trees must (a) occupy an area of at least one hectare, (b) have an average width of 30 metres or more and (c) have a canopy cover greater than 30% (Boffa Miskell Ltd, 2017). A tree is a species capable of reaching at least five metres in height, excluding those whose primary purpose is to produce fruit or nuts (Ministry for Primary Industries, 2017). Two overarching themes are used to illustrate the effect these spatial limitations have upon the economics of hill country land. In Scenario 1, the spatial limitations imposed by the ETS are adhered to. It is assumed identified planting areas over a hectare in size have an average width of 30 metres and have a canopy cover close to 100%. In Scenario 2, vegetation is modelled on a 'where needed' basis, with no regard to such spatial constraints. However, canopy cover is still expected to be close to 100%.

To identify areas that conform to the models in Table 2, the reclassed slope map is converted to polygons using ArcGIS Pro. The default simplification of polygons is left on, as this would better represent a practical planting plan. However, the 'Simplify Polygon' tool is used to further streamline the result. The Visvalingam-Whyatt algorithm is used with a tolerance of two metres and a minimum area of 25 m^2, due to its ability to retain effective polygon areas (Visvalingam & Whyatt, 1992). In effect, this removes smaller polygons within larger areas. To ensure that the resulting modelled area is somewhat practical from a fencing point of view, the Zhou-Jones retain weighted effective areas simplification algorithm was also run with a minimum area of 25 m^2 (Zhou et al., 2005). This is done in an effort to remove peripheral peninsulas and depressions within the planting plan.

Table 2: Models used for the identification of farmland suitable for the creation of a hypothetical carbon farming enterprise on Patitapu.

	Scenario 1 (ETS compliant)		Scenario 2 (Total area)	
Model Number	LRI Slope Classes	Tree Species	LRI Slope Classes	Tree Species
Control	Existing Farm System	N/A		
1	G	Pinus radiata	G	Pinus radiata
1	G	Manuka	G	Manuka
2	F+G	Pinus radiata	F+G	Pinus radiata
2	F+G	Manuka	F+G	Manuka
3	A+B+C+D+E+F+G	Pinus radiata		
3	A+B+C+D+E+F+G	Manuka		

Throughout the process, any newly formed polygon boundaries of the same slope class are removed with the 'Dissolve Boundaries' tool. All holes in polygons under the size of 5000 m^2 are removed. It is unlikely that farmers would leave gaps in fenced tree stands as the utilisation of land affects its profitability (Dodd et al., 2008). Larger areas could potentially still be grazed if farm tracks are established. Similarly, the effort required to plant and fence smaller areas would largely become uneconomic. For practicality, any paddocks which had an initial modelling of over 75% in modelled forestry were assumed to be fully included within the planting plan. In contrast, polygons in paddocks where total initial afforested area was less than 25% were removed. This allowed for a greater proportion of existing fencing to be incorporated into the model, reducing the capital expenditure required for any mass land use change.

For Scenario 1, selection of these polygons for economic modelling is dependent on their size exceeding one hectare. Therefore, polygons of differing slope classes are identified as per Table 2 before having boundaries dissolved (e.g. adjacent polygons with respective slope classes of F and G are merged to form a single contiguous polygon for Model 2). Polygons exceeding 10,000 m^2 in size are selected and exported as the model. In contrast, Scenario 2 draws upon the 'right tree, right place' philosophy adopted by the Government. Total area the polygons inhabit is used for economic modelling.

2.3.2 Model Creation for Remaining Pastoral Land

As slope affects pasture productivity, the differences in the land's carrying capacity is accounted for. LRI slope classes were assigned pasture production values (kgDM/ha/yr) as measured by Lopez et al. (2003). This study is used due to the geographical relevance to Patitapu, with the study site being located on Lower North Island hill country (Ballantrae Research Station; AgResearch Grasslands). As pasture production is correlated to three slope classes; Low Slope (0-12°), Medium Slope (13-25°) and High Slope (>25°), a review of literature by Wangui et al. (2021) is used to assign pasture production to LRI slope classes. Assigned pasture production values are summarised in Table 3.

Table 3: Slope classes employed by (Lopez et al., 2003) correlated to LRI Slope Classes (Lynn et al., 2009) as per (Wangui et al., 2021).

(Wangui et al., 2021) Slope Classes	(Lopez et al., 2003) Slope Classes	(Lynn et al., 2009) LRI Slope Classes	Assigned Pasture Production (kgDM/ha/yr)
Low	0-12°	0-3°	12,568
		4-7°	12,568
		8-15°	12,568
Medium	13-25°	16-20°	5,806
		21-25°	5,806
Steep	>25°	26-35°	4,003
		>35°	4,003

Pasture utilisation by livestock is assumed to be 85% (PA Handford and Associates Ltd & AgFirst, 2011). This is likely higher than the average pasture utilisation of a hill country farm (Wangui et al., 2021), but is indicative of the pasture management skills possessed by the owner (Mckenzie, 2015). The annual quantity of pasture required for consumption of a singular stock unit (SU) is 550 kgDM (Lincoln University Faculty of Agribusiness & Commerce, 2011; Wangui et al., 2021). Pasture production is calculated by multiplying polygon area by estimated pasture production. By removing wastage (15%) and dividing by 550 kgDM, carrying capacity per polygon is calculated. From this information, stocking rate (SR) can be determined on a per slope class and overall farm system basis. Hypothetically, as modelled forestry takes up increasing amounts of land, the remaining farming system increases in profitability due to increased stocking rates on relatively flat land.

2.3.3 Application of Economics to Models

For estimating model profitability, both the forestry and remaining farm system need to be assessed. Previously measured and modelled production levels of the two enterprises need to be converted into monetary values. Two overarching assumptions are made. Pastoral land is considered 'status quo' (Funk et al., 2014). Currently operating as a ruminant production system, any modelled changes in land use will incur any associated costs. For simplicity, all modelled land use change is expected to occur in year one (Hale et al., 2015). Practically, this may not occur due to the magnitude of proposed changes. However, this allows for the easy differentiation between initial capital investment and ongoing maintenance costs for the forestry enterprise. Secondly, all modelled forestry is assumed to be planted for the sole purpose of earning NZUs under the NZ ETS. Therefore, low maintenance costs are expected. Existing farm tracks are considered satisfactory and no pruning or thinning costs are incurred (Kim & Langpap, 2015; Manley & Maclaren, 2012; West et al., 2020). Preferential incomes such as timber harvesting for Pines, or honey/oil production for Manuka are not considered, however could be considered in future work.

2.3.3.1 Modelled Forestry

Costs for modelled forestry are split into two parts. Initial costs and ongoing maintenance costs, as summarised in Table 4. Initial costs are further divided. Establishment costs consist of seedlings ($584/ha Pines) (Pizzirani et al., 2019), labour ($324/ha Pines) (Pizzirani et al., 2019), and release spraying ($183/ha Pines, $310 Manuka) (Pizzirani et al., 2019). Natural regeneration of Manuka can be achieved if a close Manuka seed source exists (Bergin & Kimberley, 2014). Failing this, establishment can occur via direct seeding; the manual spreading of seed capsules (Douglas et al., 2007). Either way, seedlings (and associated planting labour) are not required. Any costs associated with this process (e.g. spreading of seed) is assumed to be minimal and subsequently not included in the analysis. However, it has been found that establishment often relies on competing species (e.g. grasses) being controlled. Hence, costs for the blanket spraying of retired land are taken into account. In contrast, Pines are planted as seedlings sourced from a nursery, requiring physical planting as well as a release spray.

Table 4: Estimated initial and maintenance costs for modelled carbon farming enterprise.

Forestry Costs				
Initial Costs		Pines	Manuka	
Fencing	Labour	5.2	5.2	$/m
	Materials	2.2	2.2	$/m
Total Fencing Cost		7.4	7.4	$/m
Establishment	Seedlings	584	0	$/ha
	Spraying	183	310	$/ha
	Labour	324	0	$/ha
Total Establishment Cost		1091	310	$/ha
Maintenance Costs				
Maintenance		100	20	$/ha/yr
Total Maintenance Costs		100	20	$/ha/yr

Estimated fencing costs are $7.4/m and are based on modelled polygon perimeters minus the perimeter of any whole paddocks used in the model. This figure is derived from interviews specific to the Horizon's region, which quotes labour costs ($5.2/m) and material costs ($2.2/m) for erection of an electric 4 wire fence on steepland (The AgriBusiness Group, 2016). It is expected that the modelled cost will very much be a conservative estimate. It does not account for any existing fencing that can be utilised other than where whole paddocks are identified. As fence lines on hill country often follow natural topographical features such as ridgelines, it is expected that existing fencing will play an integral part in any practical fencing plan.

The establishment and fencing costs combine to form the initial costs of the project. This is assumed to be the amount borrowed from the bank in the form of a term loan. A ten year repayment period (Beef and Lamb New Zealand Economic Service, 2022), with equal principal repayments, incurs interest of 6% pa (Research First Ltd, 2021). Ten years is an averaged (2015-2020) rural lending term for increased mortgages (Beef and Lamb New Zealand Economic Service, 2022). As interest rates have been historically low in recent years (Reserve Bank of New Zealand, 2020), a maximum rate since 2015 has been used (Research First Ltd, 2021). For practicality, accounts have been budgeted on an annual basis. That is, there is only one compounding period per annum. An overdue account interest rate of 8% is liable in the year after which it is incurred (Research First Ltd, 2021). Ongoing maintenance costs total \$100/ha/yr for Pines (New Zealand Institute of Forestry Inc., 2005) and \$20/ha/yr for Manuka (Boffa Miskell Ltd, 2017). Values include management costs and insurance for Pine forests against fire.

MPI lookup tables were used to calculate Pine carbon sequestration specific to the Hawkes Bay/Southern North Island region (Ministry for Primary Industries, 2017). Annual changes in carbon accumulation are used to determine income received. The same process is followed for Manuka, albeit with new carbon sequestration values derived by fitting measured carbon sequestration from the pilot study to existing growth rates set out in MPI's look-up tables under 'Indigenous Forest'. An initial carbon price of \$75/tCO_2 is used to determine generated revenue. This is the approximate market price as of June 2022 (Jarden Securites Ltd, 2022).

2.3.3.2 Remaining Pastoral Land

An annual economic study by Beef and Lamb New Zealand Economic Service (2022) averages farm profitability for both (a) region specific and (b) land type on a per SU basis. In this case, monetary values are specific to hard hill country in the Eastern North Island (Beef and Lamb New Zealand, 2021). To account for variation in market prices, a five-year averaged farm profit (before tax) is applied to remaining pastoral land. The resultant averaged figure (2015-2020) of \$24.72/SU takes into account farm inputs, outputs as well as business administration costs such as interest and depreciation. Income is multiplied by the quantity of stock units on the modelled system to determine profitability of the remaining pastoral land.

2.3.4 Resultant Models

The resulting net profit of ruminant production and carbon farmed forestry are summed to determine the profitability of the system as a whole in Years 1 (planting) through to Year 50 (Wangui et al., 2021). The upper limit is constrained by lack of published growth data for both Pines and Manuka (Ministry for Primary Industries, 2017). As trees exhibit differing growth rates depending on their age, system profitability will change depending on the stage in the forests life span. Therefore, taking into account the long-term profitability of the system will help determine the viability of modelled changes.

The resulting economic models are subject to analysis, including optimisation, Net Present Value (NPV), Internal Rate of Return (IRR) and sensitivity analysis. An optimisation model was run on the base polygon layer to determine the most profitable enterprise on each parcel of land (Hale et al., 2015). This was done on the basis that all enterprises were in a 'status quo' state, with economics built in accordingly (Table 4). Average changes in carbon storage from years 0-50 are used for both Pines and regenerating bush (Wangui et al., 2021).

Two discount rates are used to calculate NPV. These are 3% and 6%. The former represents an approximate long term average inflation level, whilst the latter is better aligned with current market conditions. The resulting figure will forecast the value of today's investment in the future (Abdul-Salam et al., 2022; Hale et al., 2015). This represents an estimate for the opportunity cost of the investment (Paul et al., 2013). IRR is also calculated. This indicates the profits in relation to the investment when the model's NPV is equal to zero.

Sensitivity analysis is used to illustrate the impact market price changes could have upon the viability of such systems. Due to the uncertain future of NZUs, a wide range of carbon prices are considered, with analysis of what this could mean for the property on an economic level (Funk et al., 2014). Deviations in the profitability of livestock production is also considered, with ranges reflecting maximum and minimum profitability in recent times (Beef and Lamb New Zealand Economic Service, 2022).

3 Results

3.1 Nukuhau: Manuka Sampling

Above Ground Biomass was successfully measured at 16 of the 33 grid sample locations (48.48%). Figure 6 shows which sites were measured. Seven sites are unable to be used as Manuka did not persist in these areas. Prolific undergrowth on the steep terrain, impacted accessibility. Some old farm tracks identified using historic Google Earth imagery had not been maintained during land use change. For this reason, data from ten further sites is not collected. Due to time constraints (being a pilot study), random points are unable to be checked beforehand for Manuka persistence or accessibility. Due to the small sample size, results should be interpreted appropriately. Time constraints meant a large-scale study was not practical. Raw data and subsequent analysis from sample sites is accessible in the appended excel sheet entitled 'Manuka Carbon Sequestration.xlsx'.

3.1.1 Manuka Carbon Storage

The mean Manuka carbon storage of twelve-year-old naturally regenerated Manuka is 29.92 tC/ha. This translates to the storage of 109.63 tCO_2/ha which will be used for the purposes of economic modelling. This value is 81.80% more than what MPI estimates from regenerating bush of the same age (Ministry for Primary Industries, 2017). This increase in carbon storage is scaled to growth rates provided in MPI look-up tables for indigenous forest. These values are subsequently used for the economic modelling of regenerated bush on Patitapu.

