THE EFFECT OF AN INTEGRATED CATCHMENT MANAGEMENT PLAN ON THE GREENHOUSE GAS BALANCE OF THE MANGAOTAMA CATCHMENT OF THE WHATAWHATA HILL COUNTRY RESEARCH STATION

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ABSTRACT

An integrated catchment management plan implemented in the Mangaotama catchment of the Whatawhata Research Station in 2001 demonstrated that *Pinus radiata* forestry on marginal land, along with conservation measures and intensification could produce a win-win outcome for economic output and the environment. However, greenhouse gas mitigation was never fully considered. This research investigated the effect of the plan on the land's greenhouse gas balance and carbon stocks between 2000 and 2011. Historical records, modelling with OVERSEER and CenW, literature values and field measurements were used to account for CO_2, CH_4, and N_2O from the four main land-use types: pasture, native forest, pine, and native plantings. The original land-use would have emitted a net 10.99 Gg CO_{2e} over 10yrs, whereas the new land-use sequestered a net 47.26 Gg CO_{2e} in its first 10yrs. The total carbon stocks rose by 15.9 Gg C. Forestry conversion of almost half the area explained most of this effect. Agricultural intensification increased per hectare emissions from pasture, but overall pasture emissions were lowered by over half due to the reduction in livestock numbers. The native plantings had a small impact due to the small area planted and their slower growth compared with pines. Soil carbon was lost under all land-uses, except possibly in grazed native forests, but these conclusions were hampered by a scarcity of samples. Uncertainty also surrounded the modelling of the pine forest in complex terrain, which is not yet adequately captured in CenW. A preliminary look at carbon trading suggested that it could strongly undermine the viability of the original farm system, but it could also help to fund the expensive transition to the new land-use. Overall, it was found that in addition to the benefits already shown by the integrated catchment management plan, it was also an effective way of mitigating climate change.

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ABBREVIATIONS

C	Carbon
CO_2	Carbon Dioxide
CO_{2e}	Carbon dioxide equivalents
CH_4	Methane
CUR	The current established land-use regime from 2006-2011
CWD	Coarse woody debris
DBH	Diameter at breast height
DEM	Digital Elevation Model
DM	Dry matter
DOC	Dissolved organic carbon
ETS	Emissions trading scheme
FAO	Food and Agriculture Organisation
GDP	Gross Domestic Product
Gg	Gigagram (1000 tonnes)
GHG	Greenhouse gas
GIS	Geographic Information System
GWP	Global Warming Potential
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
LUCC	Land use capability class
LW	Live-weight (for lamb and beef carcasses)
N_2O	Nitrous oxide
NPP	Net primary production
NZU	New Zealand unit
OLD	The old land-use regime, pre-2001
ppb	Parts per billion
Pg	Petagram (1,000,000 gigagrams)
ppm	Parts per million
sph	Stems per hectare
SU	Stock unit
t	Tonne
TRA	The transitional period to the new land-use regime from 2001-2006

CHAPTER 1: INTRODUCTION

1.1 INTRODUCTION

Climate change is one of the world's foremost environmental concerns. All nations and all sectors of society will be faced with its impacts and are obligated to address it. There is great concern that the costs of change are insurmountably high, given that emissions are bound to economic growth, and that the costs of inaction could be even greater (Stern, 2007). Therefore, there is an urgent need to find ways to reconcile our economic activities with this grave environmental threat.

For New Zealand to do this, the problem is in creating viable yet 'carbon friendly' agricultural systems. The agricultural sector is New Zealand's biggest emitter of greenhouse gases, contributing 46.5% of total emissions in 2009 (Ministry for the Environment, 2011). It is also a major part of the national economy, contributing 12.2% of GDP in 2010 (Ministry of Agriculture and Forestry, 2011e). The global demand for livestock products is only expected to increase due to the growing world population and its changing consumption patterns (Thornton, 2010), driving further development in this sector.

Reforestation of marginal land, especially in hill country, is one option for mitigating emissions. An integrated catchment management plan for the Whatawhata Hill Country Research station combined reforestation and intensification, demonstrating that a win-win between growing trees and better farm profitability can be achieved (Dodd et al., 2008a). This approach could provide a workable system for hill country, likely to fit the criteria of being both productive and carbon neutral. However, its effects on greenhouse gas (GHG) mitigation have never been fully quantified. There is critical need to develop these types of farm systems, so further study is needed for the insights to be drawn out, improved and applied.

The rest of this section briefly explores the basic background of these issues, and then outlines a study to investigate the greenhouse gas balance at of the Mangaotama catchment at Whatawhata.

1.2 BACKGROUND

Emissions of GHGs are driving an increase in global temperatures, which could have far reaching consequences if allowed to go too far (IPCC, 2007a). The three main problem gases are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The levels of CO_2 and CH_4 are well beyond the natural range of the past 650 millennia and are still increasing (IPCC, 2007a). Fossil fuel use, land use change and agriculture are the main sources of GHG emissions (IPCC, 2007a).

Climate change stands to have many negative impacts. These will depend on future emissions rates and the resulting temperature increase. Small temperature rises will see negative effects on both the environment and society and these effects will be severely exacerbated if temperatures climb higher (IPCC, 2007a). Substantial negative impacts on the global economy, especially on the poor, are predicted (Stern, 2007). To worsen matters, climate change poses the threat of continuing for centuries (IPCC, 2007a) and triggering abrupt and disruptive shifts in the global climate (Stern, 2007).

Food production may need to double by 2050 using the same or less area of land with greater environmental constraints, creating a need for sustainable intensification, which has been defined as producing more food with less impact from the same or less area of land (Godfray et al., 2010). In the past, global food production has grown along with population through the extensive use of fertilizers, machinery, irrigation, and chemicals, but it is becoming apparent that we have reached a critical point where the environmental costs of this approach are too large and the past linear relationship between inputs and productivity cannot be expected to continue (Pretty, 2008). New Zealand has followed this same path in agricultural intensification over the past several decades, with dramatic increases in fertilizer use and other inputs, and is expected to continue along it for the next decade at least, but it is questionable if this approach can be maintained (MacLeod and Moller, 2006). Clearly, new approaches to agriculture will be needed.

Incorporating climate mitigation into agricultural systems will need to be an important part of achieving sustainable systems. Agriculture directly affects the natural, social, and economic capital on which it relies. Sustainable agriculture will tend towards enhancing this capital, whereas unsustainable practices will act to erode them leaving less for the future (Pretty, 2008). The effects of climate change could greatly undermine this capital. Therefore, present day agricultural systems which are major contributors to climate change are of questionable sustainability. The climate outcome for New Zealand is still uncertain, but negative impacts on our agriculture are a definite possibility (Kenny, 2001). Even if climate change may appear a small threat to land-users in the short term, societal pressure, including the Emissions Trading Scheme (ETS) and consumer concerns in overseas markets, will likely place growing demand on the sector to address its emissions in order to remain competitive.

New Zealand is faced with a significant challenge in this regard. Addressing climate change without decreasing production is a difficult task for pastoral agriculture. Firstly, emissions are an inherent part of the biology of raising ruminant livestock. In a life cycle analysis of NZ lamb, 57% of total emissions from production to consumption were from enteric CH_4 and 15% were from excreta-derived N_2O. Together they made up 90% of on-farm emissions (Ledgard et al., 2010). Thus, the biggest problem is on-farm, where the main issue is the animals themselves. Reducing emissions here is a major issue, because many of the means of doing so are still in development, or may involve trade-offs with production, although some do have the potential for win-win outcomes (Pinares-Patino et al., 2009).

A second means of addressing climate change is to capture more carbon in trees and soil. Trees clearly sequester carbon in biomass as they grow. Carbon can be sequestered in the soil but the gains are likely to be small and more uncertain than those provided by forests. Unfortunately, trees can decrease pasture production, creating a trade-off between forestry and livestock production. For example, at 400 stems per hectare and 10yrs age, pasture dry matter production under trees can be down to 12% of that in the open (Hawke, 1991) significantly reducing the land's ability to support livestock. One way of reducing the severity of the trade-off is by planting marginal areas which have low pastoral value already. Hill country contains many areas suitable for this, because hill land is mostly classified as land use capability class (LUCC) V, VI and VII (MacKay, 2008), with class VII having severe limitations, making it more suited for forestry than pasture (Lynn et al., 2009).

While reforestation does require sacrificing farmland for trees, this does not necessarily mean the sacrifice of agricultural productivity or revenue. Financially, the introduction of the ETS could provide an additional revenue stream for forests (Manley and Maclaren, 2010), potentially making forestry a better option on low productivity farmland. From a food production stance, better utilization and intensification of remaining land could be used to compensate for the area lost to trees, providing the forested area sequesters enough to offset this increase in emissions. In fact, the retirement of steep-lands into forest may be needed due to high levels of erosion rendering pastoral agricultural unsustainable on them anyway (Blaschke et al., 1992).

In 2001, an integrated catchment management plan was implemented at the Whatawhata Hill Country Research Station in the Western Waikato to investigate the use of a combination of reforestation of marginal land, intensification of agricultural land, and restoration and protection of native forest fragments on the sustainability of the farm. 153ha of steep land were planted with pines and 7ha in native trees, riparian buffers were put in place, native bush fragments were fenced, and the remaining pasture was intensified for production (Dodd et al., 2008a). The regime change produced significant gains in both economic and environmental indicators (Dodd et al., 2008a). The main downside of the plan was the significant short-term costs of implementation, almost $600,000 in the first 10yrs, which posed a major barrier to implementing this sort of land use change (Dodd et al., 2008a). However, the effect of carbon trading on the plan's financial viability was never considered.

1.3 RESEARCH CONTEXT

If reforestation of marginal hill country land is to become a useful strategy for offsetting emissions without undercutting the land's agricultural uses, then effective means of doing so must be found. There are many areas that need investigation, for example: how to minimize the trade-offs between forestry and agriculture, and enhance economic and environmental synergies; the amount of land in forest actually required to offset the farm's own emissions; how much potential 'carbon surplus' for off-setting emissions from other sources can be produced; what effect does the variable topography in hill country have on GHG cycling, etc.

The Whatawhata project is a suitable case study for investigating many of these issues. It offers the comparison of different land use regimes at the same site and it has achieved a win-win outcome for reforestation, environmental protection and improved economic performance. The site's long history as a research farm provides several useful long term data sets and numerous previous studies for information.

The aim of this research was to investigate the effect of a land-use plan based on reforestation of marginal land, intensification of agricultural land and conservation of native forest remnants, in North Island hill country by looking at the Whatawhata case study over the decade from immediately before the land-use changes until the present. In particular, the aim was to assess how successful the plan was in mitigating the farm's impact on climate change. To meet this aim, the following research questions were addressed:

1) What were the net greenhouse gas balance and sizes of the carbon stocks of the catchment under the old, transitional and new land uses?
2) What was the role of each land-use change in altering the greenhouse gas balance?
3) How did the topography and management changes affect the carbon sequestration of each land-use?

1.4 THESIS ORGANIZATION

In this chapter the background of the issue and the research has been introduced. Chapter 2 reviews the literature on the major issues. In Chapter 3 the methods of the research are described. Chapter 4 presents the results. The key findings of this research are discussed, and conclusions and implications are presented in Chapter 5. Appendix A contains details on the values used in calculating the final results, and Appendix B gives details on the results of sampling in the native forests and soils.

CHAPTER 2: LITERATURE REVIEW

2.1 INTRODUCTION

This section expands upon the background material presented in the Introduction, providing a context to the issues involved in this study. The following review surveys the literature on:

- Sustainable agriculture and climate change;
- Greenhouse gas (GHG) cycling in pastoral agriculture and forest systems;
- Options for addressing climate change;
- Previous studies of agricultural GHG balances;
- Marginal lands & carbon sequestration in Hill Country;
- Trees in hill country,
- Ecosystem services and carbon trading;
- Previous carbon studies at Whatawhata.

2.2 SUSTAINABLE AGRICULTURE & CLIMATE CHANGE

Sustainability (or sustainable development) evades tight definition. Some common themes are the triple bottom line (or three pillars) of the economy, the environment and society and the issues of inter- and intra-generational equity (Kajikawa, 2008). A well-known definition of sustainable development, given in the Brundtland report, is “development that meets the needs of the present without compromising the ability of future generations to meet their own needs”. It is explained as being the goal of promoting harmony within humanity and between humanity and the natural world (World Commission on Environment and Development, 1987).

These are inherently ethical issues, meaning that ethics and philosophy must take their place alongside more scientific approaches to sustainability (Vucetich and Nelson, 2010), particularly when it comes to defining the goals of sustainability. Because much depends on fundamental distinctions in values and philosophy, the way of achieving and measuring sustainability is contested, often being divided into ‘soft’ sustainability, which views nature, society and the economy as substitutable, versus ‘hard’ sustainability, which says they cannot be interchanged

(Becker, 1997). Nonetheless, achieving sustainability broadly concerns reconciling the fields of economics, ecology, social science and ethics in a harmonious way to create a world that is good and enduring.

Sustainability has become a major need for future agricultural development, as we attempt to feed the worlds growing population within environmental limits (Godfray et al., 2010). Modern agriculture has proceeded down a path of intensification of production based on increased area and industrial inputs, which has greatly raised food production, but it is unlikely that these gains can be maintained without significant negative effects (Pretty, 2008).

Globally, livestock agriculture is responsible for many negative environmental impacts. When feed crops are included, livestock agriculture is the largest user of land covering 70% of all agricultural land and 30% of the earth's surface (Steinfeld et al., 2006). According to the FAO, livestock agriculture is responsible for deforestation, land degradation, more GHG emissions than transport (18% of global total), 8% of global water consumption, biodiversity loss, and is possibly the largest source of water pollution through eutrophication, erosion, tannery chemicals, pesticides, hormones and antibiotic use (Steinfeld et al., 2006).

New Zealand agriculture has also experienced a dramatic period of intensification. Macleod & Moller (2006) analyzed indicators of land-use change from 1961-2001 in New Zealand. Their results show there has been substantial intensification with increasing use of ecological and energy subsidies. Non-nitrogenous fertilizer use has doubled, nitrogen fertilizer use has increased 60-fold, pesticide use has probably increased by over 10-fold, irrigated land increased by 400%, and energy use would have matched these increases. Macleod & Moller (2006) concluded that these trends raise serious concerns about the economic and ecological sustainability of New Zealand agriculture, especially given inevitable rises in global oil prices and the link between intensification and degradation.

The sustainability of New Zealand's hill country pastures is even more in doubt. Deforestation has left the steepest of these areas with very high erosion rates, which undermines the pasture production of what is already economically marginal land, meaning that it is not sustainable to farm them in this way (Blaschke et al., 1992).

While there is a great need for more environmentally friendly practices, there is also a great need for increased food production due to population growth, creating a requirement for 'sustainable intensification' (Godfray et al., 2010). Agriculture directly impacts the natural, social, physical, financial and human capital on which it is reliant for its success; therefore, to be sustainable it must enhance rather than undermine these assets (Pretty, 2008). Negative effects such as erosion directly undermine production, while others, like water pollution, can undermine its social acceptance.

Climate change threatens to directly damage food production and indirectly undermine the social and economic capital which supports it. Despite the difficulties in predicting the future, it is clear that climate change poses a significant risk to the sustainability of the world as a whole. According to the IPCC, even in low global temperature increase scenarios (up to 2^{o}C) impacts could include increased water stress for hundreds of millions of people; an increased extinction risk for up to 30% of species; lowered crop growth at low latitudes; greater storm and flood damage; increased health risks from the spread of diseases, heat waves, floods and droughts. At higher global temperature increases (above 2^{o}C) the above-mentioned impacts will be worsened, and could also include increased coastal flooding for millions of people, decreased global food production, and the extinction of more than 40% of species (IPCC, 2007a). Today's actions on emissions will have long lasting consequences. Due to the inertia of the climate system, even if emissions were stabilized now, climate change would still have the potential to continue for centuries(IPCC, 2007a).

The Stern Report (2007) investigated the likely impacts of climate change on the global economy. The impact of ignoring the problem was predicted to be in the upper range of a 5-20% reduction in per capita consumption now and forever, with a disproportionate amount of this falling on the poor. In contrast, the resource cost of aiming to stabilize CO_2 at 550ppm would be an estimated -1 to 3.5% of GDP. Delaying action would have a higher price, as it would require deeper cuts in emissions and would result in greater climate change and greater adaptation costs. The risk of pushing past thresholds of abrupt change in the climate system (e.g. ice sheet collapse) added further concern for limiting future temperature rises by cutting emissions sooner rather than later.

For New Zealand agriculture, in some areas, pasture production may actually be increased through warmer temperatures and 'carbon fertilization' (i.e. higher CO_2 concentrations allowing greater photosynthesis). However, problems will likely come from a more variable and extreme climate, increased frequency and severity of drought, more floods and erosion and the spread of low quality subtropical grasses (Kenny, 2001). Just as the positive effects that are predicted for some developed countries will likely disappear as temperatures climb higher (Stern, 2007), the negative effects may well overwhelm any gains New Zealand makes if climate change goes far enough. Therefore, this issue should not be ignored in regards to creating a sustainable food system in New Zealand.

2.3 GREENHOUSE GAS CYCLING IN PASTORAL AGRICULTURE & FORESTS

2.3.1 GREENHOUSE GASES

GHGs are deemed responsible for the current pattern of global warming (IPCC, 2007a). From pre-industrial times to 2005, atmospheric CO_2 concentrations went from 280 ppm to 379 ppm, CH_4 went from 715 - 1774 ppb, N_2O went from 270 - 319 ppb (IPCC, 2007a).

Global Warming Potentials (GWP) give a simple way of comparing the gases by expressing the different lifetimes and radiative forcing of each gas in relative amounts of CO_2, called carbon dioxide equivalents (CO_{2e}), that would produce the same effect over a given time frame (IPCC, 2007a). Other metrics exist and GWP has shortcomings, especially in regard to short-lived gases, but 100yr GWPs were chosen for the Kyoto protocol (IPCC, 2007b) and it has become common to use GWPs in both government reporting and research. The 100yr GWPs from the 4th assessment report for the three main gases are: CO_2 =1, CH_4 = 25, N_2O = 298 (IPCC, 2007b). Studies that only include CO_2 may be misleading, as the large differences between gases show that even small emissions of the more potent gases can be equally important.

Accounting for these gases means understanding carbon cycling for CO_2 and CH_4 and the denitrification part of the nitrogen cycle. The diagrams below give a simplified picture of these cycles in pasture and in a managed forest (Figs. 1 and 2).The global carbon cycle includes both long-term geological cycles and shorter term biological cycles, in terrestrial and marine systems. Only the biotic terrestrial cycle can be easily manipulated by farmers, although the use of fossil fuels is technically part of the geological cycle. As pasture and forests are the main land uses in New Zealand and the main subject of this study, they are compared below.

2.3.2 CARBON STOCKS

In forests, carbon cycling is dominated by two major stocks of carbon: the trees and the soil. In contrast, pasture has just one major stock – the soil. Forests often have smaller soil carbon stocks. For example, the New Zealand national inventory gives an estimated 92.04 t C ha^{-1} for native forest soils and 88.96 t C ha^{-1} for pre-1990 plantation forest soils (Ministry for the Environment, 2011). In contrast, the national inventory gives an estimate of 117.66 t C ha^{-1} for low producing pasture. Nevertheless, the addition of the second large carbon stock in the trees will typically more than compensate for this difference, meaning that forests tend to have much larger total carbon stocks than pasture. The National Inventory estimates a value of 173 t C ha^{-1} for native forest biomass (Ministry for the Environment, 2011). Shrublands have slightly less carbon stored in plants, depending on their age. South Island shrublands have been measured at 78t CO_2 ha^{-1} (Coomes et al., 2002). Pasture plant biomass, in contrast, is estimated in the national inventory as a mere 6.75 t C ha^{-1} for high producing grassland and 3.05 t C ha^{-1} for low producing grassland(Ministry for the Environment, 2011), an almost negligible amount compared to mature forest. Given that forests have almost double the carbon stocks of pasture, conversion from pasture to forest will require a period where carbon is being actively removed from the atmosphere and locked up as this stock grows.

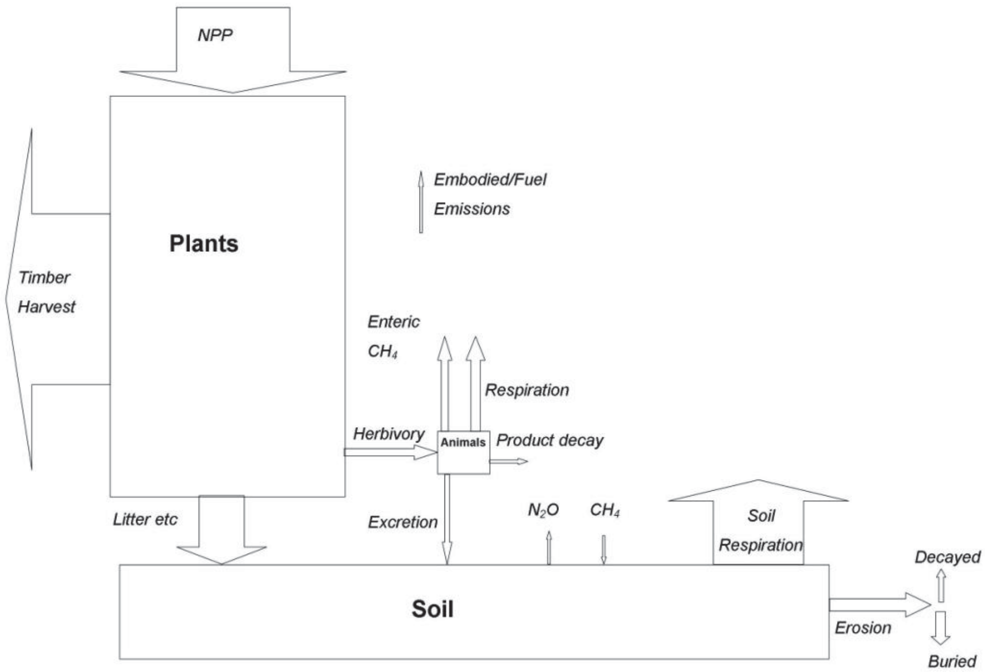

Figure 1. Greenhouse gas cycling in a forest. The size of the arrows and boxes gives an approximate indication of the typical relative sizes of stocks and flows based on the values used throughout this study. Up arrows indicate emissions, down arrows sequestration. NPP = net primary productivity.

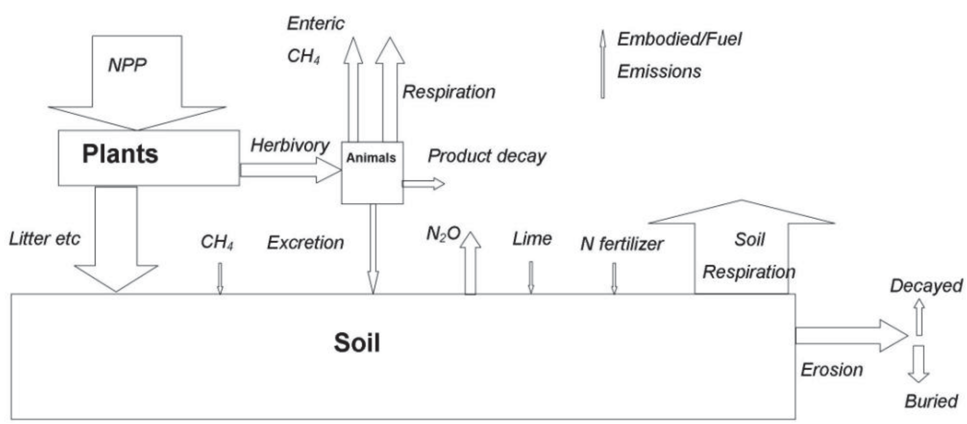

Figure 2. Greenhouse gas cycling in a grazed pasture. The sizes of the arrows and boxes give an approximate indication of the typical relative sizes of stocks and flows based on the values used in this study. Up arrows indicate emissions, down arrows sequestration. NPP = net primary productivity.

The carbon stored in animals is very small compared with that in the soil and plant pools. Typically there would be orders of magnitude less carbon in the animals. For example, in the native forest of the Orongorongo valley there were 16.4 - 24.6 kg ha^{-1} of possums (Fitzgerald and Gibb, 2001), most of which would be water. In intensively grazed pasture, even with 18 stock units (SU) ha^{-1} (a stock unit being one breeding ewe raising one lamb, used in standardizing different types of livestock) at 55 kg per SU this is still only 990 kg ha^{-1}, most of which is water. Compared with the above values for the soil and the plants, animals are a minor contributor to the total carbon stocks.

2.3.3 EMISSIONS

Pastoral farmland has much greater net emissions than forests, primarily due to the presence of ruminant grazers, fertilizer use, and the poor ability of pasture species to sequester carbon into long-term stores. The higher animal densities in pasture means greater fluxes of N_2O from the higher levels of excreta as well as from the use of nitrogen fertilizers to support the grazers. For two highly productive dairy pastures average emissions of 1.6-1.8 kg N_2O ha^{-1} yr^{-1} for ungrazed sites and 9.5-11.7 kg N_2O ha^{-1} yr^{-1} for grazed sites have been measured (Saggar et al., 2004), showing the difference the presence of grazers makes. 3.7 kg N_2O ha^{-1} yr^{-1} was found for a sheep grazed pasture (Saggar et al., 2007b). Thus, the type of stock can also be important. In comparison, ungrazed 80yr old *Kunzea ericoides* stands had N_2O fluxes of 0.7 kg N_2O ha^{-1} yr^{-1}, while in 8-12yr old stands emissions were undetectable (Price et al., 2010). In pine forest, N_2O has been measured at 0.6 kg N_2O ha^{-1} yr^{-1} (Saggar et al., 2008).

Emissions of enteric CH_4 are directly related to stock density. The average National Inventory emission factors for enteric CH_4 in 2009 were 56.2 and 11.2 kg CH_4 $head^{-1}$ yr^{-1} for beef cows and sheep, respectively (Ministry for the Environment, 2011). Respiration from animals is typically ignored, as there is no net gain of CO_2 in the atmosphere, rather it only serves to release recently fixed plant carbon. Only the emissions of N_2O, which is independent of carbon, and CH_4 which converts the CO_2 to a more potent gas are counted.

Measuring changes in soil carbon or accurately accounting for the soil's very similar sized inputs and outputs is difficult, so these are typically ignored (e.g. in the Emission's Trading Scheme), effectively assuming pasture soils are at a steady state. Photosynthesis and respiration are assumed to be cancelling each other out, so only the conversion to the more potent gases needs measuring. Some studies have tested trends in soil carbon stocks to detect if this assumption is valid. A long-term study of hill country soils detected both small increases and decreases in soil carbon stocks (Schipper et al., 2009). The size of the change differed with slope. The reasons for the changes were unclear, but may have been related to fluctuations in the climate. The small magnitude (0.27% to -0.066% change per year) highlights the difficulty of capturing this change.

Fertilizers, herbicides, fuel and other man made inputs create emissions during their production. These are called 'embodied emissions' because they do not actually occur on site but would not have been emitted had the product never been made. In addition, there are direct fuel combustion emissions from transport and management activities. Embodied emissions and combustion emissions from fuel are likely to be more evenly spread over time in pastures than in a production forest, because they are associated more with day to day operations rather than major events such as planting, pruning, thinning and harvesting. Embodied emissions in native forest may be associated with management activities as per pasture if it is grazed, or with conservation activities (e.g. weed and pest management) if it is being actively protected.

2.3.4 OTHER FLUXES

Soil CH_4 flux is usually a small net consumption of CH_4. It is smaller in pastures than in forests. It has been measured as -0.64 kg CH_4 ha^{-1} yr^{-1} in a sheep grazed pasture (Saggar et al., 2007b). In pine forests fluxes have been measured ranging from -4.2 to -14.0 kg CH_4 ha^{-1} $year^{-1}$ (Tate et al., 2006). In *Kunzea ericoides* stands it ranged from -4.73 kg CH_4 ha^{-1} $year^{-1}$ in the older stands to -0.66 kg CH_4 ha^{-1} $year^{-1}$ in the young stands (Price et al., 2010).

In pasture removals of plant biomass, apart from consumption by animals, are absent unless surplus grass is used for making hay. Harvesting of animal products will, therefore, be larger than removals of plant biomass. In production forestry the entire mature forest is likely to be cut at once, with a large fraction of the biomass removed. In native forests harvesting is much less common than it once was, but low levels of deforestation of native forest and scrub is ongoing, although much of this is to make way for plantation forests (Ewers et al., 2006). Removals of meat, wool and timber can contribute emissions from their decay as well as become sinks in landfills and long lived products. The longer life spans of wood products again add to the greater sink capacity of forests over pastures. Half-lives for wood products range from 2yrs for paper to 35yrs for saw wood (Nebel and Drysdale, 2010). It is suggested that only 18% of wood decays in landfill in the first 46 yrs, with none decaying after that (Nebel and Drysdale, 2010). In contrast, meat products are unlikely to last long.

Erosion has the potential to be both a source and a sink of carbon. Erosion is normally greater in pasture, than in forests. For example, Quinn and Stroud (2002) measured a threefold higher level of suspended sediment being exported from a pasture catchment than from a native forest catchment. Small amounts of carbon can also be lost as dissolved organic carbon in runoff. Quinn and Stroud (2002) found this to be 20% higher in the pasture catchment than in the native forest as well, probably due to the higher occurrence of riparian wetlands. The criterion that needs to be satisfied for soil erosion to constitute a carbon sink consists of two components (Berhe et al., 2007): (1) at least partial replacement of eroded carbon by newly fixed carbon in the eroded site, and (2) preservation of the carbon at the depositional site in more passive pools (with a longer residence time) than at the eroded site. Berhe et al.(2007) estimated that soil erosion and deposition stabilize at least 0.72 Pg C per year globally. Dymond (2010) concluded that erosion creates a net sink of 1.1-5.6 million t C yr^{-1} in New Zealand due to the quick movement of sediment to the oceans were 80% is assumed permanently buried, and due to the regeneration of eroded soils. However, these results may not apply to individual sites, as the assumptions may not hold in all landscapes.

2.3.5 EFFECT OF LAND USE CONVERSION ON SOIL CARBON

Although plant biomass clearly increases with afforestation, changes in soil carbon are less clear. A number of reviews have been done on this question. Guo & Gifford (2002) did a meta-analysis of soil carbon trends with land use change based on 74 studies, in 16 countries, most of which were from Australia, Brazil, New Zealand and the USA. They found an 8% increase in soil carbon with conversion from native forest to pasture, a 13% loss from native forest to plantation forest and, a 10% loss from pasture to plantation. The loss of carbon with the pasture-to-forest conversion was observed under conifers but not under broadleaved species.

Paul et al. (2002) also reviewed the effects of afforestation on soil carbon in 43 studies, covering tropical and temperate countries, again mostly from the USA, New Zealand and Australia. They found that carbon declined by 0.53% per yr on ex-pasture sites for the first ten years. The least amount of carbon accumulated in soils of long-rotation softwoods, particularly *P. radiata*, established on ex-improved pastoral land in temperate regions. Average changes were 14.1 g C m^{-2}, a very small amount relative to typical increases in forest mass.

Laganier et al. (2010) did a meta-analysis on the same issue, based on 33 studies in five climate zones, with a large number of the studies being done in the USA or in temperate areas of the world. The meta-analysis found that afforestation resulted in a non-significant increase in soil organic carbon stocks of 3% for pastures. An increase in carbon of approximately 12% was observed when *Eucalyptus* spp. and *Pinus* spp. were used, while planting broadleaved trees (excluding *Eucalyptus* spp.) was associated with an increase of more than 25%. Planting coniferous trees (excluding *Pinus* spp.) had little effect. However, the study raised a major methodological issue regarding the inclusion of the litter layer. Whether or not studies included litter explained a large portion of the variability in the data, with the inclusion of litter changing the broad picture from a loss of carbon under trees to a gain compared with pasture.

Overall, conversion from pasture to forest increases carbon stocks despite changes in soil carbon. Moreover, it is likely to reduce emissions by having lower N_2O and CH_4 fluxes and higher soil CH_4 consumption.

2.4 OPTIONS FOR ADDRESSING CLIMATE CHANGE

Combating climate change requires manipulating the global carbon cycle to reduce emissions and enhance sequestration. In general, emissions reductions can come from targetting the major sources: lowering the use of fossil fuels through alternative energy or more efficient use, better farming practices, and by halting deforestation.

As large amounts have already been emitted and emissions are still tied to economic growth, there is great interest in enhancing sequestration wherever possible. Lal (2008) outlines several means of doing so (Fig. 3) which are summarized below. Many options are available, but several are still in development, and fewer still are directly applicable to agriculture.

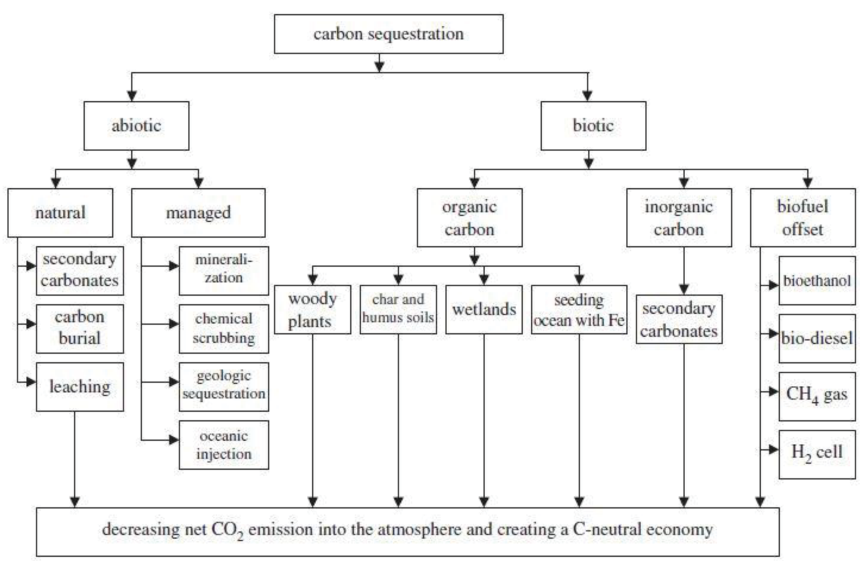

Figure 3. Methods for sequestering carbon (Lal, 2008).

Abiotic methods mainly involve engineering fixes with the potential to create large sinks, as well as some natural processes. Oceanic injection involves injecting liquefied CO_2 from industry into the deep ocean. Geological sequestration involves injecting industrial CO_2 into deep geological strata such as old oil wells, coal seams and saline aquifers. There are concerns with stability, cost, and adverse ecological impacts for both methods. Industrial CO_2 emissions can be removed and bound in stable minerals through chemical scrubbing. Energy and expense are major hurdles here. Natural processes can also occur with secondary carbonates in soil, but these have a fairly limited potential.

Biotic methods involve enhancing pre-existing biological processes that sequester carbon. Use of bio-fuels can offset fossil fuel use and potentially enhance sequestration in marginal land. Seeding the ocean with iron is aimed at enhancing photosynthetic phytoplankton growth by removing their main limiting nutrient, in the hope that it may transfer carbon to the ocean floor. This technique is controversial due to its likely effect on ocean ecology. Wetlands restoration may slow their release of carbon and turn them back into sinks. Soils often lose carbon after conversion from native vegetation. Therefore, good management may be able to raise carbon levels in soils. This can be achieved via fertilizers and manures, use of cover crops, no-tillage, bio-char and many other methods. Afforestation, restoration of degraded forests, stopping deforestation and improved management of secondary regenerating forests are all options for enhancing the growth and thus sequestration from forests.

Terrestrial biotic carbon sequestration is generally a win-win strategy, because of its multiple benefits. Engineering techniques have the potential to create much larger sinks, but are currently expensive, leaky, and may have unintended negative effects (Lal, 2008). They also require carbon to be physically captured at source which is not possible for New Zealand's outdoors livestock.

Looking more specifically at agriculture, there are three broad options that exist for mitigating GHGs (Smith et al., 2008): reducing emissions, enhancing sequestration, and avoiding emissions by use of bio-energy and not clearing new land. Native forest clearance is now low in New Zealand (Ewers et al., 2006) so avoiding clearance is a minor avenue for mitigation here. Most emissions in New Zealand pastoral agriculture are related directly to livestock, either through enteric CH_4 or N_2O from excreta and fertilizer. Therefore, the use of bio-fuels, though helpful, will only go a small way to mitigating emissions. Options for reducing on-farm emissions of ruminant CH_4 are: altering the diet, modifying ruminal fermentation, or reducing stock numbers (Pinares-Patino et al., 2009). Choices for reducing N_2O include: reducing dietary nitrogen, use of nitrification inhibitors, strategic fertilizer use and reducing soil compaction (Pinares-Patino et al., 2009). Most of these are still in development, so reducing emission is not presently possible in extensive systems (Pinares-Patino et al., 2009). This means that enhancing sequestration in trees and the soil is the only currently viable option, with trees being the only sure fire method.

