# Wild Animal Control for Emissions Management (WACEM) research synthesis

Landcare Research
Manaaki Whenua

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# Wild Animal Control for Emissions Management (WACEM) research synthesis

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## Abstract

We have quantified the current and potential carbon storage on conservation land. Conservation land was estimated to currently contain a total of 2578 Mt C (9461 Mt $CO_2$e) in vegetation and soil. We estimate that conservation land could store up to 698 Mt $CO_2$e more than at present, through reforestation (large gains per hectare), advance of existing shrubland successions (moderate gains per hectare) and recovery of existing forests from disturbances, including animal browsing, (small gains over large areas). Wild animal control plays a role in all three types of gain, but here we demonstrate that, for existing forests, it will be very challenging to quantify sequestration in existing forests that (a) can be attributed to wild animal control and (b) is additional to the sink that already exists. It is our view that the presence of a forest sink in existing forests is a strong endorsement of business-as-usual management of conservation land in forest and that this management should be maintained for the benefit of all New Zealanders. We recommend that any future effort to increase carbon sequestration through targeted wild animal control should focus on successional communities—removing wild animals from broadleaved-hardwood successions where the palatable canopy species of the future need to become established, and supporting other management actions to establish novel woody successions to indigenous forests within degraded grasslands. These actions should be evaluated alongside the carbon costs of undertaking these management activities. We note that on conservation land where natural succession does not readily occur, additional, and likely intensive, management will be required such as planting for shelter and provision of seed, with associated weeding requirements. The fastest, and most cost-effective, action to increase carbon sequestration on conservation land is to choose sites close to existing indigenous forest seed sources that are relatively warm (mean annual temperature > 9°C) and receive reasonable rainfall (> 1000 mm per annum) and retire these from grazing and to prevent fire and domestic or wild animal incursion as far as possible and allow natural regeneration to proceed unhindered. Further thought needs to be given to the carbon consequences of novel ecosystems – especially whether exotic woody species are able to play a desirable role in natural regeneration of indigenous forests.

Keywords: carbon stocks, sequestration, forest sinks, natural regeneration, additionality

# 1 Introduction

The Department of Conservation (DOC) is required to manage public conservation land for conservation purposes. Other uses are allowable provided they are consistent with conservation. The Department's 2013 Statement of Intent stresses a need for increased engagement with 'consumers, businesses and communities' in order to achieve more for conservation (DOC 2013, p. 9). Given that policy mechanisms and markets already exist for carbon revenue, carbon is a strong opportunity both for New Zealand's public good (in reducing our emissions liabilities) and for additional income through increased carbon sequestration. Given the extent of conservation land (c. 35% of the New Zealand landscape; Ministry for the Environment 2007), considerable scope exists to enhance per-hectare carbon either by a small amount over the large land area of existing forests or a large amount over the relatively modest area of currently non-forested lands. Activities that promote carbon sequestration on conservation land can also achieve desirable conservation outcomes. For example, the re-establishment of under-represented coastal forest ecosystems could increase carbon sequestration at a site and increase the representation of these now-scarce forests. It is possible that flow-on increases in recreational values could also occur. Conservation land provides a suite of 'ecosystem services' (benefits provided to humans by nature). Trade-offs with other ecosystem services are certainly possible under management scenarios that privilege carbon sequestration, but these trade-offs are not explicitly considered here.

Given that conservation land represents the largest land area in New Zealand under a single manager, DOC's management of carbon is critical for New Zealand. The Department therefore commissioned a series of reports outlining the potential for carbon management of conservation land—including identification of the most significant information gaps—and a major research programme investigating the potential for wild animal control to assist carbon sequestration in indigenous ecosystems (the Wild Animal Control for Emissions Management or WACEM programme). We know of no other agency, globally, that has investigated the potential for wild animal control to contribute to nationally increased carbon stocks – in this respect, DOC has shown international leadership. In this report we synthesise the most important findings and their implications for management. We also list chronologically all outputs from the programme along with their scientific summaries.