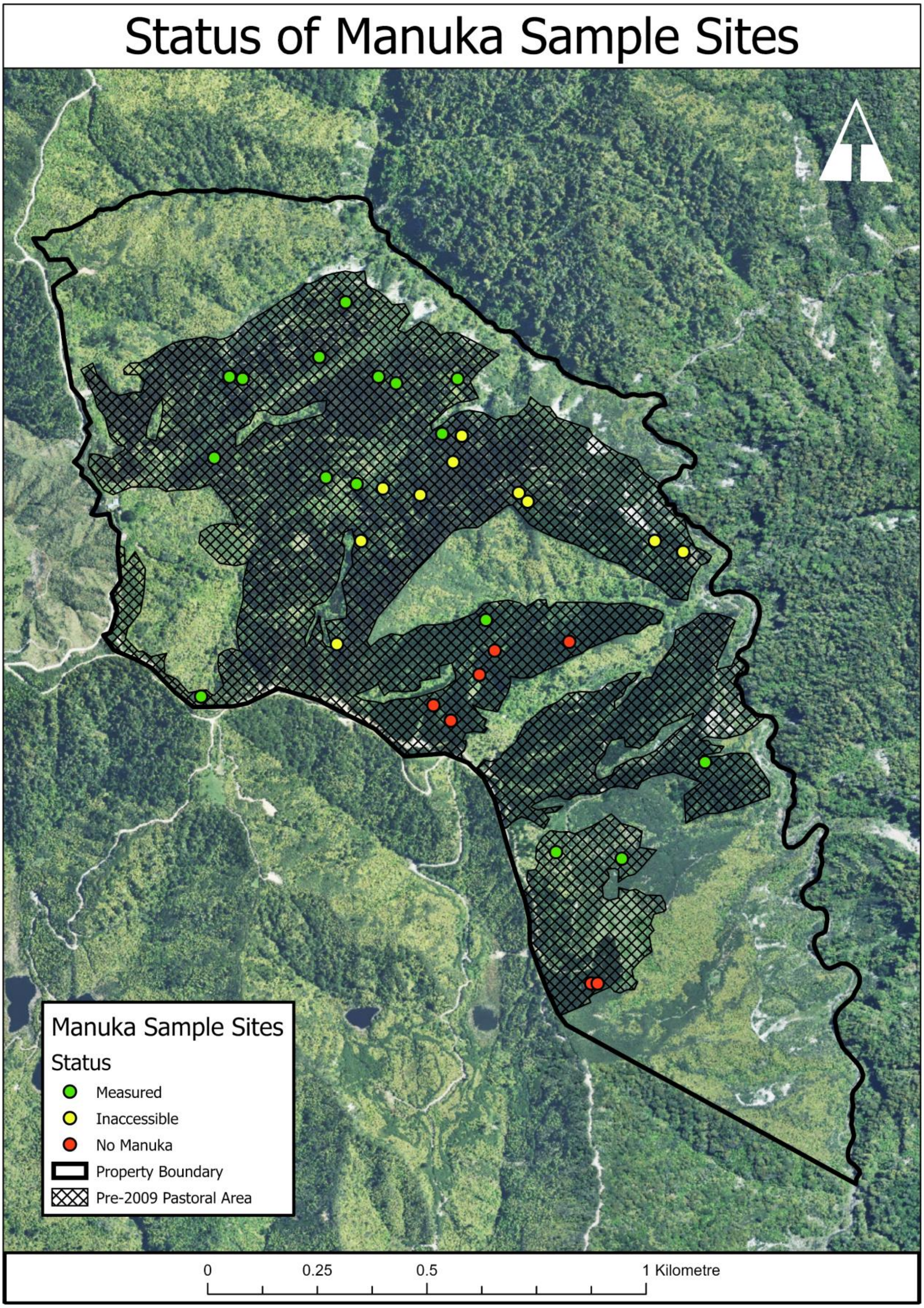

Figure 6: The status of Manuka Sample Sites on Nukuhau.

The median carbon storage is 22.50 tC/ha (82.45 tCO_2/ha), indicating that the data is right skewed (Figure 7). The range of calculated Manuka carbon storage values is large; 59.74 tC/ha between the minimum of 11.11 tC/ha and maximum of 70.85 tC/ha. This large range is due to the bimodal nature of the data as shown in Figure 7. This means that regression models or confidence intervals cannot be effectively applied to the data in its current form. Pedersen and Skovsgaard (2009) state that due to the nature of the data, "nonlinearity is often addressed, but rarely quantified" in the discipline of forestry science. Being a biological system, relationships between variables are often complex (Tozer et al., 2021), resulting in data which does not follow a recognised distribution pattern.

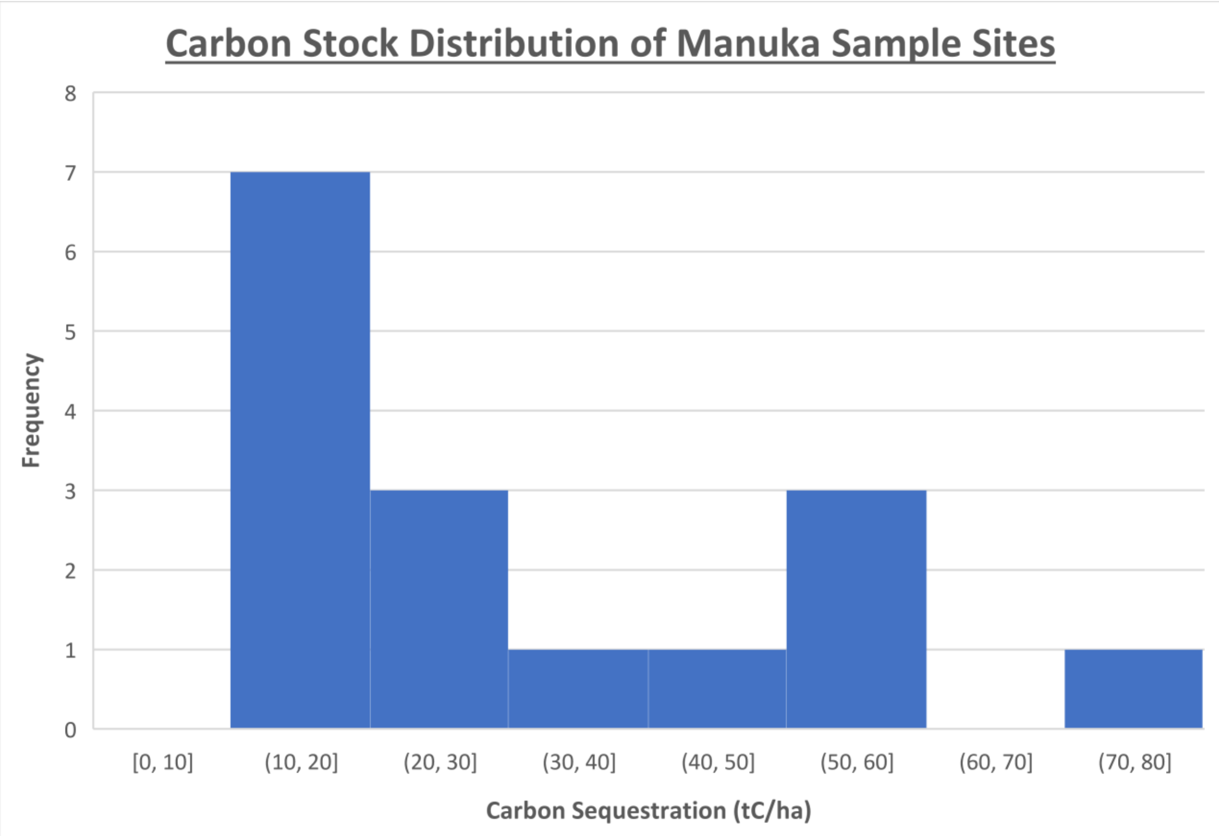

Figure 7: The distribution of calculated carbon storage among sample sites.

3.1.2 Regression using Individual Tree Data

As lack of a known distribution inhibited the use of regression to predict carbon storage of Manuka stands on a per unit area basis, raw data was collated to see if it could be used to predict carbon storage of individuals. As per Figure 8, plotting the calculated carbon storage of each tree on a histogram returns a gamma distribution (Beets et al., 2014). Applying a log(10) function to the raw data transforms it to an approximate normal distribution (Figure

9), inferring the original held a chi squared distribution (subclass of gamma distribution). A Normal QQ plot is shown in Figure 10 to demonstrate the approximate normality of the transformed data. As the assumption of normally distributed data is met, regression analysis can be used to predict variables. The author hypothesises that if data from ample sample sites is collected, the distribution would reflect that shown amongst individual trees (gamma; chi squared distribution).

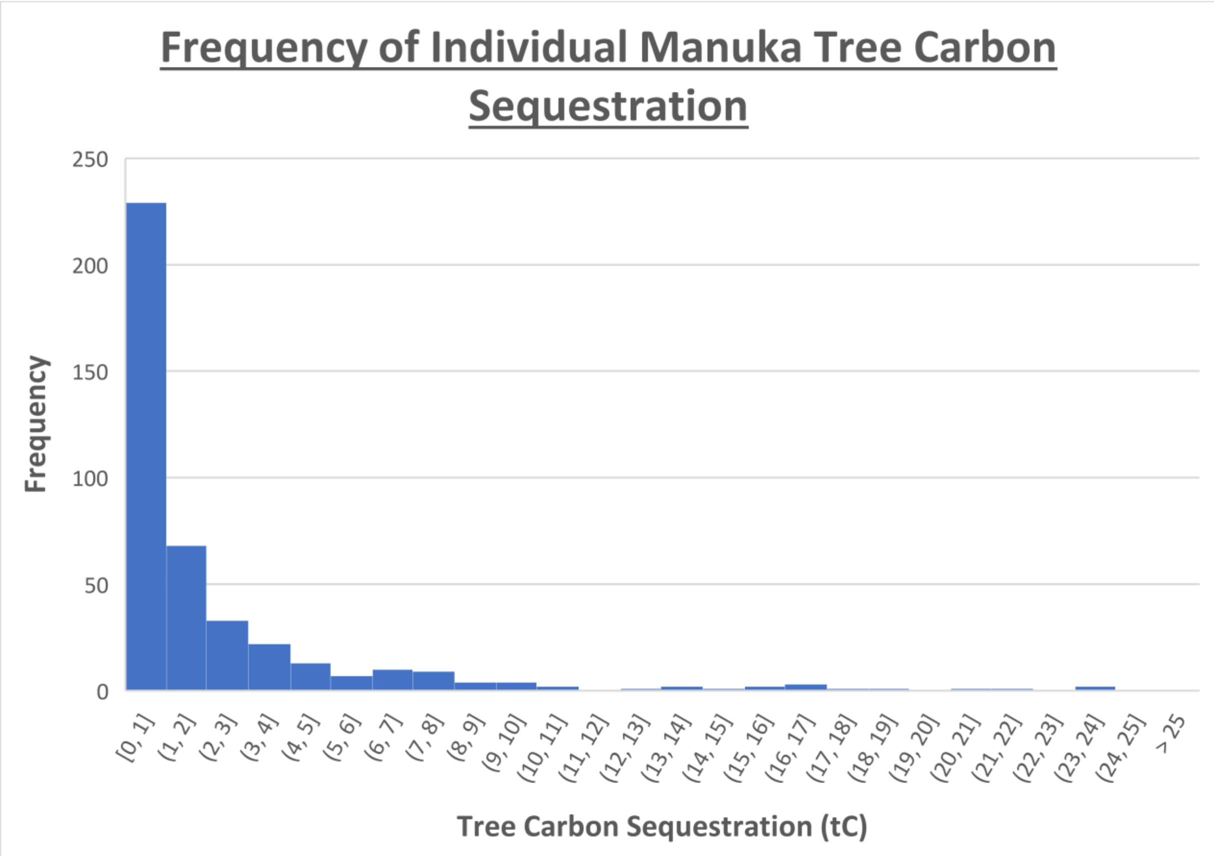

Figure 8: Distribution of individual tree carbon storage.

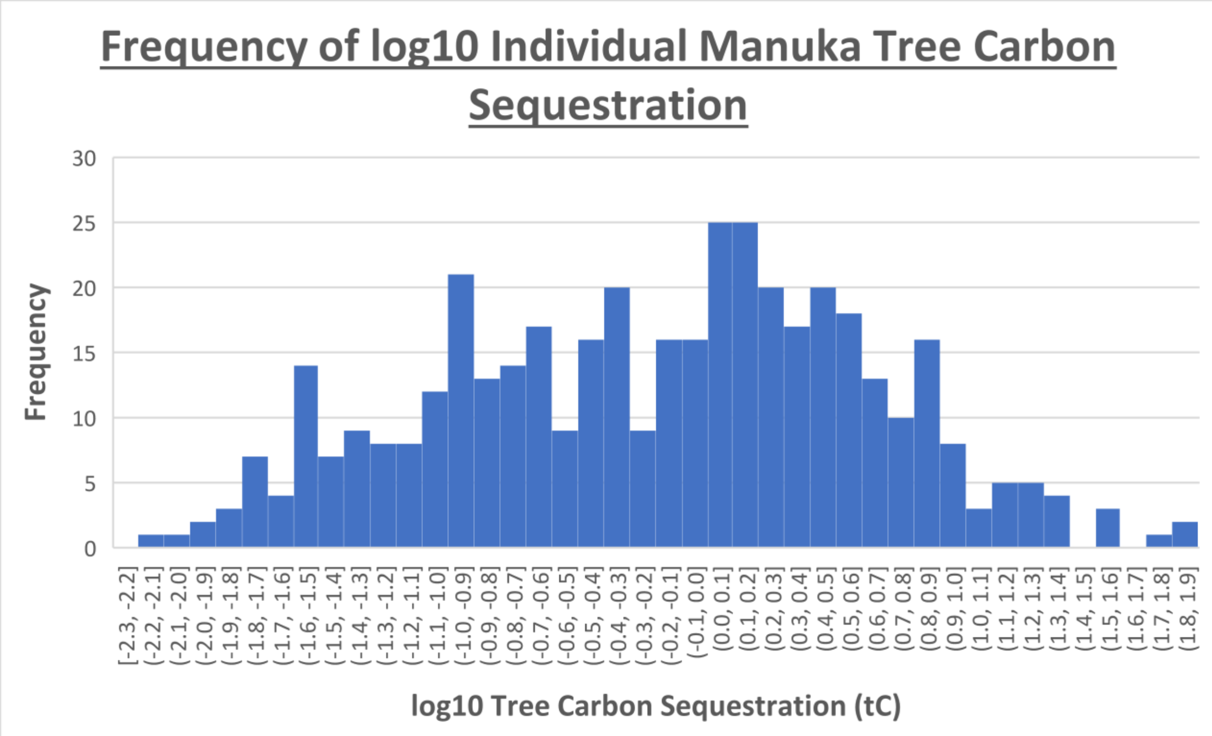

Figure 9: Distribution of individual tree carbon storage after applying a log(10) function.

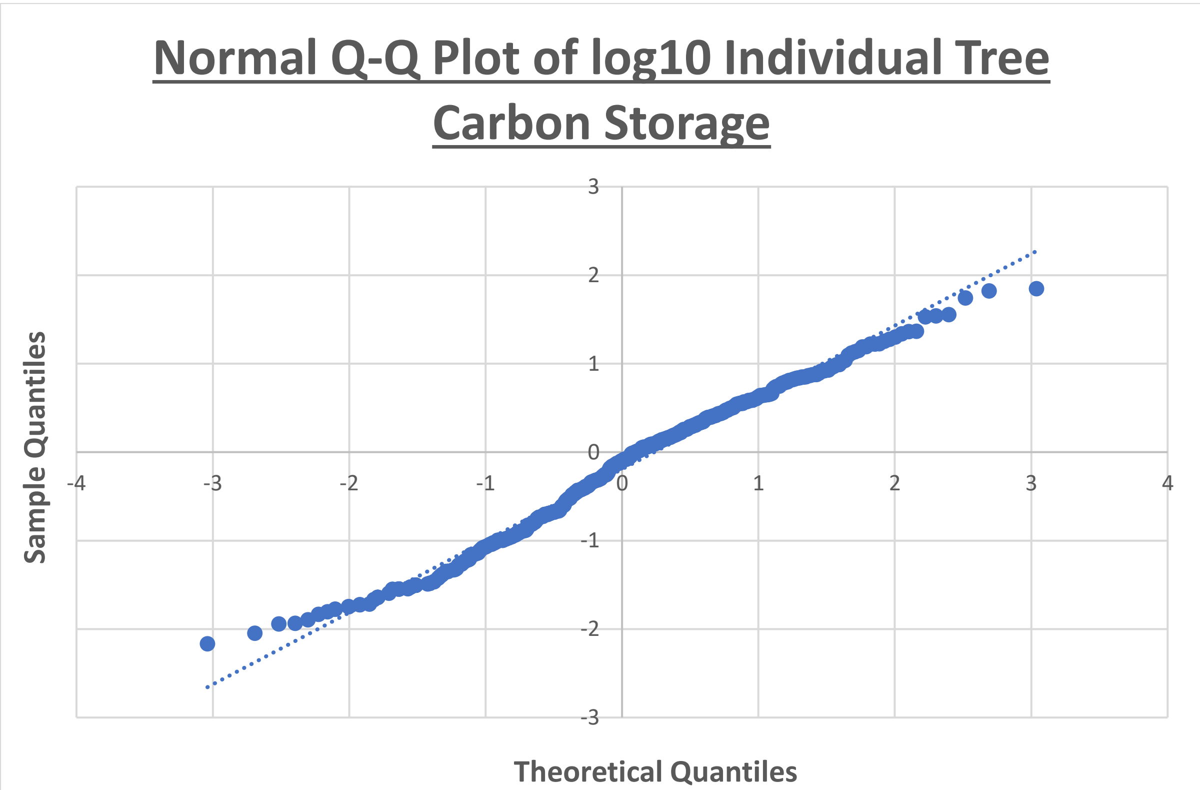

Figure 10: QQ Plot of individual tree carbon storage after applying a log(10) function. This illustrates approximate normality.