2.5 PREVIOUS STUDIES OF AGRICULTURAL GREENHOUSE GAS BALANCES

Agricultural land tends to be a net source of emissions. Intensity, soil management and woody vegetation management are among the most important factors controlling this. Full GHG studies of New Zealand agricultural landscapes are lacking, although many specific components have been better investigated. A number of GHG balances have been done for various types of agricultural land overseas. The following studies provide some examples. Typically emissions are counted as positive values (additions to the atmosphere) and sequestration as negative values (withdrawals from the atmosphere).

Flessa et al. (2002) compared the net GHG balances for conventional and organic systems at the Scheyern Research Station in southern Germany. These included emissions of all gases but not sequestration. Both systems had a mix of crops, meadows and livestock. The conventional farm emitted 4.2 t CO_2 ha^{-1}, whereas the organic farm emitted 3.0 t CO_2 ha^{-1}. Organic systems could reduce emissions, but this came with a reduction in yield showing the role de-intensification can have.

Kustermann et al. (2008) developed the REPRO model based on the Scheyern Research Station. They tested it on 18 organic and 10 conventional nearby farms. Their analysis included all main gases, emission and sequestration, and technical flows as well as natural flows. They only included cropping systems. They found organic farms had lower net emissions of GHGs on a per area basis (106 – 1875 kg $C0_{2e}$ ha^{-1} yr^{-1}) compared with conventional farms (1878-3697 kg $C0_{2e}$ ha^{-1} yr^{-1}), however, all GHG balances were highly variable. The organic systems had a higher potential for sequestration in the soil, helping to lower their net emissions, whereas conventional systems tended to deplete soil carbon.

Liebig et al. (2010) analyzed the net GHG balances of grazing in the Northern Great Plains of North America. This included all main gases, fertilizer uses and the change in soil organic carbon for three different grazing regimes. 'Heavily Grazed native pasture' (1.1 animals ha^{-1}) had a net balance of -618 kg CO_{2e} ha^{-1} yr^{-1}, 'Moderately Grazed native pasture' (0.39 animals ha^{-1}) was -783 kg CO_{2e} ha^{-1} yr^{-1}, 'Heavily Grazed Crested Wheat Grass pasture' (1.2-2.3 animals ha^{-1}) was 397 kg CO_{2e} ha^{-1} yr^{-1}. Soil carbon change was the most important factor in the net balance, again demonstrating the role of sequestration in soils as a means of offsetting emissions.

Bray & Golden (2009) developed 50 yr scenarios for net GHG balances for two different grazing areas and vegetation management regimes in Australia. They included emissions and sequestration of CO_2 and CH_4 but not N_2O from soils. The first scenario for southern-central Queensland was a 4000ha property managing approximately 1000 ha of 'Brigalow' scrub re-growth. The net balance if the re-growth was retained was -71.5 t CO_{2e} ha^{-1} (-1430 kg ha^{-1} yr^{-1}), but they would be able to run very few stock. If strips of scrub were retained then it was 5.8 t CO_{2e} ha^{-1} (116 kg ha^{-1} yr^{-1}), and if all was cleared then it was 42.8 t CO_{2e} ha^{-1} (856 kg ha^{-1} yr^{-1}). The second scenario was for 'remnant eucalypt savanna-woodland' in north Queensland. Retaining the current woodland structure emitted 7.4 t CO_{2e} ha^{-1}, allowing thickening sequestered 20.7 t CO_{2e} ha^{-1} but required a 30% decrease in livestock, thinning trees less than 20cm diameter at breast height (DBH) emitted 12.4 t CO_{2e} ha^{-1}, thinning trees <10cm DBH emitted 8.9 t CO_{2e} ha^{-1}, but both allowed an increase in livestock. These results show the large effect that management of woody vegetation can have on net emissions as well as the significant trade-offs between grazing and carbon sequestration in trees.

In New Zealand, Life Cycle Analyses have been done for different agricultural sectors, including lamb (Ledgard et al., 2010) and dairy (Lundie et al., 2009). These do not account for carbon sequestration. Neither do they report absolute emissions; instead, they take an intensity approach giving emissions in kg per unit of product. They show significant amounts of emissions occur during agricultural production and they are informative for how to improve emissions efficiency, but they do not necessarily indicate if there have been actual reductions in absolute net emissions. This is because gains made in reducing the per unit emissions may be offset by increases in the number of units made.

Tools are available for giving rough estimates for a given landscape. Look up tables are available for pre-calculated amounts of CO_2 locked in the total living and dead biomass of forests of different ages (Ministry of Agriculture and Forestry, 2011d). For example, for Waikato-Taupo region post-1989 *Pinus radiata* at age ten would store 163 t CO_2 ha^{-1} (Ministry of Agriculture and Forestry, 2011d) . Regenerating indigenous forest of the same age would be 40.2 t CO_2 ha^{-1} (Ministry of Agriculture and Forestry, 2011d). These tables do not account for site specific variation so are only a rough guide. These can be used to choose the amount of area needed to offset emissions. OVERSEER, a nutrient budgeting model, can be used to calculate farm emissions (Wheeler et al., 2008).

2.6 MARGINAL LANDS & CARBON SEQUESTRATION IN HILL COUNTRY

Marginal land refers to land that, despite being able to support agriculture, is still physically poor for agriculture and thus at the economic margins of what can be cultivated profitably at a given set of prices and other factors (Peterson and Galbraith, 1932). In a New Zealand context, LUCC VIII would be the physical limits while LUCC VII is likely to be at the margin for pastoral agriculture (Lynn et al., 2009). What is marginal for one land-use is not necessarily so for another (Peterson and Galbraith, 1932). LUCC VII is still suitable for forestry. Marginal lands are an attractive area for conversion to forestry in order to combat climate change. An advantage of these areas is that, because they are poor for agriculture, there is a smaller opportunity cost for their conversion than with highly productive farmland, and, because they have normally already been cleared, there is no loss of native vegetation carbon stocks with the conversion to production forestry.

Marginal lands provide additional challenges to their use for carbon sequestration. They are likely to sequester less carbon than more fertile sites for the same reasons they are poor for agriculture. The greater spatially variability makes the choice of suitable sites for trees more difficult. In hill country landscapes the spatial patterns of carbon cycling will be much more complex than on more productive flatter land. The steep terrain creates a complex mix of differing slopes, aspects and soils. These factors are recognized for hill country pastoral systems. For example, pasture dry matter (DM) growth rates can differ from 120 kg DM ha^{-1} day^{-1} on low slopes to 35 kg DM ha^{-1} day^{-1} on steep slopes (Saggar et al., 1999). The percentage of dry matter growth that is legumes can differ by 39% to 13% from steep slopes to stock camp sites due to nutrient transfers by stock (Ledgard et al., 1987).

The GHG balance of forests may vary considerably across a farm landscape due to differences in plant growth and other factors. Differences in *Pinus radiata* forestry site fertility are commonly measured using the 300-index or Site Index. Watt et al. (2010) analyzed the relationship between these indices and environmental variables across New Zealand. The two main environmental determinants of Site Index were Mean Annual Temperature and Fractional Available Root-zone Water Storage, accounting for 49% of the variance. Temperature and water availability may differ between slopes and aspects. Gillingham and Bell (1977) measured mean daily temperature differences as large as 8.3^{o}C between steep northern slopes and steep southern slopes for a hill country pasture. Soils and water availability may also vary considerably at these scales. Therefore, it is possible that forest productivity could vary significantly at relatively finer scales in hill country landscapes. This makes generalizations from paddock-scale studies on flat land to whole landscapes difficult. Landscape scale studies in complex topography are needed to fully understand the capacities for marginal hill country land to be used as a carbon sink.

2.7 TREES IN HILL COUNTRY

The most cost-effective way of fighting climate change using trees is to stop deforestation rather than plant new forests (Stern, 2007). Farms often contain remnant native forests; protecting these fragments could be a more efficient strategy than spending to plant forestry blocks. Native bush fragments differ significantly from continuous forest. They are subject to a range of pressures which can affect their levels of biomass and their ability to sustain themselves. Livestock grazing operates independently of the forests ability to support it, and thus suppresses regeneration with the potential to cause the complete collapse of the forest (Dodd et al., 2011). Pest mammals can also weaken regeneration by seed predation (Dodd et al., 2011). Fencing and other management activities can help restore some features, particularly regeneration, undergrowth and the sub-canopy layers, but even then native forest fragments are still unlikely to maintain themselves in their original form (Burns et al., 2011). Regeneration is dominated by shorter-lived smaller species, with shade tolerant canopy trees regenerating at a much lower rate (Burns et al., 2011). Therefore, it is likely that fragments will have significantly less ability to store and sequester carbon, especially long-term, than continuous forest. Nevertheless, they may still contribute significantly to mitigating a farm's emissions and their continued degradation may actually add to emissions.

Hill country pasture can be susceptible to invasion by weeds and native forest re-growth. This regenerating scrub is a large potential sink (Trotter et al., 2005). It has the advantage of not requiring any cost to establish, compared with planting trees or with fencing and pest control to protect bush fragments. The main cost is the lost grazing land, although if in some cases controlling re-growth may be too costly anyhow.

Agro-forestry, which can take the form of block plantings, low density plantings, and rows of trees in timber belts is recommended for enhancing the sustainability of hill country farms (Blaschke et al., 1992). Poplars are commonly used for erosion control in pastoral hill country (Wilkinson, 1999). In studies on poplar trees their biomass is often not reported. Typical amounts of carbon can be estimated using diameters and allometric equations from Lodhiyal et al. (1995). Initial plantings are usually <10cm in diameter (Wilkinson, 1999). At 100 stems per hectare (sph) this gives < 6 t C ha^{-1}. For 8-11 yr old poplars at 25-400 sph, an average diameter of 31.9 cm has been recorded (Douglas et al., 2006), giving 5-87 t C ha^{-1}. In a poplar-pasture system with trees at 37 sph and 29 year old trees, plant biomass was 23.8 t C ha^{-1} compared with 9.5 t C ha^{-1} for open pasture (Guevara-Escobar et al., 2002). These values are much lower than what is possible for a full conversion to forestry, and become comparable to shrub-lands only with older trees or denser plantings. This, combined with the fact that they are typically used for small areas of localized erosion control, means that they will have much less potential for offsetting emissions than full forestry. On the other hand, poplars do increase sequestration in a way that has fewer trade-offs with pasture growth.

2.8 ECOSYSTEM SERVICES & CARBON TRADING

Ecosystem services are the benefits that people obtain from ecosystems. They include provisioning (e.g. food and water), regulating (e.g. climate stability), cultural (e.g. aesthetics) and supporting (e.g. nutrient cycling) services (Millennium Ecosystem Assessment, 2005). They are of considerable value to agriculture, yet, in the attempt to maximize tradable provisioning services many other services are often harmed (Power, 2010). GHG mitigation is one of the few services, in the case of countries with carbon trading schemes, in which an originally unvalued service has been explicitly valued, and incorporated into the wider economy.

As GHG production is an externality, a key policy approach is putting a price on carbon to internalize the cost. Taxes and carbon trading may help reduce emissions effectively, although they are unlikely to be sufficient alone (Stern, 2007). Schemes to do this have been put in place in several nations. In 2009 the global carbon market was worth US$144 billion (Kossoy and Ambrosi, 2010). There are a number of different trading schemes now running. The European Union is currently the largest carbon market. In the European Union's trading scheme over 2008-2009 prices ranged around €8 - €30 t CO_{2e} (Kossoy and Ambrosi, 2010).

New Zealand's Emissions Trading Scheme (ETS) is the first mandatory economy-wide scheme outside of Europe (Kossoy and Ambrosi, 2010). New Zealand Units (NZUs) are to be at a set price of $25 t CO_{2e} until 2012. Agriculture is yet to be included. Post-1989 forests can opt in and receive credits for carbon after January 1st 2008, and are penalized for any drops below previous levels. In 2009 international public and private buyers purchased a total of up to 600,000 forestry NZUs on the spot market at prices believed to be around US$14 (Kossoy and Ambrosi, 2010). Alongside the ETS is the Permanent Forest Sinks Initiative, which allows forest owners to earn Kyoto Protocol units for post-1990 permanent forests (Ministry of Agriculture and Forestry, 2011c).

The ETS poses both a threat and an opportunity to hill country land owners. Substantial costs may be imposed on agricultural production but forestry may become more profitable than it was previously. The actual costs that will be incurred are unknown. Only upstream processors are in the scheme and may not pass the costs directly on, and the scheme is to be eased in over a period of several years before the full costs of emissions must be paid (Ministry of Agriculture and Forestry, 2011b).

A study on the potential impact of carbon trading on the profitability of forestry on ex-farm sites in New Zealand showed that carbon trading enhances forest profitability in a way that is complementary to the traditional income from forests (Manley and Maclaren, 2010). For a *radiata* pine clear-wood regime the maximum Land Expectation Value (LEV) was \$1223 ha^{-1} at age 24 yrs. With a carbon price of \$30 t CO_2 the maximum LEV increased to \$6647 ha^{-1} at age 30. Income from carbon trading is steady across forest growth, with the greatest cost at harvest; this is the reverse of the situation for timber. Carbon trading also affected the most profitable management regime. Longer rotations were preferred with higher carbon prices, and higher prices make no thinning preferable. A potential negative for the forest owner is that it can create the risk that, if carbon prices are high, harvesting might become too expensive due to the cost of surrendering carbon units. This can be managed by planting high value trees or retaining some of the earned credits. Manley and Maclaren (2010) concluded that the ETS stands to make forestry more affordable for a large area of New Zealand agricultural land which is currently unprofitable for forestry.

2.9 PREVIOUS CARBON STUDIES AT WHATAWHATA

The Whatawhata Research Station has been a research farm since 1949 (Dodd et al., 2008b). During this time a large number of studies have been done on animal and plant genetics, animal production, soils, pasture and stream ecology. Although many of these studies touched on carbon, GHGs, or relevant processes, very few focused on it. No attempt to complete a full GHG balance for the farm has been made before.

Permanent plots set up in the native forest fragments and plantings have been used for assessments of fragment condition (Smale et al., 2008) and for a previous attempt to quantify carbon sequestration (Richardson and Burrows, 2004). Forest plots were measured in 2000 and 2002 for the diameters of all plants with a DBH >3cm. In the plantings' plots, heights and canopy width were measured in 2002 and 2003. Total carbon in aboveground biomass was estimated using allometric relationships. Richardson and Burrows (2004) used this data to estimate aboveground carbon in the vegetation. In 2000 the aboveground carbon was 110-126.5 t C ha^{-1}. In 2002 it was 116.5-153.5 t C ha^{-1}, a 4-21% increase. Plantings were 0.025-0.05 t C ha^{-1} in 2001/02 and 0.035-0.3 t C ha^{-1} in 2003, an increase of 40-369%. However, Richardson and Burrows (2004) concluded that the results were unreliable due to errors in some plots, indicated by the growth for some sites being well beyond what was thought possible for such a short time. These errors have since been rechecked and resolved (M. Dodd, personal communication, 2011). More measurements have also been done in the fragments since that study.

During the planning stages of the new land-use plan, Way (1999) undertook modelling for the pine forests using the Agroforestry Estate Model. A number of different scenarios were run. The 50% afforestation proposal is most similar to that which was actually done in terms of area and locations planted, although tree stocking in the scenarios was less than was done (600-1000 vs. 1200 sph) and final density was less (300 vs. 400 sph). Root biomass was expected to reach 31.5-43 t ha^{-1} by year 10. Assuming 20% of total biomass is roots (Beets et al., 2007)and 50% is carbon, this would predict 78.8-107.5 t C ha^{-1} at the time of the present study. The expected yield of wood was 393.4-430.6 m^3 ha^{-1} by year 20. Yields were assumed to be the same from all LUC classes, with only small differences between them. It was predicted that the Net Present Value of the farm would be increased considerably at discount rates less than around 7%, but it would be costly in the short term. The costs of implementation would make it necessary to find outside financing. The possibility of earning carbon credits was not considered and it is expected that the extra income would strengthen the profitability for forestry on the farm.

Carbon dynamics and export in the streams was studied by Quinn and Broekhuizen (2003) prior to the changes in 2000-2002, and they predicted the likely patterns over the next 60 years. Carbon stored in wood in the streams was estimated to be 200 g C m^{-2} in pasture, 1450 g C m^{-2} in native forest and 5000 g C m^{-2} in pines. Regenerating forests were not expected to lead to large increases in stream wood for about 50-70 years. Particulate organic carbon export was not directly measured, but estimates were based on suspended sediment and total organic nitrogen export for a dry year (1996-97), giving 5-13 kg C ha^{-1} yr^{-1} for native forest, 41-51 kg C ha^{-1} yr^{-1} for pasture. During a wet year (1995-96) there was estimated to be a threefold increase in exports. This estimate did not include the export of wood. In 15-20 years regenerating riparian vegetation was expected to shade out ground cover which currently holds stream bank sediment in place, leading to a large pulse of exported carbon as the channels widen. However, this may not occur as expected. The grassed riparian corridors in the pine forest are not regenerating as anticipated, possibly due to feral pigs (J. Quinn, personal communication, 2011). A study of run-off under the old land use (Quinn and Stroud, 2002) found that sediment losses in a wet year (1995-96) were 3212 kg ha^{-1} yr^{-1} and in a dry year (1996-97) were 988 kg ha^{-1} yr^{-1}. Dissolved organic carbon (DOC) loss was 33.4 kg ha^{-1} yr^{-1} in the wet year and 25.5 kg ha^{-1} yr^{-1} in the dry year. DOC appeared to come from riparian wetlands, which are more common in pasture, hence the 20-25% higher export values in pasture than in native forest catchment (Quinn and Stroud, 2002). Sampling during the land-use changes showed the new land use reduced sediment loss by 76% and DOC loss by 33% (Dodd et al., 2008a). Values for exports may be underestimates as the measurements can miss some large pulses of sediment loss during floods.

Soil carbon and leaf litter was measured in the grazed fragments in order to detect the effects of fertilizers (Stevenson, 2004). Soils were sampled at two sites each on moderate slopes at 0-10cm and 10-20cm in the main fragments of the catchment. Nearby pastures were also sampled as a reference. Coarse non-woody litter and finer decomposed litter were also measured at some points. These measurements provide valuable soils data for the original land-use. Higher soil carbon levels occurred in the pasture than the fragments. Soil carbon measurements have been made in a number of other studies, but the sampling locations on the farm were not recorded, making it difficult to attribute the value to an area or to resample for changes. Soil fertility data across the catchment were recorded during the transition period but did not include carbon.

Soil carbon changes under pasture (0-75mm depth) have been measured every few years from 1983-2006 under different phosphate fertilizer loadings on the farm in the catchment directly adjacent to the Mangaotama Catchment (Schipper et al., 2009). No effect was found from the different phosphate fertilizer loadings which the experiment tested. They did find significant variation in carbon levels between the two slope classes - easy (10-20°) and steep (30-40°). For the first 6 years, on average, carbon increased 0.270% C yr^{-1} (1.56 t C $ha^{-1}yr^{-1}$) on easy slopes and 0.156% C yr^{-1} (1.06 t C ha^{-1} yr^{-1}) on steep slopes. After this carbon declined by -0.066% $year^{-1}$ (0.45 t C ha^{-1} yr^{-1}) on steep slopes, but there was with no significant change on easy slopes. The reasons for the change were unclear, but Schipper et al. (2009) suggested it was due to climate, in particular, a series of dry summers over the latter period of the study.

Many other studies, particularly those associated with the catchment management project recorded data of relevance, e.g., fertilizer usage, stocking rates and performance, sediment loss and so on. Those that were used in the present study are mentioned in the Methods section.

CHAPTER 3: METHODS

3.1 STUDY SITE

The study site was the Mangaotama catchment within the Whatawhata Hill Country Research Station, west of Hamilton in the Waikato region of the North Island, New Zealand (Fig 4). The catchment is 269 ha of predominately steep land ranging from 100 to 250m above sea level.

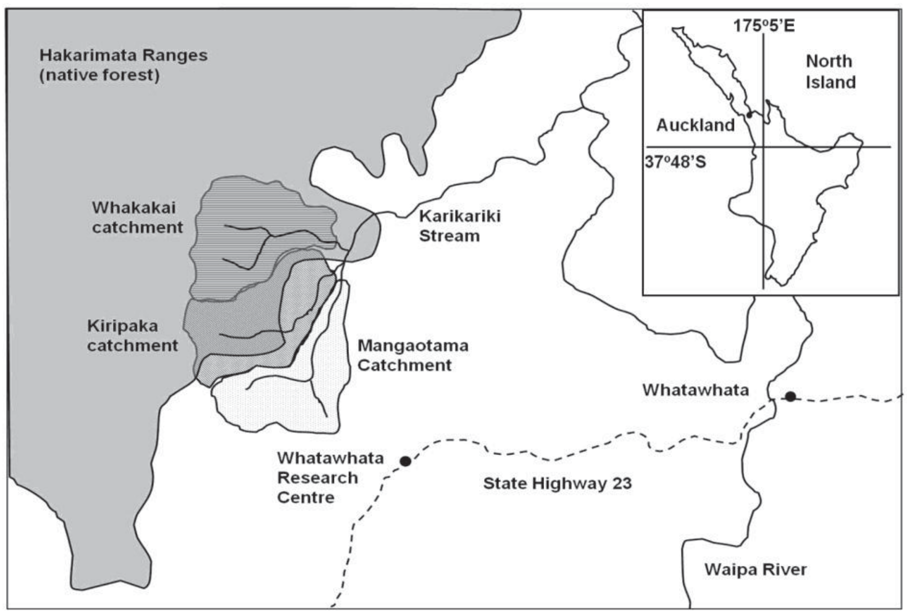

Figure 4. Location map of the study catchment (Dodd et al., 2008b).

The steep areas consist mainly of sedimentary greywacke derived soils, whereas the easier slopes carry soils containing volcanic ash of varying depths (Bruce, 1978). Kaawa hill soils and Waingaro steeplands soils clothe most of the steep slopes and Dunmore hill soils and Dunmore silt loams cover easier slopes. A small amount of Mangapiko clay loam borders the main stream. The catchment has a high density of streams and forms the headwaters of the Mangaotama stream. The streams range from 1^{st} order up to a 4^{th}order stream which exits the catchment. The land-use capability classes (LUCC) range from III to VII, with most being classes VI and VII (Fig 5).

Figure 5. Distribution of land-use capability classes within the Mangaotama catchment (adapted from maps supplied by AgResearch Ltd.).

The climate is normally moist to temperate. The mean annual rainfall between 1996 and 2006 was 1530mm, and the mean annual temperature was 13.7^{o}C between 1990 and 1999. Mean solar radiation was 13.2 MJ m^{-2} day^{-1} and mean relative humidity for 2001-2006 was 85% (calculated from data supplied by AgResearch Ltd.).

The area was originally covered in podocarp-broadleaf forest before being cleared in the 1920s for pastoral farming with the land being used as a research farm since the 1949 (Dodd et al., 2008b). Prior to 2001 the land use regime was as follows (Fig. 6): The farm was run with sheep and beef cattle with an average of 8.8 SU ha^{-1}, with a sheep to cattle SU ratio of 61:39 just prior to 2001 (Dodd et al., 2008a). 5 ha of remnant native forest persisted in some gullies. These remnants were grazed by stock and were very degraded (Smale et al., 2008).

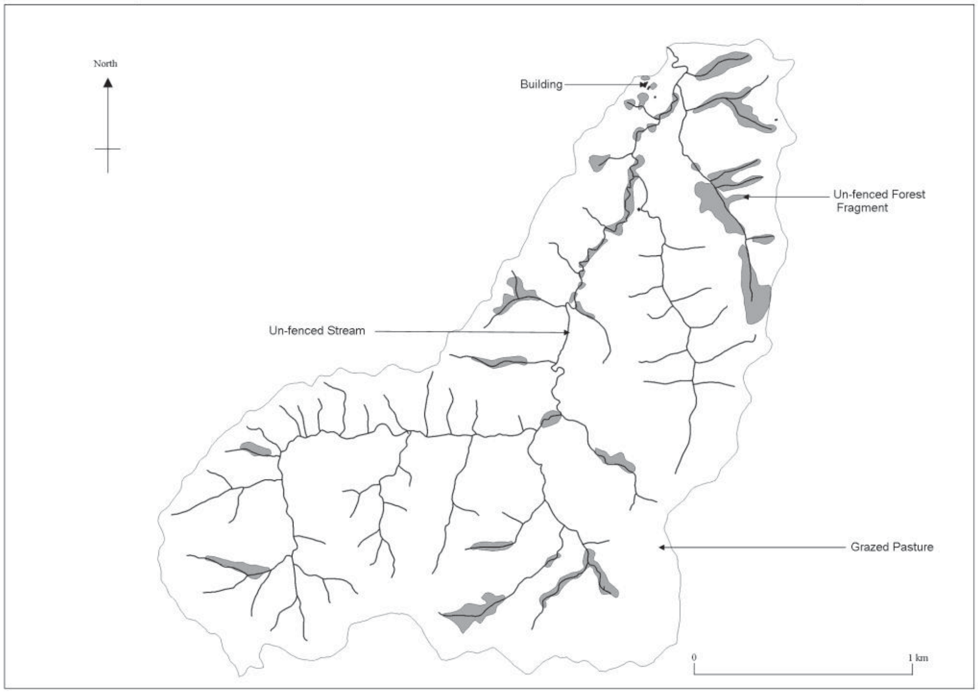

Figure 6. Land use in the Mangaotama catchment prior to 2001 (adapted from maps supplied by AgResearch Ltd.).

The area was fertilized annually with 20.7 kg ha^{-1} of phosphorous and 12.6 kg ha^{-1} of sulphur (Quinn and Stroud, 2002). Much of the land was erosion prone, with between 988-3212 kg ha^{-1} yr^{-1} of sediment being lost from the catchment (Quinn and Stroud, 2002).

In 2001 a new land-use regime was implemented as part of an integrated catchment management project (Fig 7). The aim was to improve the management of catchment to meet the desired outcomes of all stakeholders (Dodd et al., 2008b). In total, 153 ha was converted to commercial *Pinus radiata* forest planted at 1200 sph and thinned to a final density of 400 sph. This was never grazed (M. Dodd, personal communication, 2011).

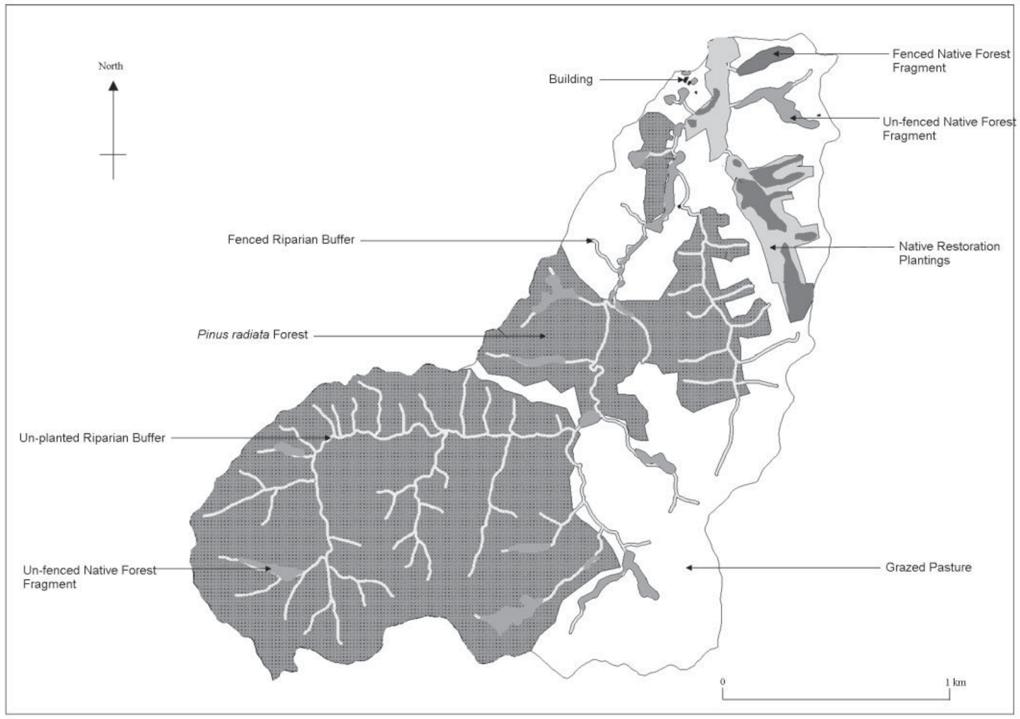

Figure 7. Land use in the Mangaotama after 2001 (adapted from maps supplied by AgResearch Ltd).

A mix of native trees and shrubs were planted over 7 ha, linking the main native forest fragments. Most of the fragments were fenced to exclude grazing. The remaining pasture was intensified. The average stocking rate was raised to 10 SU ha^{-1} with a sheep to cattle ratio of 64:36 (Dodd et al., 2008a). Fertilizer use was increased to 24 kg ha^{-1} yr^{-1} of phosphorous, 30.5 kg ha^{-1} yr^{-1} of sulphur, and 30 kg ha^{-1} yr^{-1} of nitrogen was applied to some class V land (Dodd et al., 2008a). In 2006 nitrogen fertilizer was increased to 100 kg ha^{-1} yr^{-1} of urea on all pasture (M. Dodd, personal communication, 2011).

3.2 CONCEPTUALIZATION

3.2.1 GHG BALANCE METHODOLOGIES

Constructing a full GHG balance completely from direct measurements of all fluxes is a difficult task due to the large number of stocks and fluxes and the high variability in them across space and time. For example, to measure the fluxes from the soil of CO_2, CH_4 and N_2O requires the use of field chambers and repeated measurements to capture the variation with the seasons, rainfall, soils, and other factors. This has been done in some studies for CH_4 and N_2O (Saggar et al., 2004, Saggar et al., 2007b, Saggar et al., 2008). Measurements of soil respiration face the additional problem of distinguishing root respiration, which is autotrophic and normally accounted under plant growth, from soil heterotrophic respiration. Several methods exist for eliminating roots from the soil, to eliminate root respiration, most of which are disruptive and can alter the readings (Hanson et al., 2000). Eddy covariance is a widely used technique for assessing the exchange of CO_2 between the atmosphere and plant canopy over a year, but faces limitations to it applicability, such as uneven terrain and changeable environmental conditions (Baldocchi, 2003). Erosion can be directly measured by regularly sampling suspended sediment in the streams exiting a catchment. Again, sampling needs to be regular, especially during floods when most sediment may be removed, and point measurements cannot distinguish between the contributions of different areas in the catchment. Although each stock and flow could be measured, the time, cost and effort needed to do so for all of them means this is unreasonable for an individual study. Therefore simplifications need to be made.

Net fluxes from a stock can be estimated from repeat measurements of the usually more easily measured stock. This cannot distinguish between the relative contributions of each input and output, but can indicate the net effect on the atmosphere, which is of primary concern. For example, forest biomass carbon can be measured using allometric equations relating diameter and height to biomass. Standard protocols have been made for New Zealand native forest carbon assessment (Payton et al., 2004) and allometric equations have been developed for New Zealand native forest (Coomes et al., 2002). The difference between repeat measures over time indicates if there has been a net exchange of carbon. This requires much less in the way of sophisticated equipment and can be done as often as needed for the temporal resolution desired. Issues can arise in the uncertainty of wood density values, allometric equations developed for different conditions and species, and inconsistent practices between repeat measurements.

Change in the soil carbon stock can also be measured using this same approach. Estimates of soil carbon mass requires knowing the percentage carbon, bulk density, and the depth sampled. A depth of 30cm was used in the New Zealand carbon Monitoring System (Tate et al., 2005), although different depths are often used for different studies. Greater depths will give larger estimates, but most carbon is in the top several centimeters. This method requires clear records of where past soil samples were taken, to avoid confusing differences between sites with temporal change.

Numerous models have been developed for estimating parts of the carbon cycle under specific land uses. Use of models can simplify the amount of data needed, while still providing estimates of specific areas of interest. For *Pinus radiata* there are several models available. C_CHANGE links the pine stand growth model STANDPAK and the growth portioning model DRYMAT to estimate total carbon stores in managed stands (Beets et al., 1999). It is currently used in the National Inventory (Ministry for the Environment, 2011). CenW is a comprehensive forest model linking carbon, nitrogen, energy and water dynamics which can be run for different species, including *Pinus radiata*. It incorporates a number of physiological and environmental factors for modelling forest growth and soil dynamics. An earlier version of the model was able to very successfully simulate dynamics in forest under very different conditions (Kirschbaum, 1999). Erosion models are also available. The universal soil loss equation has been adapted for use in New Zealand for predicting surface soil loss (Dymond, 2010). The NZEEM model has been developed for estimating mass movement erosion (Dymond, 2010). These erosion models can account for the different effects of slope, vegetation and soil on erosion. The problem with models is that it might not be known if the results can be trusted for a specific site. The parameters values must be sufficiently accurate and the model structure sufficiently correct to ensure they give good results. This may be a problem if the model has been developed for another place or does not include factors which are of major importance to the site.

Fluxes associated directly with livestock and technological fluxes are often less likely to vary considerably between places, and many have had national emission factors developed for New Zealand national accounting purposes. Enteric CH_4 and N_2O from pastoral soils both have national emission factors which are used for the National Inventory (Ministry for the Environment, 2011) and are employed by the OVERSEER nutrient budgeting model (Wheeler et al., 2008). N_2O emissions can be highly variable, so accurate accounting of spatial and temporal patterns can require more sophisticated modelling as is supplied in the NZ-DNDC model (Saggar et al., 2007a), but this requires many more input values. Many emissions factors for embodied emissions, fuel combustion and product decay have been derived for Life Cycle Analyses and GHG reporting. Direct measurement of embodied emissions is unreasonable for most studies due to the large number of different sources which may occur up the production line. As there should be little variation for the same types of fuel, standard emission factors for their combustion can be applied. Some nationally set values are available (Ministry for the Environment, 2009) and others have been measured in previous life cycle analyses (Barber, 2008). Agricultural and forestry products decay over time depending on what product the substance is made into, the half-life of that product and how much is sequestered long-term in landfill. Again, these measurements are beyond the scope of most studies, but are available for some products such as timber (Nebel and Drysdale, 2010).

3.2.2 SYSTEM BOUNDARIES

It is important to define system boundaries in space and time in order to clearly define what is to be included in the study. This is especially important when it comes to following the origins of imports and the fate of exports from an area. A cut off point must be defined, however arbitrarily, because one could keep tracing connections forever. The system boundaries must confine the study to the appropriate temporal and spatial scale for the problem, because patterns can change significantly at different scales (Wiens, 1989). Practical issues may limit how well this can be done in reality. For this study, the original catchment management study had already restricted the physical boundaries to the Mangaotama catchment.

The system boundary for the present study was defined as the physical limits of the catchment with the downstream limit set by a permanent water sampling weir. For inputs crossing the boundary (e.g. fertilizer and fuel) the direct emissions from combustion and the embodied CO_2 emissions associated with the item's production were included. For outputs crossing the boundary (e.g. meat, wool, sediment) only the direct emissions associated with the breakdown of the item were counted. Indirect emissions from further processing and consumption were not included. The deposition or breakdown of exported sediment was included. These off-site emissions were attributed to the particular location the item originated from or was used.

Four main land-use types where identified. Each required different approaches in analysis, were likely to differ in their GHG balance, and which provide distinct land use options for managers. The land use types were: pasture, *Pinus radiata* forestry, native plantings and native forest fragments. Some other possible land covers (described below) were not included due either to a lack of data, or due to their small contribution, or both.

Carbon stores in streams were ignored due to a lack of information on carbon in the sediment and because the estimates for carbon in wood debris (Quinn and Broekhuizen, 2003) combined with their small area suggested that total catchment carbon stores in streams would be far less than 1% of the total stores in the soil. N_2O and CH_4 emissions from streams were excluded due to a lack of information and because they were also expected to be very small based on the small area and low emission rates compared with terrestrial emissions (Wilcock and Sorrell, 2007). Stream side wetlands were also excluded, although they may have had much higher rates of denitrification. There was not enough information on their distribution to properly include them. Farm tracks, which were a mix of gravel and dirt tracks, were excluded due to their small area and treated the same as pasture. A 5 ha area had been sprayed and over-sown with a forage crop for three years prior to this study, but it was not a part of the original catchment management plan, and was deemed to have been around for too short a time and to be too small an area to justify the effort needed to properly account for it. Gorse and other woody weeds were excluded due to a lack of data and their likely small contribution. Poplar plantings were not included although this was an important aspect of the management plan. Only 1000 trees were planted (Dodd et al., 2008a) so it was expected they were likely to contribute less than a few per cent of the biomass that would be generated from the other plantings. In addition, there was no historical data on their growth. Some small sections of mature poplars and other trees were approximated by treating them as grazed native forest.