# 2 Research structure

## 2.1 Reports outlining the potential for carbon management of conservation land

Five reports were commissioned that synthesised existing information on current and potential carbon stocks of conservation land. Options for increasing either carbon stock or rates of sequestration by terrestrial vegetation were also explored. Specific focus was given to the potential for wild animal control to contribute to increased carbon sequestration on conservation land. In particular we asked: Which ecosystems are likely to particularly benefit from wild animal control and which ecosystems are especially vulnerable to reduction in wild animal control? Prioritisation was achieved through construction of a matrix of management actions and the questions that need to be answered about each potential management action (Table 1). Whilst research is required across all of the cells of the matrix, those cells targeted by the Wild Animal Control for Emissions Management (WACEM) programme are highlighted in green.

**Table 1** Knowledge required to determine whether carbon sequestration can be managed, or preferably, enhanced in natural ecosystems. Green coloured squares indicate topics addressed across this research programme

| | Management activity | | | | |
|---|---|---|---|---|---|
| Potential management action | Establish baselines | Encourage woody successions | Effective herbivore control | Manipulate other invasive species | Establish high-carbon-storage ecosystems |
| What is the potential for carbon gains? | What are the potential carbon stocks?<br>How robust do baselines need to be? | What rates of sequestration are possible? | What carbon gains are possible at selected sites? | Can invaders be categorised in terms of potential carbon stock impacts? | Can species combinations be designed to enhance carbon stocks and over what time period? |
| How are effective management actions to be implemented? | Which systems are resistant or resilient in terms of carbon? | Can seeding overcome dispersal limitation or is expensive, and carbon intensive, planting required? | Can forest carbon be maintained through pulsed recruitment? | When can exotic plants be used to enhance carbon storage? | Can critical components for high carbon systems be restored? |
| What other consequences are desirable? | What is the monetary benefit of achieving potential? | What are the consequences for water yields? | When are carbon and biodiversity gains congruent? | Does control of an invasive species reduce exotic richness? | Do high storage systems meet biodiversity goals? |
| How should benefits be quantified? | Will potential change with time? | How can additionality be determined? | Is it realistic to measure slow soil carbon changes? | Can invasive impacts on carbon stocks always be partitioned? | Can carbon benefits be modelled over relevant timescales? |

The five reports from this initial scoping study (including two *Science for Conservation* publications) also led to two publications in the *New Zealand Journal of Ecology*.

## 2.2 Research programme on wild animal control for emissions management

One methodological study was conducted alongside five studies that tested hypotheses about the potential for wild animal control to increase carbon stocks/sequestration or at least reduce the risk of carbon loss. The methodological study produced a field protocol for measuring soil carbon stocks. The other five studies targeted specific situations thought to offer the greatest potential for management of carbon on conservation land. Questions asked across several of the studies were: How big a change in carbon stocks/sequestration is required to detect significant differences resulting from wild animal control in typical experimental designs? How could wild animal control potentially benefit palatable broadleaved-hardwood forests both during succession and at maturity? Are some tree species protective against carbon loss even in the event of their widespread mortality? Are some impacts of wild animal control more apparent below ground than above? What are the ways that rates of establishment of woody succession could be enhanced through management? Do tall-tussock grasslands recover carbon following removal of grazing?

Across this programme we produced four contract reports, five published manuscripts (in *Journal of Ecology*, *Biological Conservation*, *New Zealand Journal of Ecology* and *Forest Ecology and Management* (two)) and an additional manuscript that is in preparation for publication (Peltzer et al.).