3.1.2.1 Using Regression to Predict Variables

Correlation coefficients and subsequent coefficients of determination were determined on an individual tree basis. Table 5 details the relationship between all measured and calculated variables. As basal circumference and height were used to measure tree carbon in the first instance, they are dependent on each other. However, an equation that computes tree carbon storage is able to be derived as a simplification of Equation 1. Circumference was also found to have a gamma, chi squared distribution. Hence, it was also transformed using a log base 10 function. Height data was discrete, yet appeared to have an approximate normal distribution. Both appear to be good predictors of the transformed individual tree carbon storage ($r^2 > 0.60$). Circumference and height are also colinear.

3.1.2.2 Individual Tree Carbon Storage

The two best predictors of individual tree carbon storage are log10(Circumference) ($r^2 = 0.98$) and Height ($r^2 = 0.68$). Multiple regression is used to determine if a combined model could predict individual tree carbon to a higher accuracy that any one of the variables by themselves. The goal is to provide an equation which utilises the measured inputs of Circumference and Height to calculate tree carbon levels, as a means of simplifying equation 1. Significance levels of said model are detailed in Table 6.

As the p-value is below 0.05 for both Height ($3.9626\ldots \times 10^{-200}$) and Circumference (0.0), they are deemed to be significant additions to the model. A low overall p value of zero indicates that this more complex model does a better job at predicting individual tree carbon storage than either variable by itself. Such a p value was expected due to these two variables being used to originally calculate tree carbon storage. An adjusted r^2 of 0.9973… indicates that 99.7% of variation in tree carbon storage can be explained by Circumference and Height in a combined model. The resulting equation is shown in Equation 2. *Carbon Storage* is measured in kg, *BC* (Basal Circumference) in millimetres and *Height* in metres.

$$Carbon\ Storage = 10^{((0.12586594 \times Height) - 4.9878298} \times BC^{2.01132098} \qquad (2)$$

Table 5: Coefficients of correlation (r) and coefficients of determination (r^2) between various measured and calculated variables.

Regression	r	r^2
log10(Circumference):log10(tC)	0.99	0.98
Height:log10(tC)	0.82	0.68
log10(Circumference):Height	0.73	0.54
Slope:tC/ha	-0.68	0.46
Aspect:tC/ha	-0.48	0.23
Slope:Density	0.46	0.21
Height:tC/ha	0.41	0.17
log10(Circumference):Density	-0.37	0.14
Aspect:Density	-0.36	0.13
log10(tC):Density	-0.35	0.12
log10(Circumference):Slope	-0.34	0.11
log10(tC):tC/ha	0.33	0.11
log10(tC):Slope	-0.33	0.11
Height:Slope	-0.25	0.06
Height:Density	-0.18	0.03
tC/ha:Density	-0.14	0.02
Height:Aspect	-0.08	0.01
log10(tC):Aspect	-0.04	0.00
log10(Circumference):Aspect	0.01	0.00

Table 6: Model utilising the variables circumference and height to predict the carbon storage of individual Manuka

SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.998689385
R Square	0.997380488
Adjusted R Square	0.997367984
Standard Error	0.041941407
Observations	422

ANOVA

	df	SS	MS	F	Significance F
Regression	2	280.6341456	140.3170728	79767.23175	0
Residual	419	0.737055207	0.001759082		
Total	421	281.3712008			

	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 95.0%	Upper 95.0%
Intercept	-4.987829762	0.014090711	-353.9800041	0.0	-5.015527052	-4.960132471	-5.015527052	-4.960132471
log10(Circumference)	2.011320979	0.008871497	226.717198	0.0	1.993882793	2.028759165	1.993882793	2.028759165
Height	0.125865939	0.002199639	57.22117586	3.9626E-200	0.121542236	0.130189642	0.121542236	0.130189642

This equation is a simplification of Equation 1. Due to Height and Circumference being the two inputs for the original equation, other inputs lack strength as explanatory variables to be considered for the new model. By the same notion, these variables are not independent, an assumption for model creation. However, in the context of equation simplification, this is acceptable. Equation 2 could be tested for accuracy by conducting a field trial, potentially on Patitapu.

Boffa Miskell Ltd (2017) suggest that Manuka growth rates are higher on North facing slopes, due to increased light interception and temperature. A slight correlation was found between carbon storage and aspect on a per unit area basis ($r^2 = 0.23$) (Table 5). Notably, no measurable sample sites had a Western orientation (225-315 degrees). However, slope was found to be negatively correlated with calculated total carbon storage per hectare ($r = -0.68$). The steeper the gradient, the lower the total carbon storage. Aspect and slope should therefore be taken into account when making land management decisions. LiDAR with a spatial limitation of one metre to be released in August 2022 by the Taranaki Regional Council could be utilised to determine whether evidence exists that supports these claims using a dataset of higher sensitivity.

3.1.3 Tree Height

Mean tree height is 3.91 metres when averaged amongst all individuals in the sample population. It is within 95% certainty that the population mean sits between the values of 3.77 and 4.06 metres. An interquartile range of 2.00 metres indicates that the majority of sample sites were located in similar environments (under the canopy of exotic species of similar densities). As a result, vertical growth of Manuka trees is comparable over twelve years. A larger range of 6.00 metres is due to two factors. At the lower end of this range, young trees filling in previously uninhabited areas are measured. A small number of sample sites where light is not blocked by an exotic canopy appear to have relatively fast vertical growth rates.

This infers vertical growth of 326 mm/yr. This is lower than estimates provided by Boffa Miskell Ltd (2017) of 410 mm/yr and is likely due to increased shade levels from relatively fast growing exotic trees. The same source states that a healthy Manuka tree will be 4 metres high at year 10, suggesting that vertical growth is stunted by the overarching exotic canopy. As a result, we can assume that measured carbon sequestration of Manuka is conservative by nature.

3.1.4 Basal Circumference

For the same reasons as the variation in the mean heights of trees, the basal circumference has a relatively small interquartile range of 164.75 mm and a large overall range of 1171 mm. The mean circumference is 186.42 mm; measured 100 mm off the ground. Basal circumference variable was the best predictor of AGB, and therefore an ideal proxy of total carbon stored on an individual tree basis ($r^2 = 0.98$) . Marden et al. (2020) explained 92 to 99% of variability within measured AGB with root collar diameter (RCD).

Marden et al. (2020) did also find a relatively strong relationship between RCD and tree height ($r^2 = 0.82$). This is in contrast to basal circumference only explaining 54% of variation in height measurements on Nukuhau. However, the aforementioned study was in a plantation where nursery seedlings were transplanted in an orderly spatial pattern (3 metre x 3 metre spacing). This eliminated any influence that plant density (constant 1,111 trees/ha) and age had upon such a relationship, in stark contrast to regenerated bush where density and age of vegetation is largely dependent on environmental factors.

3.1.5 Density

The mean tree density across the sample sites is 10,750 trees/ha. The mean stem density is 12,450 stems/ha, measured as divergent stems 100 mm off the ground. This indicates that the average Manuka tree has 1.16 divergent stems at measuring height. Tree density has also been thought to be correlated to tree height. A higher density of trees has been thought to encourage the vertical growth of trees as they compete for available light (Boffa Miskell Ltd, 2017), possibly at the expense of trunk diameter. This is an example of intra-species competition. As tree density is fundamentally a spatial measurement, any relationship should be dealt with accordingly. Owing to the bimodal nature of the data, linear regression can neither confirm nor refute such a relationship.

Boffa Miskell Ltd (2017) suggest that self-seeded Manuka grows in densities of over 10,000 stems/ha, collaborating collected data. Natural thinning occurs within the stand over time as trees compete for light and other resources (Lambie et al., 2021). After several decades, as primary colonisers are about to be outcompeted by understorey species, stem density is likely to find an equilibrium as low as 1,600 stems/ha (Boffa Miskell Ltd, 2017; Scott et al., 2000). Therefore, the measured density may depend on the age of the trees located inside of the sample plot. Isolating sample sites that achieved over 40 tC/ha renders a mean circumference of 300 mm and mean tree density of 9,600 trees/ha. This infers that in older stands with lower tree densities, average circumference increases more than proportionally (Smale et al., 2008), leading to an increase in carbon storage. This assumes that older sample sites record the highest carbon storage.

3.2 Patitapu: Economic Modelling

The total productive area of Patitapu is 2,238 hectares. A hard hill country property (Beef and Lamb New Zealand, 2021), 53.2% of this is steeper than 26° (Table 7). As a result, large proportions of the farm are shown to be afforested when modelling the two highest slope classes (Models 2 and 3). The same methodology could be used on another property with equal proportions of slope class to illustrate how forestry and pastoral farming can coexist. As an alternative, slope classes could be further divided.

Initially, an optimisation equation was run over all base polygons created from DEM extrapolation to identify the most profitable land use on each land parcel. This was done assuming all enterprises are in a ‘status quo’ state. Secondly, the resultant economic models are assessed to determine which combination of enterprises is the most profitable. These take into account costs associated with land use change. It is hypothesised that any inequalities in profitability amongst land uses will be reflected in recent land use change to blanket planted exotic forestry. NPV and IRR is calculated so that potential returns can be realised in the context of the current economy. Finally, a sensitivity analysis is run to investigate the affect changing market prices will have upon profits and associated land use change. Economic models and aforementioned financial indicators can be accessed in the appended excel file titled ‘Model Economics.xlsx’.

Table 7: Proportion of productive land in each LRI slope class on Patitapu.

Slope Class	Slope Angle (degrees)	Area (ha)	Percentage of Productive Area (%)
A	0-3°	79	3.5%
B	4-7°	98	4.4%
C	8-15°	315	14.1%
D	16-20°	260	11.6%
E	21-25°	295	13.2%
F	26-35°	672	30.0%
G	>35°	518	23.2%

It is important to note that for all steps of modelling, an increased carbon sequestration rate for regenerating Manuka has been used as per carbon sequestration data collected on Nukuhau. This will lift revenue from NZUs linearly (+81.80%). If default carbon sequestration rates were used, revenue would be reduced by the same amount for native regeneration enterprises.

3.2.1 Optimisation Model

The optimisation model confirmed that at the current carbon price, ‘carbon farming’ of Pinus radiata is the most profitable enterprise over all polygons (Hale et al., 2015; Lambie et al., 2021). The earning of NZUs with regenerating bush was consistently less profitable when compared to the exotic species. At any one carbon price, the most profitable species is the faster growing species. This is a direct function of the default look-up tables provided for land in forestry under 100 ha in area (Ministry for Primary Industries, 2017). It was found that for pastoral farming to become the most profitable enterprise across the entirety of the farm, the carbon price would have to drop below \$9.40/tCO_2. Based on the current carbon price, this would represent a reduction of approximately 87.5% in price.

3.2.2 Economic Models

The two models that incorporate variable slope classes are shown in Figure 11. The control (status quo pastoral system) and Model 3 (whole farm afforestation) are not illustrated due to the extremity of both. Either the whole productive area of the property is in pasture or forestry, respectively. All models have different characteristics from an economic perspective. The initial costs and subsequent year on year profits are two factors which will explain the mechanics behind recent mass land conversion to 'carbon farmed' forestry (Fernandez & Daigneault, 2017).

Models will be referenced by the same naming convention used in Table 2. Models 1 and 2 use areas in the steepest and two steepest LRI slope classes to model forestry on, respectively. Model 3 assumes whole farm blanket planting of woody vegetation. Models are referenced as Pinus radiata or Manuka. A model designated 'ETS compliant' means that spatial limitations associated with the ETS are adhered to, and capable of earning NZUs under the scheme as it stands. Models termed 'Total area' includes polygons smaller than one hectare in size. These areas are included as if they were authorised under the scheme, something which does not currently occur. The purpose of this is to demonstrate how the ETS in its current form does not promote the integration of trees on farms, but mass land use change to exotic forestry (Schirmer & Bull, 2014).

The large differences in afforested area (Table 8) is a function of high proportions of land belonging to steep slope classes (Table 7). The same methodology could be applied to other production systems with different proportions of slope class. Results from the optimisation model suggest that even higher producing flat land could be more profitable under forestry than dairy production. Alternatively, further differentiation could be made amongst slope classes. An increasingly complex model could identify areas most suited to forestry (as an environmental mitigation) by not only taking into account slope but adding variables such as hydrologically active areas and changes in soil type (Nickless et al., 2017; Tozer et al., 2021).

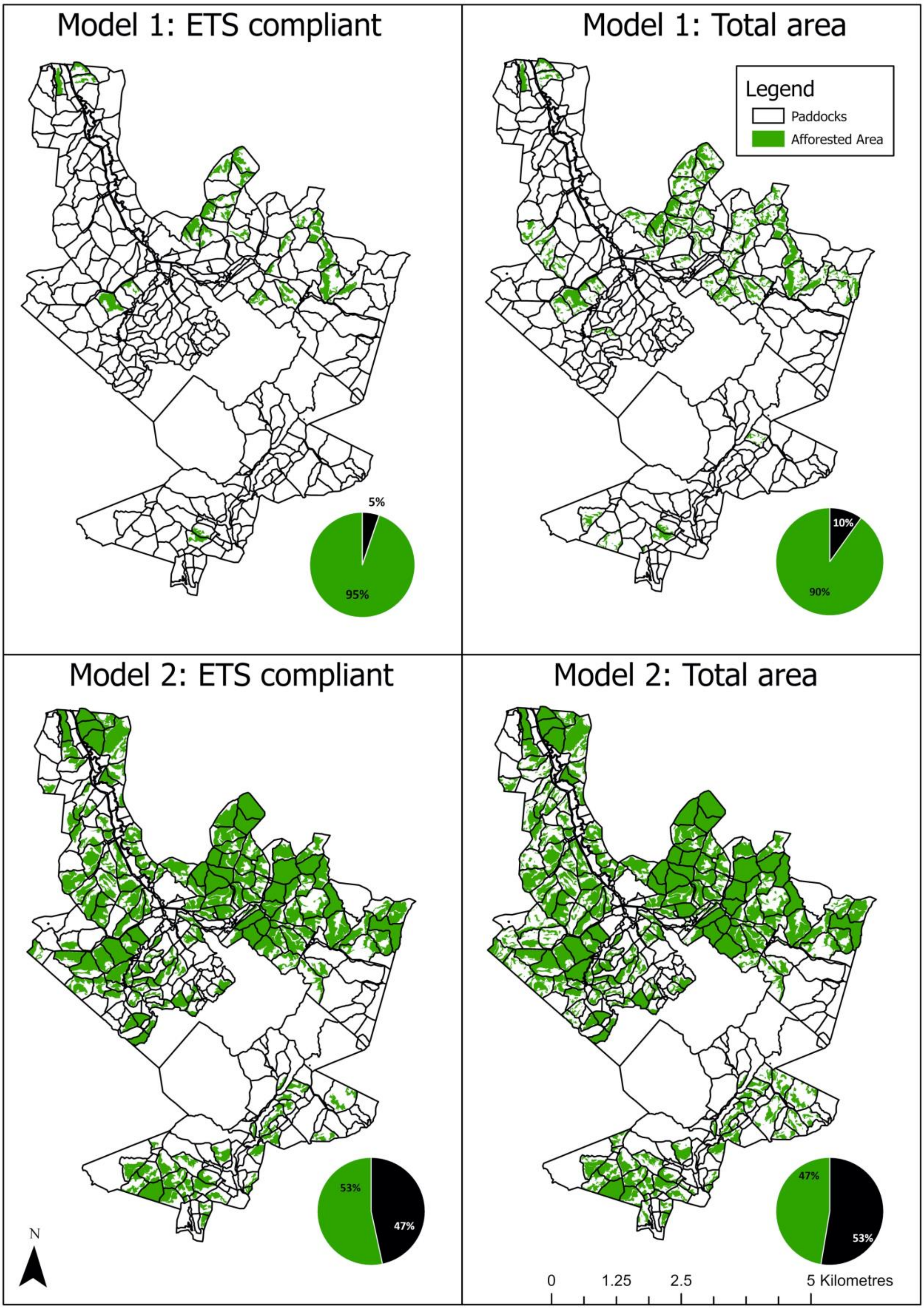

Figure 11: Resultant models showing area of land afforested (green) compared to land kept in pastoral production (black). The control (status quo) and Model 3 (whole farm afforested) are not shown due to the extremity of both.