During the period of interest from 2000 to 2011, the catchment can be seen as going through three phases likely to have differing GHG balances. These are:

- the old regime pre-2001 (OLD) which was all pasture with small bush fragments;
- the transitional period of conversion to the new regime (TRA) with the establishment of young pines and natives, and riparian and forest fencing; and
- the current regime (CUR) established since 2006, after the new trees have grown and nitrogen fertilizer has been further increased.

In reality, the current regime has undergone many idiosyncratic changes in its livestock practices post 2004, but this has been due to the Whatawhata Station's role as a research farm rather than the dictates of the management plan. Thus, for the purposes of the study it was assumed that the livestock practices from the plan were continued to the present.

All major stocks and flows of both natural and technical flows of CO_2, CH_4 and N_2O were included. Embodied emissions associated with capital flows were excluded because they were not a major part of the regime change. Emissions associated with the workers respiration, waste disposal, etc., were also excluded. Electricity for buildings was excluded because it would not have changed much with the management changes and would likely be a very small amount.

3.3 GLOBAL WARMING POTENTIALS

CO_2, CH_4 and N_2O were all expressed in CO_2 equivalents (CO_{2e}) using 100yr GWPs from the IPCC 4th assessment (IPCC, 2007b). These were: CO_2 = 1, CH_4 =25, N_2O = 298.

3.4 PASTURE

3.4.1 PASTURE EMISSIONS

This section covers the procedure used to derive the emissions from the pastoral component of the catchment.

3.4.1.1 ENTERIC METHANE & NITROUS OXIDE

OVERSEER® was used to calculate enteric CH_4 emissions and N_2O emissions from the soil associated with nitrogen inputs from animal excreta and nitrogen fertilizer. OVERSEER estimates CH_4 emissions based on the national inventory accounting method, which uses the number of stock units to estimate dry matter intake multiplied by an emission factor based on pasture digestibility (Wheeler et al., 2008). For N_2O it uses national inventory emission factors which calculate kg CO_{2e} emitted per ha, based on the amount of fertilizer input and the amount of nitrogen excreted, derived from animal nitrogen intake minus losses in agricultural products (Wheeler et al., 2008).

The pasture was divided up by LUCC and each land class was modeled separately. The inputs used and the values used to derive them are given in Tables 1 and 2. Classes III to VI were entered as ryegrass clover because they were predominately High fertility pastures whereas class VI was entered as brown-top as this was predominately low quality pasture. For the original regime the number of stock units per ha was divided into sheep and cattle using the sheep:cattle ratio. Stock unit values for class III land were unavailable, so were given the same value as class IV. For the transitional regime only the average stocking rate for the whole farm was available. This was converted into stocking rates for each LUCC by assuming the proportionality between the average and each LUCC was the same as in the old regime. Again this was split into cattle and sheep using the sheep:cattle ratio. The current stocking rates were assumed to be the same as during the transitional period. Due to the exclusion of stock from streams, the grassed riparian zones had no enteric CH_4 emissions.

Table 1. Values used in deriving the stocking rates for input into OVERSEER. (SU = stock units, LUCC = Land use capability class, OLD = old land use, TRA/CUR = transitional and current land uses, Na = not available)

Parameter		OLD	TRA/CUR	Reference
SU ha^{-1} for	III	Na	Na	Na
LUCC	IV	18.1	Na	(Dodd et al., 2008b)
	V	12.2	Na	(Dodd et al., 2008b)
	VI	9.8	Na	(Dodd et al., 2008b)
	VII	7.3	Na	(Dodd et al., 2008b)
Sheep:Cattle		61:39	64:36	(Dodd et al., 2008a)
Average SU ha^{-1}		8.8	10.0	(Dodd et al., 2008a)

Table 2. Inputs for the OVERSEER model used in the calculation of enteric methane and nitrous oxide emissions in grazed pasture.(SU = stock units, LUCC = land use capability class, OLD = old land use, TRA/CUR = transitional and current land uses)

Regime	Parameter	LUCC	Value
OLD	Sheep SU ha^{-1}	III	11.0
		IV	11.0
		V	7.4
		VI	6.0
		VII	4.5
OLD	Cattle SU ha^{-1}	III	7.1
		IV	7.1
		V	4.8
		VI	3.8
		VII	2.8
TRA/CUR	Sheep SU ha^{-1}	III	12.87
		IV	12.87
		V	8.68
		VI	6.97
		VII	5.19
TRA/CUR	Cattle SU ha^{-1}	III	7.24
		IV	7.24
		V	4.88
		VI	3.92
		VII	2.92
All	Pasture Class	III-VI	Ryegrass Clover
		VII	Brown-top
TRA	Nitrogen Fertilizer kg N ha^{-1} yr^{-1}	V	30
CUR	Nitrogen Fertilizer kg N ha^{-1} yr^{-1}	all	46.67
OLD	Wool kg^{-1} ha^{-1} yr^{-1}	III	44.53
		IV	44.53
		V	30.01
		VI	24.11
		VII	17.96
TRA/CUR	Wool kg^{-1} ha^{-1} yr^{-1}	III	52.77
		IV	52.77
		V	35.57
		VI	28.57
		VII	21.28

N_2O was calculated at the same time as enteric CH_4 using the same stocking rates. As with changes in stocking rates after 2004, it was assumed production also remained the same. Wool production for each LUCC was derived using an average wool production per stock unit of 4.1 kg yr^{-1}. Records for this value from the start of the new regime (data provided by AgResearch) were nearly constant across years, so were applied to all years. This value was then multiplied by the LUCC specific sheep stocking rate to give yearly per ha wool production values. The '% beef as male' was set as 0% for OLD, and 100% for TRA and CUR when the Angus breeding herd was replaced with a bull beef enterprise (Dodd et al., 2008a). All time periods were set as 'finishing beef' because the original system also involved a finishing component (Dodd et al., 2008b). Nitrogen fertilizer was only applied to some class V land during the transitional period (Dodd et al., 2008a), but the specific areas were not identified, so it was assumed it was applied to all class V pasture. In 2006 nitrogen fertilization was extended to all pasture at 100kg of urea ha^{-1} yr^{-1} (M. Dodd, personal communication, 2011). For the ungrazed grassed riparian zones a value of 370 kg CO_{2e} ha^{-1} yr^{-1} was derived for N_2O emissions using the N_2O global warming potential and a value of 3.4 g N_2O ha^{-1} day^{-1} for ungrazed pasture (Saggar et al., 2007b).

3.4.1.2 FUEL COMBUSTION EMISSIONS & EMBODIED EMISSIONS

It was assumed that petrol use and herbicide use for all years were the same as they were in 2011. Weed control has been ground-based spot spraying of metsulfuron on less than one hectare per year (M. Dodd, personal communication, 2011). It was excluded from the balance because this would contribute only a few kg of CO_2 to the entire catchment.

Helicopter fuel use was estimated using a fuel consumption value of 100 l hr^{-1} (McCallum, 2009) and the flying time for fertilizer application of 23.4 ha hr^{-1} was derived from the average for TSP and DAP fertilizer application from McCallum (2009). All fertilizer was assumed to be applied by helicopter, as was the nitrogen since 2006. Fertilizer was assumed to be applied in the same manner each year, in two runs, once for nitrogen and once for the other fertilizers, thus when nitrogen fertilizer was applied per ha fuel use was doubled. No inputs were attributed to the ungrazed grassed riparian zones. Table 3 gives the values used for these parameters and Table 4 lists the emission factors.

Table 3. Amounts of inputs used on pasture during the different time periods, used in calculating embodied and combustion emissions. (OLD = old land use, TRA = transitional period, CUR= current land use).

Regime	Input	Amount Used	Reference
OLD	Vehicle petrol (l ha^{-1} yr^{-1})	5.5	(M. Dodd, personal communication, 2011)
	Helicopter fuel (l ha^{-1} yr^{-1})	4.27	(McCallum, 2009)
	Phosphorous (kg ha^{-1} yr^{-1})	20.7	(Quinn and Stroud, 2002)
	Sulphur (kg ha^{-1} yr^{-1})	12.6	(Quinn and Stroud, 2002)
TRA	Vehicle petrol (l ha^{-1} yr^{-1})	5.5	(M. Dodd, personal communication, 2011)
	Helicopter fuel (l ha^{-1} yr^{-1})	4.27	(McCallum, 2009)
	Helicopter fuel on Class V (l ha^{-1} yr^{-1})	8.55	(McCallum, 2009)
	Phosphorous (kg ha^{-1} yr^{-1})	24.6	(Dodd et al., 2008a)
	Sulphur (kg ha^{-1} yr^{-1})	30.5	(Dodd et al., 2008a)
	Nitrogen on Class V (kg ha^{-1} yr^{-1})	30	(Dodd et al., 2008a)
CUR	Vehicle petrol (l ha^{-1} yr^{-1})	5.5	(M. Dodd, personal communication, 2011)
	Helicopter fuel (l ha^{-1} yr^{-1})	8.55	(McCallum, 2009)
	Phosphorous (kg ha^{-1} yr^{-1})	24.6	(Dodd et al., 2008a)
	Sulphur (kg ha^{-1} yr^{-1})	30.5	(Dodd et al., 2008a)
	Nitrogen (kg ha^{-1} yr^{-1})	46.67	(M. Dodd, personal communication, 2011)

Table 4. Emission factors for embodied emissions and fuel combustion for inputs used on pasture. Emission factor x amount of input = kg Co_{2e}.

Input	Emission Factor	Reference
Petrol (l)	2.735	(Barber, 2008)
Helicopter fuel (l)	2.671	(McCallum, 2009)
Phosphorous (kg)	0.9	(Saunders et al., 2006)
Sulphur (kg)	0.3	(Saunders et al., 2006)
Nitrogen (kg)	3.25	(Saunders et al., 2006)

3.4.1.3 EMISSIONS FROM PRODUCT DECAY

Product per stock unit per year of wool, lamb, and beef were calculated and converted into their carbon content and then expressed as CO_2. The grassed riparian zones were not grazed thus produced nothing.

It was assumed that all meat (livestock live weight) produced was completely respired within the year it is produced. Although wool is likely to have a longer lifespan, (e.g. 9 years for carpet (Petersen and Solberg, 2004)) a static approach was taken, ignoring the time delays in emissions and counting all emissions from product decay in the year of their production. This avoids misattributing emissions from previous land use practices to the new ones over the transition period. No wool was assumed to remain as a long-term sink in landfill due to a lack of information.

The same values for wool production as in OVERSEER (Table 2) were used. They were converted into carbon based on wool being 51% carbon (Crawshaw and Simpson, 2002) then expressed as CO_2.

Beef live-weight was estimated from the reported live-weight per ha values (Table 5) divided by the average SU ha^{-1} value for cattle calculated using the average stocking rates and sheep:cattle ratio (Table 1). This gave the LW produced per stock unit of cattle, so that the difference in stocking rates across the farm could be accounted for. Lamb live-weight was calculated the same way as for beef, giving the LW of lamb produced per sheep stock unit.

Table 5. Beef and lamb live-weights used in the calculation of emissions from decay of agricultural products (LW = Live-weight).

Time		**kg LW ha^{-1}**	
OLD	Beef	86	(Dodd et al., 2008a)
	Lamb	116	(Dodd et al., 2008a)
TRA/CUR	Beef	232	(Dodd et al., 2008a)
	Lamb	217	(Dodd et al., 2008a)

These values were multiplied by the stocking rate for each LUCC (Tab 2) to give live-weights per ha for each land class. The live-weights were converted into CO_2 based on meat being 21% carbon (assuming meat is 37% organic matter (Warris, 2000) and an organic matter to carbon conversion factor of 0.58).

3.4.2 SEQUESTRATION

This section covers the procedure used to derive any possible sequestration occurring in the pastoral component of the catchment.

3.4.2.1 SOIL CARBON CHANGE

Schipper et al. (2009) studied long-term trends in soil carbon on the same farm in a pasture area directly adjacent to the study catchment. They considered the imbalances they saw to be due to a difference between sequestration and emissions. The soil changes differed between slope classes and were unrelated to the different fertilizer regimes. Therefore, the values from Schipper et al. (2009) can be applied here, even though under slightly different conditions. During the period of interest, the carbon stocks declined by 450 kg C ha^{-1} yr^{-1} on steep slopes but did not change on easy slopes. These values were converted into CO_2 and counted as emissions. These soil carbon estimates were only for the top 7.5cm layer of soil only, so do not account for possible changes occurring at deeper depths (0-20cm) that were included under other land uses.

Stevenson (2004) sampled two sites in pasture in 2001 which are still in pasture today. These were re-sampled to check for any changes (see section 3.4.3.1 in this chapter). The results confirmed that the direction of the trend was consistent with Schipper et al. (2009), but provided too few data points to use for a different estimate at both depths.

3.4.2.2 SOIL METHANE FLUX

Soil CH_4 consumption was approximated by taking the value of -0.64 kg CH_4 ha^{-1} yr^{-1} from a study done for a high intensity sheep grazed pasture (Saggar et al., 2007b). For the ungrazed grassed riparian zones a value for ungrazed pasture of -0.85 kg CH_4 ha^{-1} yr^{-1} was used (Saggar et al., 2007b). Both were converted into kg CO_{2e} ha^{-1} yr^{-1} using the GWP for CH_4.

3.4.3 CARBON STOCKS

This section covers the procedure used to derive carbon stocks in the pastoral component of the catchment.

3.4.3.1 SOIL CARBON STOCKS

Pasture carbon stocks were estimated from Schipper et al. (2009) by taking their 1996 %C values for all treatments (as the treatments had no significant effect) and deriving an average % carbon value for easy and steep slopes. Schipper et al. (2009) samples were done to 7.5cm depth, so this was extrapolated to 10cm by assuming no change over the next 2.5cm. The mass of carbon was calculated by multiplying the Schipper et al. (2009) % carbon values, their average bulk density values and the 10cm sample depth. This value was then extrapolated to the second 10-20cm depth by taking from Stevenson's (2004) pasture sites the average of the subsoil carbon mass over the topsoil mass, and multiplying the topsoil estimates from Schipper et al. (2009) by this amount. These carbon masses for 1996 on both slopes and at the two depths were extrapolated through time using the average loss of carbon per year from Schipper et al. (2009). The year 2000 value was taken as the baseline.

The current soil carbon values were assessed by field measurements. These were done in the same locations as Stevenson's (2004) pasture sites, which remained in grazed pasture at the time of sampling. For each of the two sample sites, 10 cores, for 0-10cm and 10-20cm, were taken randomly around the site (i.e. 20 cores total). Cores were bulked by depth. % C was analyzed by the Waikato University laboratory using a LECO furnace.

Bulk density was measured using a core of 88.6cm^3 volume. Two samples were taken at 0-5cm depth and at 10-15cm depth for each site, 4 cores total per site. They were then dried at 105^{o}C to a constant weight. Bulk density was calculated from the dry weight and divided by the volume of the core. The soil carbon mass was calculated by multiplying bulk density, the sample depth and % carbon.

3.4.3.2 OTHER CARBON STOCKS

Carbon contained in pasture plants and livestock were ignored, because their small biomass per hectare would likely make them less than a few per cent of the soil stock, and likely within the margins of error for the soil carbon estimates.

3.5 NATIVE FOREST FRAGMENTS

3.5.1 NATIVE FOREST EMISSIONS

This section covers the procedure used to derive any emissions from the native forest component of the catchment.

3.5.1.1 ENTERIC METHANE & NITROUS OXIDE

Because all the fragments were grazed under the old regime, these emissions were calculated as for pasture, with each area of forest receiving the same value as the LUCC that it was in. In reality this will be an overestimate, because the forests were likely to be grazed less intensely than the pasture; however, there were no records on this. Under the transitional and current regime, ungrazed forests were assumed to have zero enteric CH_4 emissions. Emissions from any possible feral animals were excluded due to a lack of information. N_2O from ungrazed fragments was given an approximation of 0.7 kg N_2O ha^{-1} yr^{-1} from a study of 80yr old *Kunzea ericoides* stands which had been ungrazed for 6yrs (Price et al., 2010). This value was converted into kg CO_{2e} using the GWP for N_2O.

3.5.1.2 FUEL COMBUSTION EMISSIONS & EMBODIED EMISSIONS

Embodied emissions and combustion emissions associated with fertilizer were excluded because any actual fertilizer received was accidental, being associated with the adjacent pastoral land use. Emissions from vehicle fuel associated with managing stock were included when the forests were grazed because this was intentional; the same value as for pasture was used. Herbicide use was excluded for the same reasons as in pasture. Ungrazed forests received no inputs, thus, had no associated embodied emissions. Possible emissions associated with the pest control program were excluded due to a lack of data.

3.5.1.3 PRODUCT DECAY

Emissions via the breakdown of meat and wool produced from the grazed forests were calculated as for pasture using the value for the LUCC the forest was in. Ungrazed forests did not produce any agricultural products, thus, there were no emissions from product decay.

3.5.2 SEQUESTRATION & CARBON STOCKS

This section covers the procedure used to derive the carbon stocks and sequestration in the native forest component of the catchment.

3.5.2.1 FOREST GROWTH & PLANT CARBON MASS

Thirty nine permanent 5x10m plots had previously been established in the four main fragments along transects perpendicular to the direction of the forest gullies (Smale et al., 2008). All trees were tagged, but tree ferns were not. The plots had been previously measured in 2000, 2002, 2004, and 2008. Measurements included the diameter at breast height (DBH) of all stems >3cm and the basal area of coarse woody debris (CWD) crossing three parallel 10m lines in the plot. Tree heights, litter mass and CWD mass were not measured.

Twenty seven of the plots were re-measured in November 2011. All live trees, tree ferns (>1.3m tall) with a DBH >3cm diameter were measured and had their DBH and species recorded. Large dense thickets of lianas were excluded because they had not been measured in previous years, but this only affected one plot. Dead standing trees >10cm DBH were measured for DBH and the species was recorded if possible. They were assigned a decay class based on their appearance (Payton et al., 2004):

0 =Live fallen tree.

1 =Wood hard but not stained, visible growth rings, bark intact.

2 =Wood hard but stained, growth rings indistinguishable, bark usually sloughing off. Core of the log remains solid.

3 =Advanced wood decay with some loss of original form, bark may remain intact in some species. Wood friable, no solid core.

4 =Characters of decay class 3, but loss of shape precludes volume measurement. Logs of decay class 4 are not measured.

Heights of all tagged trees and the tree ferns and dead standing trees were measured using a clinometer and the height was calculated using trigonometry.

The heights of trees that did not have height measurements, and all the heights for years prior to 2011, which lacked height data, were back-estimated using linear regressions between DBH and height using the 2011 data from the measured trees (SAS Enterprise Guide v.4.2). This was done for each species separately. Where the regression by species was not significant at α= 0.1 level, a regression using all individuals of all species, separately only by trees and tree ferns, was used instead (Table 6).

Table 6. Slopes and intercepts from linear regressions of 2011 heights and DBH used in the estimation of unmeasured tree heights in native forest. Those marked with a dash were not significant at a 0.1 level and the values given for 'Trees' or 'Tree Ferns' were used. Na = not applicable.

Species	Slope	Intercept	r^2	p-value
Beilschmiedia tawa	0.290	6.235	0.19	0.08
Knightia excelsa	0.350	4.882	0.80	0.01
Dicksonia squarrosa	0.136	2.887	0.09	0.07
Dysoxylum spectabile	0.138	5.673	0.96	0.003
Cyathea dealbata	0.163	1.440	0.21	0.001
Carpodetus serratus	-	-	-	Na
Dacrydium cupressinum	-	-	-	Na
Dacrycarpus dacrydioides	-	-	-	Na
Hedycarya arborea	-	-	-	0.3
Kunzea ericoides	-	-	-	0.3
Laurelia noveazelandiae	-	-	-	Na
Litsea calicaris	-	-	-	Na
Leucopogon fasciculatus	-	-	-	Na
Melicytus ramiflorus	-	-	-	0.4
Nestegis lanceolata	-	-	-	Na
Oleria rani	-	-	-	Na
Phyllocladus trichomanoides	-	-	-	Na
Rhopalostylis sapida	-	-	-	Na
Streblus heterophyllus	-	-	-	Na
Cyathea medullaris	-	-	-	Na
Cyathea smithii	-	-	-	Na
Trees	0.2110	6.0512	0.34	<0.0001
Tree Ferns	0.1233	2.6147	0.16	<0.0001

Tree biomass was estimated using the following allometric equation (Coomes et al., 2002) adding 25% for root mass and using a carbon concentration of 50%:

Above ground biomass (ABG) = $0.0000598p(d^2h)^{0.946}(1-0.0019d)+0.03d^{2.33}+0.0406d^{1.53}$

Where: h= height (m), d = DBH (cm), p = species-specific wood density (kg/m^3).

Total Biomass = AGB x 1.25

Total Carbon kg ha^{-1} *= Total Biomass x 0.5*

Species-specific wood density data was taken from Richardson et al. (2009). Data for *Nestegis lanceolata*, *Streblus heterophyllus, Carpodetus serratus, Dysoxylum spectabile, Laurelia novae-zelandia, Oleria rani, Leucopogon fasciculatus* and *Rhopalostylis sapida* were unavailable, so they were given the value of 600 kg/m^3 for 'unknown species'.

The change in live carbon for each plot was expressed as a yearly change by taking the difference between the total carbon mass for each plot and the mass when it was previously measured, and dividing this by the number of years between the measurements.

All CWD with a diameter >10cm were measured for their length and two widths at both ends. A decay class was assigned as above. If decay varied along the length it was split into sections. The species was identified if possible. If not, it was simply classified as a tree or a tree fern. CWD carbon mass was calculated from the following the equations of Coomes et al. (2002):

$$Volume = ((\pi l)/32) \times ((a+b)^2+(c+d)^2)$$

Where: l=length, a & b = diameters at one end, c &d = diameters at the other end.

Carbon mass = Volume x (fresh wood density) x (decay stage modifier) x 0.5

In the case of standing dead trees, some heights were measured directly, and both ends widths were measured if they could be reached. If the top end could not be reached, the DBH was recorded. If the height could not be directly measured, the height was calculated as for the live trees. Two standing dead tree ferns did not have their height measured so it was generated from a regression of diameters and heights from the measured trees in the same class ($y=0.259x^{1.133}$, $r^2 = 0.38$). One dead standing *Kunzea ericoides* was also unmeasured so was estimated using a regression from two unknown trees ($y=0.219x+2.935$, $r^2=1$). For the pieces with a diameter and height measurement, the volume was calculated by assuming the piece was a perfect cylinder.

The 'all other log' decay stage modifier was used (Coomes et al., 2002) as none of the species in their list were present. Wood densities were the same as those used for live trees. Unidentified tree ferns were the most common debris. They were given a value of 207 kg/m3, from the average of the density for *Cyathea* *dealbata* and *Dicksonia squarrosa*, the two most common ferns. Unknown trees were given the 600 kg/m3 for 'unknown species'.

CWD basal area, rather than volume, had been measured in previous years. In order to estimate previous years' volumes, basal area was also measured in 2011. The diameter of all CWD >3cm diameter crossing the top and bottom boundaries and a parallel middle (10m) line were measured. A linear regression between basal area and CWD mass for the 2011 data was carried out in SAS Enterprise Guide v.4.2. This had a slope of 0.155, an intercept of 3.807, a p-value of 0.12, and an r^2 of 0.097. Despite this poorly significant relationship, it was used, as there was no other way to estimate previous years CWD debris mass. Thus, there is considerable uncertainty in the estimates for previous years CWD amounts.

The percentage cover of litter was visually assessed. All litter <10cm diameter was sampled from a 17.5cm x 17.5cm litter covered area near to the centre of each plot. The collected litter was then dried at 60°C for 72 hours and weighed. The weight and % cover were used to give the mass of litter for the plot.

Litter weights were converted into carbon by using a conversion factor of 0.58 assuming they were 100% organic matter. Litter mass was adjusted for the area by multiplying the portion of the area covered in litter by the weight of the litter. Litter mass had not been measured in previous years, so 2011 weight data was used for all years and adjusted using their percentage cover values. This assumed the depth of litter did not vary significantly because there was no information on this. For those plots not measured in 2011, the previous 2008 measurement value for percentage cover was used, along with the average weight for the other plots.

Because previous year's height data was estimated via regression equations, whereas the 2011 data was from field measurements, this lead to a false comparison between 2011 and the previous years. The height of an individual tree could increase or decrease not due to any actual changes but due to the discrepancy between the regression model's prediction and actual individual's height. Therefore, the yearly gain in carbon between 2008 and 2011 could not be reliably calculated.

The total plant-associated carbon stocks were calculated by summing the live plant carbon, CWD, and litter values for each plot. The yearly change in carbon from the plant-associated stocks was calculated from the difference between the samples for each plot divided by the number of years between samples. The average carbon stock and yearly change for all plots for each year were used for all native forest fragments. Carbon stocks in un-sampled years were estimated by using the value of the preceding sampled year plus the yearly amount of change. Because the 2008 – 2011 yearly amount of change could not be calculated, the average of all years' previous change rates was used for this period. As a result, the 2011 carbon stock used in the final analysis was an extrapolation and differed from that actually measured, being approximately 18 t C ha^{-1} less.

Statistical tests were done in SAS v. 9.2. Mixed-model ANOVAs were done on the calculated carbon mass and carbon gain per year for the plots for each year by slope and aspect, with the forest fragment as a random factor. Interaction effects were checked for.

3.5.2.2 SOIL METHANE FLUX

For the forest fragments an approximation of -4.73kg CH_4 ha^{-1} yr^{-1} was taken from a study of 80yr old *Kunzea ericoides* stands which had been ungrazed for 6yrs (Price et al., 2010). This was converted into kg CO_{2e} ha^{-1} yr^{-1} using the GWP for CH_4.

3.5.2.3 SOIL CARBON STOCKS

Stevenson (2004) measured soil carbon at 0-10cm and 10-20cm with two samples in each of four main areas of forest on moderate slopes in June and August of 2001, when these fragments were still grazed. The original data presenting the carbon for the two samples at the two depths in mg C cm^{-3} was available (Table 7). These were converted into t C ha^{-1}. The average of the two samples was taken, and the two depths summed, to give the soil carbon stock for each forest fragment. Areas of forest outside the main fragments were given the average value of the four fragments.

Table 7. Data used in the calculation of soil carbon stocks for the native fragments under the original land-use regime. From the study by Stevenson (2004).

Fragment name	Sample	Depth	mg C cm^{-3}
Rewarewa	1	0-10	66.96
	2	0-10	66.99
	1	10-20	31.75
	2	10-20	33.58
Kahikatea	1	0-10	53.14
	2	0-10	67.00
	1	10-20	33.24
	2	10-20	23.94
Rimu	1	0-10	44.93
	2	0-10	18.72
	1	10-20	27.37
	2	10-20	15.35
Kohekohe	1	0-10	45.12
	2	0-10	35.49
	1	10-20	47.50
	2	10-20	21.55

The current soil carbon stocks were calculated from samples taken in October 2011. Six of Stevenson's (2004) sites were re-sampled, two in the grazed forest and four in the ungrazed forest. The same field method was followed as for pasture. Significant differences between the soil carbon mass calculated for the 2001 and 2011 samples were tested for in SAS Enterprise Guide v.4.2 using paired t-tests. As no significant difference was found for grazed forests, the 2011 data was used as to calculate the soil carbon stock for all years. For the ungrazed forests the 2001 data was set as the baseline and the intervening years were estimated by assuming there was a linear change from this to the 2011 value.

3.6 NATIVE RESTORATION PLANTINGS

3.6.1 EMISSIONS

This section covers the procedure used to derive the emissions in the native restoration plantings component of the catchment.

3.6.1.1 ENTERIC METHANE & NITROUS OXIDE

The plantings were not grazed so no emissions from ruminants occurred. Any possible emissions from feral animals were ignored due to a lack of data. N_2O emissions were set at zero based on a study of 8-12yr old ungrazed *Kunzea ericoides* stands which found emissions of N_2O to be undetectable (Price et al., 2010).

3.6.1.2 FUEL COMBUSTION EMISSIONS & EMBODIED EMISSIONS

Combustion and embodied emissions were associated with the planting and early management of the restoration plantings. In 2001, 19 700 plants were planted in 7 ha (Dodd et al., 2008a), and release spraying was done (Dodd, 2002). In 2002, due to winter losses, 13% infill planting (about 365 sph) was needed (Dodd, 2002); two release sprays were carried out in 2002 with another in 2003. Emissions associated with the pest animal control were ignored.

Detailed information on the use of inputs was not readily available so, given the small contribution of these emissions, rough estimates were used. Release spray was assumed to use 3kg ha^{-1} of 'general herbicide'. Planting was assumed to be done with hand tools. Vehicle travel was included for planting and spraying activities to account for transport of plants and workers, all of which were assumed to come from Hamilton. Vehicle emissions were estimated based on the use of one large vehicle travelling 30km twice each day, with three weeks work for the initial planting and one week's work for the infill planting. Vehicle travel emissions were attributed evenly across the plantings by dividing the emissions by the area of the plantings to give a per ha value for the emissions. Sprayings were assumed to take 2hrs per ha, thus added 2 days additional travel per spray. Table 8 gives the result of these estimates and Table 9 lists the emission factors used to convert the Table 8 values into emissions.

Table 8. Amounts of inputs applied to native plantings during the planting and establishment phases used in calculating embodied and combustion emissions.

Input	2001	2002	2003
Vehicle Travel (km ha^{-1}yr^{-1})	197	94	17
Herbicide (kg ha^{-1} yr^{-1})	3	6	3

Table 9. Emission factors for embodied emissions, and fuel combustion for inputs used on native plantings. Emission factor x amount of input = kg CO_{2e}.

Input	Emission Factor	Reference
Petrol use (km)	0.310	(Ministry for the Environment, 2009)
Herbicide (kg)	18.6	(Saunders et al., 2006)

3.6.1.3 PRODUCT DECAY

The fragments were ungrazed so did not produce any agricultural products.

3.6.2 SEQUESTRATION & CARBON STOCKS

This section covers the procedure used to derive the carbon stocks and sequestration in the native restoration plantings component of the catchment.

3.6.2.1 PLANT GROWTH & PLANT CARBON MASS

Most of the plantings were of mixed native shrubs and trees, but some much smaller areas had been planted in single species stands of native trees. These stands visibly differed in size from each other and the mixed plantings. For simplicity, only the mixed plantings were included in the analysis and their values applied to all areas.

Thirty three permanent plots had been previously set up in the mixed plantings. They covered a range of slopes and aspects. Plot sizes were mostly 5 x 15m but some were 9x8m and 7.5 x 15m. They were measured in February 2002, October 2002, and 2006. Widths (of the whole plant's canopy) and heights of all trees in the plot were measured, but trees were not tagged. The trees were still small so width and height measurements had been taken, rather than DBH as in the fragments.

The trees' widths were not measured in 2006, so these were estimated by assuming that the previous measurements proportions between height and width were constant. Some trees did not have a previous measurement to use. Therefore, if the tree had been skipped in October 2002, then the February 2002 values were used. If it had not been measured in February either then the proportions of an individual of the same species either from the same plot or a plot with the same aspect and slope was used. Only one width had been taken in these years, but two perpendicular widths were needed to calculate the tree's volume, so it was assumed that both widths were the same.

Tree carbon mass was calculated as follows:

Total Carbon = Width 1 x Width 2 x height x shrub density x 1.25 x 0.5

Shrub density values were taken from Coomes et al. (2002). *Kunzea ericoides* was given the value for *Leptospermum scoparium* (1.54 kg m^{-3}) but all other spescies were assigned the value of 'other species' (1.17 kg m^{-3}) because they were not on the published list. Destructive sampling to derive shrub density values was considered inappropriate as they had been planted for conservation purposes.

Based on visual observations in 2011, it was assumed that CWD and litter would be insubstantial in the plantings so they were ignored for all years. There was no CWD or litter data for previous years with which 2011 data could be compared if it had been collected.

Statistical tests were done in SAS Enterprise Guide v.4.2. One-way ANOVAs were done on the calculated carbon mass and carbon gain per year for the plots for each year by slope, aspect, and planting section. In cases where there was unequal variance as tested by Levene's test, the Welch's variance–weighted ANOVA was used instead. Interaction effects were checked for with two-way ANOVAs.

The February 2002 measurements were assumed to be sufficiently similar to the initial conditions in 2001, so were set as the value for 2001. The October 2002 measurements were used as the value for 2002. The amount of yearly carbon change between the samples was calculated by taking the difference between them divided by the amount of time in years. Growth from 2006 to 2011 was extrapolated by assuming that each plot would maintain the same rate of yearly carbon accumulation as it had maintained from October 2002 to 2006. The average of all plots for each year was used as the carbon stock, and was applied to all planted areas.

3.6.2.2 SOIL METHANE FLUX

For the plantings an approximation of -0.66 kg CH_4 ha^{-1} yr^{-1} uptake was taken from a study of 8-12yr old ungrazed *Kunzea ericoides* stands (Price et al., 2010). This was converted into kg CO_{2e} ha^{-1} yr^{-1} using the GWP for CH_4.

3.6.2.3 SOIL CARBON STOCKS

Initial values were taken from Stevenson's (2004) for soil samples in sites which were in pasture in 2001, and were later planted. Four of these sites were re-sampled in October 2011. The same method was followed as for pasture. The differences between the 2001 and 2011 values were tested using paired t-tests in SAS Enterprise Guide v.4.2. The 2001 values were set as the baseline and the intervening years were estimated by assuming a linear change from 2001 to 2011.

3.7 PINE FOREST

3.7.1 EMISSIONS

This section covers the procedure used to derive the emissions in the Pine forest component of the catchment.

3.7.1.1 ENTERIC METHANE & NITROUS OXIDE

The forest was never grazed, thus emissions of enteric CH_4 were set at zero, with any possible emissions from feral animals excluded due to a lack of data. N_2O emissions were approximated using a value of 0.6 kg N_2O ha^{-1} yr^{-1} (Saggar et al., 2008) converted into kg CO_{2e} ha^{-1} yr^{-1} using the GWP for N_2O.

3.7.1.2 FUEL COMBUSTION & EMBODIED EMISSIONS

Information on the embodied and combustion emissions due to the nursery, planting, pruning, and thinning stages of the forest operation were not available. Based on rough estimates for these values, they were not expected to contribute more than a couple of percent at most of what was sequestered by the pine forest. Therefore, the highly uncertain estimates not used and these emissions were ignored. The thinning operation was likely to have had the largest emissions due to the use of chainsaws. The other operations mainly use hand tools, thus would have low emissions.

3.7.1.3 PRODUCT DECAY

The forest was ungrazed so did not produce agricultural products. No harvesting occurred during this time and thinning and pruning waste was left were it was cut.

3.7.2 SEQUESTRATION & CARBON STOCKS

This section covers the procedure used to derive the carbon stocks and sequestration in the Pine forest component of the catchment.

3.7.2.1 FOREST GROWTH & TREE CARBON MASS

Forest growth was simulated using CenW version 4.0. This is a comprehensive forest growth model linking flows of carbon, energy, nutrients and water in trees and soil (Kirschbaum, 1999). The parameter set for *Pinus radiata* was provided by Dr Miko Kirschbaum. Due to the difficulties of obtaining initial soil parameters via the model's equilibrium run function, an initial soil parameter set for pasture was provided by Dr Miko Kirschbaum. Because this was used, the model could not be relied on for estimating changes in the soil carbon pools.

Climate readings came from a weather station on site but only covered up to 2006. No other data was available and the climate would differ from data from the nearest city, Hamilton, which is not in hill country. For 2007-2011 the model was rerun over the old data from 1999-2003. The climate data included minimum and maximum daily temperature, daily precipitation, daily solar radiation, and relative humidity.

CenW is based on modelling forest on a flat surface. There is currently no function to adjust for the effects of slope and aspect, which were expected to be important for the Mangaotama Catchment, via their influence on the climate variables. Climate data was available for the site, but this was not across a range of slopes and aspects.

This was accommodated for by adjusting the climate files for different slope/aspect combinations. The landscape was split into 12 sections based on three slope classes, easy (0-20^{o}), steep (20-35^{o}), very steep (>35^{o}), and four aspects, North (315-45^{o}), East (45-135^{o}), South (135-225^{o}) and West (225-315^{o}) using a 5m x 5m grid square Digital Elevation Model (DEM) in ArcGIS v. 9.2® (Fig. 8).