# 3 Synthesis

## 3.1 Potential for carbon management of conservation land

Results from this research programme suggest that there is potential for conservation land to store up to 698 Mt $CO_2$e more than at present (Carswell et al. 2008). Of this figure, up to 99 Mt $CO_2$e could be Kyoto-compliant (i.e. post-1990; Mason et al. 2012). Gains could result from replacing non-forest landscapes with forest, successional change of existing shrublands into tall forest, coupled with net sequestration by existing tall forests. Some of these processes will be easier to achieve than others. For example, a recent analysis of whether New Zealand's natural (pre-1990) forests are collectively a net source or sink of carbon has shown that net sequestration is occurring in regenerating forests (Holdaway et al. 2014a). Given that most of this pre-1990 regenerating natural forest is on conservation land, significant carbon gains are occurring under 'business as usual' management. With respect to gains through non-forest-to-forest or shrubland-to-forest transitions there will be some situations where gains can be easily achieved with minimal management and others where management would be highly intensive and expensive in order to effect transition. The WACEM research programme focused on the situations where management may be required to effect transition and also considered the ability to measure the difference made through management.

### 3.1.1 Non-forest to forest gain

We estimate that approximately 600 000 ha of conservation land were not in forest at 1990 and could potentially support higher carbon indigenous vegetation than at present (Carswell et al. 2013). Out of the total 600 000 ha, the lands that would require little management intervention are those close to existing indigenous forests that are relatively warm; lands with mean annual temperature below 9°C have only a low probability of naturally regenerating forest (Mason et al. 2013a). Lands distant from existing forests and experiencing lower mean annual temperatures (e.g. the plains of the eastern South Island) would be very unlikely to regenerate indigenous forests without extensive management intervention, such as planting. The areas of conservation land available for non-forest-to-forest transition, with a predicted net increase in biodiversity during natural regeneration, are in southern Marlborough, parts of the foothills of the Southern Alps and many areas of Stewart Island (Fig. 1). Of these areas, Mason et al. (2013a) predict that southern Marlborough is unlikely to regenerate naturally into indigenous forest without relatively intensive management intervention on account of being distant from natural seed sources.

For this exercise, when defining areas with potential for post-1990 afforestation, we excluded areas where this would achieve no net benefit to biodiversity. For example, replacement of diverse tussock grasslands with a monoculture of an early-successional shrub species may not represent a desirable outcome for biodiversity if tussock grasslands are under-represented at national scale. Net benefit to biodiversity was calculated using a modification of the approach used by the Vital Sites and Actions (VSA) framework (Overton et al. 2010) to calculate 'restored significance'. The framework operationalizes the concept of 'ecological integrity', defined by Lee et al. (2005) as 'the full potential of indigenous biotic and abiotic factors, and natural processes, functioning in sustainable communities, habitats, and landscapes' and adopted by the New Zealand Department of Conservation (DOC 2011) as its primary biodiversity goal. Restored significance measures the marginal improvement in environmental representation (one component of ecological integrity) if the biological community is restored from its present condition to potential natural condition. A site will have high restored significance if the present vegetation is in poor condition relative to potential and it occurs in an environment that has low environmental representation (i.e. most of the natural vegetation has been cleared).

It is apparent that the area of conservation land that could transition to indigenous forest with little management effort is relatively limited – in fact, there are no parcels of at least 10,000 ha in extent. This is not surprising given that conservation land is already being managed for biodiversity, which will include promotion of natural regeneration to indigenous forests, where appropriate. Additionally, the land areas that could benefit the most from natural regeneration to indigenous forests (e.g., alluvial floodplains, riparian habitats and coastal ecosystems) are severely under-represented in conservation land both locally and internationally (Pressey 1994; Walker et al. 2006, 2008). The biggest areas of potential gain for both carbon and biodiversity per hectare now lie outside conservation land (Carswell et al. 2013).

We note that for many of the areas with a low probability of indigenous forest establishment, exotic woody species are establishing with ease (e.g., southern Marlborough). The establishment of conifers (wilding or otherwise) is an extremely cost effective way to increase carbon sequestration of grasslands retired from agriculture. Some evidence suggests that they can benefit biodiversity, particularly when they act as nursery sites for indigenous species (Becerra & Montenegro 2013; Norton & Forbes 2013). However, the indigenous species regenerating under pine (*Pinus radiata*) thus far appear to be predominantly sub-canopy species and not those that would comprise a tall indigenous forest in future (e.g., Allen et al. 1995). Other evidence suggests that the ability of exotic woody species to facilitate indigenous forest succession is dependent on local indigenous seed sources (Carswell et al. 2013). Until further research is conducted on long-term trajectories of exotic conifer-indigenous ecosystems, we argue that wilding conifer invasion is not consistent with conservation of the land, in terms of biodiversity benefit.