3.2.2.1 Initial Costs

The model with the highest estimated initial cost is the total area of Pinus radiata over the two steepest slope classes (Model 2, Pinus radiata, Total area) (Table 8). At a cost of over $4.1 million in capital, a large proportion (69%) is expected to be used on fencing. A conservative estimate, the model does not consider existing fencing other than where whole paddocks are utilised. This is true across all models. In practice, existing fence lines would be an integral part of any planting plan. However, a conservative figure is ideal for budgeting purposes. In this case, the initial cost of land use change is used as the loaned monetary amount; repayable over a 10-year period with interest instalments. This is built into the ongoing costs.

Table 8: The area of productive land afforested and kept in pasture under different economic models. The initial cost associated with these models is also shown.

Model Number	Woody Species	Scenario (ETS/Total)	Pastoral Area (ha)	Afforested Area (ha)	Initial Cost ($)
Control	N/A	N/A	2238.3	0.0	$0
1	Pinus radiata	ETS compliant	2121.4	117.0	$584,357
1	Pinus radiata	Total area	2017.3	221.0	$1,587,970
1	Manuka	ETS compliant	2121.4	117.0	$492,991
1	Manuka	Total area	2017.3	221.0	$1,415,366
2	Pinus radiata	ETS compliant	1194.5	1043.8	$3,096,622
2	Pinus radiata	Total area	1060.3	1178.1	$4,146,514
2	Manuka	ETS compliant	1194.5	1043.8	$2,281,412
2	Manuka	Total area	1060.3	1178.1	$3,226,438
3	Pinus radiata	ETS compliant	0.0	2238.3	$2,442,031
3	Manuka	ETS compliant	0.0	2238.3	$693,886

All models compliant with the ETS cost significantly less than Total area equivalents. This is simply due to the hypothetical removal of spatial constraints imposed by the ETS. ‘Total area’ scenarios contain fenced land parcels under a hectare in size, on top of the polygons used by ‘ETS compliant’ models. The greater the proportion of smaller pockets of afforested land, the higher the associated costs (Schirmer & Bull, 2014). However, the ‘Total area’ models will be better aligned with the ‘right tree, right place, right purpose’ ideology from a spatial point of view (Beardmore et al., 2019).

Table 8 illustrates how under any planting regime, natural regeneration (Manuka) is cheaper to implement than pine forestry (Lambie et al., 2021). This is a function of plant and labour costs associated with exotics, whereas native species are left to colonise retired land. However, spraying costs are higher for natives due to the ability of exotic species to outcompete young seedlings (Table 4). This system assumes that conditions are suitable for natural regeneration to take place. Namely, viable dispersed seed must be present, competition with exotic species must be minimised, grazing pressure must be low coupled with optimal soil and climatic conditions (Burns et al., 2011). If conditions don't allow for successful establishment of native species, nursed seedlings would have to be purchased and planted, increasing costs significantly (Lambie et al., 2021; Morales et al., 2016). Generally, native seedlings cost more, and are planted at a higher density than their exotic counterparts.

3.2.2.2 Profitability

For display purposes, profitability of models is truncated to Years one, two, five, ten, 25 and 50 (Table 9). This generally encapsulates the early period where trees are establishing (Years 1-5) before rapid initial growth (Years 5-25). Carbon sequestration rates decline over time as trees reach maturity (Years 25+) (Lambie et al., 2021). This relationship appears to be sigmoidal in nature.

Under the current pastoral system, the farm would earn $242/ha/yr. Across all models, this is the highest income earner in Years one and two for two reasons. Large initial costs associated with afforestation (Dodd et al., 2008) mean large debt repayments are payable in the first ten years. Secondly, initial carbon sequestration/growth rates are minimal during the establishment years. Therefore, models that have greater proportions of pastoral area will be better off financially in Years one and two (Dodd et al., 2021). Income from the farming system largely offset any initial losses from the forestry enterprise.

The establishment of Manuka on the steepest of slopes (Model 1, Manuka) does not alter the bottom line of the entire system. As modelled forestry is a small percentage of total area; earnings from Manuka become unassuming on a per unit area basis. However, a $45/ha difference between (Model 1, Manuka, Total area) and the control in Year ten does amount to an additional $100,724 across the property. This is in contrast to Pines which earn $259/ha in

Year ten (Model 1, Pinus radiata, ETS compliant), totalling $38,051 more than the control. Hence, even at small scales it is profitable to incorporate carbon farming enterprises into existing sheep and beef systems.

Table 9: Profitability of different models at different years. Years 1, 2, 5, 10, 25 and 50 have been chosen for display purposes.

Model Number	Woody Species	Scenario (ETS/Total)	Year 1 ($/ha)	Year 2 ($/ha)	Year 5 ($/ha)	Year 10 ($/ha)	Year 25 ($/ha)	Year 50 ($/ha)
Control	N/A	N/A	$242	$242	$242	$242	$242	$242
1	Pinus radiata	ETS compliant	$196	$205	$345	$259	$357	$338
1	Pinus radiata	Total area	$120	$139	$408	$251	$460	$423
1	Manuka	ETS compliant	$209	$210	$233	$278	$313	$248
1	Manuka	Total area	$145	$149	$195	$287	$377	$253
2	Pinus radiata	ETS compliant	($52)	$22	$1,258	$460	$1,236	$1,062
2	Pinus radiata	Total area	($137)	($58)	$1,348	$457	$1,363	$1,166
2	Manuka	ETS compliant	$65	$71	$254	$628	$844	$259
2	Manuka	Total area	($6)	$2	$215	$646	$921	$261
3	Pinus radiata	ETS compliant	($237)	($99)	$2,527	$759	$2,225	$1,850
3	Manuka	ETS compliant	$12	$14	$374	$1,120	$1,384	$130

At the current carbon price, the larger the area of modelled forestry, the larger the property's income stream. The highest profit across any model is in Year seven of (Model 3, Pinus radiata, ETS compliant), with net income of $2,915/ha (see Appendix: Model Economics.xlsx). Compared to the control this equates to almost $6 million in extra earnings for the year. Initial losses also peak in the same model in Year one with a $237/ha loss. Year two also incurs a loss of $99/ha. Net losses in the early stages are due to the large investment required for blanket land use change to exotic species and initial low carbon sequestration rates. The only models to have two years of successive losses are where large areas of Pines are planted (Model 2, Pinus radiata, Total area and Model 3, Pinus radiata, ETS compliant). This is an example of a high input, high output system, costing $4,146,514 and $2,442,031 respectively (Table 8). For most farmers, this would require external funding to achieve (Flack et al., 2022). This leads to the purchase of whole properties by large corporations who subsequently blanket plant exotic species (Schirmer & Bull, 2014). Multi-generation family farms cannot afford to undergo such a transformation (Beardmore et al., 2019; Flack et al., 2022).

Models that contain large areas of Pinus radiata (Model 2, Pinus radiata and Model 3, Pinus radiata) have an initial peak in profitability occurring in Year seven (see Appendix: Model Economic.xlsx). This demonstrates the fast initial growth of the exotic species post establishment; Pinus radiata is known to be one that outcompetes natives (Lambie et al., 2021). A sharp decline in Year nine is attributed to the inclusion of thinning by MPI's look-up tables (Ministry for Primary Industries, 2017) (see Appendix: Model Economics.xlsx). Therefore, profitability is relatively minimal in Year ten (Table 9). It is unlikely that trees on such hard hill country will be thinned in a dedicated 'carbon farming' enterprise (Manley & Maclaren, 2012) where harvesting of trees is not planned. However, where areas of 100 hectares are planted, profitability is based off field assessments (Ministry for Primary Industries, 2018), rectifying any errors in carbon accounting. This is not the case for smaller woodlots, where default look-up tables are used (Ministry for Primary Industries, 2017).

In contrast, models that include Manuka do not have thinning built into MPI look-up tables. Slow initial growth rates compared with Pines means that profitability of models is small until Years four to seven (see Appendix: Model Economics.xlsx). Only then do earnings become comparable to the existing pastoral system (control). This difference would be exacerbated if carbon sequestration rates were not scaled by data collected on Nukuhau, indicating the importance of this work. The greater the proportion of forestry, the faster this profitability increases thereafter, although this is coupled with low initial income. This is due to linearly decreasing proportions of the existing pastoral system.

In conclusion, the larger the proportion of pastoral production, the more stable the operation. Costs of any planting or retirement of land are recovered by the cash flow of the remaining sheep and beef operation. Pines cost relatively more to establish but offer disproportionate returns compared to pastoral farming and native regeneration in the long term (Lambie et al., 2021). Blanket planting the entire property in Pinus radiata is the most profitable model by some margin. This is due to (a) maximising area in the most profitable enterprise and (b) minimising costs by eliminating the need for fencing.

3.2.3 Forecasting Investment Returns

By considering two different inflation rates, the forecast return on investment can be expressed in current monetary terms. Discount rates of 3% and 6% are used to calculate NPV in Years ten and 25. 3% is used as it represents an approximate long-term average, whilst 6% is better aligned with current market conditions. This represents the opportunity cost of investment (Paul et al., 2013). IRR gives an indication of monetary returns when NPV is equal to zero.

3.2.3.1 Net Present Value

NPV for Year's ten and 25 with a 3% discount rate is shown in Table 10. Table 11 shows the equivalent with a 6% discount rate. As the control does not require capital investment, such measurements are not applicable. These figures only take the 'carbon farming' enterprise into account. As the pastoral system is considered 'status quo', profits from this enterprise are removed from analysis.

The model with the highest NPV in all situations by a large margin is blanket planted Pines (Model 3, Pinus radiata, ETS compliant) (BakerAg, 2019; Lambie et al., 2021). Ten years after planting, the investment of $2.44 million dollars is worth $21.35 million dollars with a discount rate of 6%. This NPV is almost three times as high as the next gross NPV value (Model 3, Manuka, ETS compliant). However, the initial costs of this model are less at $0.69 million dollars. As initial costs are scaled similarly to Year ten NPV, we can expect similar rates of return between both models (Table 12).

In all cases, the NPV for all (Total area) models is lower than the NPV for all (ETS compliant) models of the same vegetation type in Year ten. As 'Total area' models encapsulate land parcels of smaller area, the cost of fencing rises disproportionally to the total afforested area (Fernandez & Daigneault, 2016). For Pinus radiata, this relationship inverts in Year 25, as the carbon sequestration over a larger total area maximises returns. In Year 25, Manuka shares relatively similar NPV's. This is a function of the returns from Manuka carbon sequestration being relatively small. The margins between models is smaller, meaning differences in NPV are not as volatile as Pines (Lambie et al., 2021). Higher initial costs associated with (Total area) models will mean these have a lower rate of return (Table 12) (Fernandez & Daigneault, 2017).

Tables 10 and 11 effectively illustrate that Manuka may not provide any real returns until after Year ten. Under a 6% discount rate, (Model 1, Manuka, Total area) still models a negative NPV in Year 25, the only model to do so. This is due to the higher fencing costs associated with 'Total area' models (Fernandez & Daigneault, 2017; Schirmer & Bull, 2014) coupled with the relatively small earnings of Manuka. This illustrates how the targeted retirement of land is not financially viable to farmers (Flack et al., 2022). For any land use change to be effective on a national scale, economic incentives must exist (Beardmore et al., 2019). Such enterprises only appear to be profitable when planted on mass, contrary to the goals of the Government.

Table 10: Net Present Value of different economic models with a discount rate of 3%. Years 10 and 25 have been chosen as they respectively represent medium and long-term return periods.

Model Number	Woody Species	Scenario (ETS/Total)	Initial Cost ($)	Year 10 NPV (3%)	Year 25 NPV (3%)
1	Pinus radiata	ETS compliant	$584,357	$170,138	$2,586,202
1	Pinus radiata	Total area	$1,587,970	($687,525)	$3,876,778
1	Manuka	ETS compliant	$492,991	($546,598)	$1,033,668
1	Manuka	Total area	$1,415,366	($2,041,547)	$943,811
2	Pinus radiata	ETS compliant	$3,096,622	$5,918,266	$27,475,464
2	Pinus radiata	Total area	$4,146,514	$5,266,180	$29,596,430
2	Manuka	ETS compliant	$2,281,412	($463,656)	$13,636,183
2	Manuka	Total area	$3,226,438	($1,882,876)	$14,030,723
3	Pinus radiata	ETS compliant	$2,442,031	$21,351,254	$67,578,746
3	Manuka	ETS compliant	$693,886	$7,757,069	$37,992,917

At a discount rate of 3% in Year 25, (Model 1, Manuka, Total Area) has an NPV less than the initial cost. The initial investment worth $1.42 million reduces to $943,811, meaning a loss of $471,555 is expected when inflation is considered. When the discount rate is increased to 6%, the list expands to include (Model 1, Manuka, ETS compliant). In this model, it is expected that a loss of $1.60 million dollars would occur in today's terms. This information actively discourages farmers to integrate native vegetation into their systems.

Table 11: Net Present Value of different economic models with a discount rate of 6%. Years 10 and 25 have been chosen as they respectively represent medium and long-term return periods.

Model Number	Woody Species	Scenario (ETS/Total)	Initial Cost ($)	Year 10 NPV (6%)	Year 25 NPV (6%)
1	Pinus radiata	ETS compliant	$584,357	$23,957	$1,446,241
1	Pinus radiata	Total area	$1,587,970	($868,011)	$1,818,894
1	Manuka	ETS compliant	$492,991	($551,432)	$383,472
1	Manuka	Total area	$1,415,366	($1,955,006)	($188,834)
2	Pinus radiata	ETS compliant	$3,096,622	$4,196,694	$16,886,944
2	Pinus radiata	Total area	$4,146,514	$3,458,014	$17,780,700
2	Manuka	ETS compliant	$2,281,412	($925,266)	$7,416,361
2	Manuka	Total area	$3,226,438	($2,274,951)	$7,139,717
3	Pinus radiata	ETS compliant	$2,442,031	$16,839,713	$44,052,826
3	Manuka	ETS compliant	$693,886	$5,937,357	$23,825,233

3.2.3.2 Internal Rate of Return

The higher the proportion of pastoral land in a model, the higher the calculated IRR (Table 12). As less land requires fencing under ETS compliant models, the initial cost is less. Hence, returns are large compared to this preliminary outlay of funds. Total area models do earn more over time however they do not share a linear relationship with the original investment. The fencing of smaller areas exponentially increases the perimeter of fencing required as the total afforested area increases (Fernandez & Daigneault, 2017). This is the law of diminishing returns.