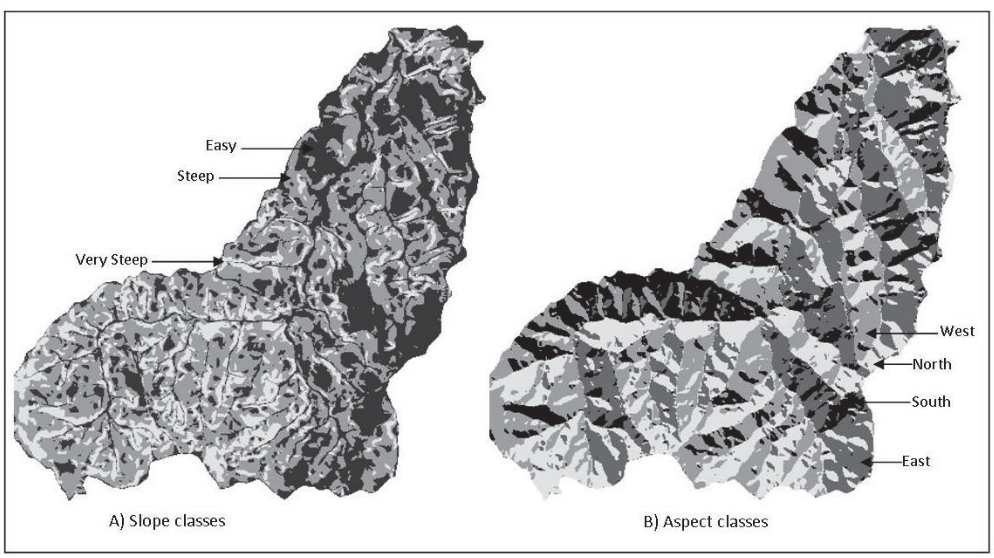

Figure 8. Slope and aspect classes that were used in modelling the pine forest. Shown for the whole catchment. When these two maps where overlaid they gave the 12 slope/aspect classes.

The DEM for the catchment and the solar radiation tool in ArcGIS v.9.2® were used to create maps of the radiation received relative to a flat slope by dividing the solar radiation map created in ArcGIS v.9.2® by the value at the largest flat area in the catchment (Figs. 9 & 10). The solar radiation maps were created for each month. The average relative solar radiation for each slope/aspect section was calculated in ArcGIS v.9.2® for each month (Table 10). The CenW climate file's solar radiation data was multiplied by this value (depending on which month it was) to give the adjusted solar radiation for that slope/aspect section. Minimum and maximum temperatures were adjusted in the same way by assuming that the temperature of each grid square was independent and controlled only by solar radiation. Precipitation was adjusted by multiplying the rainfall by the cosine of the slope/aspect section's average slope. Relative humidity was adjusted for temperature using the relationship between temperature and saturation vapor pressure from Goff and Gratch (1946):

$$\begin{aligned} Log_{10}S = \; & -7.90298\,(373.16/T-1) \\ & + 5.02808\,Log_{10}(373.16/T) \\ & - 1.3816x10^{-7}\,(10^{11.344(1-T/373.16)}-1) \\ & + 8.1328x10^{-3}(10^{-3.49149(373.16/T-1)}-1) \\ & + Log_{10}(1013.246) \end{aligned}$$

Where: T= temperature in Kelvins, S = saturation vapor pressure, in hPa.

The vapor partial pressure was assumed constant across the landscape. It was calculated by multiplying the original relative humidity by the saturation pressure from the above equation using the mean of the unadjusted maximum and minimum temperatures. The vapor partial pressure was divided by the temperature corrected saturation pressure from the above equation using the mean of the adjusted max and minimum temperatures. Values above 100% relative humidity were set as 100%.

$$R \times S = V$$
$$V/S_{adj} = R_{adj}$$

R = Relative humidity, *S* = Saturation vapor pressure, *V* =Vapor partial pressure, $_{adj}$ denotes temperature corrected values.

Figure 9. Relative solar radiation map for January. Darker shades are lower values (relatively less radiation). Minimum value = 0.017. Maximum value = 1.11.

Figure 10. Relative solar radiation map for July. Darker shades are lower values (relatively less radiation). Minimum value = 0.025. Maximum value = 1.79.

Table 10. Relative solar radiation and slope values for the landscape divisions used in the adjustment of the climate input data to account for topography in the CenW model used in modelling the pine forest.('Flat' was not used in the model but is shown for comparison).

Landscape section	**Average Relative Solar Radiation**												**Average Slope**
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec	
Easy North	1.04	1.07	1.13	1.19	1.26	1.30	1.27	1.21	1.14	1.08	1.04	1.03	13.428
Easy South	0.98	0.95	0.90	0.84	0.77	0.73	0.74	0.81	0.88	0.94	0.97	0.99	12.893
Easy East	1.01	1.02	1.02	1.03	1.04	1.03	1.02	1.02	1.02	1.01	1.01	1.01	13.523
Easy West	1.01	1.01	1.02	1.03	1.04	1.04	1.03	1.02	1.02	1.01	1.01	1.01	13.071
Steep North	0.99	1.06	1.16	1.29	1.43	1.49	1.45	1.33	1.19	1.08	1.00	0.97	27.116
Steep South	0.86	0.80	0.70	0.57	0.46	0.42	0.43	0.52	0.66	0.77	0.85	0.88	27.497
Steep East	0.95	0.95	0.97	0.98	1.00	1.01	0.99	0.98	0.97	0.95	0.94	0.94	27.404
Steep West	0.94	0.94	0.95	0.97	0.99	0.99	0.98	0.97	0.96	0.94	0.94	0.94	26.698
Very Steep North	0.90	0.99	1.14	1.32	1.51	1.59	1.54	1.38	1.18	1.02	0.92	0.87	40.199
Very Steep South	0.70	0.61	0.48	0.34	0.30	0.30	0.29	0.31	0.43	0.57	0.68	0.72	41.333
Very Steep East	0.83	0.84	0.85	0.87	0.89	0.89	0.87	0.86	0.85	0.84	0.83	0.83	40.762
Very Steep West	0.82	0.83	0.85	0.86	0.89	0.89	0.89	0.87	0.85	0.83	0.82	0.82	40.297
Flat	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.000

Exact pruning and thinning dates could not be obtained, so for the years when it was done, percentage of trees pruned and pruning lift of each operation used in the original pine modelling (done before the pines were planted) were used (Way, 1999). The 'Gorse covered land' scenario from Way (1999) was used as its stocking rates were closer to what was actually done. For the thinning, the scenario matched with the actual year it was thinned (year 9). Because of the difference in stocking rates, the portion of trees pruned was simplified from Way (1999) to 25% each time. Because the exact date for these operations was not known, they were set as being in the 1st month of the year. Table 11 gives the values used and Table 12 gives some additional inputs and their sources.

Table 11. Pruning and thinning values used in calculating the inputs for the harvesting events in CenW.

Year after planting	Pruning Lift (m)	Tree Height (m)	Branches cut per tree (%)
5	2.2	10	20
7	4.5	14	30
9	6.5	16	40

Pruning lift height was converted into an estimated percentage of branches cut by running the model for flat land to get the heights for the relevant year and treating each tree as a cylinder with branches the whole way down. Thus the portion of branches cut per tree was the pruning lift height divided by the tree height which was rounded to the nearest ten percent.

The actual input for the percentage of branches cut was the branches cut per tree multiplied by 25%. No wood or fine material was ever removed so these were set at zero.

Table 12. Additional inputs into the CenW model.

Parameter	Section	Value	Reference
Atmospheric N input	All	6 kg ha^{-1} yr^{-1}	(Quinn and Stroud, 2002)
Initial Tree Density	All	1200 sph	(Dodd et al., 2008a)
Final Tree Density	All	400 sph	(Dodd et al., 2008a)

The relative size of the thinned stems, and maximum weed height were derived from field measurements carried out in 2011 (Table 13). Approximately 16 trees were measured for DBH at 2 sites for each slope/aspect class (east and west slopes were treated as one class for each slope as there was no difference in the solar radiation or average slopes). At these sites the height of the tallest weed was visually estimated and approximately 10 cut stems were measured as close to 1m from the cut base as possible. Slope and aspect were measured at one point for each site, however this could differ across the area. An effort was made to try and keep as much within the class as possible. Only one site could be accessed for the Very Steep East/West class so extra trees were measured there to compensate. From this data the average DBH of standing and cut stems, relative size of cut stems to standing stems and maximum weed height for each class were calculated.

The 'fertility adjustment at start up' function was not used despite some differences between observed DBHs and modeled ones. The purpose of this adjustment is to account for soil fertility factors the model does not include. However, even enormous adjustments could not always make the modeled DBHs match the field data. At the same time, even small adjustments resulted in large changes in the modeled carbon mass (despite the lack of change in DBH). Because this adjustment had an enormous effect on the results and the modeled and observed values were not wildly different, this approach for accounting for differences in soils and other factors was not used.

The simulation was run for 10 years, starting with the 1st August 2001. Output was given for each year. Total carbon mass in trees was calculated from the sum of Sapwood, Heart-wood, Bark, Foliage, Fine roots, Coarse-roots, Branches, Reproductive tissues, Carbohydrates, and Reserves multiplied by 0.5. Total carbon mass in litter was taken from the sum of Metabolic carbon, Structural carbon, and Coarse Woody carbon for both surface and soil layers. Tree and litter carbon were summed and the difference between years gave the net carbon sequestered or lost by these pools. An additional run was done out to 31 years with harvesting at year 30 (100% of wood cut and removed). The model automatically re-ran the climate data for the additional years that lacked any climate records (being in the future).

Table 13. Field measurements of the *Pinus radiata* forest. Each class had a maximum of two sampling sites. The slope and aspect for both sites are shown, separated by a comma. Slopes and aspects are only approximate values for the area.

Class	Slope	Aspect	Average DBH (cm)	Average cut stem diameter (cm)	Relative size of cut stems	Average Max weed Height (m)
Easy South	15,19	160, 190	32.3	27.4	0.85	2.3
Steep South	25, 30	180, 220	30.7	26.6	0.87	2.3
Very Steep South	40, 40	180, 200	28.8	25.1	0.87	2.3
Easy East/West	11, 17	280, 290	36.2	28.0	0.77	1.8
Steep East/West	35, 35	40, 300	34.4	29.5	0.86	1.5
Very Steep East/West	41	60	29.3	24.0	0.82	2.0
Easy North	18, 17	310, 30	34.5	30.2	0.88	2.3
Steep North	28, 23	335, 330	34.7	28.5	0.82	2.0
Very Steep North	40, 45	340, 340	30.6	28.6	0.93	1.9

3.7.3.2 SOIL METHANE FLUX

An approximation of -9.1 kg CH_4 ha^{-1} yr^{-1} was used (Tate et al., 2006) and was converted into kg CO_{2e} ha^{-1} yr^{-1} using the GWP for CH_4.

3.7.3.3 SOIL CARBON STOCKS

The current values were obtained from field measurements taken in October 2011. Six sites were measured using the same method as for pasture. These covered one easy and one steep slope on Northern, Southern and Eastern aspects. One site had previously had the % carbon measured in 2000 when in pasture. This was re-sampled. However, because bulk density had not been measured originally, a nearby pasture site with the same slope and aspect was measured for bulk density. It was assumed that the nearby pasture site had not changed significantly over time and would be similar to the original sampling site.

Emissions or sequestration from changes in soil carbon stocks were estimated by taking the estimates for the original pasture value in 2000 and the measured values for 2011 for steep and easy slopes, and using the difference to calculate the yearly change.

3.8 EROSION

Sediment export values could not be applied directly, because the landscape included mixtures of land uses. A rough solution was achieved by setting constant erosion rates from the pastures and forests (Table 14). The average sediment loss under the old regime was set as the erosion rate for pasture (calculated from data supplied by AgResearch Ltd.). Erosion had previously been found to be in the order of threefold less under trees (Quinn and Stroud, 2002) so the value for pasture was divided by 3 to give a value for land under trees. Sediment was converted to carbon by assuming 7% carbon and then expressed in CO_{2e}. It was assumed that 80% was sequestered permanently with the rest emitted (Dymond, 2010).

Table 14. Emissions and sequestration due to erosion under forests and pasture.

Land-use	Emissions (kg CO_2 ha^{-1} yr^{-1})	Sequestration (kg CO_2 ha^{-1} yr^{-1})
Forest	48.81	199.65
Pasture	146.80	587.20

3.9 NET BALANCE CALCULATIONS

The land-uses were mapped in ArcGIS v.9.2® using a combination of GIS layers provided by AgResearch Ltd. A smaller catchment size of 269 ha was used rather than the original 296 ha catchment, because there was no DEM available for the larger area. Only a section downstream of the main catchment sampling weir, in which no major land use changes had occurred, was missed by doing this. Some of these maps did not fully match the actual land use changes, so were adjusted by comparison with aerial photos (supplied by AgResearch).

The results from the calculations described in the above methods (Appendix A) were expressed in kg CO_{2e} per $25m^2$ (or kg C per $25m^2$ for carbon stocks) so they could be entered into 5m x 5m grid square rasters. Homogenous areas of the landscape were identified for each year (e.g. Steep LUCC VI pasture or Easy North Pines). The carbon values for each of these areas were summed to give a single value to be entered into the raster. Emissions were counted as positive values (i.e. contributions to the atmosphere), and sequestration as negative values (i.e. removals from the atmosphere).

5m x 5m grid square raster layers were made for each of these homogenous sections for each year. These raster maps were slope adjusted by dividing them by the cosine of the slope derived from the DEM. The values for the slope adjusted rasters were then summed for the catchment and the output for each land use was summed to give the yearly net balance (or carbon stock) for the catchment.

CHAPTER 4: RESULTS

4.1 THE ROLE OF EACH MAJOR LAND USE CHANGE

The new management plan dramatically altered the distribution of the land-use types within the catchment. Grazed pasture decreased from 94.5% of the total area to only 35.6%. Pines were planted on 50.7% and native shrubs on 3% of the land. 5.1% was retired for ungrazed riparian zones. Originally, 5.5% was occupied by grazed native forest fragments. Most of this was then fenced, leaving grazed forests covering only 1.3% of the area, with ungrazed forests occupying the remaining 4.1%. The following results are presented in order of their contribution to the overall catchment's land area under the new plan.

4.1.1 CONVERSION TO PINE FORESTRY

4.1.1.1 PINE GROWTH CHANGES WITH SLOPE AND ASPECT

Simulated pine growth was slightly lower on the Very Steep Eastern and Very Steep Western slopes, and much less on the southern slopes. All other slopes and aspects were very similar to each other (Table 15). In 2011, the Very Steep Southern slopes had only 44% of the average carbon mass of the non-southern slopes, while the Steep Southern slopes had 67% and the Easy Southern slopes had 89% of this.

Table 15. Modelled yearly changes in pine biomass carbon (t C ha^{-1}) from planting in 2001 to 2011 for each slope/aspect class.

Slope	Aspect	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Easy	**East**	-2.6	4.2	6.6	11.5	17.3	18.2	18.0	15.6	18.6	10.6
Steep	**East**	-2.6	4.0	6.6	11.1	18.0	18.7	18.8	16.5	18.8	11.1
Very Steep	**East**	-2.5	3.1	4.9	8.3	15.6	17.2	18.2	15.9	17.7	11.8
Easy	**West**	-2.7	4.2	6.7	11.6	17.4	18.3	18.0	15.6	18.6	10.6
Steep	**West**	-2.6	4.0	6.5	11.0	18.0	18.6	18.8	15.4	18.7	11.3
Very Steep	**West**	-2.5	3.1	4.8	8.2	15.6	17.2	18.3	15.9	17.7	11.9
Easy	**North**	-3.0	4.5	7.3	12.4	17.5	17.7	17.1	15.8	18.4	9.9
Steep	**North**	-3.1	4.8	8.6	13.1	18.0	17.3	16.9	16.3	17.9	9.1
Very Steep	**North**	-3.2	4.6	8.4	12.3	18.2	17.5	17.2	17.6	17.8	9.1
Easy	**South**	-2.5	3.2	4.5	8.3	14.4	16.2	17.5	13.4	17.6	10.8
Steep	**South**	-2.4	2.0	3.0	4.7	9.5	11.7	13.8	11.4	13.1	8.5
Very Steep	**South**	-2.4	0.3	2.0	3.9	6.3	7.5	8.2	7.3	7.7	5.1

There were some discrepancies between the field-measured DBHs and the modeled ones (Table 16). However, the discrepancy was not overly large, with all modeled DBH being within ±3.6cm of the observed DBH confidence intervals. Linear regression between the mean observed and predicted DBH for the values of all the slope/aspect classes showed the relationship to be significant (y = 0.998x-2.141, p= 0.03, r^2 = 0.5).

Table 16. Modelled (Predicted DBH) and measured (Field DBH) diameters at breast height for pine forests by slope/aspect class. Together with the differences between GIS-derived slopes (Model average slope), and slope and aspect measured in the field (Field site slopes, Field site aspects). Predicted DBHs marked with * are outside the 95% confidence intervals for the measured DBHs, shown in brackets. Field site aspects marked with * are outside the aspect class used in the model (i.e. these were borderline sites). Aspect classes in the model were: North = 315-45˚, East/West= 45-135˚ & 225-315˚, South =135-225˚. Field site slopes and aspects are approximate for the general sampling area. Where two values are given for slope or aspect, they are for different sites.

Class	Model Average slope (Degrees)	Field site slopes	Field site aspects	Field DBH	Predicted DBH
Easy North	13.43	18, 17	310*, 30	34.5 (± 1.96)	29.66*
Easy West	13.07	11, 17	280, 290	36.2 (±1.62)	36.70
Easy East	13.52	11, 17	280, 290	36.2 (±1.62)	36.70
Easy South	12.89	15,19	160, 190	32.3 (±2.05)	30.35
Steep North	27.12	28, 23	335, 330	34.7 (±1.97)	32.49*
Steep West	26.70	35, 35	40*, 300	34.4 (±1.89)	31.12*
Steep East	27.40	35, 35	40*, 300	34.4 (±1.89)	31.12*
Steep South	27.50	25, 30	180, 220	30.7 (±1.67)	27.07*
Very Steep North	40.20	40, 45	340, 340	30.6 (±1.74)	28.26*
Very Steep West	40.30	41	60	29.3 (±2.26)	32.55*
Very Steep East	40.76	41	60	29.3 (±2.26)	32.55*
Very Steep South	41.33	40, 40	180, 200	28.8 (±1.69)	23.48*

4.1.1.2 STOCKS & FLOWS IN PINE FORESTS

By 2011 the pine forest biomass was the largest stock of carbon (Fig. 11). Next largest was the soil. Forest growth was by far the largest flux with all other fluxes, except the loss of soil carbon, being of negligible importance. Soil carbon losses were large, but were more than compensated for by the growth of the forest.

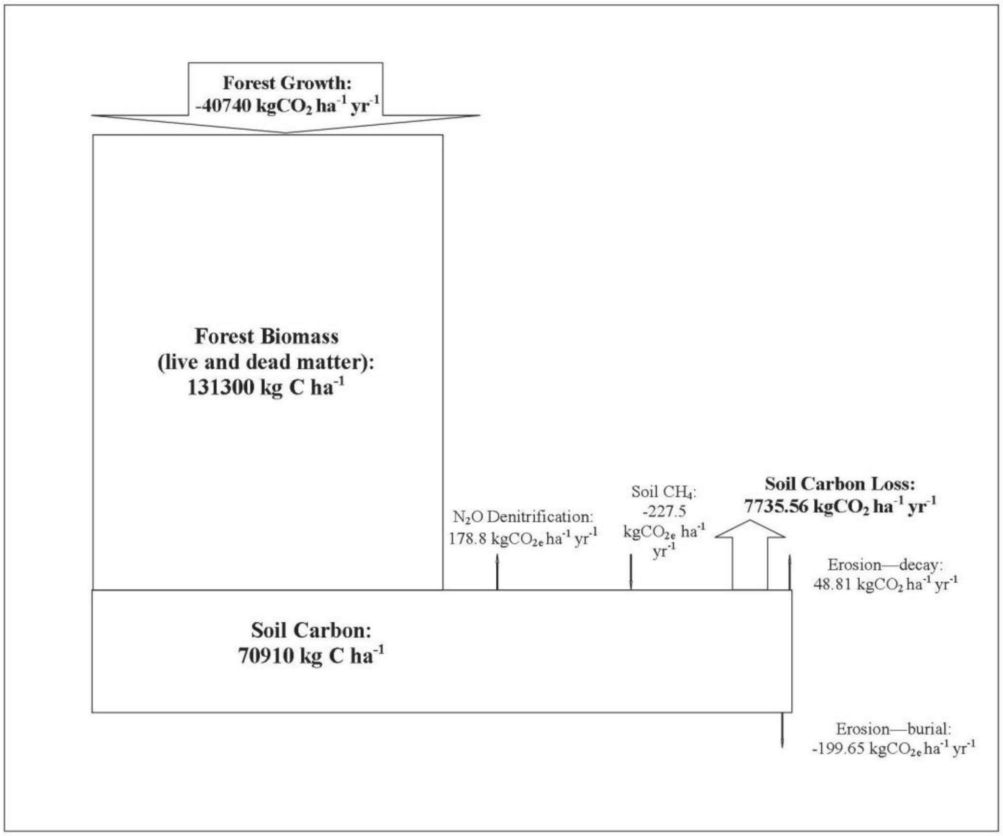

Figure 11. Carbon stocks and flows for pine forest in 2011 (example using values for steep Eastern slopes). Positive values are emissions, negative values are sequestration. Erosion refers to the fate of sediment exported from the catchment.

Initially, after planting in 2001 the soil carbon stocks were slightly larger, and the forest biomass was small. At first, the forest biomass was actually in decline due to the decay of initial litter debris, but this was quickly reversed. Growth rates steadily increased until around 2006, when they tended to level off (Fig. 12). Although growth rates dropped off in 2011, when the model was run for a further 30 yrs, the long-term growth rate fluctuated around the 2011 levels (Fig. 12). By 2008 forest biomass had become comparable to the soil carbon stock, and overtook it afterwards. Soil carbon loss was constant throughout the 2001-2011 period, leading to a gradual shrinking of the soil pool.

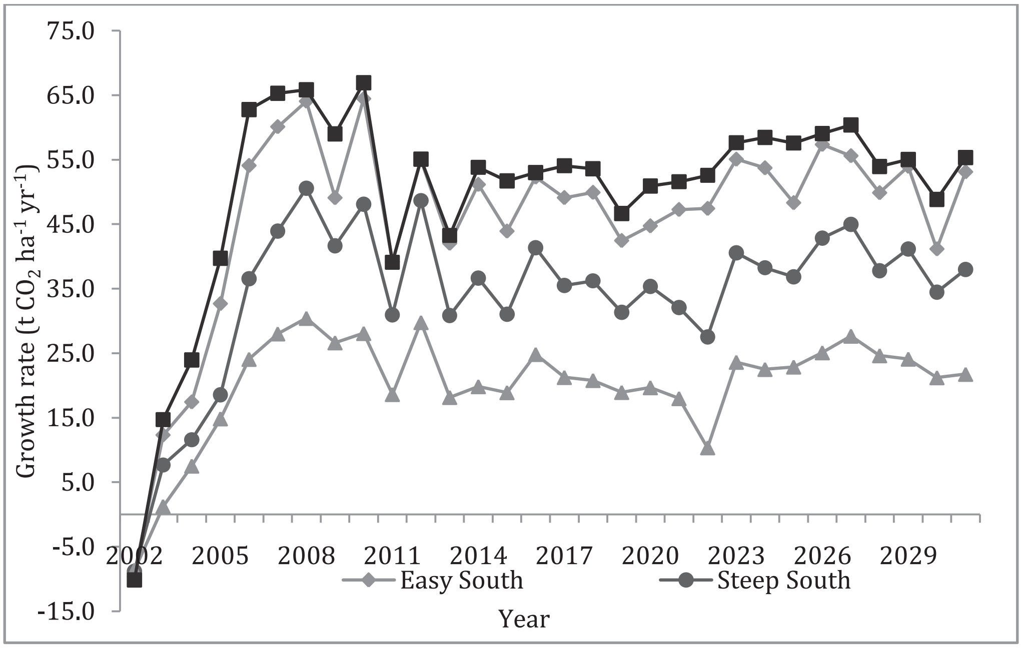

Figure 12. Growth rates of pine forest on different slope/aspect classes over a 30yr rotation (not including harvest). 'Other' is the mean for all non-southern slopes which showed considerable overlap in growth rates. Easy South was generally at the lower end of the range for 'Other' slopes.

4.1.1.3 CONTRIBUTION OF THE PINES TO THE CATCHMENT GHG BALANCE

The pine forest controlled the dynamics of the overall GHG balance in the catchment to such an extent that the other land uses had little effect (Fig 13). Conversion of pasture to pine forest initially led to a major spike in emissions, adding 2685 t CO_{2e} yr^{-1} in 2002, more than double of what was originally emitted from the catchment. This jump in emissions can be attributed to the loss of soil carbon at 1134 t CO_{2e} yr^{-1} and to the decay of litter not being matched by the growth of the trees. The initial litter in CenW was automatically set at around 10 t C ha^{-1}, most of which was in the form of below-ground resistant carbon from roots. However, this may be too large for what would actually be in the pasture, thus this initial jump in emissions is likely to be an overestimate.

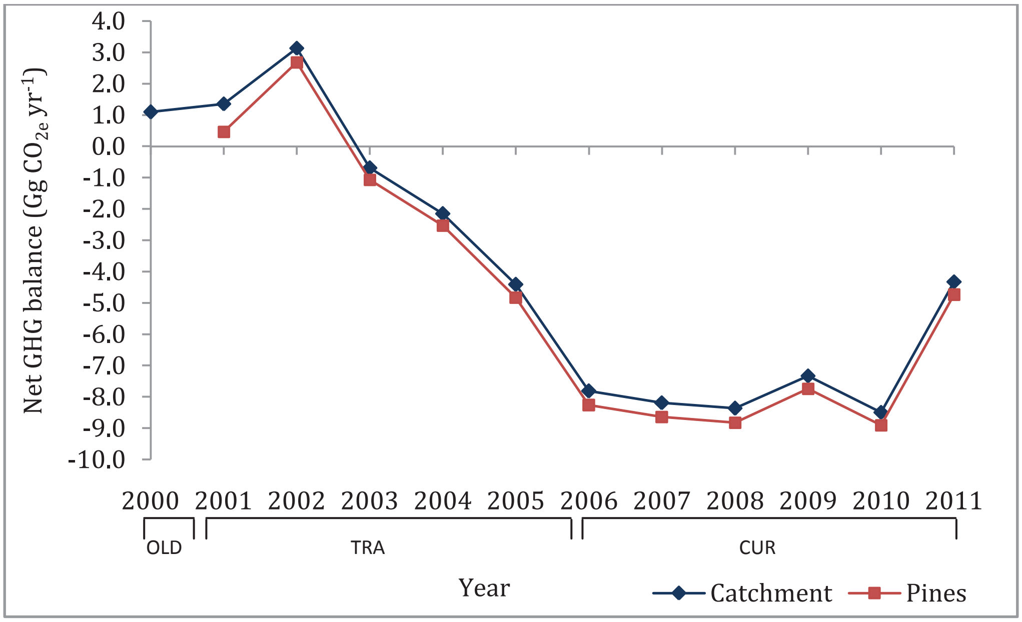

Figure 13. Changes in the net greenhouse gas balance for the whole catchment and for the pine forest, from before the land-use changes in 2001 until 10yrs after. Positive values are net emissions and negative values are net sequestration of greenhouse gases. OLD – old management regime (before conversion to pines and other land-use changes), TRA – transitional period (pines planted and establishing), CUR – current management regime (established pine forest).

After 2002 the forest steadily increased its rate of sequestration before leveling out after 2006, reaching a maximum of 8905 t CO_{2e} yr^{-1}, before decreasing slightly after the thinning in 2010. The lower growth rates and the higher net GHG balance in 2011 appear to have been primarily due to the thinning suppressing the growth of the pines. When thinning was removed from the model a much smaller drop was seen, and growth was higher than it would otherwise have been for the next few years (Fig 14). The pines offset 834.6% of the pasture's net emissions over the 10 year period between 2001 and 2011.

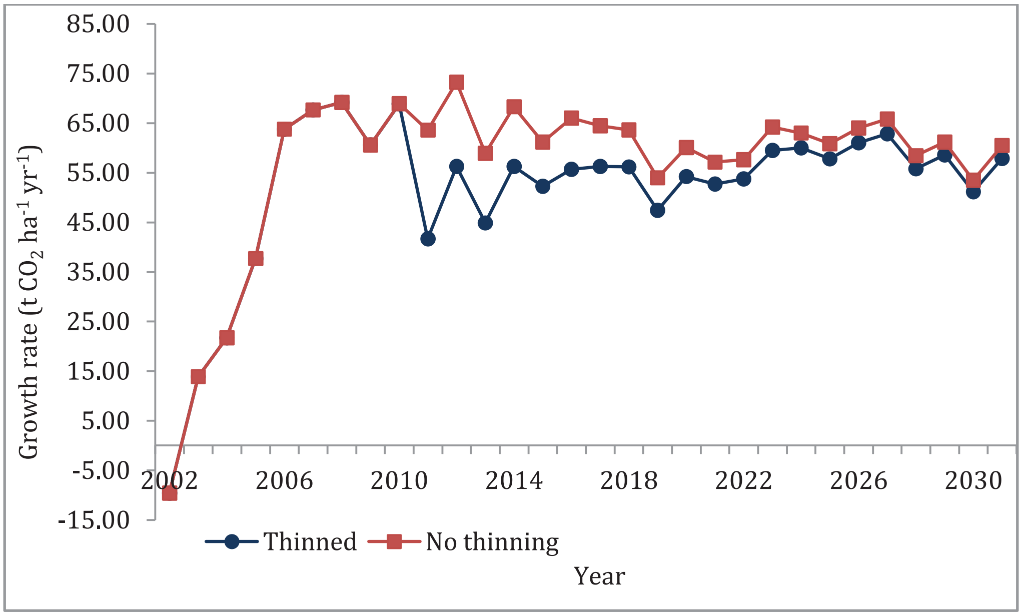

Figure 14. Growth rates over a 30yr rotation (not including harvest) with thinning in 2010 and without thinning included in the model, using the Steep East class as an example.

4.1.2 PASTURE REDUCTION AND INTENSIFICATION

4.1.2.1 STOCKS & FLOWS IN PASTURE

In pasture (Fig. 15) the soil carbon stocks were larger than in any of the forests except the native plantings. Per ha emissions were highest in the current regime, slightly lower in the transitional period and lowest under the old regime. Soil carbon losses were smaller than in the forests and only occurred on steep slopes, but this is likely to be an underestimate. Due to this loss, the carbon stocks on steep slopes are gradually declining. Other fluxes were of negligible importance.

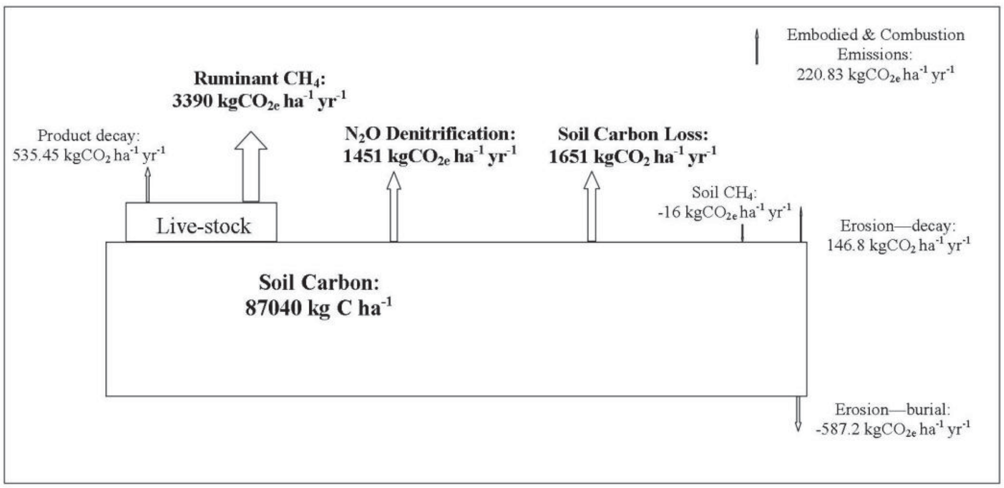

Figure 15. Stocks and flows for pasture in 2011 (using LUCC V steep slopes values). Livestock were not counted in the stocks. Positive values are emissions, negative values are sequestration. Erosion refers to the fate of sediment exported from the catchment.

On a per ha basis, the three main emission fluxes were ruminant CH_4, N_2O from the soil, and, on steep slopes only, the loss of soil carbon. Together these made up between 87% and 93% of total emissions from pasture (Table 17).

Ruminant CH_4 was the largest source of emissions. Despite increasing when the intensification took place, the relative importance of ruminant emissions declined slightly. N_2O from the denitrification of fertilizer and live-stock nitrogen inputs was the second most important source of emissions, although it was more than matched by soil carbon losses on steep slopes. The contribution of N_2O increased whenever more nitrogen fertilizer was applied, as during the transitional period for LUCC V, and again for all LUCCs during the current regime. Soil carbon losses made up a larger proportion of the emissions on poorer quality LUCCs, because these were more lightly grazed yet soil losses were assumed to be the same as elsewhere. The only sequestration occurring in the pasture was the tiny amounts from the consumption of methane in the soil and from eroded sediment which is assumed to be exported to the deep oceans and buried.

Table 17. Contribution of ruminant CH_4, N_2O and soil loss to total per ha emissions from each LUCC and each time period in pasture. Note: this is based on Steep slopes. Soil loss is not occurring on Easy slopes. OLD = the old regime pre-2001, TRA = the transitional period with higher stocking rates, CUR = the current regime higher fertilizer inputs.

		% contribution to total per ha emissions				
		III	IV	V	VI	VII
CH_4	**OLD**	53.3	53.3	48.2	45.0	41.5
	TRA	52.5	52.5	46.2	45.1	40.7
	CUR	50.2	50.2	45.2	42.1	37.4
N_2O	**OLD**	20.1	20.1	18.2	17.0	14.9
	TRA	19.8	19.8	19.8	17.0	15.4
	CUR	21.6	21.6	20.7	20.0	19.2
Soil carbon loss	**OLD**	19.5	19.5	26.2	30.4	35.8
	TRA	17.3	17.3	22.5	27.4	33.3
	CUR	16.5	16.5	22.0	25.6	30.6

4.1.2.2 PASTURE CONTRIBUTION TO THE CATCHMENT GHG BALANCE

The pasture was the only land use that was always a net source. Emissions from the pasture were responsible for the slightly reduced sink capacity of the whole catchment in comparison with the pines alone (Fig. 13). The pasture originally produced net emissions of 1209 t CO_{2e} yr^{-1} which was reduced by 58% once the land-use changes were complete (Fig 16). The increased use of nitrogen fertilizer in 2006 raised emissions by 37.63 t CO_{2e} yr^{-1}.

The ungrazed grassed riparian zones in the pasture and pine forests produced a very small 1.4 t CO_{2e} yr^{-1} sink. Between 2001 and 2011, the ungrazed riparian zones offset 0.2% of the pastures net emissions. These areas were essentially carbon neutral as the only fluxes they had were very small and based on very uncertain estimates. In reality, some regeneration may be making them a stronger sink, although the presence of stream side wetlands may have markedly increased their N_2O emissions (especially for those still receiving runoff from the pasture), but this could not be determined due to a lack of data.