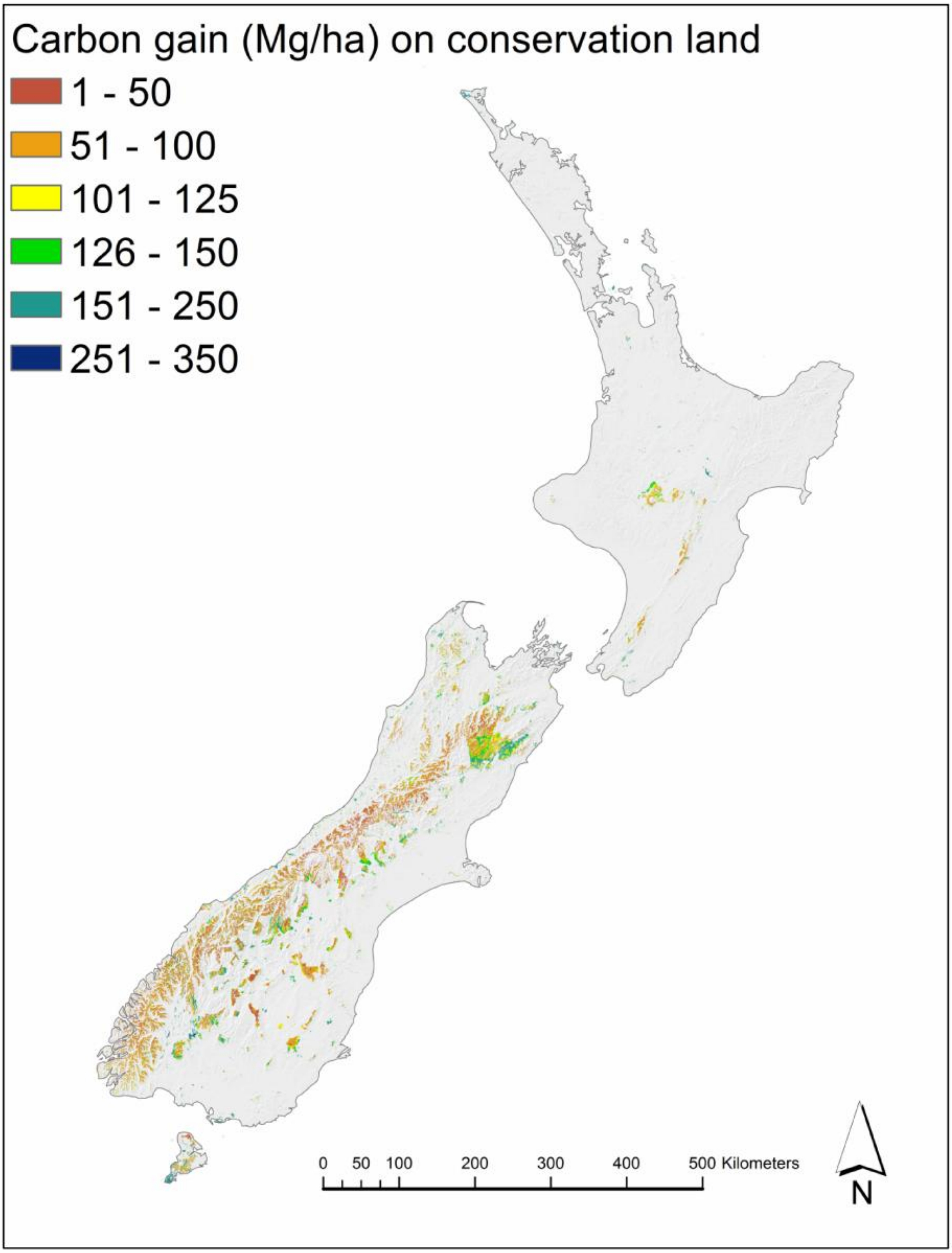

**Figure 1** Potential post-1990 gains in carbon using non-forest to forest conversion of conservation land. Note that only areas with a net increase in ecological integrity (assessed through 'restored significance'; Overton et al. 2010) are shown.