Manuka consistently reports lower IRR than Pine (Table 12). This is a direct function of the relatively slow growth of natives compared to exotic species. As modelled figures take into account the increased carbon sequestration rates provided by the pilot study, current earnings under the ETS would be less. Natural inflation would likely outcompete this model as shown in Tables 10 and 11. For (Model 1, Manuka, Total area), arguably the model most aligned with government policy, this would mean an expected IRR below average inflation levels. In contrast, (Model 1, Pinus radiata, Total area) provides the lowest IRR of all Pine models at 12% (Table 12).

Table 12: The Internal Rate of Return of different economic models.

Model Number	Woody Species	Scenario (ETS/Total)	Initial Cost ($)	IRR (%)
1	Pinus radiata	ETS compliant	$584,357	17%
1	Pinus radiata	Total area	$1,587,970	12%
1	Manuka	ETS compliant	$492,991	9%
1	Manuka	Total area	$1,415,366	5%
2	Pinus radiata	ETS compliant	$3,096,622	26%
2	Pinus radiata	Total area	$4,146,514	23%
2	Manuka	ETS compliant	$2,281,412	16%
2	Manuka	Total area	$3,226,438	14%
3	Pinus radiata	ETS compliant	$2,442,031	50%
3	Manuka	ETS compliant	$693,886	53%

Demonstrating the monetary attractiveness of blanket planting, (Model 3, Pinus radiata) offers an IRR of 50% (Lambie et al., 2021; Spiekermann et al., 2021). Model 3, Manuka has an IRR of 53% due to the relatively small initial cost. This would effectively involve removing all stock units from the property and letting it naturally revert to indigenous bush (Tozer et al., 2021). If retirement of the land from pastoral uses was decided on as an alternative income stream, it is likely that the enterprise which produced the highest NPV would be decided on (Model 3, Pinus radiata) (Table 11). If cash up front is preferred (i.e. farmer retirement), the property would usually be sold.

3.2.4 Sensitivity Analysis

A sensitivity analysis is conducted for every model and year reported in Table 9. The goal is to determine the effect that varying carbon and red meat prices have upon the overall profitability of models. A large range of carbon prices is utilised due to the uncertainty surrounding the market (\$25-\$150/tCO_2). Profitability per stock unit is also varied by ± 60%; reflecting peaks and troughs of the previous ten years.

Due to no inclusion of forestry in the control, changes in carbon price do not affect the bottom line. An increase of 60% to stock unit profitability (\$39.56/SU) yields \$387/ha/yr, whilst a decrease of the same value (\$9.89/SU) brings in \$97/ha/yr. Figures between these values are fairly typical of a hard hill country pastoral farm operation, depending on variables such as market prices, climate and on farm production (lambing percentages).

Conversely, Model 3 contains no pastoral farming. Therefore, farm margins are not dictated by changes in the red meat market. What can be seen as a worst-case scenario for the carbon sector with a fall back to \$25/tCO_2, yields \$675/ha for 25-year-old Pines and \$448/ha for 25-year-old Manuka. Both are significantly higher than the existing systems profitability (\$97-\$387/ha). Conversely, if the carbon price was to double again to \$150/tCO_2 as some commentators suggest (New Zealand Productivity Commission, 2018), \$4,550/ha for Pines or \$2,789 for Manuka could be made in the same year. This constitutes a respective 18.8 and 11.5 times more than the potential earnings of the current control scenario. If the carbon price continues to steadily increase, wholesale conversion to blanket planted exotics may rise exponentially.

Under heavily afforested Pine models (Model 2, Pinus radiata and Model 3, Pinus radiata), an increase in carbon price does not adequately compensate for associated high initial costs and low initial carbon sequestration (Schirmer & Bull, 2014). Even at \$150/tCO_2, losses between \$200/ha (Model 3, Pinus radiata, ETS compliant) and \$34/ha (Model 2, Pinus radiata, ETS compliant) are recorded in Year 1 assuming the profitability of any pastoral area remains constant. In simpler terms, such large land conversion would still incur substantial losses despite vastly increased carbon prices.

This is in contrast to regenerated Manuka which records profits between \$37/ha (Model 2, Manuka, Total area) and \$103/ha (Model 2, Manuka, ETS compliant) given the same variables. Early carbon sequestration of Manuka (as measured on Nukuhau) effectively breaks even with costs of natural regeneration when the carbon price is minimal. In this situation, (Model 2, Manuka, ETS compliant) earns more than blanket grown Manuka

(Model 3, Manuka, ETS compliant) due to the pastoral system being more profitable during establishment years. (Model 2, Manuka, Total area) earns less overall with less pastoral area. Model 3, Manuka, ETS compliant sits between these figures with $94/ha, boosted since no additional fencing is required for this model. However, earnings increase after establishment (Table 9), indicating that Manuka production is financially viable on hard hill country. It offers an alternative to Pines if large capital investments are not workable for the business.

Not under any combination of market prices does (Model 1, ETS compliant) for either Pines or Manuka fall below $50/ha in Year 1. With 104.1 hectares in forestry, (Model 1, Pinus radiata, Total area) falls to a loss of $26/ha under the worst possible combination of market prices (Table 8). These models contain forestry as 5% and 10% of the total productive area of the property respectively (Figure 11). Due to the relatively small proportions of forestry in these models, earnings from Manuka and Pines are comparable (i.e. Model 1, Pinus radiata and Model 1, Manuka) when looked at holistically as a function of whole farm profitability. Due to smaller initial investment, (Model 1, ETS compliant) is more profitable than (Model 1, Total area) at any combination of market prices. This relationship inverts when carbon sequestration rates increase and debt repayments finish.

4 Discussion

4.1 Nukuhau: Manuka Sampling

Collected Manuka carbon storage data greatly differs from that described by MPI (+81.80%). This is somewhat aligned with the findings of Dodd et al. (2021) who predict that default look-up tables underrepresent carbon storage by 24% for established indigenous forest (over 50 years old) and by up to 270% for planted indigenous species. This could be due to varying establishment rates of woody vegetation. An argument is made that farmers, wishing to maximise profitability will not leave land unforested or ungrazed. The bimodal data seen between sample sites is investigated, with the conclusion that differences are due to varying levels of light interception. Finally, an argument is made for the review of Manuka carbon sequestration rates of which drive the profitability of regenerating land through the ETS (Dodd et al., 2021; Lambie et al., 2021).

4.1.1 Establishment of Target Species through Natural Regeneration

When measuring Manuka carbon storage, sample sites where no Manuka existed (Figure 6) were removed from the analysis. This provides an explanation as to why estimated carbon storage is higher than that reported in MPI look-up tables (Ministry for Primary Industries, 2017). It is not unlikely that gaps could naturally occur in retired, regenerating bush. Initial establishment issues could be due to a variety of factors such as lack of viable seed, competition amongst species, animal grazing, the soil environment, climate or a mixture of factors (Boffa Miskell Ltd, 2017; Burns et al., 2011; Carver & Kerr, 2017; Douglas et al., 2007).

In contrast, Funk et al. (2014) suggest that the age of naturally regenerated forestry in any one study area has a sigmoidal relationship from year zero through to year ten, when total canopy cover was expected. This is similar to estimates provided by Marden et al. (2020), who estimates total canopy cover to be achieved between years 6.5 and nine. This provides evidence to suggest, that carbon storage estimates from the Ministry for Primary Industries (2017) are not viable in the longer term when complete canopy cover is achieved. Continued establishment of primary colonisers will eventually vegetate the entire land available, whether by Manuka or other species.

4.1.1.1 Seed Viability and Germination

In the first instance, seed of the target species must be viable. If natural regeneration is the preferred establishment technique, a close by seed source of considerable quantity must exist (Carver & Kerr, 2017; Davis et al., 2013a), as well as an effective dispersal mechanism. In the case of Patitapu this is achieved through the large sections of covenanted indigenous bush as well as sporadic stands of native vegetation. Wind acts as the dispersal instrument for fine Manuka seed (Funk et al., 2014; Norton et al., 2020). Seed must not be perished. The ability of a species to naturally regenerate may depend on the ability of viable seed to break 'seed dormancy' (Douglas et al., 2007). Viable seed does not translate to successful germination. Manuka has historically relied on fire to break the seed capsule, releasing seed (Battersby et al., 2017; Boffa Miskell Ltd, 2017). Exposure to higher temperatures and sufficient light has proved sufficient (Mackay et al., 2002). Other species rely on stratification, introduction of light, physical abrasion or chemical interactions to break seed dormancy (Douglas et al., 2007).

4.1.1.2 Interspecies Competition

Target species must quickly establish themselves to outcompete other species, namely exotic grasses (Douglas et al., 2007; Lambie et al., 2021). Weeds initially shade forestry species; establishment often requires chemical control of competing species (Carver & Kerr, 2017). If the required control is not achieved, Manuka seedlings will simply not be able to survive. Once established, competition for light encourages fast vertical growth. As trees mature, the competition for light becomes less prevalent due to relative canopy heights. This is replaced by competition for nutrients and water as roots expand downward and outward (Scott et al., 2000). If tree density is high, individual trees may outcompete their counterparts, leading to natural thinning.

4.1.1.3 Grazing Pressure

To establish themselves, seedlings must not be grazed by ruminants or pest species such as hares or deer (Carver & Kerr, 2017; Norton et al., 2020; Smale et al., 1995). Practically, protection from the former can be achieved through fencing for stock exclusion (Bassett et al., 2005; Boffa Miskell Ltd, 2017; Burns et al., 2011; Morales et al., 2016) or decreasing stocking pressure (Funk et al., 2014). The control of pest species is difficult, and often expensive (Morales et al., 2016). The use of bait is common for smaller pest species, whilst larger targets require culling (Husheer et al., 2005; Wright et al., 2012). Recent studies provide evidence that Manuka is relatively unpalatable to stock (Boffa Miskell Ltd, 2017), and is therefore likely to establish where other species will not.

4.1.1.4 Soil Factors

The ability of Manuka to establish on areas of low fertility (Olsen P < 10) is well documented (Battersby et al., 2017; Douglas et al., 2007; Reis et al., 2017; Scott et al., 2000; Stephens et al., 2005; Wellington et al., 2022). Boffa Miskell Ltd (2017) even detail the ability of Manuka to germinate in rock crevices, where topsoil does not exist. In saying this, although largely unquantified, soil fertility is likely to have an impact upon the initial growth rates of regenerated species. Increased nutrient availability leads to higher growth rates given all other conditions are optimised (Boffa Miskell Ltd, 2017; Nickless et al., 2017). It is the ability of Manuka to flourish in lower fertility areas and outcompete other species that leads to it becoming the prevailing species. A wide range of soil pH values are tolerated by Manuka (Boffa Miskell Ltd, 2017).

The abundance of Mycorrhizae fungi in soils is beneficial for Manuka growth. Mycorrhizae fungi forms symbiotic relationships with Manuka roots (Stephens et al., 2005). Effectively increasing root surface area, trees receive greater quantities of nutrients and water (Davis et al., 2013b). In exchange, the plant provides an energy source to the fungi in the form of carbohydrates. Areas previously in pasture with a natural affinity for Mycorrhizae will provide benefits for P uptake and associated growth of young trees (Douglas et al., 2007; Nickless et al., 2017; Stephens et al., 2005). Areas of low fungal inoculation may result in poor establishment rates. If found to be an issue, fungi can be manually applied to the soil (Boffa Miskell Ltd, 2017; Davis et al., 2013a).

There is evidence to suggest that Manuka establishes itself more successfully on compacted soil (Bassett et al., 2005). Douglas et al. (2007) attribute this to the relative fineness of Manuka roots compared to other woody species. This could have implications on hill country, where the likes of mass movements and damage from stock movements have left the soil comparatively compact. Manuka is known to be amongst the first few species to establish themselves on slip scars (Ross et al., 2009), where relatively compact unfertile subsoil predominates. However, data provided by (Marden et al., 2020) demonstrates that whilst this may be true, tree vigour compared to trees grown on stable sites is lower. This impacts growth rates and associated carbon sequestration.

4.1.1.5 Climatic Factors

Manuka seedlings do not survive in a shaded environment. Frosts have also been known to kill young trees, however tolerance increases with maturity. Trees are reasonably tolerant of wind. Early growth of young trees has also been found to be positively affected by increased temperature and rainfall (Boffa Miskell Ltd, 2017). Therefore, it is thought that higher initial growth rates are found on North facing slopes, due to increased light and temperature (Lambie et al., 2021). South facing slopes may suffer from relatively poor germination rates.

4.1.2 Economic Modelling

For the purposes of the models subsequently run, it is assumed that canopy cover is at or near 100%, to ensure optimal utilisation of the land. If this does not occur, it would be likely that the farmer supplements the natural regeneration process with the planting of nursery seedlings. If sample sites found not to contain Manuka are considered to have no carbon storage capacity, the mean carbon storage becomes 20.82 tC/ha (76.28 tCO2/ha). This is still 26.50% more than estimated with MPI lookup tables for regenerating native bush of the same age (Ministry for Primary Industries, 2017). This further brings results into line with Dodd et al. (2021).

4.1.3 Variation in Measurements

A bimodal distribution in total carbon storage amongst sample sites was found. Potential bias could of also been introduced to the study via the limitations of equipment and experiment design (Pedersen & Skovsgaard, 2009). It must be noted that the pilot study's goal was to provide an average carbon sequestration rate for subsequent economic modelling of Manuka. It was not designed to disprove existing datasets nor has the scale to do so. However, some discrepancies are raised which may warrant further work.

4.1.3.1 Bimodal Distribution

Pedersen and Skovsgaard (2009) propose that there are four major sources of variance in predicting tree volume from allometric equations. These include (a) systematic measurement error, (b) an uncertain least square regression estimate, (c) use of an inferior allometric equation and (d) sampling design. Covered below, potential measurement error may have been introduced, however is unlikely to be the cause of a bimodal distribution. Regression has not been able to be utilised to predict carbon storage on a per unit area basis due to data distribution. No evidence has been found by the author that refutes the accuracy of allometric equations used by Beets et al. (2014). Therefore, the cause of the bimodal distribution is likely to be attributable to study design. In this case, the sampling of two populations.

The bimodal distribution of carbon storage is likely due to relative exposure to light. Under a recently closed canopy of exotic species, Manuka growth rates are stunted, before eventually dying. This is in contrast with Manuka stands of a comparable age growing in high densities exhibiting similar vertical growth rates (Boffa Miskell Ltd, 2017). Historic imagery reveals that all Manuka sample sites which had measured carbon stocks of over 40 tC/ha all were subject to increased light levels at some point in their life. Therefore, vertical and horizontal (trunk expansion) growth is not limited, leading to increased carbon storage. The two sample sites with the largest measured carbon stock did not compete with any exotic species from establishment until measurement. As a result, the data shows two peaks (Figure 7). One from Manuka grown under the canopy of exotic species and one from Manuka grown with partial/full access to sunlight. The former appears to have a gamma distribution (Beets et al., 2014) whilst too few sample sites exist of the latter to determine if it is normally distributed. As available light levels were not measured, sub-groups cannot be extrapolated for linear regression modelling.

As Manuka seedlings do not tolerate shade (Boffa Miskell Ltd, 2017), it can be assumed that all measured trees growing within an exotic canopy are of similar age to the exotic tree crop. In contrast, areas with full light interception would likely be composed of trees of different ages. This is a function of the sigmoidal relationship of tree age over the time it takes to achieve a full canopy cover (Funk et al., 2014). This is in contrast with Battersby et al. (2017) who state that all trees in a Manuka stand are likely of the same age, however this is probably when compared to a climax forest community, where differences in age between native trees can be centuries. This somewhat explains the difference in mean tree circumference of Manuka, with the range in basal diameter decreasing by 36% to 131 mm when sites measuring over 40tC/ha are removed from the analysis (secondary data peak).