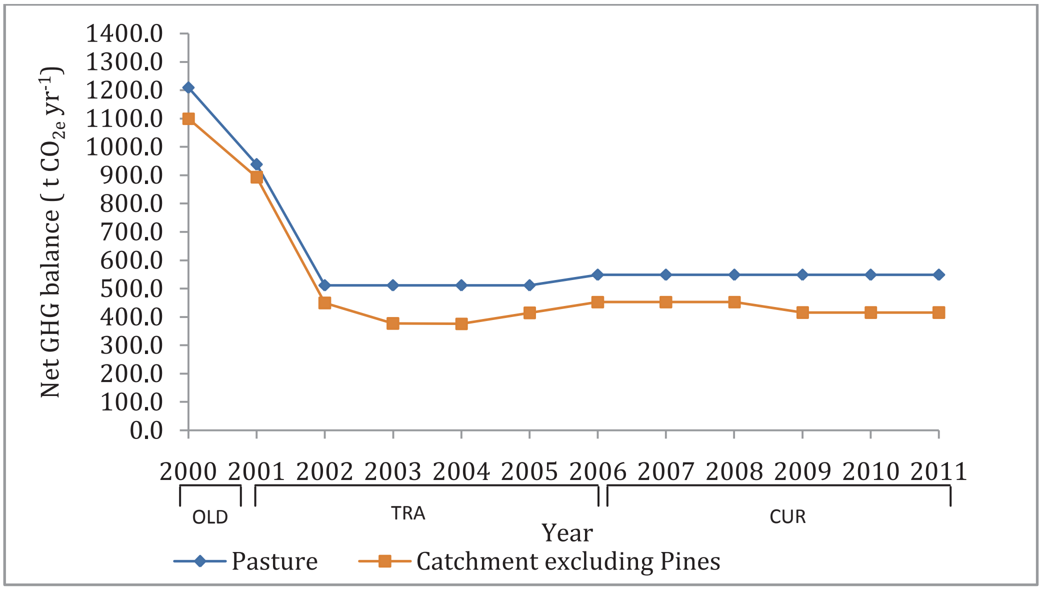

Figure 16. Changes in the net greenhouse gas balance for the whole catchment excluding the pine forests and for the pasture alone. Positive values are net emissions of gases. OLD – old management regime (before conversion to pines and other land use changes), TRA – transitional period (pines planted and establishing), CUR – current management regime (established pine forest, increased N fertilizer).

4.1.3 NATIVE FOREST FRAGMENTS

4.1.3.1 EFFECT OF SLOPE & ASPECT ON NATIVE FOREST FRAGMENTS

Forest carbon mass was significantly affected by the slope class and, in 2011, by the aspect (Table 18). However, this was only at a 0.1 level of significance for most years. This can be attributed to higher levels of carbon on the steep slopes with easy and moderate slopes generally quite similar (Table 19). The effect of aspect in 2011 was dubious. Northwestern slopes had the highest mean, but only 3 samples (versus 12-11 for north and south) and an enormous 95% confidence interval of ± 800.5 t C ha^{-1}.

Sequestration in biomass was significantly affected by aspect between 2000 and 2002 and by slope between 2002 and 2004 (Table 18). When the aspect was significant (2000-02), the most different class was the Eastern aspect which had a mean of -33.15 t CO_2 ha^{-1} yr^{-1} (versus 17.29 t CO_2 ha^{-1} yr^{-1} for Northwest, 11.52 t CO_2 ha^{-1} yr^{-1} for North and 11.96 t CO_2 ha^{-1} yr^{-1} for South). However, the Eastern aspect only had two replicates, making this result very uncertain. When the slope effect was significant (2002-04), this could be attributed to lower growth on the easy slopes of 4.55 t CO_2 ha^{-1} yr^{-1}, versus 13.47 t CO_2 ha^{-1} yr^{-1} on the moderate slopes and 14.57 t CO_2 ha^{-1} yr^{-1} on the steep slopes.

Table 18. P-values for mixed model ANOVAs for native forest fragments' total carbon mass in biomass (live mass, coarse woody debris, and litter) between the year 2000 to 2011, and the yearly change in carbon in live plants over the three sampling intervals used. SLOPE: Easy (n=6), Moderate (n=16), Steep (n=15). ASPECT: East (n=2), North (n=13), Northwest (n=7), South (n=15). 2011 values do not include plots from the 'Kohekohe' Forest. Values in bold are significant at α=0.1.

	Total Carbon Mass					**Total Carbon Change**		
	2000	**2002**	**2004**	**2008**	**2011**	**2000-02**	**2002-04**	**2004-08**
Slope	**0.094**	**0.090**	**0.041**	**0.056**	**0.007**	0.211	**0.026**	0.441
Aspect	0.641	0.633	0.467	0.534	**0.072**	**0.053**	0.198	0.909

Table 19. Total carbon mass (live biomass, coarse woody debris and litter) in t C ha^{-1} for the native forest fragments by slope class. 95% confidence intervals are in brackets. Replication in 2011 is Easy =5, Moderate = 12, Steep = 9. Prior to 2011 replication was Easy =6, Moderate =16, Steep = 15.

	Year				
Slope	**2000**	**2002**	**2004**	**2008**	**2011**
Easy	98.97 (± 50.3)	108.13 (±52)	110.61 (±53.8)	111.41 (±53.7)	143.45 (±100.8)
Mod.	96.99 (±23.1)	101.24 (±27.6)	105.15 (±30.4)	111.18 (±32.7)	162.91 (±54.8)
Steep	138.77 (±63.2)	144.61 (±68.3)	151.67 (±74.2)	157.21 (±74.2)	235.57 (±165.8)

No significant effect of slope or aspect was found on the coarse woody debris (CWD), with the p-values being 0.75 and 0.81, respectively. Litter mass was significantly affected by both slope and aspect in all years, except 2011 (Table 20). The effect of slope can be attributed to a tendency for lower levels of litter to be found on steep slopes (Table 21). Eastern aspects tended to have much higher levels of litter, but this was limited to only two samples (Table 22). Apart from this, North aspects tended to have higher levels of litter mass than Northwest and South.

Results for interaction effects between slope and aspect were judged to be unreliable as only 3 of the 10 combinations had more than 4 samples available and the possible combinations were poorly covered. The effect of removing grazing or planting around the fragments could not be tested due to a lack of proper replication at the level of whole fragments.

Table 20. P-values for mixed model ANOVAs for litter mass in the native forest fragments. SLOPE: Easy (n=6), Moderate (n=16), Steep (n=15). ASPECT: East (n=2), North (n=13), Northwest (n=7), South (n=15). Values in bold are significant at α=0.1.

	Year				
	2000	**2002**	**2004**	**2008**	**2011**
Slope	**0.002**	**0.032**	**0.024**	**<.0001**	0.657
Aspect	**<.0001**	**0.009**	**0.002**	**<.0001**	0.644

Table 21. Mean litter mass (t C ha^{-1}) for the native forest fragments by slope. Easy (n=6), Moderate (n=16), Steep (n=15).

	Year				
Slope	**2000**	**2002**	**2004**	**2008**	**2011**
Easy	3.91 (± 4.1)	3.59 (± 3.6)	3.70 (± 4.5)	4.14 (± 4.7)	4.92 (± 6.2)
Moderate	4.05 (± 3.3)	3.58 (± 2.4)	3.32 (± 2.5)	5.55 (± 5.5)	3.11 (± 1.1)
Steep	2.66 (± 1.3)	2.36 (±1.3)	2.67 (± 1.4)	2.54 (± 1.3)	2.97 (± 1.5)

Table 22. Mean litter mass (t C ha^{-1}) for the native forest fragments by aspect. East (n=2), North (n=13), Northwest (n=7), South (n=15).

	Year				
Slope	**2000**	**2002**	**2004**	**2008**	**2011**
East	13.89 (± 138.0)	9.64 (± 87.2)	11.01 (± 107.9)	23.0 (± 244.4)	3.53 (± 3.2)
North	3.94 (± 2.3)	3.70 (± 2.2)	3.37 (± 2.0)	4.1 (± 2.7)	4.39 (± 2.6)
Northwest	2.07 (± 2.9)	1.92 (± 2.4)	2.06 (± 2.7)	2.1 (± 3.0)	2.22 (± 3.0)
South	2.32 (± 0.8)	2.23 (± 0.9)	2.35 (± 1.1)	2.5 (± 0.7)	2.94 (± 1.0)

4.1.3.2 STOCKS AND FLOWS IN NATIVE FOREST FRAGMENTS

The native forests had larger carbon stocks in the forest biomass than the pines, and similar-sized soil stocks. Much more carbon was contained in the forest biomass than in the soil. Sequestration from the forest's growth was much smaller than that by the pines. Forest growth fluctuated slightly over the years. In the grazed forests (Fig. 17) ruminant methane and nitrous oxide were an important source of emissions. However, these emissions were probably overestimated, because forest fragments would not have been grazed as intensely as the open pasture, but were assumed to be grazed the same due to a lack of data. Even so, emissions from grazing were still generally offset by the growth of the forest. In ungrazed forests (Fig. 18) the loss of soil carbon cancelled out a substantial portion of the sequestration from the trees. Despite this, the ungrazed forest fragments still had larger soil carbon stocks than the grazed forests. Other fluxes were of negligible importance.

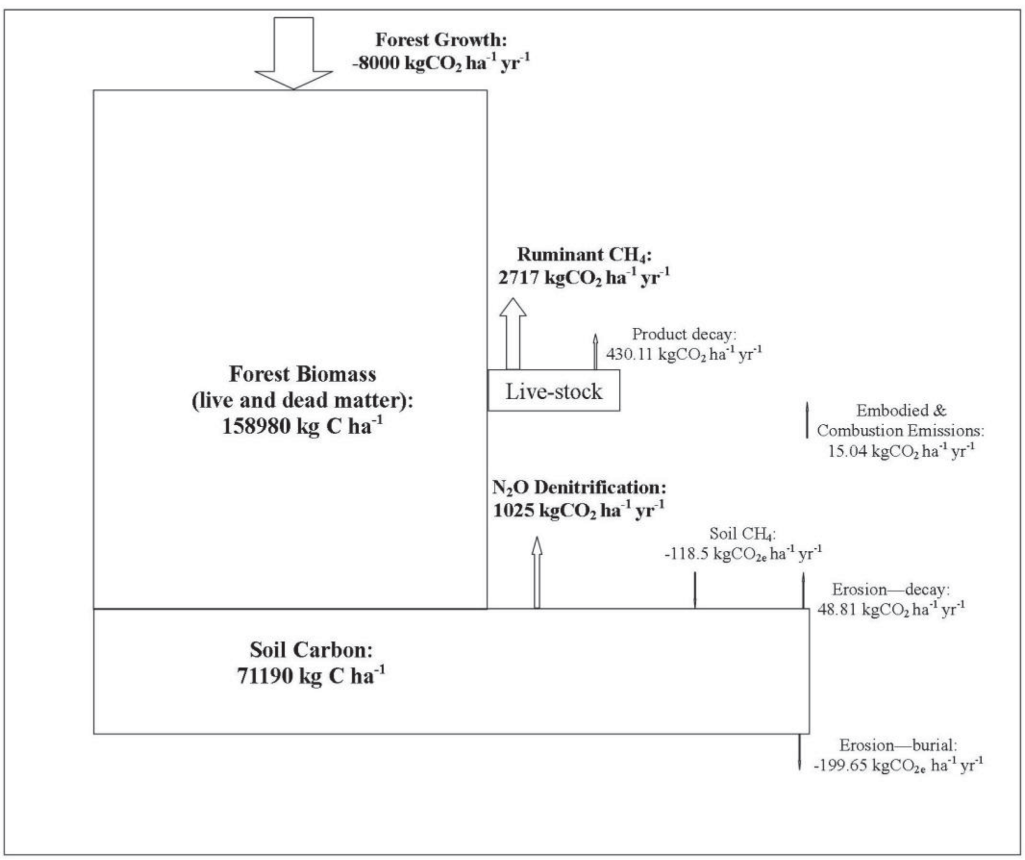

Figure 17. Stocks and flows in grazed native forest in 2011 (example for LUCC VI). Livestock were not counted in the stocks. Positive values are emissions, negative values are sequestration. Erosion refers to the fate of sediment exported from the catchment.

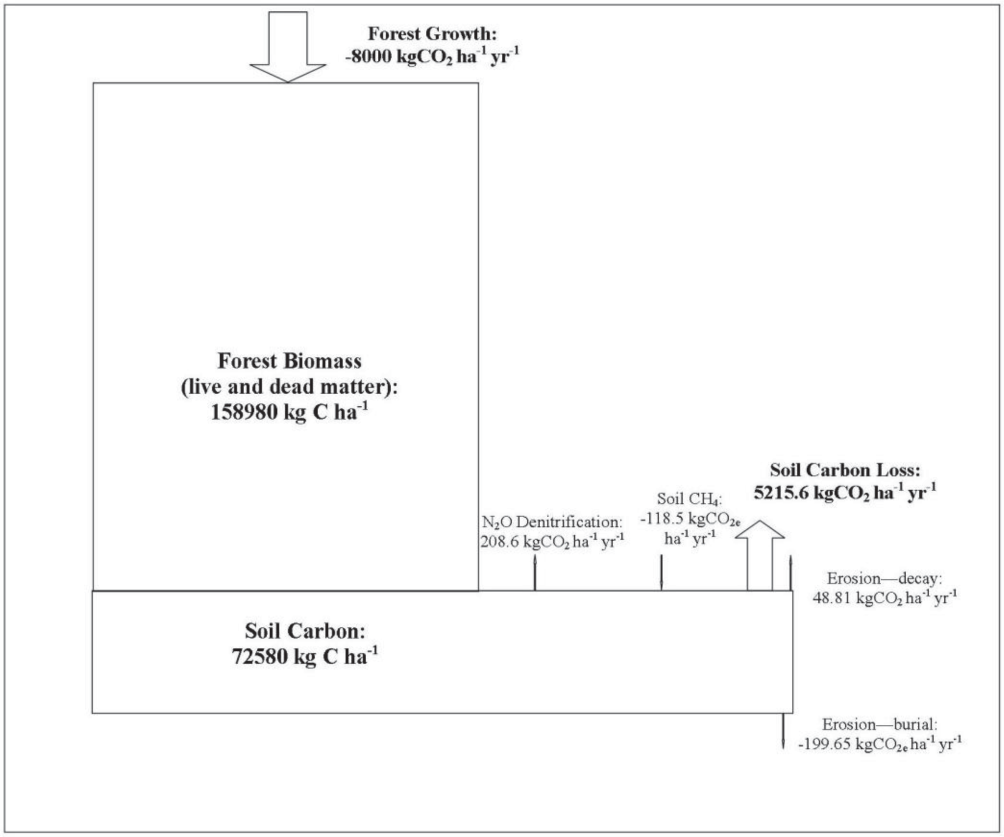

Figure 18. Stocks and flows for ungrazed native forest in 2011. Positive values are emissions, negative values are sequestration. Erosion refers to the fate of sediment exported from the catchment.

4.1.3.3 NATIVE FOREST FRAGMENT BIOMASS CARBON STOCKS & CHANGES

Live carbon stocks steadily increased between 2000 and 2011, sequestering on average between about 5.9 to 12 t CO_2 ha^{-1} yr^{-1} (Table 23). The size of the increase from 2008 to 2011 is misleading, due to previous years using regressions to estimate heights and 2011 using actual data.

Table 23. Average live carbon mass in native forest fragment in 2000-2011, and yearly change in live carbon for the three sampling intervals used for all plots. Unsampled plots for 2011 (n=12) used 2008 values in calculating the mean. 95% confidence intervals in brackets. (Note: the comparison between 2011 and prior years is misleading due to differences in methods).

	Year/Time period	Mean
Live Carbon Mass t C ha^{-1}	**2000**	109.45 (±25.65)
	2002	114.89 (±27.87)
	2004	119.63 (±28.55)
	2008	124.01 (±30.75)
	2011	150.23 (±41.20)
Live Carbon Change t CO_2 ha^{-1} yr^{-1}	**2000-02**	10.02 (±7.34)
	2002-04	12.00 (±4.11)
	2004-08	5.84 (±2.61)

CWD stocks were much smaller than the carbon stocks in live biomass. There were less CWD in the ungrazed ‘Rewarewa’ fragment than in the grazed ‘Rimu’ fragment, however, their confidence intervals overlapped (Table 24). In both the grazed and ungrazed fragments measured in 2011 the majority of the CWD was from dead tree ferns, although the grazed areas had proportionally more from tree ferns. Of all the CWD measured in the grazed ‘Rimu’ fragment, 81.2% was from tree ferns, and 18.8% from trees. In comparison, for the ungrazed ‘Rewarewa’ and ‘Kahikatea’ fragment, 62.2% of all CWD was from tree ferns, and 37.8% was from trees. The litter stocks were smaller than the CWD, and showed no clear trend over time (Table 25).

Table 24. Average amounts of Coarse Woody Debris in the native forest fragments in 2011. N = number of plots. 95% confidence intervals are in brackets. 'Rewarewa' includes the fragment sometimes referred to as 'Kahikatea'.

Site	Treatment	N	Mean t C ha^{-1}
Rewarewa	Ungrazed	15	4.65 (± 2.55)
Rimu	Grazed	12	6.96 (± 3.32)

Table 25. Mean litter mass in native forest fragments, 2000-2011. 95% confidence intervals are in brackets.

Year	2000	2002	2004	2008	2011
Mean (t C ha^{-1})	3.62 (±2.30)	3.60 (± 1.59)	3.06 (± 1.19)	4.09 (± 2.21)	3.22 (± 0.99)

The removal of grazing from some fragments has had a visible impact on the understory's growth. However, this has not translated into more carbon sequestration in the ungrazed fragments. There are some differences between the carbon sequestration in grazed and fenced fragments, but neither was consistently higher than the other (Table 26). Grazing has appeared to have had its greatest impact on the undergrowth, which is dominated by the smallest trees and unmeasured ground cover, thus contributing little to the total carbon stock of the forest. On average, the vast majority of the plots' live carbon mass was contained in the larger trees (Table 27).

Table 26. Summary statistics for differences in the total change in carbon stocks in plant-associated carbon pools (live biomass, coarse woody debris and litter) between grazed and ungrazed native forest fragments. Negative values are losses of carbon (emissions) whereas positive values are a gain in carbon (sequestration).

	Grazed (t CO_2 ha^{-1} yr^{-1})			Fenced (t C O_2 ha^{-1} yr^{-1})		
Time period	**2000-02**	**2002-04**	**2004-08**	**2000-02**	**2002-04**	**2004-08**
Mean	3.30	10.83	3.27	14.09	6.42	7.16
Median	9.58	7.52	-1.39	9.54	6.68	8.73
Std. dev.	30.20	23.97	12.92	25.43	15.60	8.66
Minimum	-84.78	-51.53	-16.37	-23.30	-40.59	-12.07
Maximum	44.77	40.99	27.30	93.95	25.91	21.58

Table 27. Mean contribution of different size classes of trees and tree ferns to the total live carbon mass in native forest fragments, for all plots measured in 2011.

Size Class	% of plot live carbon mass
<10cm DBH	2.66
10-30cm DBH	54.24
>30cm DBH	44.82

Total carbon stocks increased steadily over time. Inclusion of CWD and litter into the balance slightly changed the live carbon sequestration rate in all time periods (Table 28), with the biggest difference during 2004-08, when sequestration was reduced by 1.09 t C ha^{-1} yr^{-1}. There was considerable variation between plots. Some areas sequestered more carbon than the pine forests, while others lost much more carbon than would be emitted by the same area of pasture. The largest sites had an order of magnitude more carbon than the smallest. The median carbon mass is lower than the mean, indicating that the forests are skewed towards lower biomass.

Table 28. Summary statistics for total plant associated (live, coarse woody debris, litter) carbon mass in native forest fragments between the year 2000 and 2011, and the yearly change for the three sampling intervals used. 95% confidence intervals of the mean are in brackets. (Note: the comparison between 2011 and prior years is misleading due to differences in method; 2011 uses 2008 data to fill missing values).

	Year/Time Period	Mean	Median	Min	Max
Total Carbon Mass t C ha^{-1}	**2000**	117.92 (± 25.71)	92.92	22.04	440.20
	2002	123.50 (± 27.79)	93.80	22.06	485.28
	2004	127.87 (± 28.65)	95.89	21.22	488.24
	2008	134.14 (±30.33)	104.58	15.04	511.76
	2011	158.98 (±41.45)	115.81	10.61	678.63
Total Carbon Change t CO_2 ha^{-1} yr^{-1}	**2000-02**	10.24 (±8.59)	9.54	-84.78	93.95
	2002-04	8.00 (±5.91)	6.68	-51.53	40.99
	2004-08	5.76 (±3.27)	6.83	-16.37	27.30

4.1.3.4 CONTRIBUTION OF THE NATIVE FOREST FRAGMENTS TO THE CATCHMENT GHG BALANCE

The along with the native restoration plantings the native forest fragments made up the difference between the pasture and the non-pine catchment. Prior to 2001, when all the native forest fragments were grazed, they sequestered a net 110 t CO_{2e} yr^{-1} (Fig. 19). Most were then removed from grazing in 2001, which reduced the contribution of the grazed forest category. Despite the ungrazed forests making up the vast majority of the biomass since 2001, both forest types sequestered similar amounts. This large reduction in the ability of ungrazed forests to sequester carbon can be explained by the 66 t CO_{2e} yr^{-1} loss of soil carbon occurring in them; this loss is not apparent in the grazed forests, but that may simply be due to a lack of samples. If this loss was not occurring, ungrazed forests would have been a much larger sink, sequestering up to 131 t CO_{2e} yr^{-1}, and would have been a more important sink than the native plantings.

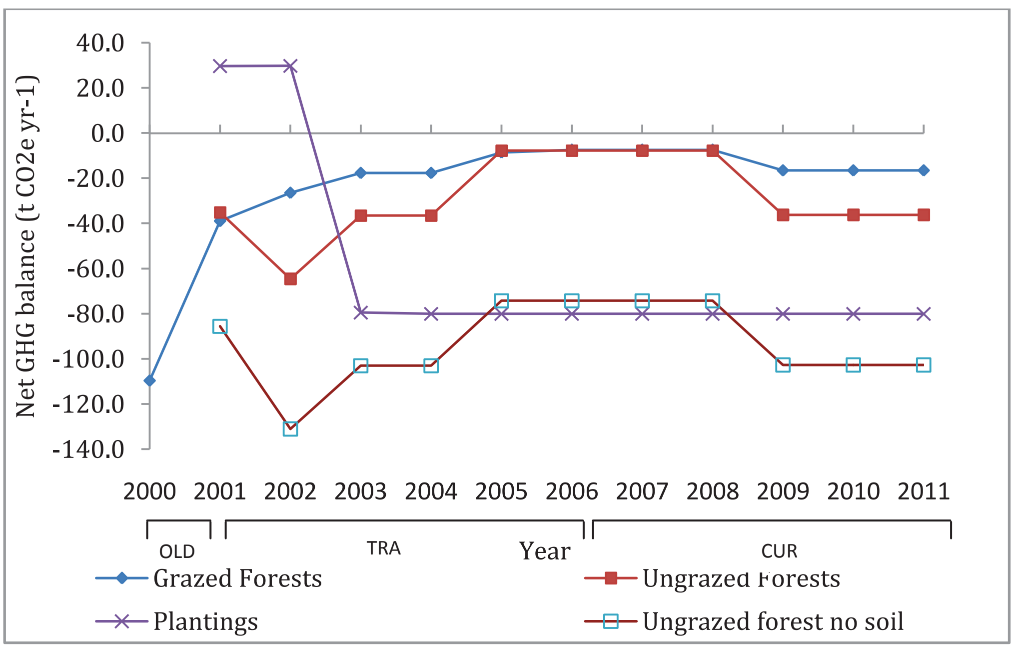

Figure 19. Changes in the net greenhouse gas balance for grazed and ungrazed forest fragments, and the native plantings for the whole catchment from before the land-use changes to 10yrs after. 'Ungrazed forest no soil' gives the values for the net sequestration in ungrazed forests when soil carbon losses are excluded. Positive values are net emissions and negative values are net sequestration. OLD – old management regime (before conversion to pines and other land use changes), TRA – transitional period (pines planted and establishing), CUR – current management regime (established pine forest increased N fertilizer).

Under the old regime, prior to 2001, the net sequestration by native forest fragments offset 9% of the net emissions from the pasture (in addition to the livestock emissions occurring within the fragments). Under the new land use regime, the emissions offset from 2001-2011 was reduced to 7.9%, with 5% from the ungrazed fragments and 2.9% from the grazed ones. This reduction can be attributed to the loss of soil carbon occurring in the ungrazed fragments. If this loss of soil carbon were not actually occurring, then the percentage offset by the forests would have risen to 19.2%, with 16.4% from the ungrazed fragments. This gain would occur mainly due to the loss of grazing-associated emissions in the forests.

4.1.4 NATIVE RESTORATION PLANTINGS

4.1.4.1 EFFECT OF SLOPE, ASPECT & SECTION ON NATIVE PLANTINGS

There was no significant effect of slope or aspect on plant biomass in the native plantings, or any interaction effect between the two from 2002 to 2006 (Table 29). In 2002 there was a significant effect of the planting section (the blocks they were originally planted in – which had some differences in planting density, etc.) with one section having a higher mean (0.56 t C ha^{-1} in February 2002). This did not have any significant effect on the biomass growth and there was no longer any significant difference between sections by 2006.

Table 29. P-values for ANOVAs on live carbon in native plantings mass from February 2002 to 2006 and the yearly carbon change in 2002 and from 2002 to 2006. SLOPE: Flat (n=3), Easy (n=14), Steep (n=16). ASPECT: Flat (n=3), East (n=11), West (n=9), North (n=5), South (n=5). SECTION: 10 different planting areas (n=1-5). Values in bold are significant at α=0.05.

	Live carbon Mass			Live carbon change	
Parameter	**Feb-02**	**Oct-02**	**2006**	**2002**	**2002-06**
Slope	0.55	0.77	0.63	0.85	0.62
Aspect	0.51	0.18	0.45	0.19	0.46
Section	**0.01**	**0.04**	0.62	0.15	0.69
slope x aspect	0.67	0.42	0.86	0.27	0.87

Live carbon in the plantings biomass was initially very low, but increased rapidly over the first few years (Table 30). The native plantings had much lower stocks of carbon in biomass than the native forest fragments, but were soon able to achieve higher growth rates (compare Table 30 vs. Table 28).

Table 30. Summary statistics for native plantings – live carbon mass and yearly change in carbon mass. 95% confidence interval for the mean is in brackets.

	Year/Time Period	Mean	Median	Min	Max
Live carbon mass t C ha^{-1}	**Feb 2002**	0.28 (±0.06)	0.26	0.05	0.75
	Oct 2002	0.76 (±0.16)	0.69	0.21	2.52
	2006	16.74 (±3.86)	14.27	1.91	57.80
Live carbon change t CO_2 ha^{-1} yr^{-1}	**Feb-Oct 2002**	2.61 (±0.66)	2.24	-0.04	9.80
	2002-06	14.86 (±3.49)	12.37	1.50	51.67

4.1.4.2 STOCKS AND FLOWS IN NATIVE PLANTINGS

The native restoration plantings had much smaller stocks in the forest biomass than the mature native fragments and the established pine forest, but much larger stores of soil carbon (Fig. 20). They sequestered more carbon than the native forest fragments, but were losing similar amounts of soil carbon. It must be noted than the forest biomass stocks and growth shown in Figure 20 are estimates only, based on the extrapolation of data from 2002 to 2006. In reality growth may have slowed or increased since then. Again, other fluxes were of negligible importance. Embodied and combustion emissions occurred during the planting and early establishment phase, but these were minor fluxes of 61-141 kg CO_{2e} ha^{-1} yr^{-1}.

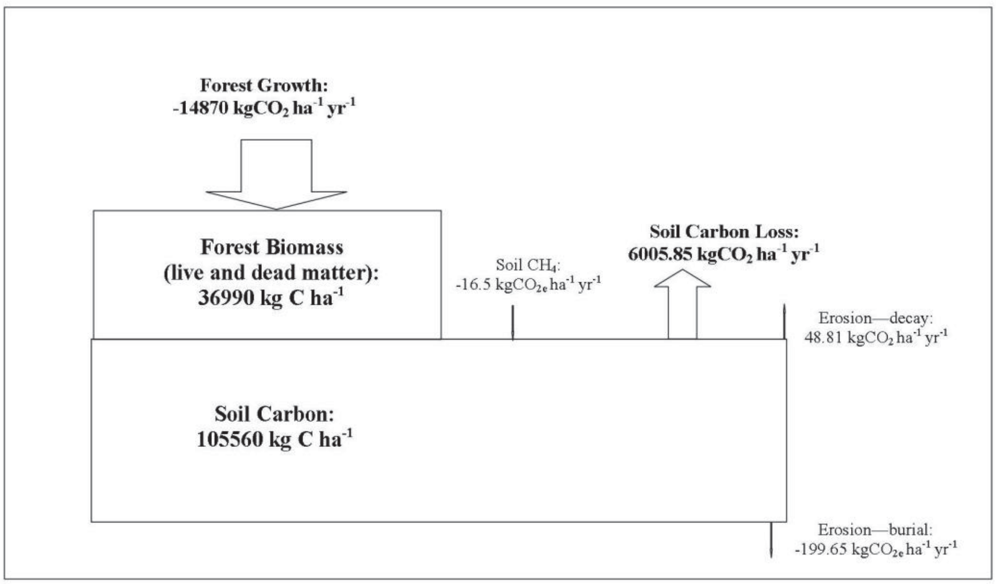

Figure 20. Stocks and flows for native restoration plantings in 2011. Positive values are emissions, negative values are sequestration. Erosion refers to the fate of sediment exported from the catchment.

4.1.4.3 CONTRIBUTION OF THE NATIVE PLANTINGS TO THE CATCHMENT BALANCE

The native plantings were responsible for most of the reduction in net emissions between the pastoral component and the rest of the non-pine catchment (Fig. 16), sequestering a maximum of 80 t CO_{2e} yr^{-1} (Fig 19). In the first 2 yrs the plantings were a net source of 30 t CO_{2e} yr^{-1}, which was due to the 53 t CO_{2e} yr^{-1} soil carbon loss not being matched by plant growth. From 2003 to 2006 growth was high enough to overcome this loss, and the growth was assumed to continue to be this high after 2006. From 2001 to 2011 the native restoration plantings offset 10.5% of the pasture's net emissions.

4.2 COMPARISON OF FOREST TYPES WITH EACH OTHER, EMISSIONS TRADING SCHEME LOOK-UP TABLES, & PROJECTIONS FOR PINE GROWTH

Once established, the pines grew much faster than the native plantings or forest fragments, even on the southern slopes where they had lower growth rates (Fig. 21). Native plantings grew slower than the pines, but faster than the native forest fragments. The reverse was true of the carbon stocks, with the native fragments having the highest stocks per ha (more than any of the pine forest classes) and the native plantings the lowest (Fig. 22). The plantings were an estimated 65% of the mass of the slowest growing pines in 2011. In Figs. 21-24, all non-southern slopes for pine forest have been averaged, because, in general, they were very similar in terms of sequestration values, and only the southern slopes were consistently lower. The Emissions Trading Scheme look-up table for pine uses the Waikato/Taupo region value.

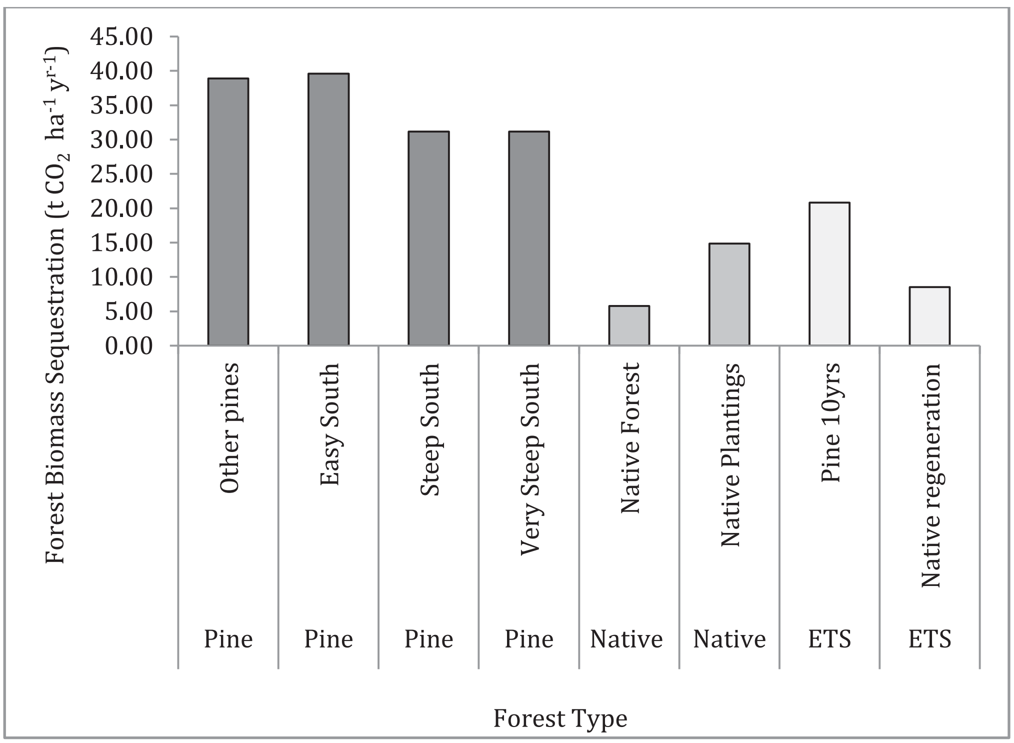

Figure 21. Carbon sequestration rate in forest biomass (live and dead plant matter) for different forest types in 2011, and the Emissions Trading Scheme (ETS) lookup table values for forests at age 10. 'Other pines' is the average of all non-southern slope/aspect classes in the pine forest.

Both the pines and native plantings had much higher carbon sequestration rates and biomass than the estimates from the ETS look-up tables (Figs. 21, 22). ETS values for pine sequestration were only 53.49% of the average for non-southern slopes, and the ETS estimates for carbon stocks were only 34.58% of the average for non-southern slopes. The ETS value for native regeneration's sequestration rate was 57.33% of that estimated for the native plantings, and the only 29.34% of the present estimate for the carbon stocks.

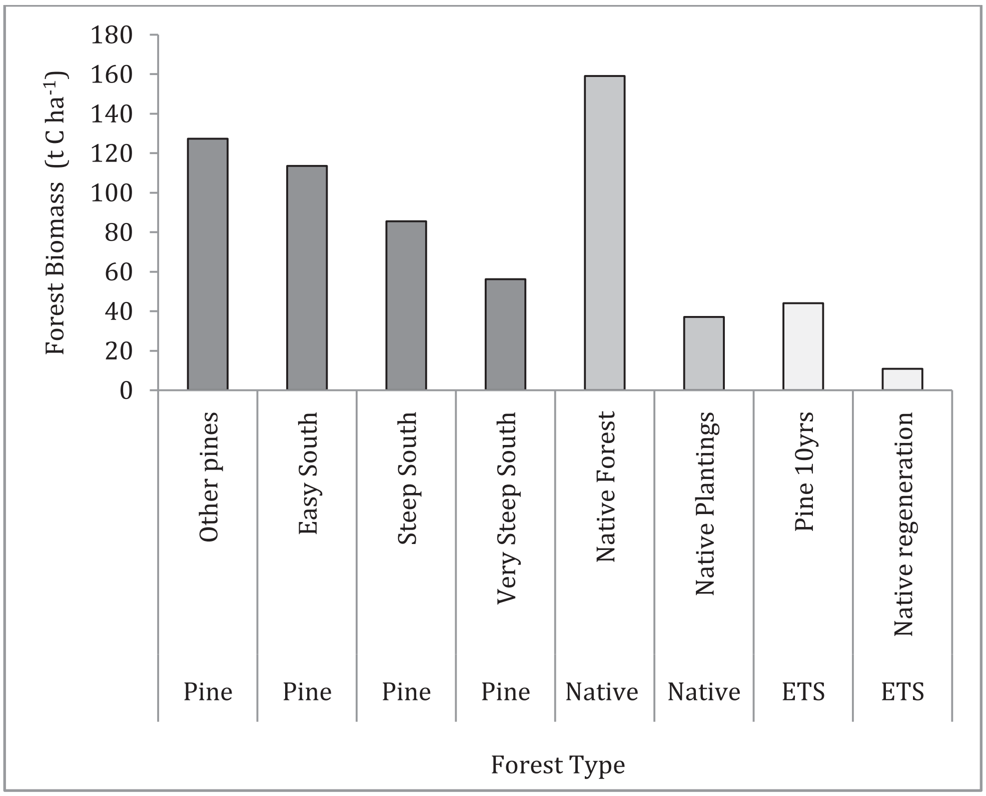

Figure 22. Carbon in forest biomass (live and dead plant matter) for different forest types in 2011, and the Emissions Trading Scheme (ETS) lookup table values for forests at age 10. 'Other pines' is the average of all non-southern slope/aspect classes in the pine forest.

Pine growth was modeled out to 30 years (i.e. projecting growth a further 20 years into the future, by extended the model run in CenW to cover the longer time frame, and re-running it over the previously used climate data when necessary), with harvesting at year 30 and no additional thinning or pruning. Over this extended time period the carbon stocks in the pine forest grew well beyond those currently in the native fragments, however, on very steep southern slopes they achieved only slightly higher levels (Fig 23). Again, after 30 years non-southern slopes were still similar and southern slopes still had much lower biomass. After harvest each slope/aspect class lost on average 74% of their biomass. The ETS predicts the stock at 30 years for this region to be only 220 t C ha^{-1}, and that only 56% would be lost at harvest (Ministry of Agriculture and Forestry, 2011d), both of which are much lower than the present results predict.