### 3.1.2 Net sequestration by existing regenerating forests

We recently concluded that New Zealand's naturally regenerating (pre-1990) forests are absorbing more carbon through increased biomass than is released to the atmosphere during mortality (Holdaway et al. 2014a). The net sequestration is largely driven by normal successional processes whereby low-statured early-successional forest is progressively replaced by larger trees that contain more carbon per hectare. For example, the Ministry for the Environment estimates that approximately 50 000 ha of land changed from 'grassland with woody biomass' (grassland with scattered trees) to 'natural forest' (land with at least 30% cover of tree species per hectare) between 1962 and 1989 (MfE 2013). This comprises only change detectable from satellite imagery. Wiser et al. (2011) suggest that 45% (c. 670 000 ha) of New Zealand's total pre-1990 shrubland area (c. 1.5 Mha) shows evidence of recruitment of tree species. These shrublands typically comprise bracken (*Pteridium esculentum*), gorse (*Ulex europaeus*), tree fern (*Dicksonia squarrosa*) and mānuka (*Leptospermum scoparium*) mixed with exotic grasses. The indigenous tree species regenerating within these communities include kānuka (*Kunzea ericoides*), māhoe (*Melicytus ramiflorus*), marble leaf (*Carpodetus serratus*), kāmahi (*Weinmannia racemosa*) and five-finger (*Pseudopanax arboreus*). A previous analysis has suggested that succession of these shrublands to tall forests could result in storage of at least an additional 500 t $CO_2$e/ha (Holdaway et al. 2012).

As with non-forest-to-forest gains, some management could help effect the transition in certain situations. An investigation of the potential for enhancement planting of mānuka/kanuka shrubland with conifers or other tall canopy trees has shown that planting is only useful where there are no existing seed sources of taller trees (Paul et al. 2013). There was no evidence for an increased rate of sequestration with enhancement planting over 20 years, but it does appear as if the longer term sequestration could be increased by the addition of these larger species. Further, Paul et al. (2013) showed that, to maximise survival and growth of the canopy species, enhancement planting must occur before the mānuka/kānuka shrublands are 4 m in height.

Given that net sequestration is already occurring in regenerating forests, additional per-hectare gains resulting from additional management would be very difficult to quantify. This is because any increase in sequestration rate as a result of management will be small relative to the baseline sequestration rate and is likely to be statistically non-significant due to large levels of natural between-plot variability. The difficulty in measuring change that results from management is illustrated in Fig. 2. To ascribe a change in sequestration rate of existing forests to management, it is necessary to find a statistically significant difference between the red and blue lines in the figure. Given that between-plot natural variability (generally expressed as 95% confidence limits) is generally larger than the difference between the two lines, this will be very difficult. Nonetheless, Holdaway et al. (2012) suggest that there are some scenarios where per-hectare gains could be measured. Therefore, most of the sections that follow focus on per-hectare gains and attempt to quantify the amount of gain that can be detected through management of existing forests.