The measuring of naturally regenerated Manuka carbon storage was conducted under an exotic canopy because a physical date could be assigned to land retirement, and subsequently, an approximate age of regenerated bush could be inferred. In this case, the planting of an exotic canopy provided this information. The alternative method to determine age is stem analysis (Beets et al., 2014), however the landowner did not want destructive techniques used. Most regenerated Manuka on retired farmland would not be under an exotic canopy. Although sufficient data was not collected to make inferences, it is likely that mean carbon sequestration of Manuka in full light conditions is significantly larger than MPI predict. This supports the notion that that MPI look-up tables underestimate carbon sequestration of Manuka. This warrants longer term studies, that may include the quantification of light interception by Manuka.

4.1.3.2 Potential Sources of Bias

Bias may have been introduced because of inaccessible sample site locations. As displayed in Figure 6, areas unable to be accessed are largely concentrated on a North facing hill face. Areas were inaccessible due to (a) higher densities of Manuka and undergrowth and (b), steepness. As tree density has been shown to show little correlation to carbon storage on an individual tree (r^2 = 0.12) or per unit area (r^2 = 0.02) basis, it is likely any introduced bias is minimal. In contrast, slope has been shown to explain a higher quantity of variation with an r^2 value of 0.47. Although a potential source of significant bias, high slope angles preventing foot traffic were not necessarily located at the sample sites themselves but at the extremities of hill faces (adjacent to waterways and ridgelines). It has been noted that due to the nature of the study, time restraints prevented reconnaissance of sample sites before the trial site visit.

The TruPulse 200L rangefinder calculated tree height to the nearest metre. Considering basal diameter is measured to the nearest millimetre, this could have caused substantial variation in modelled carbon storage. The value and range of this potential bias is unknown (Sullivan et al., 2018). For the purposes of the study, the ability to non-detsructively measure biomass was essential. With the equipment available to the author at the time, the rangefinder provided the best combination of accuracy, cost, and time efficiency. The use of rangefinders to measure tree height is common within forestry science, however their application is usually reserved for taller species (Sullivan et al., 2018).

Manuka trees over a height of one metre have equal statistical weighting. In terms of tree height, basal circumference and tree density, calculation of means could have been unintentionally biased. This was done to better understand the overall carbon storage capabilities of regenerated native bush (Searle & Chen, 2017). However, this could impact the statistical relationship between the three variables and carbon storage, with younger trees decreasing the mean from each sample location. Long term, these trees may be outcompeted by taller, deeper rooted trees in terms of light, water, and nutrients (Lambie et al., 2021). Their inclusion into the statistical analysis may overestimate their importance in predicting other variables, including the statistical relationship between those in question.

4.1.4 Evidence Prompting Review of Native Regeneration under MPI Look-up Tables

A limiting number of sample sites coupled with bimodal data distribution prevents any meaningful statistical relationships between tree measurements and carbon storage being found. However, results do suggest a substantial difference to those reported in MPI look-up tables (Dodd et al., 2021). West et al. (2020) state that these reference tables are conservative in nature. This is significant as it is these tables which predetermine earnings from retired land under 100 hectares (Ministry for Primary Industries, 2017) which is allowed to either naturally regenerate or is planted with seedlings. By underpredicting the carbon storage capacity of primary colonisers of retired land, the resultant profitability gap between Pines and Manuka grows larger (Lambie et al., 2021). As the opportunity cost of planting natives grows, property owners will revert to the planting of exotic species (Norton et al., 2020).

This may mean that the establishment of long-term trials in a variety of environments are required (Dodd et al., 2021). Region, topographical features, local climate, light interception, soil fertility and plant cultivar are all variables which should be considered. Due to the amount of data associated with Pines, the same look-up tables are able to differentiate expected carbon stocks depending on the region they are located in (Ministry for Primary Industries, 2017). This should be the overall goal; to form a similar database for different native tree species detailing regional differences in carbon sequestration over time (Carver & Kerr, 2017; Lambie et al., 2021).

Furthermore, this study fails to take the potentially significant carbon storage which is held in roots (Marden et al., 2020) and other pools (Lambie & Dando, 2019). Taking these pools into account would further exacerbate the differences between MPI lookup tables and measured data (Dodd et al., 2021). A Manuka root:shoot ratio of 0.18 (Marden et al., 2020; Scott et al., 2000) would increase carbon storage proportionally, assuming similar carbon composition of roots. If applied to the mean carbon storage of sample plots, carbon storage could increase to a total of 35.31 tC/ha (129.39 tCO_2/ha). This would represent a value 114.57% higher than currently stipulated, meaning regenerating forestry owners could be being undercompensated by over 50% for the carbon sequestration capabilities of their forest. Referencing (Scott et al., 2000), Stephens et al. (2005) state “carbon accumulation by *Leptospermum scoparium* is rapid and similar to that of plantation forestry”. Whilst this may only be true in the short-term proceeding germination, payment for carbon sequestration services must be suitably aligned with alternatives (e.g. production forestry) to incentivise the retirement and subsequent natural regeneration of land unsuitable for pastoral grazing.

A study conducted by Lambie and Dando (2019) estimated carbon inputs from leaf litter from Manuka and Kanuka (*Kunzea ericoides*) in the Southern Wairarapa to be 2,448 ± 127 kgC/ha/yr, however this value was largely dependent on location and environment. Degradation of organic matter will reduce the quantity of carbon held in leaf litter over time; as this is a short-term pool. The initial decomposition of organic matter by soil organisms generates carbon dioxide as a result of respiration (Brady & Weil, 2007). As a result, the soil’s carbon dioxide emissions will increase with increased rates of decomposition. This process depletes the soil of carbon, but is usually found in equilibrium with the return of dead plant and animal material to the soil. This is due to the population dynamics of soil organisms; which is limited by the quantity of available feed (Ettema & Wardle, 2002; Fraser et al., 2012). The effect regenerating bush has upon soil carbon levels compared to previously pastoral land should be investigated as this potentially represents a relatively stable form of carbon storage as a result of land use change (Minasny et al., 2017).

Depending on the environment, other native species that act as primary colonisers may be able to establish easily via natural regeneration or direct seeding (Douglas et al., 2007). Rogers and Walker (2005) studied the distribution of native flora, finding substantial variation between species. Comparison of the carbon sequestration capabilities of different species in areas of regional importance would be beneficial. There is also evidence suggesting that regionally specific cultivars of native species have naturally evolved (Rogers & Walker, 2005; Stephens et al., 2005), with different variations being better suited in different environments (Koot et al., 2022). Genetic diversity is the result of natural selection, where DNA changes over time to best suit the conditions (Boffa Miskell Ltd, 2017; Stephens et al., 2005). Therefore, different Manuka cultivars outperform others in different conditions such as higher altitudes or on poorly drained soil (Stephens et al., 2005). It is for this reason that locally sourced seed should be used in restoration projects (Douglas et al., 2007). Plant breeding has also produced different cultivars (Boffa Miskell Ltd, 2017); however such practise is usually targeted towards higher earning enterprises such as honey production. Comparing the carbon sequestration of these different varieties may not yet be a priority, but is something worth investigating in the future as ecological succession promotes the growth of a diverse range of species (Funk et al., 2014).

4.1.5 Incentivising Native Reversion as an Alternative to Exotic Forestry

To incentivise the planting of native species over Pines, as a minimum standard, it is essential that landowners get remunerated according to the carbon sequestration that occurs on their land. Norton et al. (2020) propose that for maximum impact, landowners should be incentivised and rewarded for planting natives (Beardmore et al., 2019). Additionally, they suggest removing the one hectare minimum area under the ETS, as small pockets of native vegetation are significant in terms of biodiversity management (Smale et al., 2008). Advantages of native species is that they have comparatively longer lifespans than Pines, and also provide an optimal environment for the establishment of successional species (Boffa Miskell Ltd, 2017). Therefore, carbon sequestration will be a long-term income earner compared to harvested exotic species.

Some Government schemes have encouraged the retirement and subsequent planting of land unsuitable for pastoral production. The Afforestation Grant Scheme (AGS, 2015-2020) and the Permanent Forestry Sink Initiative (PFSI, 2006-2018) (Funk et al., 2014; Marden et al., 2020) which have been subsequently replaced by the One Billion Trees programme and updates to the ETS are examples. Cumulatively, these schemes have been estimated to target 1.45 million hectares of marginal agricultural land or short rotation forestry for retirement into indigenous forest (Marden et al., 2020).

The One Billion trees scheme has provided monetary support for tree planting. As of 30th June 2021 the initiative had directly funded 48,265,291 trees, two thirds of which have been natives (Ministry for Primary Industries, 2021c). However, funding is dependent on spatial limitations, similar to the ETS. A one-hectare minimum area, 30 metre average width and minimum density levels all have to be satisfied to be eligible. This immediately detracts from the 'right tree, right place, right purpose' policy adopted by the Government. As a result, forestry and pastoral farming have become very separate enterprises. Whilst both may occur on a single property, very rarely are natives integrated into the landscape (e.g. on CSAs). It is agreed that to achieve successful outcomes for freshwater and greenhouse gas legislation, farmers must be rewarded and incentivised to integrate native vegetation into their systems (Beardmore et al., 2019; Norton et al., 2020). This differs to the directly funded blanket planting of exotic species. The following economic study analyses how integration of regenerating bush could be implemented into the landscape, detailing the effect these spatial limitations have upon farm profitability and associated environmental outcomes.

4.2 Patitapu: Economic Modelling

Data demonstrates that fundamentally, the ETS challenges the governments 'right tree, right place, right purpose' philosophy. Even if Manuka carbon sequestration rates are refined, the large potential returns of blanket planted exotic forestry (Lambie et al., 2021; Leining et al., 2020) encourages mass conversion of land. Sensitivity analysis shows that even in different market conditions, this would likely still occur. However, supplementary incomes that run parallel to 'carbon farming' enterprises can reduce the impact of such volatility. An argument is also made for silvopastoral systems, where forestry and ruminant production are spatially aligned (Mackay-Smith et al., 2021).

4.2.1 Impacts of the Emissions Trading Scheme in its Current Form

It has been shown that the ETS in its current form does not align with government policy. The blanket planting of Pinus radiata provides returns higher than other modelled land uses. Family run operations are priced out of land retirement due to the size of capital investment required (Flack et al., 2022) and ETS ineligibility (Ministry for Primary Industries, 2017). Hence, corporations and external investors with surplus funds purchase whole properties to plant in exotic species (Schirmer & Bull, 2014). These forests are managed differently to production forestry, where logs are harvested (BakerAg, 2019; Kim & Langpap, 2015). The associated loss of pastoral land and skilled labour means rural communities degrade (Paul et al., 2013; West et al., 2020). By attaching a monetary incentive to biodiversity, expected life span and aesthetics, these trends may be able to be reversed in favour of native species (Lambie et al., 2021; Nghiem & Tran, 2016; Nguyen & Nghiem, 2016; Spiekermann et al., 2021). The mechanism behind which on-farm sequestration is monitored and rewarded in the future is currently a topical debate in New Zealand's political scene (Climate Change Commission, 2022; He Waka Eke Noa, 2022). The result of change could promote or harm the ongoing integration of native species amongst pastoral systems.

4.2.1.1 Initial Investment Associated with Carbon Farming Enterprises

The large financial cost associated with whole farm conversion to Pinus radiata prices family run operations out of these enterprises (Dodd et al., 2008; Flack et al., 2022). Most family run businesses cannot afford to face successive years of losses, despite the allure of disparate future earnings through the ETS. With this said, multi-generational family run businesses are often sentimentally attached to pasture-based food production and do not condone whole farm conversion to forestry (Flack et al., 2022; Ryan et al., 2022; Schirmer & Bull, 2014). Rather, they prefer to integrate trees into the landscape if it makes financial sense to do so. The 'Total area' scenario offers an alternative, with smaller areas planted in woody vegetation in low pasture producing areas. However, the ETS currently does not consider these smaller areas as capable of earning NZUs (Dodd et al., 2021). This is why the majority of large scale exotic 'carbon farming' operations are achieved by collectives or established companies with large quantities of external funding (Schirmer & Bull, 2014).

From an economical point of view, utilising an exotic cover crop to nurse native species makes sense (Lambie et al., 2021; Nghiem & Tran, 2016). The large earnings associated with exotic carbon sequestration more than compensates for the establishment and maintenance of native species suitable to grow under an existing canopy. However, the ETS in its current form excludes areas under one hectare in size, so plantings are only practical on a large scale (Dodd et al., 2021). Any willingness to plant native species for biodiversity, longevity or aesthetic purposes becomes redundant when surrounded by large areas of existing exotic forestry (Lambie et al., 2021). It is likely that due to differences in carbon sequestration rates, NZUs may be repayable after mature exotic trees cease growing and eventually topple (Dodd et al., 2021).

4.2.1.2 Rural Community Degradation

After the establishment phase, trees are left without being thinned and pruned (BakerAg, 2019; Kim & Langpap, 2015; Manley & Maclaren, 2012). This means that often labour associated with traditional production forestry enterprises does not exist (Hale et al., 2015). Recently, BakerAg (2019) quantified labour requirements of such systems using an averaging approach. Pastoral farming generates 7.4 jobs/1000ha, production forestry 2.2 jobs/1000ha (non-harvest years) and carbon farming 0.6 jobs/1000ha. With no more employment opportunities offered, rural communities degrade (Miller & Buys, 2014; West et al., 2020). Pastoral farmers not only employ general labour where required, but contractors to undertake specialist tasks (Paul et al., 2013). Fencers, shearers, veterinarians, scanners, environmental consultants and accountants to name a few (BakerAg, 2019). Due to the large existing areas of farmland in woody vegetation (Pannell et al., 2021), forestry consultants and wood harvesting firms are still often involved in such systems. In comparison, large carbon forestry systems seldom require on site management. Food security suffers not only as a function of lost land, but lost social capital (Dodd et al., 2021).

This loss of social capital directly reduces rural community size as people pursue employment elsewhere (Tozer et al., 2021; West et al., 2020). In turn this reduces the need for preferential businesses that support those living rurally (Miller & Buys, 2014). The likes of schools, health centres and sports teams close. Volunteer services such as fire brigades and community groups cease to exist. Criminal activity such as stock theft, vandalism and drug production increases as a result of decreased vigilance (Federated Farmers, 2022). Roads degrade quicker where production forestry allows for product removal (Miller & Buys, 2014)

4.2.1.3 Forest Management of Carbon Farming Systems

The incorporation of thinning into MPI look-up tables assumes that the management practice is applied in all forests (Nguyen & Nghiem, 2016). In a sole carbon farming operation, thinning and pruning are not undertaken as they are fundamentally destructive to the carbon sequestration cause (BakerAg, 2019; Manley & Maclaren, 2012); atleast in the short term (Kim & Langpap, 2015). This actively penalises smaller woodlots which are left unmanaged, perhaps with the goal of facilitating natural regeneration under the canopy. Forests over 100 hectares have no such issue, undertaking field measurement (Ministry for Primary Industries, 2018). This is an example of how the ETS in its current form discourages the targeted planting of trees in the right place e.g. CSAs. The incidence of forest fires increases under this style of management (Kim & Langpap, 2015).