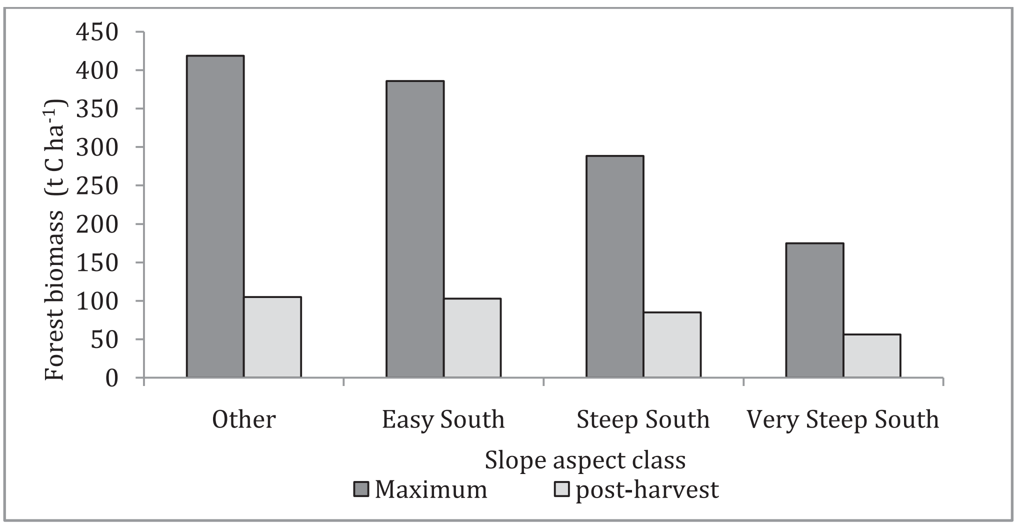

Figure 23. Pine forest biomass as projected at 30 years old, and the remaining biomass after harvest at 30 years. 'Other' is the mean of all non-southern slope/aspect classes.

The average sequestration rates from year 10 to 30 in the pine forests were slightly higher than in the first 10yrs, except for the very steep southern slopes which were slightly lower (Fig. 24).

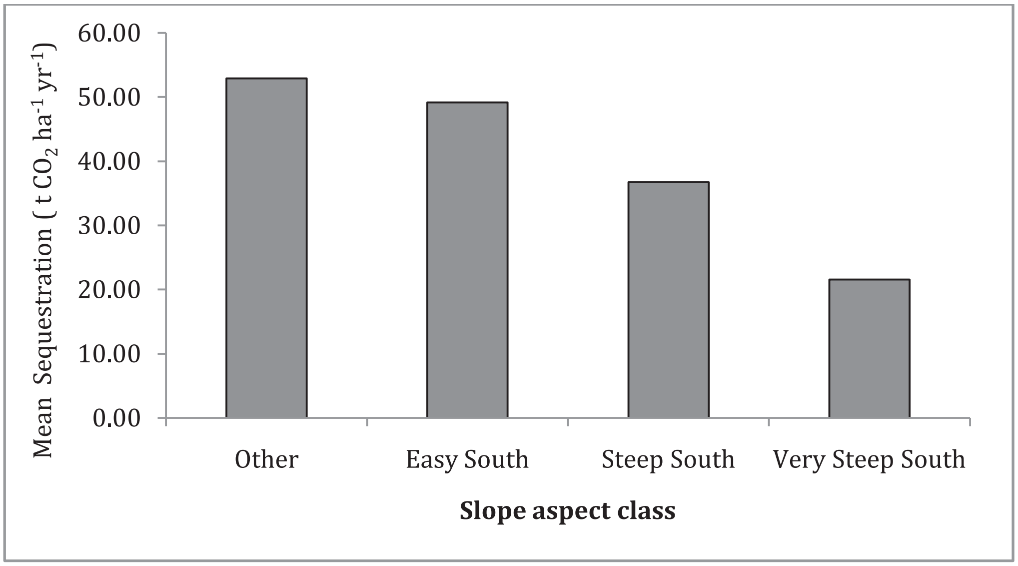

Figure 24. Average sequestration rate in pine forests between year 10 and 30, as predicted by CenW. 'Other' is the mean of all non-southern slope/aspect classes.

4.3 SOIL CARBON

Large losses of soil carbon appear to be occurring under most of the land-uses. The change for the native plantings indicated no significant difference when both depths were combined, but when split into soil depths the topsoil lost a significant amount of carbon (Table 31). The grazed pasture is also losing carbon, with the decline proportionally much larger in the subsoil (Table 31). Ungrazed forests appear to be losing large amounts of carbon from their topsoil, but this is only significant at a 0.1 level (Table 31). No analysis could be done with the pines as there were no previous soil samples. The one sample point available suggested an increase in soil carbon, but if the 2001 pasture values were applied to pine forest areas, it would suggest a decline. Analysis by slope and other factors was hampered by the small sample size (Table 32).

Some of these losses are very large. The pasture sites that were re-sampled lost an average of 2.47% yr^{-1}, almost a quarter of the total over the 10yrs. When the pasture was split into the two depths the subsoil was losing proportionally more at 3.16% yr^{-1}, whereas the topsoil is losing 2.09% yr^{-1}. The ungrazed native forest fragments lost 1.77% yr^{-1}. However, this loss is only occurring in the topsoil, which means the topsoil is losing 3.72% yr^{-1}. The topsoil of the native plantings lost 2.18% yr^{-1}.

Based on the 2011 samples for the full 0-20cm depth, the native plantings had the largest mean soil carbon stock (105.6 ±28.42 t C ha^{-1}). This was followed by the pasture and pines (83.1 ±4.71 t C ha^{-1} and 80.2 t C ha^{-1} respectively), then by the grazed and ungrazed native forests (71.2 ±17.77 t C ha^{-1} and 72.6 ±10.10 t C ha^{-1} respectively). The subsoil tended to have in the range of half to two-thirds less carbon than the top-soil for all the land uses. Flatter slopes tended to have higher carbon levels in all land uses except the native plantings, which showed the opposite trend. However, the lack of replication for some land uses, when split up by slope class, made it difficult to determine the true effects of slope.

Table 31. Changes in soil carbon by land-use and soil depth between 2001 (pre-land-use change) and 2011. N = sample size, 'Change' is the difference between the two means and the p-value is from paired t-tests for the difference between the two means. NA = Not applicable. The p-values in bold are significant at α=0.1. 95% confidence intervals of the mean are in brackets. 'top' = 0-10cm depth. 'sub' =10-20cm depth, each sample was calculated from a compound of 10 cores for carbon % and mean of two bulk density cores per depth.

Land-use	**Soil**	**2001 Mean t C ha^{-1}**	**2011 Mean t C ha^{-1}**	**N 2001**	**N 2011**	**Change**	**p-value**
Native planting	top	82.7 (±5.28)	64.7(±17.55)	4	4	-18.02	**0.03**
-"-	sub	40.9 (±12.33)	40.9 (±29.36)	4	4	0.015	1.00
Grazed native forest	top	31.8 (±166.56)	43.0 (±13.25)	2	2	11.18	0.57
-"-	sub	21.4 (±76.33)	28.2 (±101.96)	2	2	6.82	0.71
Ungrazed native forest	top	56.3 (±14.72)	43.3 (±14.28)	6	4	-20.96	**0.06**
-"-	sub	31.9 (±9.60)	29.3 (±8.74)	6	4	-1.36	0.46
Pine	top	48.9	48.3 (±13.05)	1	6	-0.6	na
-"-	sub	27.4	31.8 (±12.65)	1	6	4.4	na
Pasture	top	70.9 (±67.01)	56.1 (±51.10)	2	2	-14.79	**0.05**
-"-	sub	39.5 (±36.96)	27.0 (±20.55)	2	2	-12.49	**0.07**

Table 32. Changes in soil carbon by land-use and slope between 2001 (pre-land-use change) and 2011. N = sample size, 'Change' is the difference between the two means and the p-value is from paired t-tests for the difference between the two means. NA = Not applicable. The p-values in bold are significant at α=0.1. Total sampling depth is 20cm, each sample was calculated from a compound of 10 cores for carbon % and the mean of two bulk density cores at two depths. Mod. = moderate.

Land-use	**Slope**	**2001 Mean t C ha^{-1}**	**2011 Mean t C ha^{-1}**	**N 2001**	**N 2011**	**Change**	**p-value**
Native planting	Easy	125.4	77.4	1	1	-48.0	na
-"-	Mod.	99.8	118.3	2	2	18.6	0.33
-"-	Steep	145.4	132.3	1	1	-13.1	na
Pine	Easy	76.3	91.6	1	3	15.3	na
-"-	Steep	na	68.8	0	3	na	na
Grazed native forest	Mod.	34.1	80.25	1	1	46.2	na
-"-	Steep	72.3	62.12	1	1	-10.2	na
Pasture	Mod.	110.4	83.1	2	2	-27.3	**0.001**
Ungrazed native forest	Easy	89.5	83.8	2	1	-5.7	na
-"-	Mod.	87.6	68.8	4	3	-18.7	**0.069**

4. 4 CATCHMENT NET GREENHOUSE GAS BALANCE & CARBON STOCKS

The new land use plan transformed the Mangaotama catchment from a net source of emissions to a net sink within 2 yrs. The original land use (pre-2001) would have emitted 10.99 Gg CO_{2e} for the whole of the catchment over 10yrs, whereas the new land use sequestered 47.26 Gg CO_{2e} in its first 10yrs. Originally, the only area of the landscape which was a net sink was the grazed native forest fragments, and this was vastly overwhelmed by the emissions coming from the pasture (Table 33). During the transitional period, the pine forest added a large sink which canceled out the emissions from the pasture. The additional net sinks in the native forest fragments, native plantings and riparian zones added a relatively small amount to the catchment net sequestration. During the current period, the pine forest grew into a sink that was orders of magnitude larger than the other land uses; the net emissions from the pasture reached only 7% of the absolute value of the net sequestration in the pine forest, down from 57% in the transitional period. Net sequestration in the native plantings matched only 1% of the net sequestration in the pines.

Table 33. Total net balances of each land use during each time period; positive values are net emissions, negative values are net sequestration. OLD – 1yr long, old management regime (before conversion to pines and other land-use changes). TRA – 5yrs long, transitional period (pines and natives planted, fragments and riparian zones fenced). CUR – 6yrs long, current management regime (established pine forest and increased fertilizer use). Na = not applicable. 'no soil' = soil carbon loss excluded.

	Net GHG balance, Gg CO_{2e}		
	OLD	**TRA**	**CUR**
Pines	Na	-5.27	-47.10
Pasture	1.21	2.98	3.29
Grazed native forest fragments	-0.11	-0.11	-0.07
Ungrazed native forest fragments	Na	-0.18	-0.13
Ungrazed native forest fragments (no soil)	NA	-0.50	-0.53
Native plantings	Na	-0.18	-0.48
Grassed riparian zones	Na	-0.01	-0.01
Whole Catchment	1.10	-2.77	-44.50

The catchment's total carbon stocks rose by 15.9 Gg C, from 30.54 Gg C in 2000 to 46.45 Gg C in 2011 (Fig. 25). This change was almost entirely due to the pine forests, which increased by 14.3 Gg C. The carbon stock in pasture declined by 236 t C (not including the pasture reclassified due to the land use changes), carbon stocks in the native forests rose by 182 t C and in the native plantings by 180 t C (Fig. 26), but these all had only a small effect in comparison with the pines.

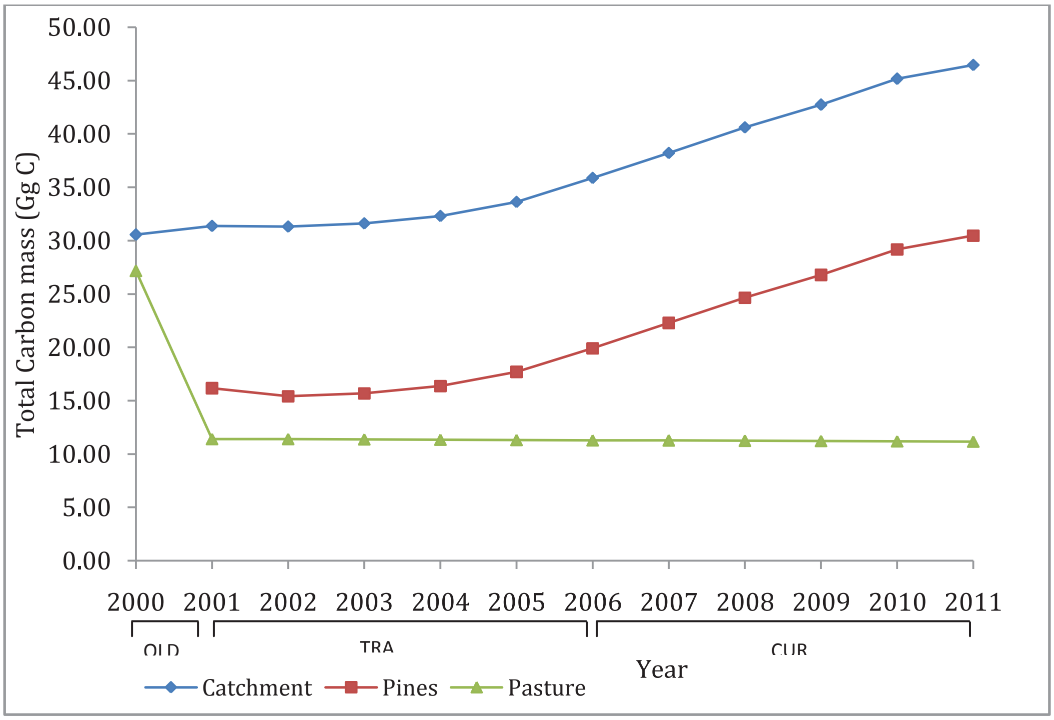

Figure 25. Change in carbon stocks for the whole Mangaotama catchment and for the two major land uses before the land use changes (OLD) and for the following 10yrs. Stocks include both soil and biomass pools. Pasture includes ungrazed riparian zones. OLD – old management regime (before conversion to pines and other land use changes), TRA – transitional period (pines planted and establishing), CUR – current management regime (established pine forest).

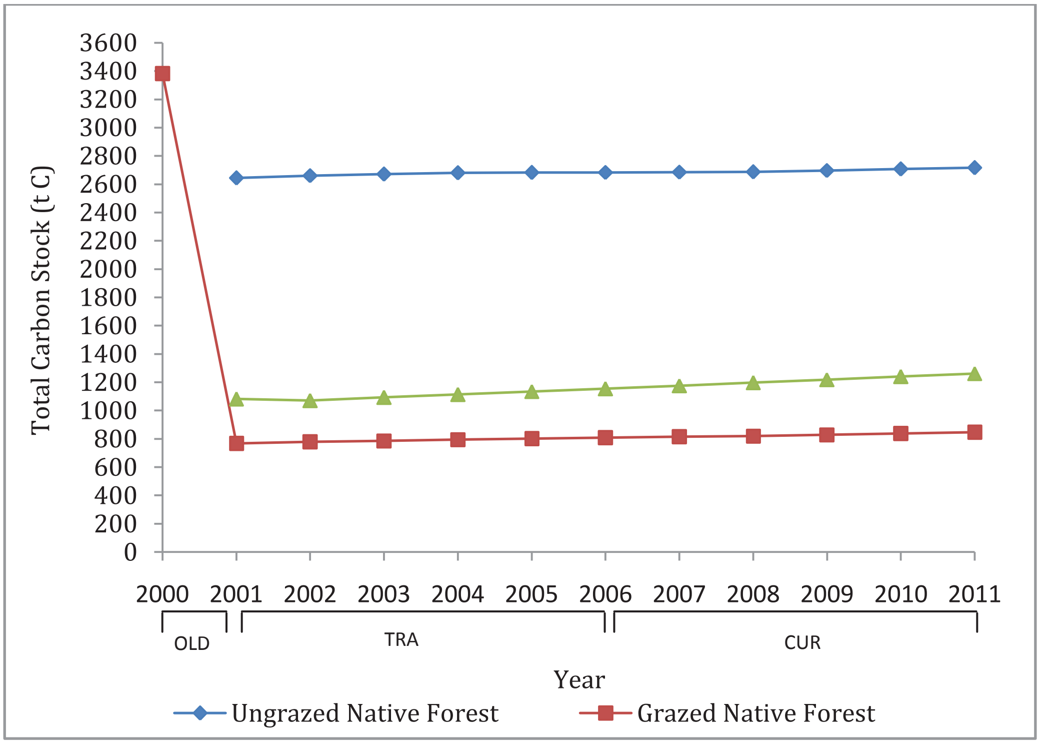

Figure 26. Changes in the total carbon stocks for the native forest fragments and native plantings for the whole Mangaotama catchment from before the land use changes (OLD) until 10yrs after. Includes both plant biomass and soil pools. OLD – old management regime (before conversion to pines and other land use changes), TRA – transitional period (pines planted and establishing), CUR – current management regime with the established pine forest.

CHAPTER 5: DISCUSSION

5.1 SUMMARY: EFFECT OF THE CATCHMENT MANAGEMENT PLAN

In addition to the environmental and economic benefits already demonstrated, this plan was an effective way of mitigating the farm's impact on climate change. The plan quickly transformed the farm from a net source of emissions into a much larger sink. The vast majority of this effect was due to the conversion of around half of the farm in to *Pinus radiata* forestry. The pine forest dominated the farm's total GHG balance so much that the presence of the pastoral component became of little importance. Reforestation of the steeper, poorer quality land was more than able to compensate for the intensified emissions from the remaining productive land. The extent of this compensation was so large, that even though the farm is not 'industry typical' because it has a greater portion of steep land than is normal, it could easily remain a net sink with even double the amount of flatter pasture in the catchment. Therefore, this study suggests that partial forestry conversion in hill country farms could be a viable means for helping the agricultural sector meet the otherwise difficult demands of New Zealand's climate change obligations. The Mangaotama farm met both the pressure to increase its profitability and more than off-set its own emissions.

The other land-use changes were dwarfed by the forestry conversion; nevertheless, they did have a significant impact on the net GHG balance of the farm. The most important aspect was the reduction in livestock numbers due to the retirement of pastoral land and the change to a more productive farming system. The per hectare emissions of the pasture actually increased as a result of intensification, but this increase was not on a par with the net decrease of emissions due to the loss of half of the pasture land to trees.

Compared with the pine forest, the native plantings and native forest fragments were less effective at offsetting the catchment's total emissions due to their much smaller total area and slower growth rates. Even so, they still offset between a fifth and a third of the pasture's emissions (the latter being the case if the possible loss of soil carbon in ungrazed fragments is ignored). Under the original system the native forest fragments managed to offset the emissions occurring from the grazing in them, plus some of the emissions from the rest of the farm (although the more heavily grazed fragments could be net sources). Native vegetation can, therefore, be an important part of mitigating some of a farm's emissions.

Creating ungrazed riparian zones was the least important change. The only real gain was from the reduced grazing emissions. The ungrazed riparian zones were estimated to produce a negligibly small sink, but this was based on very uncertain values applied from studies of different sites. However, this study might not have picked up improvements due to potential regeneration and reduced nitrogen inputs into water logged soils with a high capacity for emitting N_2O.

Many whole-farm system studies of GHGs in livestock agriculture have been done. Crosson et al (2011) reviewed these for beef and dairy cattle production. Most of these studies differ substantially from the present study, focusing solely on the livestock component of the farm, ignoring potential carbon sequestration in trees, and very rarely considering potential sequestration in the soil. As a result, the mitigation options they assessed were concerned with manipulating the livestock system only. In contrast, in the present study the management of trees was a primary mitigation strategy, alongside changes to the livestock system, and soil carbon changes were included. The focus on livestock by the studies in Crosson et al (2011) means many of them done a more in depth analysis of mitigation options in pasture than was done for the Mangaotama. However, the present study gives a more holistic overview of the effects of changes, particularly in the use of vegetation, on the whole farm landscape. The use of trees in agriculture to offset GHGs is an attractive yet often ignored option (Schoeneberger, 2009). However, in New Zealand sheep and beef farmers may be relatively open to the idea of planting trees on marginal land to offset their emissions costs under the ETS (Rosin et al., 2011), and currently the means of reducing emissions in extensive agriculture limited (Pinares-Patino et al., 2009). Consequently the work on the Mangaotama helps to fill an important gap.

Some general findings and issues identified in Crosson et al. (2011) are relevant, despite the differences between the present study and most others in this field. First, uncertainty and a lack of comparability between studies were issues. These studies often involved the use of poor quality data on farm systems, differing system boundaries, and different sources for emission factors. The present study shares this problem of uncertainty in its estimates and difficulty in comparing it with others. Second, in Crosson et al. (2011) intensification always led to an increase in emissions on an area basis, but could decrease emissions per unit of output, provided that the productivity gain was larger than the increase in emissions. In the Mangaotama, production of beef was increased from a smaller area (Dodd et al., 2008a), meaning emissions per unit output would have declined. Also, when the pasture is considered alone, per ha emissions were increased due to the intensification, again as seen elsewhere. However, when the whole farm landscape was considered together, intensification was compatible with a large decrease in net emissions for the whole area, in contradiction of the increases in emissions seen in the studies in Crosson et al. (2011). Accounting for the amount of area needed and the presence of carbon sinks in the wider landscape explains this difference. Third, carbon sequestration in the soil, although not normally tested for, was often expected to substantially offset emissions. The reverse was found to be true in the Mangaotama. The long-term data from Schipper et al. (2009) show that the soil can fluctuate between being a source and a sink of carbon, and during the time of the present study was a major emissions source. Soil carbon cannot be assumed to always be a potential sink.

5.2 INDIVIDUAL LAND USES, SENSITIVITY ANALYSIS & UNCERTAINTIES

5.2.1 PINE FORESTS

The growth rates for the pines were above average for New Zealand as a whole. For example, the average annual biomass gain for New Zealand's planted forests, excluding harvest, has been estimated at 6.4 t C ha^{-1} (23.5 t CO_2 ha^{-1}) (Hollinger et al., 1993), which is well below the maximum rates simulated for the Mangaotama, and more similar to those found for the southern slopes. As a result, the size of the sink achieved here may be larger than what would be achieved in some other parts of the country.

Converting steep marginal areas of the farm to forestry may be an attractive option, but thought must be given to which areas are chosen. Modeled pine growth was reduced on the very steep southern slopes to less than half of that elsewhere. Southern slopes were much colder and darker, especially in winter, than other areas. Even if this was an underestimate, they are still likely to be somewhat lower than other areas. These results suggest that planting the steepest southern slopes could need up to double the area, compared with planting other slopes, to achieve the same emissions off-set.

5.2.1.1 SENSITIVITY ANALYSIS

A sensitivity analysis done on the Steep Eastern class (taken as typical of non-southern slopes) in CenW showed that the most important factors contributing to the growth of pines were the presence or absence of thinning, the maximum weed height, relative solar radiation, and large decreases in rain (Table 34). All other tested factors produced less than 3% change. Not thinning at all increased the biomass by over 9% by year 30, but merely changing the number of trees thinned had little effect. Postponing thinning only had an effect at year 10 because it was the same as not thinning for this date. Taller weeds suppressed early growth but this effect largely disappeared by year 30. Large changes, especially decreases, in the relative solar radiation produced the largest change in biomass.

In tests of the original CenW model it was found that thinning lead to substantially greater wood production but had little effect on Net Primary Productivity, and that the removal of canopy matter may only be important in cases where radiation is limiting (Kirschbaum, 1999). These tests also showed that growth increased with rainfall, but only by a small amount if radiation was limiting, and growth increased with higher radiation up to an optimum before declining again due to increased evapo-transpiration (Kirschbaum, 1999).

Table 34. Sensitivity analysis of silvicultural and other inputs, and climate adjustment parameters for the Steep East class in CenW. Values above 3% in bold. Negative values indicate lower biomass than in the original model. Changes to the date of thinning also involved shifting the last pruning to coincide with the new date, as per the original model.

Parameters changed	**% Difference from Original Model Biomass**	
	yr 10	**yr 30**
No pruning	2.43	0.66
Harsher pruning (double the % branches cut)	-2.54	-0.77
1yr earlier thinning	-2.62	2.15
1yr later thinning	**5.57**	-2.06
Thinning to 500sph	0.68	1.19
No thinning	**4.63**	**9.30**
Thinning to 300 sph	-0.71	-1.20
Relative size of cut stems, upper limit (100%)	0.02	-0.19
Relative size of cut stems, lower limit (73%)	-0.18	-0.74
-1m max weed height (1m)	**3.51**	0.39
+1m max weed height (3m)	**-4.88**	-0.55
+10% relative solar radiation	-1.65	**-4.38**
-10% relative solar radiation	**-5.1**	-2.92
+30% relative solar radiation	**-13.86**	**-23.33**
-30% relative solar radiation	**-29.13**	**-28.12**
+10% rain	0.37	0.52
-10% rain	-0.61	-0.96
+30% rain	0.53	1.21
-30% rain	**-3.07**	**-4.47**

A sensitivity analysis of the climate factors was also done for the Very Steep Southern Class (Table 35), which had by far the lowest biomass in the model and the lowest relative solar radiation values. The biomass in this class responded much more strongly to changes in the relative solar radiation than the Steep Eastern Class, thus the response to this parameter is much stronger at low levels than at higher levels. Again, rainfall had little to no effect.

Table 35. Sensitivity analysis of climate parameters for Very Steep Southern class in CenW.

Parameters Changed	% Difference from Original Modelled Biomass	
	yr 10	yr 30
+10% relative solar radiation	**14.62**	**18.57**
-10% relative solar radiation	**-21.42**	**-20.15**
+30% relative solar radiation	**36.78**	**44.25**
-30% relative solar radiation	**-71.95**	**-65.61**
+10% rain	-0.63	-0.58
-10% rain	0.16	0.00
+30% rain	-2.55	-1.91
-30% rain	-0.17	-0.87

5.2.1.2 PINE FOREST MODEL VALIDATION

Validation tests of the first version of CenW against experimental data from a *Pinus radiata* forest in Australia concluded that it was able to successfully simulate carbon, energy, nitrogen and water across five very different treatments for 4-5 years (Kirschbaum, 1999). It showed excellent agreement between observed and modeled values, explaining 97.4% of the observed variation for total stem and branch wood, and 96.6% of the observed variation in mean stem diameter (Kirschbaum, 1999).

An attempt was made to compare the CenW model against field data for the Mangaotama site by comparing modeled and observed DBHs. Many of the modeled DBHs for the pines fell outside the confidence intervals of the field-measured ones, but not by far. However, the comparisons between the model and field DBH measures were hampered by several factors. First, movement in the pine forest was extremely difficult due to the recently thinned trees, which often meant samples could not be taken in ideal locations. The slopes and aspects for the field sites were only very rough estimates for the site, as individual trees could differ in micro site slope, and the aspect could change rapidly in an area. An effort was made to keep within the class and to get enough samples, but sometimes this meant borderline sites had to be done which partially covered a different class, or some distance needed to be travelled at each site to get enough trees within the one class. Due to the accessibility problems, some trees measured were near to the forest edge, thus violating the models assumptions of a closed canopy. Slopes may have been consistently steeper from those in the model in the case of Easy North, Easy South, and Steep West/East. Second, the exact thinning date for each particular site sampled was not known. Thinning had a large impact on the average DBH in the model (resulting in a large jump due to the removal of small trees), despite having much less effect on the actual biomass. These factors plus the small sample size and restricted range of sampled locations may explain the deviations between the model and the field observations. Despite these problems there was still a significant relationship between the observed and modeled DBHs.

5.2.1.3 CLIMATE ADJUSTMENTS

The greatest uncertainty in the pine forest modelling comes from the climate adjustments, a fact indicated by the sensitivity analysis and by the fact that the CenW does not currently feature any means for carrying this out. The relative solar radiation step was the most important part. The relative solar radiation value was used in the model to carry out the climate adjustment which determined received radiation, temperature and relative humidity. A lower value for relative solar radiation meant darker and colder conditions whereas a higher value meant sunnier and warmer conditions, relative to a flat surface.

Given that the non-southern sections were generally similar, and that the relative solar radiation had a much greater effect at lower values, the biggest issue was with the southern slopes. It is possible that the biomass for the southern slopes was underestimated (see discussion below). Nevertheless, this uncertainty is likely to have had a relatively small impact on the overall balance, considering that basing the simulation solely on average slopes and excluding northern and southern slopes altogether only raised the estimate by a maximum of 14% and was sometimes no different at all (see the section 5.4.2, table 40). Whether this difference will continue to increase over time as it did in the first 10 years is not known. As the pines had such a large effect on the net GHG balance in the catchment, the climate adjustment was likely to be one of the greatest sources of uncertainty in the study.

There were several assumptions which may have affected the ability to accurately adjust the climate data for the topography. For the temperature, it was assumed that solar radiation was the sole factor influencing temperature at any one point. Factors such as wind exposure and altitude were not taken into account, although the range of altitudes covered was not overly large. Values for isolated 5m x 5m blocks, or very thin strips, identified by the GIS maps but surrounded by large areas of a different slope/aspect class were more likely to be influenced by the surrounding air than by that patch's specific radiation. Despite these problems, the temperature values may not have been far off. A study of temperatures at Whatawhata in pasture on steep (30^{o}) southern and northern slopes found that on clear days in January southern slopes were 83% as warm as northern slopes and in July they were 38% as warm (Gillingham and Bell, 1977). This compares well with the present study, where the same values were estimated as 87% in January and 30% in July. However, Gillingham and Bell (1977) found that on overcast days this relationship was different, with southern slopes never more than 20% colder than northern slopes. The modeled temperatures did not factor in the effect of cloud cover on relative temperatures, meaning temperatures would have been repeatedly underestimated for southern slopes.

The same problem will have applied to the solar radiation itself. Total irradiance is primarily the product of direct irradiance, from the sun, and diffuse sky irradiance, which is scattered throughout the atmosphere and depends on both the sun's position and atmospheric conditions (Duguay, 1993). Southern slopes which do not face the sun will often receive minimal direct irradiance, but they will still get diffuse sky irradiance. A study which developed a technique for calculating solar radiation on surfaces of a given slope and aspect found that vertical (90^{o}) northern facing slopes (in the northern hemisphere, thus facing away from the sun) received more radiation on cloudy days than on clear days, due to increased diffuse radiation, whereas slopes facing the sun received less radiation on cloudy days (Allen et al., 2006). Therefore, cloud cover narrows the difference between northern and southern slopes, as was the case for temperature. Again, this may have led to underestimated growth on the southern slopes.

Humidity and rainfall faced some problems too. The relative humidity adjustment assumed a constant actual vapor pressure across the landscape, while this might differ between some sites, resulting in slight differences in evapo-transpiration. The rainfall adjustment did not account for the prevailing wind direction, which could result in the aspect having an effect in addition to the slope.

Another issue was the need to split the landscape into categories for the sake of tractability. Averages had to be applied to these categories, thus some of the spatial variability in slope and aspect was lost. In addition to climate, other factors are likely to vary with topography, such as nutrient availability from differences in soils, and these could not be captured by the model. Another problem was the lack of climate data from 2006 onwards. Climate data for earlier years had to be used to fill the gap, meaning the actual climate after 2006 will have differed from that used in the model.

Given the lack of long-term field data and any current comprehensive treatment of topography in the CenW model, these adjustments are likely to provide sufficient estimates until improvements are made in the model to develop its ability to handle these factors.

5.2.2 PASTURE

Rhodes et al. (2011) investigated the GHG emissions of a New Zealand hill country sheep and beef farm using the OVERSEER and FARMAX models. Rhodes et al. (2011) estimated emissions from the farm to be 1946 t CO_{2e} yr^{-1}. The Mangaotama's pasture originally had net emissions that were just over half of those from Rhodes et al. (2011), but the Mangaotama also had about half the area. Rhodes et al. (2011) also found that shifting the cattle away from a breeding herd system and increasing the number of sheep increased total emissions by 13% but also increased productivity, which is similar to the outcome from the livestock changes in the present study.

Comparison between the Mangaotama's emissions and other sheep and beef farms is primarily an issue of stocking rates and nitrogen fertilizer use, mediated by feed quality and soil types, multiplied by the size of the farm. In 2002, stocking rates for intensive sheep and beef in New Zealand were between 10.2-12.6 SU ha^{-1} (Parliamentary Commissioner for the Environment, 2004). The Mangaotama had an average of 8.8 SU ha^{-1} under the old regime, which increased to 10 SU ha^{-1} under the new regime (Dodd et al., 2008a), putting the catchment at the lower end of what is typical. For a sheep grazed pasture (16-18 SU ha^{-1}, 36.8 kg N fertilizer ha^{-1} yr^{-1}) in the Manawatu region, annual N_2O emissions have been measured at 3.7 kg N_2O ha^{-1} yr^{-1} (1102.6 kg CO_{2e} ha^{-1} yr^{-1}) (Saggar et al., 2007b), which was in the mid to low end of the range of the values used in the present study (689-2159 kg CO_{2e} ha^{-1} yr^{-1})(Saggar et al., 2007b). A dairy pasture in the Manawatu region had much higher emissions at 32 g N_2O ha^{-1} day^{-1} (3483.02 kg CO_{2e} ha^{-1} yr^{-1}), which exceeded the values used in the present study. Based on this, the per ha emissions from the Mangaotama catchment are likely to be average, or somewhat higher than for comparable sheep and beef farms, but likely lower than for dairy.

A sensitivity analysis of OVERSEER (Table 36) showed that the modeled estimates of methane and nitrous oxide emissions from ruminant CH_4 and N_2O from OVERSEER were most sensitive to changes in total livestock numbers, nitrogen fertilizer inputs and pasture quality; all the other parameters were largely unimportant. For every 1% increase in stock there was a nearly 1% increase in emissions. For every 1 kg increase in nitrogen fertilizer there was a nearly 0.5% increase in N_2O emissions, but because CH_4 was unchanged by fertilizer additions the impact on total emissions was much smaller. Switching from ryegrass-clover to poorer quality brown-top pasture raised emissions by just over 5%.

Table 36. Sensitivity analysis for OVERSEER. Based on Class V land for the old land-use regime. Values above 3% in bold. Negative values indicate lower emissions than in the original model.

Parameter Changed	**% difference from the original model**		
	CH_4	**N_2O**	**Total (CH_4 +N_2O)**
+10% all stock	**9.85**	**10.12**	**9.92**
-10% all stock	**-9.81**	**-10.12**	**-9.90**
+30% all stock	**30.33**	**31.15**	**30.55**
-30% all stock	**-30.33**	**-31.15**	**-30.55**
Sheep:Cattle = 40:60	-0.59	-1.22	-0.77
Sheep:Cattle = 50:50	-0.30	-0.61	-0.38
Sheep:Cattle = 70:30	0.30	0.61	0.38
Sheep:Cattle = 80:20	0.63	1.31	0.81
+10% wool produced	0.00	-0.26	-0.07
-10% wool produced	0.00	0.26	0.07
+30%wool produced	0.00	-0.79	-0.22
-30%wool produced	0.00	0.87	0.24
100% beef male	0.00	0.00	0.00
50% beef male	0.00	0.00	0.00
No beef finishing	0.00	2.71	0.74
+30kg fertilizer N ha^{-1} yr^{-1}	0.00	**15.01**	**4.11**
+50kg fertilizer N ha^{-1} yr^{-1}	0.00	**24.96**	**6.84**
+100kg fertilizer N ha^{-1} yr^{-1}	0.00	**50.00**	**13.70**
Brown-top grass	**5.24**	**5.76**	**5.38**

As most of the parameters in Table 36 were relatively well known, and some of the more uncertain ones had little effect on the model, the main uncertainty is with the OVERSEER model itself. The OVERSEER model's estimates of enteric CH_4 and N_2O are based on national inventory emission factors. The variability across the country may make the use of a single national emission factor for N_2O inappropriate (de Klein et al., 2003). The spatial and temporal variability of emissions will certainly be much greater than was captured here. In particular, wetlands – which may have had higher N_2O emissions – were not treated separately due to a lack of information. This means potential emission reductions, for example from retiring areas prone to water-logging with a high capacity for denitrification, will not have been picked up by this study. The impact of other pasture management practices, such as set stocking versus rotational grazing, or the timing of fertilizer application, also could not be assessed.

The pine conversion had such an overwhelming effect on the net GHG balance of the catchment that uncertainties in the pasture estimates would need to be very large to have an impact on the net balance. Thus, these estimates for pastoral emissions are sufficient for giving an indication of the amount of GHGs that needed to be off-set, if not the fine scale spatial variation in emissions and how they themselves can be reduced. Greater consideration of the impact of pasture management on emissions will be needed in further studies of the catchment.