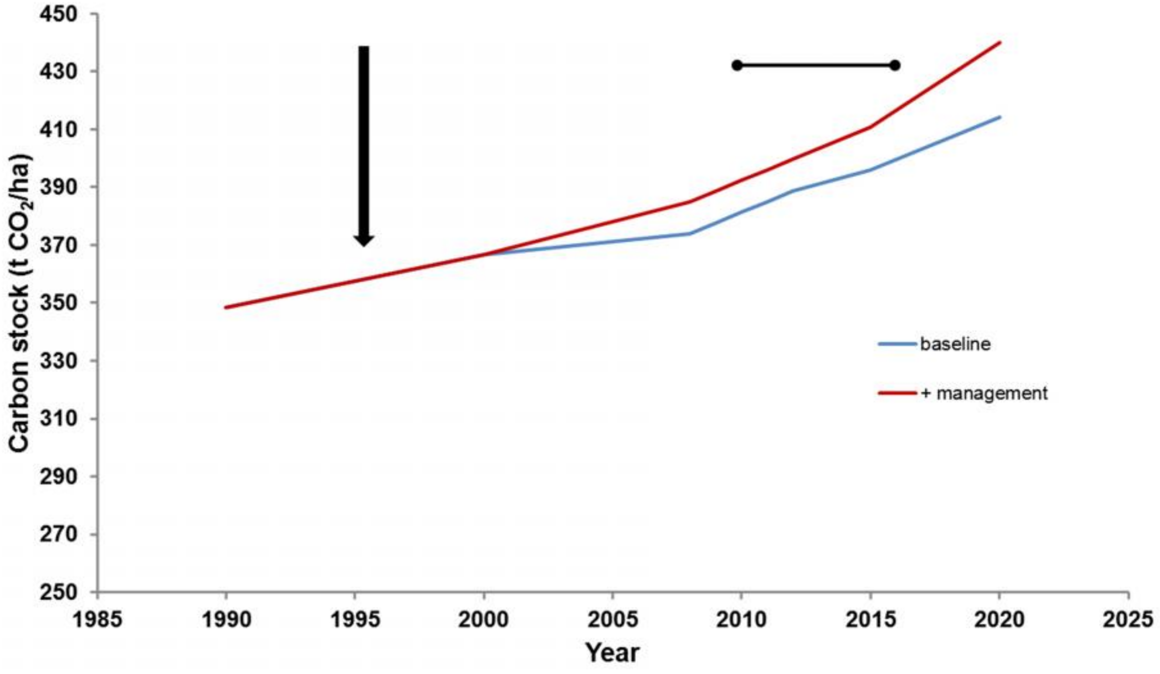

**Figure 2** Stylised trajectory of carbon stock increase in a typical New Zealand forest recovering from disturbance (namely most New Zealand forests, which are currently recovering from either anthropogenic (e.g. logging, fire) or natural disturbance (e.g. vulcanism, windthrow, earthquake)) showing the baseline of carbon stock increase (in blue), a major management input (such as extensive wild animal control depicted by the heavy arrow at 1995), a hypothesised improvement in carbon sequestration (in red) and the period over which carbon stocks are generally measured (horizontal black line between 2010 and 2015).

## 3.2 Wild Animal Control for Emissions Management

### 3.2.1 How big a change in carbon stock/sequestration is required before we can detect significant differences resulting from wild animal control in typical experimental designs?

The results obtained from the ecosystems studied within the WACEM research programme, with and without experimental manipulation, emphasise the difficulty in ascribing change in either carbon stocks or sequestration rate to management, especially wild animal control. Holdaway et al. (2012) show through power analysis that an effect size of > 0.5 t C/ha/year is required to keep plot numbers below 100 for a single study (plot numbers required increase with increasing between-plot variability). We note that the average background sequestration rate (all pools except soil) for pre-1990 regenerating forests is 1.39 t C/ha/year (Holdaway et al. 2014a). Currently, when tall forests are combined with regenerating forests, the net sequestration for all natural forests and shrublands is indistinguishable from zero (at +0.34 (95% CI −0.09 to 0.77) tC/ha/year). An increase of sequestration of 0.5 t C/ha/year across all conservation lands would change this result but would be very difficult to achieve. However, at local scales, there are naturally uniform ecosystems (e.g., even-aged non-successional shrublands) which may not require such a large proportional increase in sequestration in order to be measurable. Given that New Zealand's public conservation lands contain our largest above-ground carbon stock, it is important that management continues to protect this stock to avoid the risk of it becoming a net source of carbon in the future.