4.2.1.4 Potential Solutions

In all models run, Pine tree scenarios accumulate larger profits than their Manuka counterparts. A monetary value needs to be attached to the biodiversity, longevity and aesthetic characteristics of native species to disrupt this trend (Nghiem & Tran, 2016; Nguyen & Nghiem, 2016; Spiekermann et al., 2021). Financial incentives to plant long term woody species in areas at risk of elevated freshwater contamination could potentially better align land use with government goals, depending on their magnitude (Spiekermann et al., 2021). Without this, natives are always going to be economically unviable when compared to fast growing exotic species (Lambie et al., 2021). The current colinear approach means that in any situation Pines are more profitable than natives to plant. This reflects recent mass land use change. Breaking this collinearity would see different land uses being the most profitable on different land classes. For example, natives could become the most lucrative enterprise on erosion prone land (Model 1, Manuka, Total area) where harvesting or pastoral production is not practical.

At the time of writing, a proposed piece of legislation named the Overseas Investment (Forestry) Amendment Bill is undergoing the reading process (New Zealand Government, 2022). It is hoped that its implementation will provide significant barriers to overseas investors looking to convert existing farmland to exotic forestry. However, submissions reflect fears that the legislation is not strong enough to achieve the desired effect (Federated Farmers, 2022).

4.2.1.5 Current Discussion regarding on Farm Carbon Accounting

There is current debate as to how on farm sequestration should be measured and rewarded. There are two main current proposals. The recognition of smaller pockets of forestry not encapsulated by the ETS under an independent scheme (He Waka Eke Noa) which targets farm GHG emissions is the first. Secondly, recognition of sequestration of all forestry under a refurbished ETS is also a possibility (Climate Change Commission, 2022; He Waka Eke Noa, 2022). A recent study by Wang et al. (2022) argues that agriculture's inclusion into the ETS would reduce national emissions at a rate 4.4 times more than currently experienced. However, the same study recognises that a carbon pricing mechanism such as the ETS not only promotes forestry as a land use, but increases the costs associated with agricultural production. Therefore, any move would have widespread implications to the land use on New Zealand hill country.

Both pricing mechanisms effectively tax farmers for the greenhouse gases emitted from their systems, most importantly biogenic methane (Climate Change Commission, 2022; Leining et al., 2020). The mechanism (or lack of) behind the recognition and reward of farm sequestration not currently managed by the current ETS could hasten or slow mass land use change to exotic forestry. What many fail to take into account is the carbon sequestration capabilities of ETS ineligible on-farm vegetation (He Waka Eke Noa, 2022). If farmers are taxed for the emissions they generate, it is only fair that they are reimbursed for the emissions they sequester.

4.2.2 Market Price Volatility

The optimisation model and sensitivity analysis question the volatility of both carbon and red meat market prices. Compared to the carbon market, the red meat sector seems relatively stable. This appears to be a function of legislative interference in the carbon market since its conception (Kerr et al., 2021; Monge et al., 2016). For these reasons a relatively large range of carbon prices have been tested. Variation in pastoral farming encompasses the minimum and maximum range of the past ten years (Beef and Lamb New Zealand Economic Service, 2022).

4.2.2.1 Red Meat Sector

Prices for lamb and beef (on a per kg basis) have increased by 52% and 69% respectively since 2004 (Stats NZ & Ministry for Primary Industries Economic Intelligence Unit, 2022). Despite not doing so linearly, peaks and troughs have generally remained within these extremes (Figure 12). The quoted five year average figure of $24.72/SU is the farm profit before tax, meaning average farm expenses (cash and depreciation) are built into this number (Beef and Lamb New Zealand Economic Service, 2022).

The schedule associated with the red meat of sheep and cattle (Henry, 2017), is influenced by a number of factors. Several of which combine at any one time to form the current market price. Average export prices since 2004 are shown in Figure 12. Lamb remains the highest export earner in all reported years on a per unit mass basis. Mutton prices appear to be colinear, albeit at a lower level. Beef and veal price levels generally remain between these two extremes (Stats NZ & Ministry for Primary Industries Economic Intelligence Unit, 2022). The wool price has not provided any significant profits for low to medium micron wool producers over recent years, due to low demand and increased labour costs.

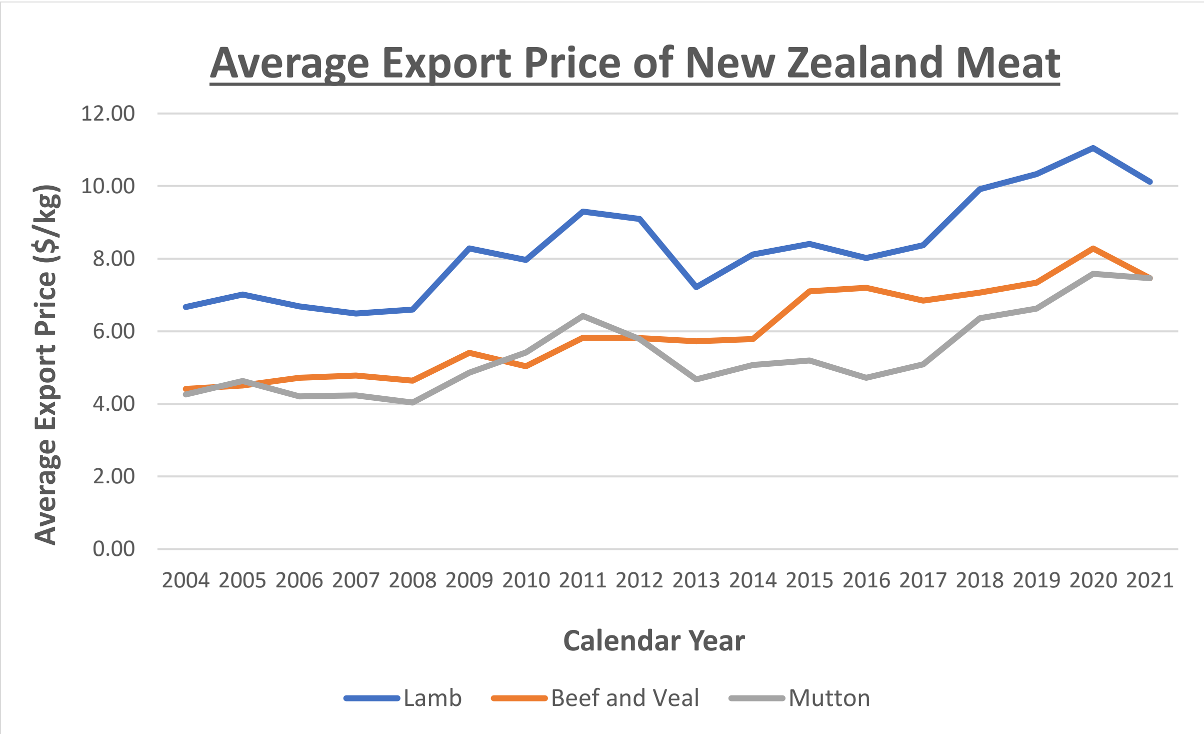

Figure 12: Average export prices of meat products commonly produced on New Zealand hill country pasture.

Due to New Zealand's small global contribution, the market price achieved is generally set by our ability to supply product to overseas markets (Clare et al., 2002). Demand for pasture-fed protein is plentiful (Morris, 2009). Production costs also impact market prices. This includes the costs associated with legislative compliance. Morris (2009) quantifies the cost of production between $50-$70 for a typical 17kg sheep carcass. It is likely that this cost has risen due to increased environmental legislation and raised input costs such as fertiliser.

Premiums are available to those who go above these environmental requirements and market their goods accordingly (Payen et al., 2020).

4.2.2.2 Carbon Sector

The quoted current carbon price of $75/t$CO_2$ is derived directly from market supply and demand for NZUs. In only two years, this value has increased by approximately 130% (Jarden Securites Ltd, 2022). The speed at which this price is increasing is attributed to increased demand of NZUs matched with the reactionary mass plantings of exotic species further fuelling supply. In the future, oversupply and legislative changes may cause further volatility. The sensitivity analysis takes into account a large range of carbon prices ($25-$150/tCO_2). This is due to the uncertain future of the market (Leining et al., 2020; New Zealand Productivity Commission, 2018).

Previously linked to international markets, the New Zealand carbon price dropped to an all-time low of <$2/t$CO_2$ in early 2013 (Best & Cowie, 2022; Monge et al., 2016). The announcement that the New Zealand market will delink from international markets in 2015 caused the carbon price to uncouple from that of international units (Kerr et al., 2021). Due to the relatively low quantity of NZUs available compared to international units, demand increased, raising the price. There is evidence that this caused ‘carbon leakage’ as the carbon price increased (Climate Change Commission, 2022; Kerr et al., 2021). This is when products that would otherwise be produced in an area with a strong carbon market are outsourced to areas with a less developed carbon market, possibly with greater emissions per unit produced (Diaz-Rainey & Tulloch, 2018). This is detrimental to the goals outlined by the Paris Agreement.

Since 2015, the price has risen quickly to the current carbon price of c. $75/t$CO_2$ in a relatively linear fashion (Best & Cowie, 2022). Some commentators forecast that this trend will continue. The New Zealand Productivity Commission (2018) consider that the carbon price would have to be between $157/t$CO_2$ and $250/t$CO_2$ by 2050 to achieved net zero carbon emissions. However, if the NZ ETS was ever relinked with international markets, the associated price could quickly recouple with that of other markets (Kerr et al., 2021; Leining et al., 2020).

Until 2021, the NZ ETS did not have a cap on the quantity of NZUs able to be supplied (Rontard & Hernandez, 2022). This was for administrative ease, allowing the supply and demand dynamic to dictate the carbon price (Leining et al., 2020). High carbon prices were found to incentivise increased supply of NZUs, rather than reduce national emissions. The ETS encourages the use of forests to act as carbon sinks (Diaz-Rainey & Tulloch, 2018), rather than reduce net emissions (Leining et al., 2020; Rontard & Hernandez, 2022). The introduction of a cap on the quantity of NZUs available would hypothetically be better at minimising emissions, albeit more expensive to implement. The effect that this introduced cap has upon market prices is recent and still being investigated. Paul et al. (2013) expects the price to rapidly increase with caps implemented. The author believes the cessation of rapid market growth around March 2022 is due to the government largely controlling the sale of NZUs through caps on supply and price (cost containment reserve and auction reserve price) (Kerr et al., 2021; Leining et al., 2020). This way, the government can largely control the carbon price and the subsequent afforestation rate to ensure they meet their national and international greenhouse gas commitments.

As factors such as biodiversity, biomass longevity and aesthetics are not financially rewarded, yet carbon sequestration is, Pines will always be more profitable to plant than the tagetted planting/regeneration of native species (Lambie et al., 2021). The relatively high cost of fencing and maintenance only compound the issue (Morales et al., 2016). This is the bottom line; land use change will reflect the current or expected earnings of an enterprise. Therefore, it is recommended that the government provide financial incentives for the values that native species promote (Lambie et al., 2021; Nguyen & Nghiem, 2016). If balanced correctly, a fine equilibrium could exist between production forestry (exotic species), pastoral production and integrated native plantings.

4.2.3 Supplementary Incomes

Modelling has focused on 'carbon farming' with woody vegetation. However, there is also an opportunity for the earning of supplementary incomes (Lambie et al., 2021; Pizzirani et al., 2019). Sustainable harvesting of Pinus radiata timber (Manley & Maclaren, 2012), Manuka oil production and Manuka honey production (Boffa Miskell Ltd, 2017; Tozer et al., 2021) are three potentially lucrative enterprises that may be successfully paired with 'carbon farming' enterprises (BakerAg, 2019). If managed correctly, these have the potential to increase the bottom line and make targeted plantings increasingly profitable.

4.2.3.1 Timber

Production forestry on relatively flat land has proved highly lucrative (Lambie et al., 2021). This is evident by the scale of operations in areas such as the Central Plateau (Hale et al., 2015). Increasing carbon prices have seen exotics planted on mass on the steepest of land. Relatively low pastoral production in these areas mean that the margin in profits is largest. These areas are therefore colonised with exotics before other higher producing grasslands. However, results from economic modelling on Patitapu infer that even the likes of dairy production on flat land could be more profitable in exotic forestry. Due to several limitations, 'carbon farmed' forests on hill country are managed differently than their production forest counterparts (Manley & Maclaren, 2012). Literature pertaining to forestry on hill country is sparse, due to the contemporary nature of the land use.

Steep, irregular terrain makes the harvesting of logs relatively unprofitable due to the specialist equipment and expertise involved (Visser & Harrill, 2017). Such areas are often physically isolated, with accessibility of large machinery and logging trucks an issue (Visser & Stampfer, 2015). In many cases, existing backcountry farm tracks are not suitable. Overarching vegetation, steepness, undercutting, river crossings, slippery surfaces and general lack of upkeep make then unusable for vehicles of their size (New Zealand Forest Owners Association, 2020). The alternative is heavy investment in decent roading, assuming this can be practically achieved. In all likelihood, this is a task which doesn't make sense economically or practically on hard hill country.

Market accessibility (Clare et al., 2002) also inhibits the financial feasibility of log production on steep hill country (New Zealand Institute of Forestry Inc., 2005). The further that product has to travel to reach an established market port or processor (Kim & Langpap, 2015), the lower the potential earnings (Lambie et al., 2021). As hard hill country is often extremely isolated (such as Patitapu), any unrealised earnings are automatically reduced. With the scale of recent plantings (driven by the carbon price), there is caution regarding the size of the available workforce come harvesting. There is potential that the expected increase in harvestable logs, hill country foresters may struggle to attract the necessary expertise and equipment.

Subsequently, most exotic species planted on hard hill country are done so solely for the earning of NZUs. Therefore, they are managed in a way as to optimise carbon sequestration (Manley & Maclaren, 2012). Thinning and pruning rarely occur in such systems (Kim & Langpap, 2015). As the mass planting of exotics on steep land is a relatively new phenomena, MPI look-up tables still account for thinning (Ministry for Primary Industries, 2017). This is apparent in all Pinus radiata models, with a reduction in the rate of carbon sequestration occurring in Year 9 due to modelled thinning.

If the harvesting of logs does prove financially feasible, such activity is allowed. MPI have recently introduced a 'rolling average' methodology of carbon accounting (Leining et al., 2020). Under the premise that harvested trees are promptly replanted, the landowner no longer has to pay back NZUs. However, earnings of NZUs do not continue to accumulate. Earnings of NZUs would only accumulate until the average carbon storage of the rotation is met. Income thereafter would only be generated from the trading of existing credits, not the earning of new ones. A financial assessment between production forestry on flat land and 'carbon farming' on hill country would prove useful in understanding the motives behind tree planting by landowners on different land classes. Utilising varying 'timber prices' and 'carbon prices' could help determine at what point that harvesting is viable on different slope classes (Manley & Maclaren, 2012).

If accessibility, access to markets and appropriate forest management allows, production forestry typically earns an IRR in the region of 4-9% (New Zealand Institute of Forestry Inc., 2005). Mead (1995) puts it in the range of 7-10%. These ranges do not compare to the double digit IRR's recorded from the earning of NZUs through Pinus radiata (Table 12). However, sustainable harvest could provide an alternative income stream. Albeit, this would remove any possibility of native regeneration under the exotic canopy.

Manuka has proven a good source of firewood, known for its long burning properties. Another common use is in the cooking of wood smoked meats. Many stand by Manuka's excellent smoking characteristics, infusing game and fish with a unique flavour (Boffa

Miskell Ltd, 2017). However, these uses are niche and demand is low. There is no existing market for Manuka as a timber source.