5.2.3 NATIVE FOREST FRAGMENTS

Richardson and Burrows (2004) previous study of the Mangaotama fragments found aboveground carbon in the vegetation of the fragments to be 110-126.5 t C ha^{-1} in 2000 and, 116.5-153.5 t C ha^{-1} in 2002. Adding 25% for below ground biomass would put this at 137.5- 158.13 and 145-197.7 t C ha^{-1}, respectively. These values are higher than the averages from the present study, being outside the 95% confidence interval in both years. The present study differed in several respects from Richardson and Burrows (2004). In particular, the current study had actual height measurements and site specific regressions for heights, and was corrected for the errors Richardson and Burrows (2004) had encountered, as well as covering a longer time period. This may explain some of the differences between the two estimates.

Richardson and Burrows (2004) identified measurement errors in the data because they produced unusually high growth rates, with normal gains of biomass thought to be around 1.7 t ha^{-1} yr^{-1} (Hall, 2001). These errors have since been corrected (M. Dodd, personal communication, 2011) and the average values obtained for the present study were close to this typical rate, but some were slightly higher. Some higher values should be considered reasonable as this is not typical continuous forest. The fragments have received ongoing inputs of fertilizer (Stevenson, 2004) which could be affecting growth. Edge habitats in fragments have often been shown to have higher densities than interior habitats (Young and

Mitchell, 1994, Davies-Colley et al., 2000) and these fragments are dominated by edges.

The Mangaotama fragments were highly degraded prior to the conservation activities (Smale et al., 2008). Unsurprisingly, the carbon mass of the native forest fragments appears to be lower than what is typical for New Zealand forests. On average, live carbon mass was between 109 and 150 t C ha^{-1} at Mangaotama. Richardson et al (2009) found live mean tree biomass to be 315 t ha^{-1} across New Zealand native forests, which, when adjusted for carbon and below ground biomass is 197 t C ha^{-1}. Using the LINKNZ model, Hall (Hall, 2001) simulated forest aboveground biomass for a site near Christchurch to be 150-240 t C ha^{-1} in mature native forest. Adjusting this for below ground carbon would give 187-300 t C ha^{-1}. In Nelson beech forest, 479 t ha^{-1} of live tree biomass was recorded (Hart et al., 2003), which when converted to carbon gives 239.5 t C ha^{-1}.

Comparisons with studies including soil carbon pools are more difficult, due to differences in the soil depth included. Forests in a transect across the South Island were found to have a total carbon stock of 290 t C ha^{-1} in all pools, including soils to 1m (Coomes et al., 2002). Hart et al (2003) found total ecosystem storage of carbon, including soils 0-60cm, in Nelson beech forest to be 344 t C ha^{-1}.

Although the average biomass carbon stocks for the Mangaotama native fragments were generally lower than forests elsewhere, when they were split up by slope class, the steep slopes, in 2011 at least, had 235.57 t C ha^{-1}, which is closer to that observed in other NZ native forests. The flatter slopes had much lower carbon biomass values, suggesting that the flatter slopes were being suppressed, presumably due to grazing. Flatter slopes are more easily accessed by stock, and are more likely to have regeneration suppressed and suffer other damage. As was seen in the analysis of the pine forest, steep slopes can gain relatively more light, creating the potential for better growth. However, this would only occur on those slopes facing the sun and southern slopes would suffer a reduction in growth. No effect of aspect was found, discounting this as a likely explanation. The fact that this difference was found between slope classes, despite no effect of removing grazing, suggests that the time-scales for the recovery of carbon stocks degraded by grazing are longer than 10 years. Other pressures may have also contributed to a generally lower carbon stock at

Mangaotama, including the influence of fragmentation, and legacy effects of the original land clearance and subsequent selective logging.

Coarse woody debris mass at Mangaotama was also lower than what is typical for New Zealand native forests. Richardson et al (2009) found mean deadwood mass in New Zealand native forests to be 54 t ha^{-1} ranging from 0 to 550 t ha^{-1}; which is equivalent to 27 t C ha^{-1} and 0-275 t C ha^{-1}. In contrast, the Mangaotama had 1.9 to 7 t C ha^{-1} on average.

The much lower CWD mass may reflect both the lower tree biomass to begin with, as well as enhanced decomposition. A study on the microclimatic effects of a forest-pasture edge in New Zealand native forest found that when the wind was blowing from the pasture into the forest elevated temperatures could extend 40m into the forest. Soil temperature was also elevated at the edge but typically reached forest conditions within 5m (Davies-Colley et al., 2000). The likely higher temperatures in the Mangaotama fragments than in typical continuous forest may favor faster decomposition and result in a smaller stock of debris. In the grazed fragments, trampling by livestock might also aid in breaking down the debris. Most of the debris were from tree ferns, which are less dense and visibly more fragile than similar sized debris from trees. Given this, it is possible they are more susceptible to being broken up by trampling, leading to faster decomposition and a smaller debris stock.

The litter mass had the opposite trend of what was observed for tree biomass, with a greater mass of litter on flatter slopes. This may simply be the case of litter tending to move down slope and accumulating on the flatter slopes. The results for the litter contain a large amount of uncertainty, being based on visual assessments of percentage cover, and extrapolated back in time for litter mass, which was based on one sample per plot. Thus, the trends for litter must be taken with caution.

The results of the ANOVAs assessing the effects of slopes and aspects must also be taken with caution. The plot sizes may have been too small to adequately asses the vegetation, with typical plots for native forests for carbon monitoring being several times larger (Payton et al., 2004). Individual plots may have been responding to small-scale variations in carbon distribution, leading to very high variation between plots.

No effect of the change in grazing management was clearly evident in the native forest fragments. There was no noticeable difference in forests carbon mass or rate of sequestration between grazed and ungrazed fragments, although the effect of grazing could not be tested statistically due to a lack of replication. This lack of a clear difference comes despite a marked improvement in the sapling (<3 cm DBH) numbers in the fragments removed from grazing, from a density of 35 per ha in 2000 to 4082 per ha by 2004 (Dodd et al., 2008a). The likely reason for this is that, despite having a visible effect on the undergrowth, removal of grazing has not had any effect on the large established trees which make up the vast bulk of the mass. Additionally, the undergrowth is not included in standard assessments of forest carbon stocks, such as the New Zealand carbon monitoring system that was adjusted for use in the current study (Payton et al., 2004). Possible effects may appear in the long term as the new seedlings and saplings mature and increase the mass of the ungrazed forests or, as the grazed forests undergo a slow loss of their mature trees without any regeneration occurring due to the suppression from grazing.

The only difference between grazed and ungrazed forests was the change in soil carbon, but with only two sample points available for the grazed forest there is considerable uncertainty about whether this was an effect of grazing or not. It is possible that soil carbon levels may be elevated above normal levels in grazed fragments from the inputs of manure from livestock, or that some factor is causing a decline for all forest fragments which simply wasn't picked up in the grazed forest. Further study with larger sample sizes is needed to reach firmer conclusions on this question.

Despite the slight possibility that fencing fragments could decrease their ability to sequester carbon due to soil carbon loss, fencing may still be the wisest option in the long run. Continued grazing would be expected to eventually destroy the forest whereas protecting it will enhance its long term viability (Dodd et al., 2011). As the plots which lost biomass show, there is the potential for the fragments that are in decline to be a much greater emissions source than the pasture. The destruction of the forest would involve large emissions as the carbon stock decomposed and the landscape would lose what was a carbon sink.

Adding native plantings around the native forest fragments would be expected to reduce edge effects and improve the habitat for interior forest tree species. Fragmentation exposes edges which have greatly different environments due to the exposure to the adjacent cleared habitat (Collinge, 1996). There is more light, higher temperatures, lower humidity, and greater susceptibility to wind at the edge. Effects can extend tens of meters into the patch. The effects of edges can depend on the edge's position, e.g. aspect. Size and shape greatly influence the extent of edge effects, long thin and small fragments, such as the ones in this study, will be dominated by edge habitat. The plantings should be acting as a buffer reducing edge effects.

Unfortunately, this effect of the plantings could not be formally tested for due to a lack of replication. It is also possible that the reduction of edge effects may not actually improve the carbon sequestration of the forest edges; previous studies have found elevated density and basal area at forest edges, indicating greater biomass and growth rates might occur at the edge (Davies-Colley et al., 2000, Young and Mitchell, 1994). Like the grazing, the buffers simply may not affect the large established trees, thus having no detectable short-term impact. Nonetheless, like the removal of grazing, the restoration plantings may allow the creation of interior habitat and buffer it from disturbance and so improve the long-term viability of the patch.

The minor fluxes of N_2O from ungrazed forests and CH_4 consumption by soils may differ from the values used here, and are likely to vary spatially. The N_2O levels may initially have been higher due to surplus nitrogen left over from when the forest was grazed, but this may have dropped off eventually. A great deal of effort would be needed to accurately account for these fluxes, but the effect of doing so would be small because they contribute very little to the overall GHG balance.

The native forest estimates have several uncertainties. Tree heights, CWD debris mass, and litter mass had to be estimated for previous years from current data. The regression for CWD was a very poor fit, so changes in this stock over time are very uncertain. The regressions for the tree heights did not always fit well, especially in the case of tree ferns where height and diameter are not necessarily well related. The tree height regression could have been improved by taking the heights of all stems in multi-stem trees to avoid the incorrect association of small stems with large heights. However, this would have been hard to achieve in the field, as clearly identifying the top of the tree through the canopy was already difficult. Wood density values had to be taken from another study (Richardson et al., 2009) and were lacking for some common species. The allometric equations were also developed for another site (Coomes et al., 2002). More work is needed developing these equations for native species, along with more comprehensive data sets on wood and shrub densities for differing regions and species. Many of these uncertainties would be faced by any similar studies on native forests. A major strength of the current study was the long-term data set available for the forest growth, even if it did lack some information.

5.2.4 NATIVE RESTORATION PLANTINGS

Richardson and Burrows (2004) previous study of the Mangaotama fragments found aboveground carbon in the vegetation of the plantings to be 0.05 t C ha^{-1} in 2002 and 0.3 t C ha^{-1} in 2003, which is much lower than in the present study. The present study did not have access to the 2003 data, and it is unclear which of the 2002 data sets Richardson and Burrows (2004) used. The use of different allometric equations is the likely cause of the discrepancy between the two estimates.

Carbon sequestration in native restoration plantings has not been well reported on. A study for the timber value of totara trees at a coastal site near Auckland found that at 11yrs mean DBH was 9.7cm, mean wood density was 446 kg m^{-3} and mean height was 5.5m (Bergin et al., 2008). The trees were planted at the same density as in the present study, which is 2m x 2m or 2500 stems ha^{-1}. Using the same allometric equations as for the native plantings in the methods chapter, this works out to be 27 t C ha^{-1}, which is comparable to what was predicted for the mixed shrubs in the present study.

The carbon mass for the restoration plantings was lower than what has been found in studies of shrub-lands, most likely due to the young age of plantings. Shrub-lands in a transect across the South Island were found to have a total carbon stock of 163 t C ha^{-1} (Coomes et al., 2002), which is closer to the Mangaotama's mature forest fragments. Bray (1991) found that after 17yrs bracken regrowth on ex-pasture achieved a total biomass of 125 t ha^{-1}, equivalent to 63 t C ha^{-1}. Total carbon in vegetation for regenerating manuka/kanuka dominated shrub-land in Tongariro National Park was 65 t C ha^{-1} in 25 year old shrub-land and 136 t C ha^{-1} for 35 yr old shrub-land, giving a growth rate of about 4 t C ha^{-1} yr^{-1} in the first 35 years (Scott et al., 2000). This growth rate is similar to that achieved in the restoration plantings in this study. Trotter et al (2005) found mean net carbon accumulation rates for manuka/kanuka shrub-land in the range 1.9 to 2.5 t C ha^{-1} yr^{-1}, when averaged over the active growth phase of about 40 years.

The MAF look-up tables predict a much lower biomass for the native plantings. At 5yrs old the plantings were estimated at 16.74 t C ha^{-1} whereas the lookup tables would put this at only 2.13 t C ha^{-1}, meaning the plantings where essentially 7yrs ahead in the look-up tables (Ministry of Agriculture and Forestry, 2011d). The higher value may be due to the plantings having had a head start over naturally regenerating shrub-lands, or simply the inaccuracy of using nationwide averages for particular sites. Depending on which is the cause, it may be desirable to either develop regional averages or allow provision for plantings by treating them as older when using MAF look-up tables.

Differences between planting sections may have had an effect in the early stages, but seem to have disappeared over time. Comparisons of the effects of slope and aspects and their combinations were hampered by the small sample size. Untested factors may have also been important, such as altitude, which had a strong effect on frost survival in the early stages (Dodd, 2002). Plot sizes may have been too small to adequately asses the vegetation, with typical plots for native forests being several times larger (Payton et al., 2004). However, this may have been less of a problem in the plantings, as they were likely to have been more homogenous at this point than mature forest would be.

Shrub canopy widths needed estimating in some years and shrub density data had to be taken from another study at a different site and for other species (Coomes et al., 2002). Shrubs could have been measured for density, but it was deemed inappropriate to destructively sample them as they had been planted for restoration purposes. Growth in the native plantings after 2006 was an estimate based on previous growth rates. It was estimated within reasonable bounds, but the actual growth would have differed. The small size of the plantings means that this has not had a major impact on the overall result.

Large losses of soil carbon from the top-soil were seen under the native plantings. This may be temporary and stabilize as the trees grow large enough to create sufficient quantities of litter to maintain the topsoil. Given that the adjacent forest fragments have much lower soil carbon levels, it could simply be declining back to what it would have been under native forest.

5.2.5 SOIL CARBON

The estimates of the soil carbon change are likely to be the second biggest point of uncertainty in this study. Only a small number of samples could be obtained for all land-uses both before and after the land-use changes. The adjacent long-term fertilizer trial (Schipper et al., 2009), from which results were extrapolated for pasture soils, did not include the same range of soils and slopes as contained in the whole catchment. The ten year gap between samples also meant there was no way of identifying when the biggest losses occurred. It was assumed that the losses were constant through time. This meant that as soon as an area was converted to trees it was counted as having this constant rate of loss. The result of this was that the pines and plantings were a source of emissions during the early years when tree growth was low. However, if soil carbon loss was also initially low then these net emissions may not have actually occurred.

The soil carbon losses measured were in the order of 2-3% yr^{-1}. This was a much larger value than those found in Schipper et al. (2009), who measured soil carbon changes in a nearby pasture and found average gains of 0.270% yr^{-1} on easy slopes and 0.156% yr^{-1} on steep slopes, and later an average loss of 0.066% yr^{-1} on steep slopes. The limits imposed by small sizes may partially explain these unusually high results.

The present studies results from the pasture soils were not used in calculating the overall balance due to the uncertainty created by the very small sample size. Instead, the values from Schipper et al (2009) were used. The few pasture soil samples from the present study suggest that applying the values for soil loss in pasture from Schipper et al (2009) was appropriate because the direction of change was the same. However, the Schipper et al. (2009) values may have been an underestimate compared with the values used for the other land-uses, because Schipper et al. (2009) did not include the full range of depths as included in the present study. The present study suggests that these lower depths are also losing carbon, but provides too few samples to quantify this well. Nevertheless, even using the lower value of soil carbon loss in pasture, emissions of CO_2 from the soil were at a comparable level to those from N_2O in pasture. This highlights the importance of including soil carbon changes alongside the more commonly counted emissions of ruminant CH_4 and N_2O.

The trend for the loss of carbon in the ungrazed native forests was only significant at a 0.1 level and there were only two samples in the grazed forest, leaving considerable uncertainty in these findings. Given the large impact this loss could have on the native forest's net balance, it is important to quantify the soil carbon more reliably. If these findings are accurate, it could weaken the case for protecting native forest fragments from grazing solely for the sake of short-term carbon sequestration, as the fragments would sequester more if there were no loss of soil carbon. However, it is not at all clear what is causing the possible decline.

Estimates for the original soil stocks under the pine forests were based on extrapolation from other areas in pasture. Using this value suggested a decline in soil carbon under the pines. The one point that did have a prior sample suggested an increase, but this site was not likely to be typical, being on the forest edge on a very gentle slope. Given this problem, the pine forests have the greatest uncertainty in their estimate for the loss of soil carbon. As they make up almost half the catchment, this is a considerable area of uncertainty. Nevertheless, the soil carbon loss assumed in the calculations was more than overwhelmed by the tree growth, although it was an important source of emissions soon after establishment.

The issue of soil changes with pine conversion has been widely investigated elsewhere. For example, Davis and Condron (2002) reviewed New Zealand paired site studies of afforestation in grassland, and concluded that afforestation will lead to a short-term loss of around 4.5 t C ha^{-1} from the top-soil (0-10cm) in the first 10-20 yrs. For the whole catchment calculations the pine forest soils were assumed to lose 0.91 t C ha^{-1} yr^{-1} on easy slopes and 2.11 t C ha^{-1} yr^{-1} on steep slopes, which is larger than the average loss in Davis and Condron (2002) but includes an additional 10cm depth. The present study's top-soil samples for pasture and pine differed by 7.8 t C ha^{-1}. This may be partially attributed to initial differences between the Mangaotama's pasture and pine site with the pines generally on the poorer quality land.

5.2.6 EROSION

The estimate of erosion was very inexact. Spatial and temporal variation was poorly captured, and deposition within the catchment couldn't be assessed. A better view awaits a more comprehensive sediment budget for the catchment. It is also uncertain how much of a sink erosion contributed for the Whatawhata farm, because the portion of carbon that was buried in the deep oceans is not known and had to be assumed. Nevertheless, from these estimates, erosion contributed a very small amount to the overall balance. Therefore, the shortcomings here are of minor importance.

5.2.7 UNACCOUNTED FOR COMPONENTS

Several things were excluded completely from the analysis due to a lack of information or resources. These gaps introduce an additional area of uncertainty, but generally concern minor points. Some land-uses were not included, but were typically minor or sufficiently similar to ones that were.

The management activities were likely to influence the GHG balance and carbon stocks of the streams. In particular, the inclusion of wood and stream-bed sediment would have slightly increased the catchment's total carbon stocks. Stream wood carbon stocks for the area have been estimated as 2 t C ha^{-1} in pasture, 14.5 t C ha^{-1} in native forest, and 50 t C ha^{-1} in pine forest (Quinn and Broekhuizen, 2003), but the amount in sediment was not known. Due to the small size of the streams including this would have a very small effect. Emissions of N_2O and CH_4 from streams and wetlands were also likely. Small lowland streams in agriculturally developed New Zealand catchments are known to be a net source of CH_4 and N_2O, but are a small contributor to the whole catchment when livestock emissions are considered (Wilcock and Sorrell, 2007). Even so, riparian management could offer another minor avenue for mitigation.

A comparison between the native plantings and natural regeneration would have been useful. There have been substantial areas of regeneration on other parts of the Whatawhata farm and future work could usefully compare the efficacy of natural regeneration versus the costly native plantings. However, at this stage, areas which have been allowed to regenerate appeared to have much less vegetation than the plantings, and only covered a small part of the catchment. An assessment of the poplar plantings would also have been helpful as these are already a common way of introducing trees into pasture landscapes, and were a part of the management plan. However, only the denser areas of large mature trees were likely to be of much overall significance, and these were approximated as grazed native forest.

Data was not obtained on the embodied and combustion emissions associated with the pines, but these are expected to be minimal in comparison with the pine's growth. Pasture and livestock biomass were not included, making the carbon stock estimates marginally lower, but this would only be by a small amount compared with the soil stock.

5.2.8 LIMITS TO APPLICATION ELSEWHERE

Overall this research takes a case study approach. Only some individual land-uses within the catchment had replicate measures for comparisons of landscape position, and these were hampered by small sample sizes when split into different slopes and aspects. The lack of replication means the study cannot conclusively generalize to all hill country in New Zealand. Other sites will have many differences in soil, climate, farm systems, etc., and may have differing outcomes should a similar management plan be implemented. Despite these drawbacks, the Whatawhata site can be considered representative of North Island hill country and broadly demonstrates what can be achieved through combined reforestation, native restoration and intensification. Under conditions similar to those at Whatawhata it is reasonable to expect broadly similar outcomes.

It should be noted, that this strategy to achieve catchment-scale sustainability for climate change is unlikely to be applicable to areas with minimal marginal land, or where the upper limit for further intensification has already been reached. The area available for forestry could be too small to offset emissions from pasture, or there would be no ability to further intensify the remaining pasture to offset the loss of land.

5.3 SCENARIOS

A range of different land-uses could have been implemented, some of which may be able to achieve better results for GHG mitigation than what was actually done. This section discusses a number of scenarios based on the data from the present study, which provide a comparison to the actual land-use changes.

5.3.1 SCENARIOS TESTED

Seven scenarios were run. All but one scenario used the original land use in 2000 as a starting point, which is equivalent to an 'all pasture' scenario.

All Pines: The entire area (apart from pre-existing forests) was planted in pines. The same silvicultural regime was followed as described in this study. It was assumed the catchment was run only as a forestry operation, so no grazing was done anywhere.

All Native Forest: The whole catchment area was assumed to be covered by native forest of the same condition as the current forest fragments. No grazing was done. No baseline pasture situation was used, i.e., it was treated as if the original forest was never cleared, just degraded.

All Native Plantings: The entire catchment was planted in a mix of native shrubs and trees as with the native plantings in this study. No grazing was done anywhere. The values for the native plantings from this study were used.

Native Riparian Buffers - 15m wide on all streams: All streams (including 1st order streams) were planted with 15m wide native riparian buffers. The values for the native plantings were used in the buffers. Grazing was assumed to be at the same stocking rates as the old regime, with none in the buffers.

Native Riparian Buffers – 15m wide on the main stream: The main 4^{th} order stream was planted with 15m wide native riparian buffers. The values for the native plantings were used in the buffers. Grazing was assumed to be at the same stocking rates as the old regime, with none in the buffers.

Headwater Pine Forestry Conversion: Only the headwater sub-catchment section of the current pine forest was assumed to be planted in pines. This was 92 ha or 34% of the catchment, instead of the 57% which was actually put in pine. The same silvicultural regime was followed. Grazing was assumed to be at the same stocking rates as the old regime.

Pastoral Intensification Only: The pasture's stocking rates and fertilizer use were increased to the maximum level they reached under the new regime; this scenario was applied to the whole catchment with no tree planting or fencing of the fragments done.

5.3.2 SCENARIOS RESULTS & IMPLICATIONS

Planting the entire catchment in pines would approximately double the sequestration achieved by the current management changes (Fig. 27). This land-use plan would have the biggest impact on reducing net emissions of all those considered, but it would result in the elimination of all agricultural production from the land.

Planting a reduced area of pine forest in the steepest headwater sub-catchment would still be sufficient to make the catchment a net sink with a maximum sequestration rate of 5.6 Gg CO_{2e} yr^{-1} (Fig. 27), which is 62% of what was achieved under the existing land-use changes. This scenario would retain even more of the agricultural land than the existing changes, and would incur fewer costs due to the smaller area in pine and the lack of native plantings.

Figure 27. Scenarios for changes in the net greenhouse gas balance for the whole catchment with planting the whole area in pines (pines only) or with planting only the steepest headwater sub-catchment in pines (Headwater Pines).

If the catchment was covered by a degraded native forest of the same standard as the current fragments, then it would sequester 740 t CO_{2e} yr^{-1} (Fig. 28), which is substantially less than in any of the pine planting scenarios. Planting the catchment with a mix of native shrubs and trees would do better, with a maximum of 2635 t CO_{2e} sequestered each year (Fig. 28), but would still be much less than the all pine scenario, and comparable to planting the headwaters with pines. Thus, natives alone have no advantages in the first ten years compared with pines, although they would likely have more conservation value.

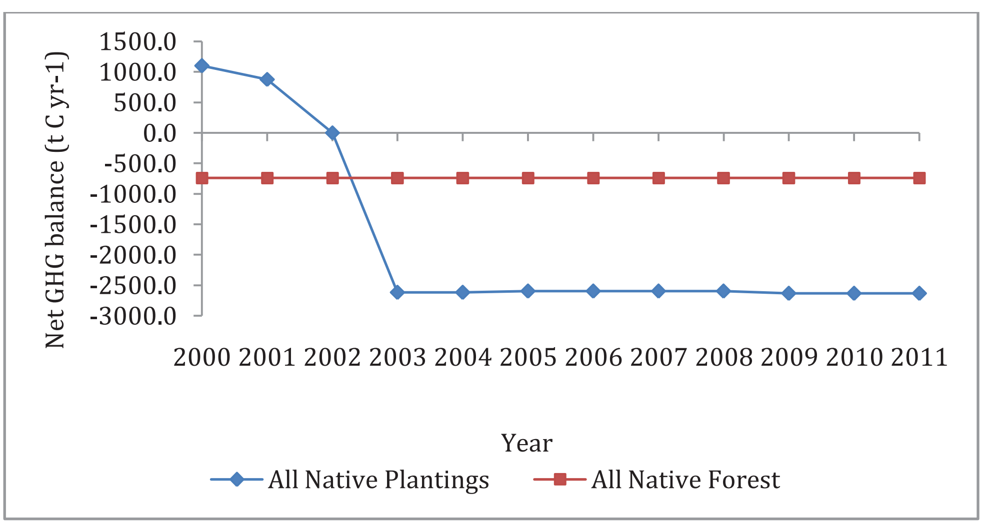

Figure 28. Scenarios for changes in the net greenhouse gas balance for the whole catchment if planting the whole area in natives (all native plantings) and if the whole area was degraded native forest (all native forest).

Planting native riparian buffers only along the main 4th order stream would be too small in total area to have a large impact on the net emissions of the catchment, and would reduce them by only 7.6 % of the baseline (Fig. 29). Planting all streams with native riparian buffers would be a large enough area to reduce the net catchment emissions by 67% (Fig. 29), however, the area and cost involved would likely make it impractical. Thus, riparian planting for GHG mitigation alone is not sufficient to make the farm carbon neutral, although it does provide some synergies between mitigation and water quality improvement.

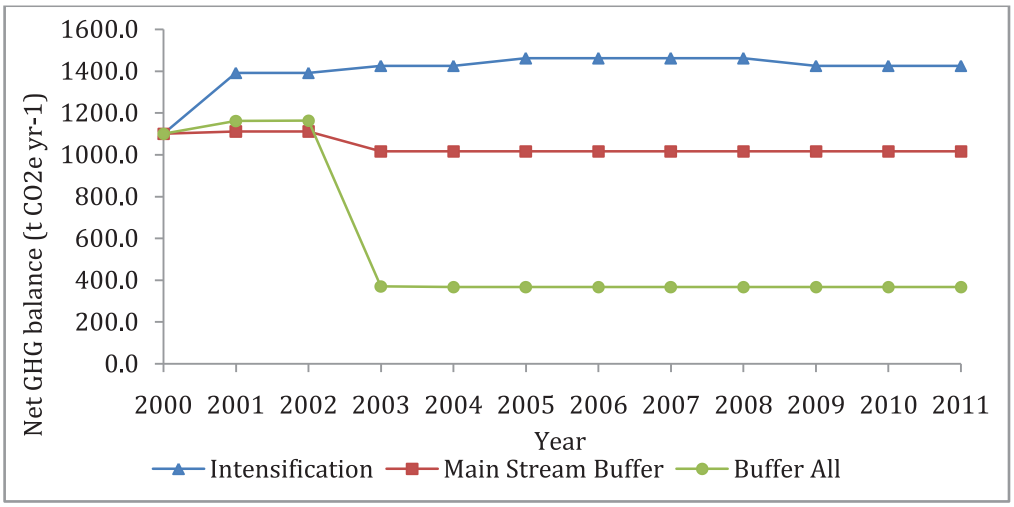

Figure 29. Scenarios for changes in the net greenhouse gas balance for the whole catchment with pasture intensification only, planting 15m riparian buffers on the main stream only (7.27ha) and planting 15m buffers on all streams (55.29 ha).

The intensification only scenario highlights the danger of simply focusing on 'carbon efficiency', that is the amount of emissions per unit of product. The new more intensive livestock system increased its productivity per ha more than its emissions. Using the values from the methods for steep class V land, net pastoral emissions per unit of agricultural product (beef, lamb and wool) went from 18.37 kg CO_{2e}/kg pre-2001, to 10.29 kg CO_{2e}/kg after 2001. Therefore, the new system increased its carbon efficiency. Likewise, this livestock system, applied to the entire catchment area, would also be more carbon efficient, producing more output per ha. It would also increase the catchment's net emissions by almost a third, to a net balance of 1461 t $C0_{2e}$ yr^{-1} (Fig. 29). Therefore, despite being more efficient, it would have a bigger negative impact on climate change. Thus, both the increase in carbon efficiency and the reduction in stock numbers must be combined for this scenario to be truly sustainable intensification.

5.4 VALUATION

With the introduction of the Emissions Trading Scheme (ETS), GHGs have received a monetary value, which could impact the financial viability of New Zealand farms, when agriculture is bought into the ETS. This section applies different methods of valuing GHGs for the catchment, and discusses the implications of carbon trading for the viability of the Whatawhata land-use plan.

5.4.1 VALUATION METHODS

The value of GHGs was calculated using four different methods at three prices (\$15, \$25, and \$60 per ton CO_{2e}). The prices were chosen to reflect a reasonable range of prices, based on the approximate range of prices experienced in the European Union's emissions trading scheme from mid 2008 to mid 2010, the likely price forestry NZUs have been trading at, and the current set price of \$25 per unit (Kossoy and Ambrosi, 2010).

In the Full Balance calculation, a dollar value was put on the carbon flows from this study to give the full value of all GHGs. In addition, three variants of the simplified methods that could be used in the ETS were done. These differed in how the pines were accounted for. In the first, a comprehensive field survey, the results for the pines growth from the current study were used. The aim was to simulate a detailed assessment of the forest.

The second variant, a simplified field survey, used the average slopes and main aspect (east and west, treated as one class) to model the pines. Forests greater than 100 ha will require the use of a field measurement approach based on permanent plots and MAF's carbon calculator, which is still in the process of being introduced (Ministry of Agriculture and Forestry, 2011a). This may include a survey of all slope and aspect classes, or, for time and cost reasons, a simpler approach could be used which might be based on the average slope and main aspect. To simulate a simplified field survey, CenW had to be run again using inputs from the average slope (28.52^{o}) and main aspect class (west and east, treated as one class). The average relative solar radiation for each month was calculated (Table 37), climate adjustments and modelling were done in CenW as before, and the output for each year was applied to the whole pine forest area and summed in GIS.

The third approach used the ETS look-up tables. If the pine forest were below 100 ha, look-up tables would have to be used. The total forest area in the catchment is not far above the 100 ha cut-off point, so this valuation method was done for comparison. Post-1989 Waikato/Taupo *Pinus radiata* values from the look up tables were used (Ministry of Agriculture and Forestry, 2011d).

Apart from the above variants, the general ETS methodology differed from the Full Balance calculation in the following aspects:

- The native forest fragments were excluded, because they are pre-1990 indigenous forest (Ministry of Agriculture and Forestry, 2011a).
- The sequestration in the native plantings was calculated using look up tables, because they are less than 100ha. Post-1989 indigenous forest values from the look up tables were used (Ministry of Agriculture and Forestry, 2011c).
- Soil carbon changes were not included.
- Embodied emissions costs would be borne by the industry producing them, so were not included.
- For pasture, only ruminant CH_4 and N_2O emissions were included.

Processors of agricultural products and fertilizer suppliers will be responsible for the agricultural emissions under the ETS, rather than individual farms (Ministry of Agriculture and Forestry, 2011b). Thus, the methods used in the ETS for calculating emissions from pasture differ substantially from the present study. For simplicity, it was assumed that the ETS method would give the same emissions estimate as the present study's results for ruminant CH_4 and N_2O emissions. It was assumed that the costs for all emissions are passed on directly to the farmer and that there is no phase-in period, so the full cost is borne immediately.

Table 37. Relative solar radiation values used in calculating the Climate Adjustment for the Average Slope method

month	Relative Solar Radiation
1	0.93
2	0.94
3	0.96
4	0.98
5	1.00
6	1.00
7	0.99
8	0.98
9	0.96
10	0.94
11	0.93
12	0.93

5.4.2 VALUATION RESULTS & IMPLICATIONS

The farm surplus under the old regime in 1998-99 was $25,800 (Dodd et al., 2008b).The yearly cost of emissions from the original land-use would be large enough to absorb the farm surplus at $25 per t CO_{2e} (Table 38), and this could be completely eroded if prices rose higher. The Full Balance and ETS valuation methods produced similar results for the cost of emissions under the old regime; with the ETS method giving a 9.4% lower valuation of the net costs. The ETS would significantly undermine the profitability of conventional hill country farming at Whatawhata. These results are similar to estimates for 17 South Island sheep and beef farms where at $25 per t CO_{2e} the median full liability of the ETS for a farm was $30,336.25 (Rosin et al., 2011). In interviews with the farmers it was agreed that this full cost would force them to sell the farm (Rosin et al., 2011).

The actual impact of the ETS will not reach this level for some time, because of the slow phase in period. Participants are eligible for free units beginning at 90% of an average baseline of emissions, which will be phased out at 1.3% per yr from 2016 (Ministry of Agriculture and Forestry, 2011b). The economic impact will also be affected by price fluctuations in carbon, meat, and wool.

Table 38. Yearly cost of emissions under the baseline land-use in the Mangaotama catchment at three carbon prices. ETS uses the emissions trading scheme methods assuming all costs passed on to the farmer. Full balance uses all flows from the present study.

Method	Carbon price		
	$15	$25	$60
ETS	14,948	24,913	59,791
Full Balance	16,504	27,507	66,017

Forestry conversion of marginal land under the ETS has the potential to turn this net cost into a net gain. The class VII land was of marginal value under the old regime and losing money, with an economic farm surplus of -$3 ha^{-1} (Dodd et al., 2008b). Under the new regime, the conversion of this marginal land to pine forestry helped to turn the farm into a net carbon sink, which could potentially generate over $800,000 in the first ten years at $25 per t CO_{2e} (Table 39).

However, the ETS valuation was very dependent on the methods used to assess the pine forest (Table 39). For example, ETS surveyed Pines method would over-value the catchment by 21% compared to the Full Balance valuation (Table 39).

If the look-up tables were used, the ETS valuation would be only 43% of the Full Balance valuation, due to the low estimates for the pine forest growth in the look-up tables. This discrepancy is so large, that at low to medium carbon prices the land-use change cannot be funded by the ETS alone, and becomes a net cost (although future income from timber may counter this).

The differences between the present studies result and those derived from look-up tables is of course unsurprising given that the tables use regional averages. However the magnitude of the difference may have significant financial implications for landowners considering the cost and benefits of being forced to use look up tables. This may be a only a small problem for forest blocks well below the 100 ha threshold, but if pine growth is expected to exceed that given in the look-up tables by a magnitude that outweighs the added cost of field measurement then there is an incentive to plant over the 100 ha threshold and ensure that more accurate measurements are taken. The ETS look-up tables also underestimated the growth of the native plantings. Considering the high cost of planting them, and that they are treated the same as natural regeneration, there are no incentives to plant natives for the sake of carbon credits.

Table 39. The net cost of greenhouse gases from 2000 to 2011 (value of carbon credits minus the implementation costs of $597,840) at three carbon prices. Full balance uses all flows from the present study, ETS uses the emissions trading scheme methods assuming all costs passed on to the farmer. Surveyed Pines uses the Pine forest values from the present study, Look up table pines uses lookup table values for the pine forest. Positive values are a cost to the land-owner, negative values are a gain.

Method	Carbon Price per t CO_{2e}		
	$15	**$25**	**$60**
Full Balance	-93,484	-554,366	-2,167,454
ETS surveyed Pines	-277,899	-861,724	-2,905,115
ETS lookup table Pines	301,770	104,390	-586,440

When the pine forest was assessed using values only from the average slope and the main aspect class, the estimate of the pine sequestration in biomass for the first 5yrs was very close or slightly less than if the more comprehensive method was used (Table 40). After 5yrs the simpler method overestimated sequestration by 8-14%. Thus, this simplification in method does not sacrifice anywhere near as much accuracy as the use of look-up tables. However, this method would have lead to a higher overestimate if there had been a greater portion of steep southern areas in the forest. These are the most different from the average situation and can have substantially lower growth rates, whereas most other slope/aspect classes are very similar.

Table 40. The effect of using the average slope and main aspect in calculating sequestration by the pine forest versus using all slope/aspect classes. % difference gives the Average slope method as a percent of the All classes method. Shows sequestration for the pine forest biomass for the actual area planted.