The analysis from Holdaway et al. (2012) reveals that different numbers of plots will likely be required between vegetation types to measure carbon stocks with reasonable precision. An indication is given in Table 2—note that number of plots required decreases as carbon stock increases and between-plot variability associated with each vegetation class decreases. Again, it is clear that moving vegetation of low carbon stock to a class with higher carbon stock will yield the biggest net change in carbon stock as a result of management. For example, movement of vegetation from any of the shrubland classes to any of the forest classes will result in at least a tripling of carbon stock. The challenge will be proving that the succession was not occurring in the absence of management of wild animals —pre-management monitoring is essential and should be coupled with monitoring of control plots (not subjected to the same management actions) while sequestration is increased for the rest of the forest.

**Table 2** Estimated number of monitoring plots (*N*) needed to obtain a given level of precision (percentage of the mean) in carbon stock estimates (including live stem and CWD pools only) for different forest types. Forest types are based on Wiser et al. (2011). Indicative mean carbon stocks and standard deviations are calculated from the LUCAS natural forest dataset. Table reprinted from Holdaway et al. (2012)

| Vegetation type | Carbon stock (Mg C/ha) | | | Sample size (*N*) required for varying levels of accuracy | | | |
|---|---|---|---|---|---|---|---|
| | *N* Plots | Mean | SD | *N* (5%) | *N* (10%) | *N* (20%) | *N* (30%) |
| **Shrubland** | | | | | | | |
| Mānuka shrubland | 31 | 42 | 32 | 923 | 231 | 58 | 26 |
| *Schoenus–Dracophyllum* subalpine shrubland | 19 | 40 | 62 | 3666 | 917 | 229 | 102 |
| *Ozothamnus–Dracophyllum* montane shrubland | 31 | 20 | 19 | 1330 | 333 | 83 | 37 |
| Matagouri shrubland | 28 | 12 | 18 | 3493 | 873 | 218 | 97 |
| Sweet vernal – Yorkshire fog successional shrubland | 61 | 33 | 35 | 1699 | 425 | 106 | 47 |
| Wheki–mānuka shrubland/low forest | 26 | 49 | 38 | 914 | 228 | 57 | 25 |
| **Broadleaved–podocarp forest** | | | | | | | |
| Kāmahi–podocarp forest | 105 | 236 | 138 | 525 | 131 | 33 | 15 |
| Māhoe forest | 52 | 182 | 121 | 674 | 168 | 42 | 19 |
| Silver fern – māhoe forest | 79 | 149 | 94 | 612 | 153 | 38 | 17 |
| Tawa forest | 69 | 235 | 139 | 540 | 135 | 34 | 15 |
| **Broadleaved forest** | | | | | | | |
| Kānuka forest and tall shrubland | 51 | 66 | 49 | 857 | 214 | 54 | 24 |
| **Beech–broadleaved–podocarp forest** | | | | | | | |
| Kāmahi–Southern rātā forest and tall shrubland | 49 | 166 | 90 | 451 | 113 | 28 | 13 |
| Kāmahi forest | 48 | 287 | 149 | 417 | 104 | 26 | 12 |
| Pepperwood–hardwood forest and successional shrubland | 58 | 242 | 124 | 402 | 100 | 25 | 11 |
| Kāmahi–silver fern forest | 46 | 199 | 97 | 371 | 93 | 23 | 10 |