4.2.3.2 Honey

Honey from Manuka has been found to contain a compound named methylgloxyl (MGO), which gives honey from this species a ‘Unique Manuka Factor’ (UMF) (Boffa Miskell Ltd, 2017; Mavric et al., 2008). It is the antibacterial properties of MGO which attract high market prices for Manuka honey compared to honey from other species (Rueckriemen & Henle, 2018). A minimum of 40 hectares of pure Manuka is said to be required to keep resulting MGO concentrations high enough to attract higher premiums (i.e. minimise contamination from other sources) (Boffa Miskell Ltd, 2017). Manuka honey sold for medicinal purposes has the highest MGO contents. MPI have developed standards for Manuka honey in an attempt to stop the false marketing of honey from alternate sources (Rueckriemen & Henle, 2018).

Different cultivars have been found to produce honey of higher MGO contents; attracting special premiums. Regional differences in Manuka cultivar produces honey of different MGO concentrations (Rueckriemen & Henle, 2018). Alternatively, specially bred cultivars are generally nursed in a facility before being transplanted (Marden et al., 2020). However, a mix of breeds may lead to a longer flowering time, maximising honey production (Nickless et al., 2017). Hives must be managed so that bees survive when target species (Manuka) are not flowering, as well as over the winter period (Boffa Miskell Ltd, 2017). This often entails physically moving hives to multifloral areas and providing supplementary feed.

Planting density impacts tree allometry. High density plantings commonly used in erosion schemes (>10,000 trees/ha) limit space for horizontal growth. Lower density plantings are optimal for honey production as trees are encouraged to spread laterally, creating a ‘bushy’ shrub where flower quantities are optimised (Marden et al., 2020). As trees grow, woody vegetation produces increasingly less flowering sites over time. Due to these changes in tree physiology, expected honey yields decrease over time. Honey production from Manuka stands has only shown to be profitable between years three and 15 in cases where vegetation is not actively managed to maintain flowering (Boffa Miskell Ltd, 2017).

As honey production requires specialist labour and equipment, landowners often contract out Manuka laden land to apiarists. Landowner's often earn between 10-30% of the income generated from honey sales (Boffa Miskell Ltd, 2017). Alternatively, if enough land was owned (such as Model 3, Manuka, ETS compliant), landowners could invest in equipment themselves. Initial hives and bees to establish a colony cost approximately $1,100 each, with $300 per year as maintenance (Boffa Miskell Ltd, 2017). Over one hectare, one hive could produce around 31.1kg of Manuka honey. Selling for an average of $55.36/kg (Ministry for Primary Industries, 2021b), a hive could effectively pay for itself in a singular year.

4.2.3.3 Oil

An established market exists for Manuka oil. Leaf extracts have been shown to display antibacterial properties (Boffa Miskell Ltd, 2017; Mathew et al., 2020; Wicaksono et al., 2018). It has been found that regional differences in cultivars yields oil of different compositions. Therefore, some regions may produce oil of higher quality, generating higher returns (Boffa Miskell Ltd, 2017). More research and testing are required to quantify these differences and the economic effect it could have upon production systems.

Many limitations of Manuka oil manufacture are similar to that of timber production. Expenses increase linearly to slope due to increased manual labour costs. Mechanical harvesting is increasing in uptake, however is currently only trialled in relatively accessible, low contour land (Boffa Miskell Ltd, 2017). Accessibility is an issue on steep hill country as is accessibility to markets and processors (Clare et al., 2002). It is unlikely that farmers with patches of retired regenerated land would own specialist processing equipment which is capable of extracting product (Kim & Langpap, 2015). Usually a third party processor would form an agreement with the farmer to harvest and produce Manuka oil. Boffa Miskell Ltd (2017) state that distillers will pay $500-$600/t for high quality younger growth. Yields are dependent on many factors, not least with the goals of the landowner in mind (Pizzirani et al., 2019). How such a system could work in unison with ETS eligible land needs further investigation due to the destructive nature of harvesting.

4.2.4 Silvopastoral Systems

Silvopastoral systems could potentially provide benefits for meeting New Zealand's greenhouse gas reduction targets and environmental standards whilst working in unison with existing ruminant production systems (Mackay-Smith et al., 2021). Instead of spatially separating livestock and forestry enterprises like many other agroforestry operations, they are integrated. Benefits and drawbacks exist for the adoption of such a system. An argument for further research of silvopastoral systems in a New Zealand context is made (Tozer et al., 2021). A redeveloped framework by Mackay-Smith et al. (2021) is used to do so.

4.2.4.1 Perceived Benefits of Silvopastoral Systems

The provision of shelter to livestock is a benefit of silvopastoral systems (Abdul-Salam et al., 2022; Spiekermann et al., 2021; Tozer et al., 2021). Shelter (sun, rain, wind etc) from the elements during climatic extremities means less stress on animals, translating to higher productivity (Beardmore et al., 2019; Mead, 1995). In sheep and beef operations, weaned lambing and calving percentages would be expected to increase as survivability would be positively impacted (Pollard, 1999).

The use of exotic species for animal fodder during periods of low or bad quality pastoral cover is common practise in silvopastoral systems (Hussain et al., 2009; Spiekermann et al., 2021; Tozer et al., 2021). The palatability, nutritional quality and fodder production of Poplar and Willow have been researched in detail (Kemp et al., 2003). Exotic trees also respond well to systematic defoliation (Tozer et al., 2021). Therefore, the utility of these species if included in a silvopastoral system increases during the likes of summer droughts.

As separation between forestry and pasture is not required under such a system, fencing is not required. A benefit of transitioning to a silvopastoral system, is that large infrastructure investments are not required. Fencing takes up the majority of initial costs in natural regeneration models, amounting to over 86% in every model (Table 13). This is due to minimal plant and labour costs due to vegetation naturally establishing. The implementation of such a system would save the same amount of cash in an exotic forestry system. However, the removal of fencing questions the viability of woody vegetation establishment when grazing animals are also present (Burns et al., 2011; Morales et al., 2016). Either grazing pressure needs to be drastically reduced or plant protection methods need to be physically installed. Both are costly.

Table 13: Modelled fencing costs associated with economic models. The implementation of silvopastoral systems could drastically reduce initial costs.

Model Number	Woody Species	Scenario (ETS/Total)	Initial Cost ($)	Fencing Cost ($)	Percentage of Initial Cost (%)
1	Pinus radiata	ETS compliant	$584,357	$456,725	78
1	Pinus radiata	Total area	$1,587,970	$1,346,855	85
1	Manuka	ETS compliant	$492,991	$456,725	93
1	Manuka	Total area	$1,415,366	$1,346,855	95
2	Pinus radiata	ETS compliant	$3,096,622	$1,957,833	63
2	Pinus radiata	Total area	$4,146,514	$2,861,235	69
2	Manuka	ETS compliant	$2,281,412	$1,957,833	86
2	Manuka	Total area	$3,226,438	$2,861,235	89
3	Pinus radiata	ETS compliant	$2,442,031	$0	0
3	Manuka	ETS compliant	$693,886	$0	0

4.2.4.2 Perceived Drawbacks of Silvopastoral Systems

One potential setback would be having to heavily reduce stock carrying capacity during the formative years of woody vegetation if vegetation was left unprotected (Tozer et al., 2021). Once established to a degree that would prevent trees from being damaged/consumed by livestock, animals could be reintroduced to the farming system. Sheep may be reintroduced ahead of cattle for this reason (Tozer et al., 2021). However, the removal of livestock for any period of time would greatly impact the farms cashflow. In addition to this, initial growth rates of forestry will be slow, as trees establish themselves. Therefore, initial earnings under the ETS will be slow. The result is several years of heavy financial losses as costs (e.g. planting and mortgage repayments) would outweigh any income (Abdul-Salam et al., 2022). Only landowners with significant outsourced funding could achieve such a change (Flack et al., 2022). In such a system, exotic species are often used over natives due to their fast growth characteristics, minimising the period of stock exclusion. Most landowners wishing to transition to a silvopastoral system do so gradually. By doing so, the cashflow of the remaining business can be somewhat maintained. This would compensate for any losses associated with land use change. Physically protecting plants is an alternative method of ensuring plant survival (Morales et al., 2016), however is also costly. A recommendation would be to prioritise the planting of woody vegetation in areas susceptible to high environmental losses, as to comply with freshwater legislation.

Although not as much an issue for the aerial application of farm inputs such as fertiliser, farm accessibility may be impacted by trees. Terrestrial vehicles such as trucks, tractors and all-terrain vehicles would likely not experience the relative freedom in pathway when travelling between paddocks. If terrain was suitable for their use, ground spreaders would be unable to achieve a uniform spread due to physical impediment of trees. If such a system was implemented, it would be essential to have well maintained farm tracks, with plantings planned around practicality on farm.

4.2.4.3 Research Priorities

More work is required to determine the role that silvopastoral systems could play in New Zealand's primary sector. An existing framework by Wood (1990) for assessing the relevance of different woody species in potential silvopastoral systems is improved by Mackay-Smith et al. (2021). Using Kanuka and Poplars as examples, the potential for their use in New Zealand silvopastoral systems is investigated. Several facets where more research is required are highlighted, indicating the imbalance in literature between exotic and native species.

Hill country sheep and beef systems are prone to drought (Hussain et al., 2009). Where limited by water, pasture production has been shown to increase under the canopy of Kanuka in the Wairarapa (Mackay-Smith et al., 2022b). In a shaded environment, water would be held in the soil for longer due to lower temperatures evaporating less water from the soil profile. Higher organic matter inputs (litterfall) (Lambie & Dando, 2019) also creates better soil structure, retaining greater quantities of water (Mackay-Smith et al., 2022b). As trees are commonly utilised as stock camps, elevated nutrient levels are present (Hussain et al., 2009; Mackay-Smith et al., 2022a), meaning they are not limiting to pasture growth. In contrast, where water stress is not likely to be constricting, the opposite is true. Shade inhibits pasture growth. Hussain et al. (2009) found decreases in pasture production between nine and 24% underneath a dense poplar and willow canopy. More research is required to quantify the interactions between trees, soil, pasture, animals and climate; particularly with native species (Mackay-Smith et al., 2021; Tozer et al., 2021).

How silvopastoral systems can help biodiversity and environmental goals in a New Zealand context requires further investigation (Mackay-Smith et al., 2021). Natives are expected to promote native fauna habitats (Tozer et al., 2021) by connecting pockets of established natural forests found throughout many hill country sheep and beef farms (Burns et al., 2011; Morales et al., 2016; Pannell et al., 2021). Many exotics (Pinus radiata a notable exception) are deciduous and have a relatively small impact (Mackay-Smith et al., 2021). Trees spaced too far apart have been shown to not have the same effect on soil conservation, due to roots not being spatially interlinked (Douglas et al., 2013; Marden et al., 2020). Therefore, planting density must positively correlate to slope (or other variables affecting environmental losses such as soil type) to have a positive effect. A good body of work exists for exotic species commonly spaced planted on New Zealand hill country farms such as Poplars, less so for native species such as Kanuka (Mackay-Smith et al., 2021). How natives interact with the soil (e.g. water usage) and subsequently impact environmental losses also need to be investigated (Mackay-Smith et al., 2022a).

From a management perspective, the literature base behind natives is considerably less developed than exotics such as Poplar and Willow (Mackay-Smith et al., 2021). The use of the aforementioned exotic species as an alternative fodder source over periods of low pasture quantity/quality has been discussed. However, the palatability and nutritive value of native vegetation for grazing ruminants is largely unknown (Tozer et al., 2021), Manuka and Kanuka being relatively unpalatable exceptions. If proved to be a potential source of forage, how native trees cope with pruning and cutting need to be analysed (Tozer et al., 2021). In terms of establishment, protection methods from grazing livestock and associated costs for natives need to be financially quantified (Burns et al., 2011; Morales et al., 2016). This will affect initial costs, which are well researched for the space planting of exotic species (Mackay-Smith et al., 2021). The minimum planting density required by both native and exotics to achieve a 30% canopy cover needs to be quantified by individual species, as to be eligible under the ETS in its current state. As discussed, the carbon sequestration rate (growth rate) of natives needs to be reassessed (Dodd et al., 2021).

Mackay-Smith et al. (2021) also account for cultural outcomes in their revamped framework for assessing tree species for their applicability in different silvopastoral systems. Natives are expected to provide greater value from a cultural perspective, as they are indigenous (Wangui et al., 2021). Tree longevity also needs to be taken into account. In a silvopastoral system, secondary succession underneath is unlikely due to grazing pressure (Husheer et al., 2005).

This may impact the expected life of species commonly regarded as pioneer species (Manuka and Kanuka), which will affect expected carbon storage. Contrasting natives and exotics in their ability to provide animal shelter would provide insights to the associated pastoral system. Mostly evergreen, it is expected that natives would provide the greatest benefit over colder, wetter periods (Mackay-Smith et al., 2021).

5 Summary

This work provides evidence in support of our hypothesis that an economic disparity between exotic and native plantings aswell as existing pastoral systems exists on New Zealand hill country. The pilot study found that carbon sequestration by regenerating native forest could be significantly underestimated in default carbon look-up tables. Although gaps in regenerating bush are not accounted for, the author argues that landowners would want to maximise value from the land, filling any spaces with nursed seedlings. Conditions must allow for the successful establishment of primary native colonisers such as Manuka to be eligible under the ETS. Although the study lacked the range and scale to categorically refute the use of default carbon look-up tables, it does warrant further work. It is key that landowners are remunerated fairly accounting to the carbon sequestration capabilities of their forests, especially if natives are to be incentivised to landowners over exotic species.

Economic modelling shows that the blanket planting of Pines vastly outcompetes both pastoral farming and natural regeneration on New Zealand hill country in terms of long-term profitability, even when increased carbon sequestration of Manuka is considered. Optimisation, NPV, IRR and sensitivity analysis reciprocate these findings at different inflation rates and market prices. However high initial costs mean these profits are likely to be unrealised by family run operations, instead attracting large corporations or collective investment. Instead, farming businesses look to retire land unsuitable for pastoral production, often with native colonising species such as Manuka establishing. Modelling shows that if areas not currently recognised by the ETS are accounted for, overall profitability increases, albeit with increased initial costs. Therefore, if spatial limitations imposed by the ETS were lifted, such land retirement could be a viable option for primary producers. Any policy developments regarding whether and how total on farm carbon sequestration can be measured and rewarded could have significant implications for hill country land use.

This work shows how the mass conversion of hill country farmland to blanket planted exotic forests is a direct consequence of the ETS in its current form. Freshwater legislation provides no such financial incentive to plant exotic forestry on mass. It is considered that reported government ideology promoting the integration of trees into the landscape is merely for public perception. An underlying motive to meet national and international greenhouse gas commitments as quickly as possible may exist, with long-term implications ignored as a result.

Potential solutions which may align land use with government philosophy include the assignment of a financial incentive to new plantings which promote not only carbon sequestration, but other significant elements. By rewarding the delivery of biodiversity, longevity and aesthetic values of forests, native regeneration could become the dominant land use on land unsuitable for harvesting or pastoral production. Silvopastoral systems could also provide income from both carbon and red meat/wool sectors where the spatial integration of trees and existing pastoral systems is possible. More research is required to understand how these systems could impact farm profitability in a New Zealand context. As demonstrated by this work, the initial costs associated with land use change will provide significant barriers to family run farm businesses looking to implement changes, even if large future returns exist.

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7 Appendix

Spreadsheet data is appended in electronic format. Please follow the attached link to access:
https://1drv.ms/f/s!AnrnuUQ33GdoaV3dRI-3yxP5Gl8

Data is open access and free to use. It is asked that if utilised, the source is appropriately acknowledged.

Available spreadsheets:

Manuka Carbon Sequestration.xlsx

Contains raw data from Manuka sample sites and subsequent analysis.

Model Economics.xlsx

Contains assumptions, profitability, NPV, IRR and sensitivity analysis.

Hardcopy readers are referenced to the electronic version of this document.