Year	**Average Slope Method, t CO_{2e}**	**All Classes Method, t CO_{2e}**	**% difference**
2001	-13.31	-13.31	100
2002	1490.25	1550.33	96
2003	-2218.16	-2201.86	101
2004	-3448.87	-3663.55	94
2005	-5954.58	-5960.44	100
2006	-10189.05	-9396.93	108
2007	-10797.23	-9772.03	110
2008	-11082.36	-9953.14	111
2009	-9736.52	-8875.36	110
2010	-10994.88	-10039.93	110
2011	-6717.97	-5872.64	114

The implementation cost was identified as a major barrier that could prevent land-owners from adopting the more sustainable practices demonstrated at the Whatawhata farm (Dodd et al., 2008a). The cost of the land use change in the first 10yrs was almost $600,000, with an estimated annual debt servicing of $53,000, while the average farm surplus was only $36,480 during the first 3yrs, showing the inability for the farm itself to fund the changes (Dodd et al., 2008a). The conservation measures of the new land-use plan would be unlikely to fund themselves from emissions trading, so would be reliant on income from the forestry venture. However, the eventual harvest of the pine forest poses a potential problem. A large portion of the income from carbon trading must be retained to offset the large cost involved with surrendering carbon units at harvest.

These results are only a preliminary look at the effect of the ETS on the financial viability of this land use change. A more comprehensive analysis including livestock and timber costs and income, and with the effects of fluctuating prices is needed to give a better understanding. The lowest price given here may actually be too high. In the first months on 2012, prices have fallen as low as $7-9 per NZU (Commtrade, 2012). The valuation done in this study is not a full economic analysis, but it does suggest that carbon credits from forestry could provide the needed catalyst for crossing financial barrier of transitioning to a more sustainable land-use in hill country.

Other studies have suggested that carbon trading will increase the profitability of forestry. Maraseni (2011) assessed the influence of a carbon price on the relative profitability of maize-peanut cropping, pasture, and plantation gum trees for inland Queensland. To do so, they assessed the emissions, soil carbon trends, and tree biomass for the three land-uses and calculated their net present values. Cultivation was normally the most profitable, but with the carbon price included, plantation forests became the most profitable. Plantations were only best at higher carbon prices and discount rates below 6%. In New Zealand, an investigation on the likely impact of the ETS on forestry (Manley and Maclaren, 2010) concluded that it would increase forestry's profitability and make it viable in many areas where it currently is not. The reasons why it enhanced the profitability of forestry were because of the time value of money (revenue is constant whereas the costs are distant), and because not all carbon credits earned need to be surrendered at harvest (Manley and Maclaren, 2010). Given that forestry alone can improve the viability of hill country farms, the use of carbon trading further strengthens the case for its use.

5.5 CONCLUSIONS

Conversion of marginal land to pine forestry, combined with intensification of the remaining pasture areas along with conservation plantings, turned the Mangaotama catchment from a net source of GHG in to a much larger net sink. The size of this transformation was so large that any uncertainties in the estimates would not be able to substantially alter this broad finding. However, this study does show the difficulties involved in obtaining adequate data for a comprehensive GHG balance, even for a site with a long history as a research farm.

Most of the shift in the GHG balance was due to the increased carbon sequestration by the pine forest, followed distantly by the reduction in stock numbers and then by the sequestration in the native plantings. The native forest fragments offset a small but significant portion of the farm's emissions, however, fencing and planting around them had no detectable effect on their growth. There is a possibility that the now ungrazed areas of native forest were losing soil carbon, meaning the removal of grazing may have had a negative effect on their carbon balance, but this conclusion was uncertain due to limited sample numbers. The native plantings also appeared to be losing soil carbon, but this was offset by tree growth. It was likely that the pines were also losing soil carbon, but this conclusion was hampered by a lack of previous soil samples.

Topography may be having an important effect in the pines, but no detectable effect in the native plantings. Pines on southern slopes sequestered much less carbon; however, this may be an underestimate of their growth. In the native forest fragments flatter slopes had lower carbon stocks, suggesting that grazing has suppressed growth in accessible areas and that the carbon stock's recovery will take longer than 10 years.

Given the size of the net sink produced, a smaller area of pines could have been planted. Small synergies between riparian planting for water quality and GHG mitigation could occur, but to achieve large gains would require prohibitively large areas to be planted.

ETS methods undervalue emissions, and the look-up tables massively undervalue sequestration in the forests for this site. The use of even simplified site-specific methods would give a much more accurate estimate. Carbon trading has the potential to impose a significant burden on conventional farms of this type; it could even eliminate the farm surplus if all costs were passed on to the farmer. On the other hand, the ETS has the potential to help land owners to cross the financial hurdle to implementing what has already been demonstrated as a more sustainable system.

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APPENDIX A

The following tables give the final values for the emissions, sequestration, and carbon stocks for each land-use as derived from the methods. These values were used in calculating the final results for the whole catchment.

Table A-1. Data Values Derived for Pasture Emissions from the Pre-2001 baseline to 2011.In kg $CO_{2e}ha^{-1}\ yr^{-1}$. Na = not applicable. Roman numerals denote LUCC, Easy and Steep refer to slope classes.

Category	Section	baseline	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Ruminant Methane & Nitrous Oxide	III	6207.00	6900.00	6900.00	6900.00	6900.00	6900.00	7169.00	7169.00	7169.00	7169.00	7169.00	7169.00
	IV	6207.00	6900.00	6900.00	6900.00	6900.00	6900.00	7169.00	7169.00	7169.00	7169.00	7169.00	7169.00
	V	4183.00	4841.00	4841.00	4841.00	4841.00	4841.00	4938.00	4938.00	4938.00	4938.00	4938.00	4938.00
	VI	3361.00	3742.00	3742.00	3742.00	3742.00	3742.00	4011.00	4011.00	4011.00	4011.00	4011.00	4011.00
	VII	2602.00	2784.00	2784.00	2784.00	2784.00	2784.00	3053.00	3053.00	3053.00	3053.00	3053.00	3053.00
	Ungrazed	na	370.07	370.07	370.07	370.07	370.07	370.07	370.07	370.07	370.07	370.07	370.07
Soil Carbon Change	Easy (10^o-20^o)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Steep (30^o-40^o)	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50	1651.50
Embodied/ Combustion	All	48.83	57.74	57.74	57.74	57.74	57.74	220.83	220.83	220.83	220.83	220.83	220.83
	Class V	na	166.65	166.65	166.65	166.65	166.65	na	na	na	na	na	na
	ungrazed	na	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Erosion decay	All pasture	146.80	146.80	146.80	146.80	146.80	146.80	146.80	146.80	146.80	146.80	146.80	146.80
Product decay	III	403.62	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40
	IV	403.62	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40
	V	272.06	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45
	VI	218.54	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11
	VII	162.79	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39
	ungrazed	na	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table A-2. Data values derived for Pasture Sequestration from the Pre-2001 baseline to 2011. In kg $CO_{2e}ha^{-1}$ yr^{-1}

Category	Landscape Section	baseline	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Soil Methane	All pasture	16.00	16.00	16.00	16.00	16.00	16.00	16.00	16.00	16.00	16.00	16.00	16.00
	Ungrazed	21.25	21.25	21.25	21.25	21.25	21.25	21.25	21.25	21.25	21.25	21.25	21.25
Erosion buried	All pasture	587.20	587.20	587.20	587.20	587.20	587.20	587.20	587.20	587.20	587.20	587.20	587.20

Table A-3. Data Values Derived for Pasture Carbon Stocks from the Pre-2001 baseline to 2011. In t Cha^{-1}

Category	Landscape Section	baseline	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Soil Carbon Stock	Easy	101.56	101.56	101.56	101.56	101.56	101.56	101.56	101.56	101.56	101.56	101.56	101.56
	Steep	91.99	91.54	91.09	90.64	90.19	89.74	89.29	88.84	88.39	87.94	87.49	87.04

Table A-4. Data values derived for native forest emissions from the pre-2001 baseline to 2011.In kg CO_{2e} ha^{-1} yr^{-1}. NA = not applicable. Roman Numerals indicate the LUCC.

Category	Section	baseline	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Ruminant Methane & Nitrous Oxide	III	6207.00	6900.00	6900.00	6900.00	6900.00	6900.00	7169.00	7169.00	7169.00	7169.00	7169.00	7169.00
	IV	6207.00	6900.00	6900.00	6900.00	6900.00	6900.00	7169.00	7169.00	7169.00	7169.00	7169.00	7169.00
	V	4183.00	4841.00	4841.00	4841.00	4841.00	4841.00	4938.00	4938.00	4938.00	4938.00	4938.00	4938.00
	VI	3361.00	3742.00	3742.00	3742.00	3742.00	3742.00	4011.00	4011.00	4011.00	4011.00	4011.00	4011.00
	VII	2602.00	2784.00	2784.00	2784.00	2784.00	2784.00	3053.00	3053.00	3053.00	3053.00	3053.00	3053.00
Nitrous Oxide	Ungrazed	na	208.60	208.60	208.60	208.60	208.60	208.60	208.60	208.60	208.60	208.60	208.60
Soil Carbon Change	Grazed	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Ungrazed	na	5215.59	5215.59	5215.59	5215.59	5215.59	5215.59	5215.59	5215.59	5215.59	5215.59	5215.59
Product decay	III	403.62	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40
	IV	403.62	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40	794.40
	V	272.06	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45	535.45
	VI	218.54	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11	430.11
	VII	162.79	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39	320.39
	All ungrazed	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Embodied/ Combustion	All grazed	15.04	15.04	15.04	15.04	15.04	15.04	15.04	15.04	15.04	15.04	15.04	15.04
	All ungrazed	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Erosion decay	All Fragments	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81

Table A-5. Data values derived for native forest sequestration from the pre-2001 baseline to 2011. In kg CO_{2e} ha^{-1} yr^{-1}. Forest Growth in t CO_2 ha^{-1} yr^{-1}.

Category	**Section**	**baseline**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
Erosion Buried	All	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65
Soil Methane	All	118.50	118.50	118.50	118.50	118.50	118.50	118.50	118.50	118.50	118.50	118.50	118.50
Forest Growth (All Live+CWD +litter)	All	10.23	10.23	10.23	8.02	8.02	5.76	5.76	5.76	5.76	8.00	8.00	8.00

Table A-6. Data values derived for Native Forest carbon stocks from the pre-2001 baseline to 2011. In t C ha^{-1}.

category	**Section**	**baseline**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
Forest Carbon (All Live+CWD+litter)	All	117.92	120.71	123.50	125.68	127.87	129.43	131.00	132.57	134.14	136.32	138.50	140.68
Soil Carbon	Grazed	71.19	71.19	71.19	71.19	71.19	71.19	71.19	71.19	71.19	71.19	71.19	71.19
	Ungrazed	88.21	86.79	85.37	83.95	82.52	81.10	79.68	78.26	76.84	75.42	74.00	72.58

Table A-7. Data values derived for native restoration plantings emissions from Planting in 2001 to 2011.In Kg CO_{2E} ha^{-1} yr^{-1}.

Category	Section	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Embodied/ Combustion	All	117	141	61	0	0	0	0	0	0	0	0
Erosion Decay	All	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81
Soil Carbon Change	All	6005.85	6005.85	6005.85	6005.85	6005.85	6005.85	6005.85	6005.85	6005.85	6005.85	6005.85

Table A-8.Data values derived for native restoration plantings sequestration from the planting in 2001 to 2011. In kg CO_{2E} ha^{-1} yr^{-1}. Forest growth is in t CO_2 ha^{-1} yr^{-1}

Category	Section	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Erosion Buried	All	199.648	199.648	199.648	199.648	199.648	199.648	199.648	199.648	199.648	199.648	199.648
Soil Methane	All	16.5	16.5	16.5	16.5	16.5	16.5	16.5	16.5	16.5	16.5	16.5
Forest Growth	All	2.60	2.60	14.87	14.87	14.87	14.87	14.87	14.87	14.87	14.87	14.87

Table A-9. Data values derived for native restoration planting carbon stocks from the planting in 2001 to 2011. In t C ha^{-1}.

Category	Section	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Forest Carbon	All	0.28	0.76	4.81	8.86	12.91	16.74	20.79	24.84	28.89	32.94	36.99
Soil Carbon	All	121.92	120.29	118.65	117.01	115.38	113.74	112.10	110.47	108.83	107.19	105.56

Table A-10. Data values derived for pine forest emissions from planting in 2001 to 2011. In Kg CO_{2E} $ha^{-1}yr^{-1}$. Soil carbon changes in t CO_2 ha^{-1} yr^{-1}.

Category	Section	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Nitrous Oxide	All Pines	178.8	178.8	178.8	178.8	178.8	178.8	178.8	178.8	178.8	178.8	178.8
Erosion Decay	All Pines	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81	48.81
Soil Carbon Change	Easy (0-20^o)	3.32	3.32	3.32	3.32	3.32	3.32	3.32	3.32	3.32	3.32	3.32
	Steep (20-40^o)	7.74	7.74	7.74	7.74	7.74	7.74	7.74	7.74	7.74	7.74	7.74

Table A-11. Data values derived for pine forest Sequestration from planting in 2001 to 2011. Erosion and methane in kg $CO_{2E}ha^{-1}yr^{-1}$.Forest growth in t CO_2 ha-^{1} yr^{-1}.

Category	Section	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Erosion Buried	All Pines	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65	199.65
Soil Methane	All Pines	227.5	227.5	227.5	227.5	227.5	227.5	227.5	227.5	227.5	227.5	227.5
Forest Growth (All Live+dead gained)	Easy East	0.00	-9.54	15.41	24.22	42.21	63.49	66.79	66.06	57.25	68.26	38.90
	Steep East	0.00	-9.54	14.68	24.22	40.74	66.06	68.63	69.00	60.56	69.00	40.74
	Very Steep East	0.00	-9.18	11.38	17.98	30.46	57.25	63.12	66.79	58.35	64.96	43.31
	Easy North	0.00	-11.01	16.52	26.79	45.51	64.23	64.96	62.76	57.99	67.53	36.33
	Steep North	0.00	-11.38	17.62	31.56	48.08	66.06	63.49	62.02	59.82	65.69	33.40
	Very Steep North	0.00	-11.74	16.88	30.83	45.14	66.79	64.23	63.12	64.59	65.33	33.40
	Easy South	0.00	-9.18	11.74	16.52	30.46	52.85	59.45	64.23	49.18	64.59	39.64
	Steep South	0.00	-8.81	7.34	11.01	17.25	34.87	42.94	50.65	41.84	48.08	31.20
	Very Steep South	0.00	-8.81	1.10	7.34	14.31	23.12	27.53	30.09	26.79	28.26	18.72
	Easy West	0.00	-9.91	15.41	24.59	42.57	63.86	67.16	66.06	57.25	68.26	38.90
	Steep West	0.00	-9.54	14.68	23.86	40.37	66.06	68.26	69.00	56.52	68.63	41.47
	Very Steep West	0.00	-9.18	11.38	17.62	30.09	57.25	63.12	67.16	58.35	64.96	43.67

Table A-12.Data values derived for pine forest carbon stocks from the planting in 2001 to 2011. In t C ha^{-1}.

Category	Section	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Forest Carbon	Easy East	10.21	7.56	11.80	18.44	29.97	47.29	65.52	83.49	99.07	117.67	128.27
(All Live + dead matter)	Steep East	10.21	7.64	11.69	18.26	29.37	47.41	66.13	84.92	101.40	120.17	131.30
	Very Steep East	10.21	7.68	10.82	15.69	23.97	39.60	56.80	75.04	90.96	108.68	120.45
	Easy North	10.21	7.22	11.74	19.09	31.46	48.91	66.58	83.71	99.52	117.88	127.79
	Steep North	10.21	7.08	11.91	20.49	33.59	51.62	68.95	85.85	102.19	120.09	129.17
	Very Steep North	10.21	7.02	11.59	19.97	32.25	50.40	67.91	85.16	102.71	120.47	129.53
	Easy South	10.21	7.71	10.95	15.42	23.75	38.10	54.28	71.76	85.15	102.73	113.52
	Steep South	10.21	7.80	9.84	12.86	17.60	27.10	38.82	52.59	63.96	77.07	85.55
	Very Steep South	10.21	7.86	8.18	10.17	14.08	20.41	27.93	36.17	43.45	51.10	56.20
	Easy West	10.21	7.56	11.80	18.47	30.04	47.41	65.68	83.68	99.29	117.91	128.52
	Steep West	10.21	7.66	11.67	18.13	29.17	47.21	65.85	84.65	100.01	118.75	130.08
	Very Steep West	10.21	7.67	10.76	15.54	23.71	39.33	56.53	74.79	90.72	108.40	120.29
Soil Carbon	Easy (0-20^{o})	101.56	100.66	99.75	98.84	97.94	97.03	96.13	95.22	94.32	93.41	92.51
	Steep (20-40^{o})	91.99	89.88	87.77	85.66	83.55	81.45	79.34	77.23	75.12	73.02	70.91

APPENDIX B

Results tables for native forest plots, and soil samples.

Table B-1. Native Forest Live Plant Carbon mass from the year 2000 to 2011 and yearly change in live plant carbon mass over the three periods used by permanent plot.

Category					**Live Carbon Mass t C ha^{-1}**					**Live Carbon Change t C ha^{-1} yr^{-1}**			
Forest	Slope	Aspect	Planted	Grazed	2000	2002	2004	2008	2011	00-02	02-04	04-08	08-11
kahikatea	steep	NW	yes	fenced	427.54	474.06	475.92	497.87	667.27	21.64	0.93	5.56	58.34
kahikatea	moderate	S	yes	fenced	78.97	78.72	89.39	91.58	96.13	-0.13	5.58	0.64	0.82
kahikatea	moderate	S	yes	fenced	158.25	209.74	212.31	220.78	305.69	25.75	1.29	2.12	27.15
kahikatea	moderate	N	yes	fenced	154.91	171.80	182.69	199.22	275.88	8.00	7.44	4.13	25.55
kahikatea	steep	S	yes	fenced	85.62	77.11	87.41	92.15	110.33	-0.16	4.82	1.19	6.06
kahikatea	easy	NW	yes	fenced	50.66	53.69	55.07	61.22	72.92	1.62	0.69	1.33	3.90
kohekohe	steep	E	no	fenced	154.07	158.83	168.85	188.40	.	0.24	7.15	4.19	.
kohekohe	moderate	NW	no	fenced	100.59	106.39	115.03	127.27	.	2.90	4.32	3.06	.
kohekohe	steep	S	no	fenced	168.08	175.44	182.35	194.89	.	3.68	2.02	3.13	.
kohekohe	moderate	S	no	fenced	78.53	93.36	77.39	92.80	.	4.17	-4.73	3.69	.
kohekohe	steep	NW	no	fenced	75.28	77.17	83.46	89.59	.	0.95	3.14	1.17	.
kohekohe	steep	NW	no	fenced	96.58	96.58	111.01	106.27	.	0.00	7.21	-1.19	.
kohekohe	steep	S	no	fenced	41.22	53.61	47.51	61.43	.	4.62	-0.52	2.23	.
kohekohe	easy	NW	no	fenced	76.98	84.99	89.53	98.45	.	3.11	2.27	2.56	.
rewarewa	moderate	N	yes	fenced	78.56	80.68	82.87	84.95	99.61	2.13	1.09	0.52	8.19
rewarewa	moderate	N	yes	fenced	155.99	160.58	163.30	175.73	259.22	2.85	1.36	3.49	28.47
rewarewa	easy	N	yes	fenced	134.19	132.94	137.83	148.61	181.15	-0.62	2.44	2.70	11.86
rewarewa	moderate	SW	yes	fenced	14.51	16.40	17.75	3.93	.	1.98	1.64	-0.27	.
rewarewa	moderate	N	yes	fenced	14.63	14.63	14.45	20.11	.	0.00	-0.09	1.42	.
rewarewa	moderate	NE	yes	fenced	26.71	26.71	26.71	27.26	34.43	0.00	0.00	0.14	2.39

rewarewa	steep	S	yes	fenced	288.40	295.80	296.65	317.30	421.70	3.70	5.28	5.16	31.14
rewarewa	easy	N	yes	fenced	165.71	174.31	184.92	177.53	238.14	4.30	5.31	-1.55	17.11
rewarewa	moderate	N	yes	fenced	59.00	66.27	70.55	78.75	99.76	4.93	2.14	2.05	7.00
rewarewa	moderate	S	yes	fenced	63.04	49.01	51.92	42.51	59.35	0.02	3.74	-0.64	1.69
rewarewa	moderate	N	yes	fenced	60.15	62.93	75.65	62.18	70.20	1.39	6.36	-1.51	2.59
rimu	moderate	NW	no	grazed	102.49	107.97	129.97	127.23	154.30	2.74	11.00	-0.68	5.89
rimu	steep	S	no	grazed	51.31	48.61	49.92	42.54	42.36	-1.35	0.66	-0.93	-0.06
rimu	steep	N	no	grazed	250.70	248.50	265.83	288.46	305.88	1.55	7.55	4.60	5.81
rimu	steep	N	no	grazed	139.53	154.57	176.87	174.31	212.78	3.97	11.15	3.48	11.52
rimu	moderate	S	no	grazed	167.84	156.37	173.81	180.86	219.98	-4.33	8.72	2.12	13.04
rimu	steep	S	no	grazed	19.72	18.40	20.00	11.45	15.68	-0.66	0.80	-0.14	1.41
rimu	moderate	S	no	grazed	112.95	122.54	131.93	154.04	203.72	5.35	6.73	5.53	17.28
rimu	steep	S	no	grazed	58.66	62.52	64.21	46.83	37.21	1.73	0.85	-1.23	0.22
rimu	moderate	E	no	grazed	91.37	54.76	23.30	26.72	.	-18.30	0.32	0.85	.
rimu	steep	S	no	grazed	25.35	23.95	35.94	22.35	.	4.33	6.00	-4.32	.
rimu	steep	S	no	grazed	199.52	203.98	209.14	224.28	306.88	2.23	2.58	4.77	29.12
rimu	easy	N	no	grazed	47.96	58.48	59.11	47.58	45.19	5.26	0.31	0.37	-0.79
rimu	easy	N	no	grazed	118.32	144.37	137.21	135.09	179.83	2.32	-3.58	0.61	14.91
rimu	moderate	N	no	grazed	74.55	84.03	87.81	94.08	111.12	4.74	3.47	1.52	9.90

TableB-2. Coarse Woody Debris Mass for native forest fragments plots as calculated from regression with basal area from the year 2000 to 2008 and as from direct measurement in 2011 and the measured basal area used for the regression.

	Category				CWD mass t Cha^{-1}					CWD Basal Area $m^2\ ha^{-1}$
Forest	**Slope**	**Aspect**	**Planted**	**Grazing**	**2000**	**2002**	**2004**	**2008**	**2011**	**2011**
Kahikatea	moderate	N	yes	Fenced	3.81	4.00	3.85	4.05	3.19	4.7
Kahikatea	moderate	S	yes	Fenced	5.37	5.61	5.75	7.05	2.63	21.9
Kahikatea	easy	NW	yes	Fenced	3.81	4.54	4.73	4.47	0.42	5.1
Kahikatea	moderate	S	yes	Fenced	4.43	3.81	4.47	8.22	0.37	17.4
Kahikatea	steep	NW	yes	Fenced	3.81	3.93	3.98	4.52	1.99	2.3
Kahikatea	steep	S	yes	Fenced	5.58	3.81	3.81	4.90	2.73	2.1
kohekohe	steep	NW	no	Fenced	4.05	4.22	3.81	3.81	.	.
kohekohe	easy	NW	no	Fenced	3.81	3.81	3.84	4.30	.	.
kohekohe	moderate	S	no	Fenced	4.43	3.81	3.81	3.81	.	.
kohekohe	moderate	NW	no	Fenced	4.35	3.81	3.81	3.81	.	.
kohekohe	steep	S	no	Fenced	5.89	3.81	8.06	5.81	.	.
kohekohe	steep	E	no	Fenced	4.16	4.84	4.85	3.81	.	.
kohekohe	steep	NW	no	Fenced	4.93	4.55	4.59	5.80	.	.
kohekohe	steep	S	no	Fenced	3.81	3.81	4.35	4.60	.	.
rewarewa	moderate	SW	yes	Fenced	6.77	5.15	5.32	7.11	.	.
rewarewa	moderate	NE	yes	Fenced	6.48	5.63	5.85	6.28	3.68	14.6
rewarewa	moderate	E	yes	Fenced	7.02	5.12	4.76	10.35	.	.
rewarewa	moderate	N	yes	Fenced	5.95	6.01	6.23	6.04	4.87	1.3
rewarewa	moderate	N	yes	Fenced	3.96	4.28	4.32	4.64	5.25	6.0

rewarewa	moderate	N	yes	Fenced	5.52	7.79	7.56	7.07	20.26	13.7
rewarewa	easy	N	yes	Fenced	4.96	7.10	6.74	5.85	10.49	17.5
rewarewa	moderate	N	yes	Fenced	4.60	5.16	5.82	5.79	4.02	13.2
rewarewa	moderate	S	yes	Fenced	5.67	7.01	5.30	9.85	6.41	44.2
rewarewa	steep	S	yes	Fenced	7.69	6.78	7.29	6.45	3.15	18.0
rewarewa	easy	N	yes	Fenced	3.85	3.89	3.87	4.00	0.37	0.5
rimu	steep	S	no	Grazed	4.11	4.65	5.71	7.43	15.01	11.0
rimu	moderate	S	no	Grazed	5.04	5.01	5.38	5.18	6.01	11.9
rimu	steep	S	no	Grazed	3.81	3.89	3.92	3.89	1.42	.
rimu	moderate	NW	no	Grazed	3.81	3.81	3.81	3.85	4.59	0.3
rimu	moderate	N	no	Grazed	3.81	8.02	7.87	9.34	16.47	33.4
rimu	steep	S	no	Grazed	6.54	9.07	8.91	7.88	10.48	21.1
rimu	moderate	S	no	Grazed	3.81	4.49	4.59	4.06	8.69	7.0
rimu	moderate	E	no	Grazed	7.54	6.20	6.58	10.20	.	.
rimu	steep	E	no	Grazed	3.81	3.81	4.86	6.27	.	.
rimu	easy	N	no	Grazed	4.41	4.02	3.81	5.32	3.91	6.1
rimu	steep	S	no	Grazed	6.78	7.46	7.50	15.58	1.46	13.9
rimu	easy	N	no	Grazed	3.81	3.89	3.90	3.85	9.16	22.0
rimu	steep	N	no	Grazed	3.81	4.88	4.58	5.04	0.01	22.0
rimu	steep	N	no	Grazed	3.81	3.81	3.81	4.87	6.26	10.8

Table B-3. Mass of litter samples in 2011 from native forest fragments, unadjusted for % cover of litter.

Forest	Grazing	**Planted**	**slope**	**Aspect**	Litter mass t C ha^{-1}
rimu	grazed	no	steep	S	2.71
rimu	grazed	no	moderate	S	4.06
rimu	grazed	no	steep	S	2.89
rimu	grazed	no	moderate	NW	2.48
rimu	grazed	no	moderate	N	5.83
rimu	grazed	no	steep	S	2.51
rimu	grazed	no	moderate	S	2.00
rimu	grazed	no	easy	N	1.23
rimu	grazed	no	steep	S	8.01
rimu	grazed	no	easy	N	17.45
rimu	grazed	no	steep	N	5.01
rimu	grazed	no	steep	N	0.78
Kahikatea	fenced	yes	moderate	N	5.59
Kahikatea	fenced	yes	moderate	S	2.65
Kahikatea	fenced	yes	easy	NW	1.26
Kahikatea	fenced	yes	moderate	S	6.49
Kahikatea	fenced	yes	steep	NW	10.41
Kahikatea	fenced	yes	steep	S	3.44
rewarewa	fenced	yes	moderate	NE	3.73
rewarewa	fenced	yes	moderate	N	10.75
rewarewa	fenced	yes	moderate	N	4.68
rewarewa	fenced	yes	moderate	N	3.67
rewarewa	fenced	yes	easy	N	7.33
rewarewa	fenced	yes	moderate	N	3.08
rewarewa	fenced	yes	moderate	S	4.80
rewarewa	fenced	yes	steep	S	8.66
rewarewa	fenced	yes	easy	N	4.45

Table B-4. Litter mass for native forest fragments from 2000 to 2011

Category							**Litter mass t C ha^{-1}**		
Forest	**Grazing**	**planted**	**slope**	**Aspect**	**2000**	**2002**	**2004**	**2008**	**2011**
kohekohe	fenced	no	steep	E	3.02	2.77	2.52	3.78	3.78
kohekohe	fenced	no	moderate	NW	0.76	0.76	1.01	0.25	0.25
kohekohe	fenced	no	steep	S	3.28	3.28	2.52	2.67	2.67
kohekohe	fenced	no	moderate	S	0.76	1.76	2.52	2.77	2.77
kohekohe	fenced	no	steep	NW	0.40	0.25	0.76	0.50	0.50
kohekohe	fenced	no	steep	NW	0.03	0.15	0.25	0.20	0.20
kohekohe	fenced	no	steep	S	2.02	0.76	2.52	4.54	4.54
kohekohe	fenced	no	easy	NW	3.02	3.53	2.52	2.22	2.22
rimu	grazed	no	steep	S	2.03	1.76	1.49	1.22	1.08
rimu	grazed	no	moderate	S	2.84	2.84	1.22	1.62	2.03
rimu	grazed	no	steep	S	1.45	2.31	1.74	1.30	2.31
rimu	grazed	no	moderate	NW	0.99	1.12	1.24	1.66	1.86
rimu	grazed	no	moderate	N	1.75	1.75	1.46	1.17	2.33
rimu	grazed	no	steep	S	2.13	1.88	1.75	1.83	2.00
rimu	grazed	no	moderate	S	0.70	1.20	0.50	0.74	0.60
rimu	grazed	no	moderate	E	24.75	16.50	19.50	42.25	3.28
rimu	grazed	no	steep	S	3.75	0.75	0.75	2.25	0.76
rimu	grazed	no	easy	N	0.25	0.12	0.10	0.18	0.18
rimu	grazed	no	steep	S	5.20	6.01	6.81	4.56	3.20
rimu	grazed	no	easy	N	10.47	8.72	11.34	12.04	16.05
rimu	grazed	no	steep	N	1.25	0.75	2.25	0.90	3.51

rimu	grazed	no	steep	N	0.04	0.08	0.12	0.08	0.05
Kahikatea	fenced	yes	moderate	N	0.84	0.84	1.96	0.00	4.47
Kahikatea	fenced	yes	moderate	S	1.19	0.66	1.19	1.83	2.12
Kahikatea	fenced	yes	easy	NW	0.44	0.31	0.31	0.69	1.13
Kahikatea	fenced	yes	moderate	S	0.97	0.97	0.52	4.09	5.19
Kahikatea	fenced	yes	steep	NW	8.85	7.29	8.33	9.37	9.37
Kahikatea	fenced	yes	steep	S	1.72	1.72	1.38	2.24	2.75
rewarewa	fenced	yes	moderate	SW	0.75	0.50	0.50	4.00	1.01
rewarewa	fenced	yes	moderate	NE	12.00	25.50	3.17	4.00	0.75
rewarewa	fenced	yes	moderate	N	10.50	12.00	2.02	12.25	1.76
rewarewa	fenced	yes	moderate	N	8.60	6.99	8.60	9.14	9.14
rewarewa	fenced	yes	moderate	N	3.98	3.51	4.40	3.28	3.98
rewarewa	fenced	yes	moderate	N	1.84	2.20	1.84	1.84	2.76
rewarewa	fenced	yes	easy	N	6.60	6.60	6.16	6.38	7.04
rewarewa	fenced	yes	moderate	N	2.47	2.31	1.85	2.16	2.96
rewarewa	fenced	yes	moderate	S	1.92	1.92	3.36	3.75	4.32
rewarewa	fenced	yes	steep	S	4.76	5.63	6.93	2.60	7.79
rewarewa	fenced	yes	easy	N	2.67	2.23	1.78	3.34	2.89

Table B-5. Soil samples for different land uses from 2001 and 2011.

Land-use 2000	Land-use 2001	Slope	Aspect	Depth	Bulk density (g/cm3) 2001	Carbon (t/ha)	Bulk density (g/cm3)	Carbon (t/ha)	N (g/cm3) 2011
grazed Forest	grazed Forest	steep	south	100-200mm	0.94	27.37	0.93	20.16	0.21
grazed Forest	grazed Forest	steep	south	0-100mm	0.88	44.93	0.95	41.96	0.36
grazed Forest	grazed Forest	mod	west	100-200mm	1.01	15.35	0.94	36.21	0.33
grazed Forest	grazed Forest	mod	west	0-100mm	0.87	18.72	0.91	44.05	0.32
grazed Pasture	grazed Pasture	mod	northwest	100-200mm	1.07	42.40	1.07	28.62	0.24
grazed Pasture	grazed Pasture	mod	northwest	0-100mm	0.94	65.65	0.84	52.11	0.41
grazed Pasture	grazed Pasture	mod	northwest	100-200mm	1.01	36.58	0.98	25.38	0.24
grazed Pasture	grazed Pasture	mod	northwest	0-100mm	1.03	76.19	1.02	60.15	0.45
Pasture	Native Planting	steep	northwest	100-200mm	0.94	45.92	0.70	67.18	0.53
Pasture	Native Planting	steep	northwest	0-100mm	0.96	86.33	0.55	78.17	0.65
Pasture	Native Planting	easy	southeast	100-200mm	0.64	44.59	1.06	26.02	0.27
Pasture	Native Planting	easy	southeast	0-100mm	0.68	80.77	0.93	51.35	0.46
Pasture	Native Planting	mod	south	100-200mm	0.89	29.33	0.98	39.80	0.30
Pasture	Native Planting	mod	south	0-100mm	0.92	84.55	0.71	66.20	0.53
Pasture	Native Planting	mod	west	100-200mm	0.87	43.61	0.90	30.51	0.24
Pasture	Native Planting	mod	west	0-100mm	0.81	79.14	0.84	63.01	0.51
Pasture	Pine	easy	east	100-200mm	1.00	27.43	1.06	33.22	0.28
Pasture	Pine	easy	east	0-100mm	0.90	48.90	0.91	56.77	0.49
Pasture	Pine	easy	north	100-200mm	.	.	1.07	27.20	0.24
Pasture	Pine	easy	north	0-100mm	.	.	0.91	41.33	0.36
Pasture	Pine	easy	south	100-200mm	.	.	1.09	54.17	0.45
Pasture	Pine	easy	south	0-100mm	.	.	0.87	62.05	0.57

Pasture	Pine	steep	east	100-200mm	.	.	1.02	31.60	0.31
Pasture	Pine	steep	east	0-100mm	.	.	1.02	58.37	0.51
Pasture	Pine	steep	north	100-200mm	.	.	1.08	18.82	0.21
Pasture	Pine	steep	north	0-100mm	.	.	0.98	30.83	0.29
Pasture	Pine	steep	south	100-200mm	.	.	0.91	25.97	0.25
Pasture	Pine	steep	south	0-100mm	.	.	0.74	40.73	0.33
grazed Forest	ungrazed Forest	easy	north	100-200mm	1.09	33.24	1.13	36.22	0.37
grazed Forest	ungrazed Forest	easy	north	0-100mm	0.78	53.14	0.73	47.61	0.38
grazed Forest	ungrazed Forest	mod	south	100-200mm	0.93	23.94	1.05	22.78	0.26
grazed Forest	ungrazed Forest	mod	south	0-100mm	0.96	67.00	1.01	49.64	0.40
grazed Forest	ungrazed Forest	easy	northwest	0-100mm	1.16	45.12	.	.	.
grazed Forest	ungrazed Forest	easy	northwest	100-200mm	1.02	47.50	.	.	.
grazed Forest	ungrazed Forest	mod	northwest	100-200mm	1.09	21.55	.	.	.
grazed Forest	ungrazed Forest	mod	northwest	0-100mm	1.16	35.49	.	.	.
grazed Forest	ungrazed Forest	mod	east	100-200mm	0.94	31.75	1.12	29.15	0.32
grazed Forest	ungrazed Forest	mod	east	0-100mm	1.10	66.96	1.00	45.97	0.39
grazed Forest	ungrazed Forest	mod	east	100-200mm	0.92	33.58	1.09	28.91	0.32
grazed Forest	ungrazed Forest	mod	east	0-100mm	1.13	69.99	0.85	30.04	0.32
Pasture	ungrazed Pasture	mod	northwest	100-200mm	1.02	35.83	.	.	.
Pasture	ungrazed Pasture	mod	northwest	0-100mm	1.05	76.13	.	.	.
Pasture	ungrazed Pasture	easy	south	100-200mm	1.01	34.18	.	.	.
Pasture	ungrazed Pasture	easy	south	0-100mm	0.83	74.70	.	.	.