| Vegetation type | Carbon stock (Mg C/ha) | | | Sample size (*N*) required for varying levels of accuracy | | | |
|---|---|---|---|---|---|---|---|
| | *N* Plots | Mean | SD | *N* (5%) | *N* (10%) | *N* (20%) | *N* (30%) |
| **Beech–broadleaved forest** | | | | | | | |
| Silver beech–broadleaf forest | 64 | 237 | 119 | 391 | 98 | 24 | 11 |
| Silver beech – red beech – kāmahi forest | 94 | 281 | 93 | 168 | 42 | 10 | 5 |
| Marbleleaf–pepperwood–wineberry forest and successional shrubland | 22 | 184 | 125 | 709 | 177 | 44 | 20 |
| Kāmahi–hardwood forest | 80 | 270 | 157 | 521 | 130 | 33 | 14 |
| **Beech forest** | | | | | | | |
| Black/mountain beech forest | 21 | 150 | 60 | 245 | 61 | 15 | 7 |
| Silver beech – red beech – black/mountain beech forest | 19 | 275 | 106 | 229 | 57 | 14 | 6 |
| Black/mountain beech forest (subalpine) | 20 | 166 | 40 | 87 | 22 | 5 | 2 |
| *Hoheria glabrata–Olearia ilicifolia* – Hard fern low forest and successional subalpine shrubland | 25 | 89 | 76 | 1110 | 277 | 69 | 31 |
| Black/mountain beech – silver beech forest/subalpine shrubland | 63 | 186 | 85 | 322 | 80 | 20 | 9 |
| **All forest and shrubland (including unclassified plots)** | 1239 | 172 | 137 | 981 | 245 | 61 | 27 |

### 3.2.2 How could wild animal control potentially benefit palatable broadleaved-hardwood forests both during succession and at maturity?

Across the studies, we have found no effect of wild ungulates on total above-ground carbon levels in broadleaved-hardwood forests. This has been investigated through:

- A nationwide comparison of control versus exclosure plots in broadleaved-hardwood forests (where red deer (*Cervus elaphus*) and other ungulates have been excluded)
- Repeated sampling (over 30 years) of a broadleaved-hardwood forest in Te Urewera National Park that is known to have high deer (Cervidae) densities
- Installation of new exclosures around canopy tree species within successional kanuka forest
- In-depth examination of a mortality event that may mimic that occurring with high densities of browsing mammals (more likely possums (*Trichosurus vulpecula*), in this instance) in a mature broadleaved-hardwood forest

Nonetheless, we do find evidence that browsing animals are changing the species composition in the understorey of broadleaved-hardwood forests (Peltzer et al. in prep.) and that this may have implications for longer-term successional trajectories (Richardson et al. 2014). Related work has shown that ungulate exclusion increases the recruitment of palatable species where forests have been subject to recent disturbance (Mason et al. 2010) and that leaf palatability increases during broadleaved succession (Mason et al. 2011). Therefore, it is reasonable to expect that more intensive control of ungulates could lead to an enhanced rate of succession in these forests at the multi-decadal timescale.

When seedlings of canopy species were fenced in Te Urewera National Park, the fencing resulted in significantly increased height growth and significantly reduced mortality, especially of highly palatable species. When permanent plots in a successional kānuka community in the same location were remeasured c. 30 years after being declared ‘successional’, it was found that kānuka was being replaced by tree ferns rather than canopy species (Richardson et al. 2014). Even though the high-carbon kānuka was being replaced by tree ferns that contain much less carbon per individual, there was no statistically significant decrease in above-ground carbon per unit area. Although evidence that deer browsing is causing this stalling of succession is merely circumstantial, a shift in species composition towards tree ferns could eventually lead to a decrease in biomass (along with associated carbon loss) at the centennial timescale. More time (decades to a century) will be required to assess whether this is happening, and maintenance of our new paired exclosure plots for canopy seedlings will be required to test whether ungulate browsers are responsible for the change in species composition.

Across all the studies in broadleaved-hardwood forests, we found no evidence that ungulate control either protects existing carbon in mature broadleaved-hardwood forests or increases the carbon stocks (and associated sequestration rates) of regenerating broadleaved-hardwood forests, in the short term (decadal timescale). However, we did find evidence that ungulate control increases the abundance of seedlings and saplings of palatable tree species, which increases the carbon in understorey individuals, and improves canopy seedling growth and survival (Peltzer et al. in prep.). It seems likely that eventually this would translate into improved carbon stocks in a future canopy. However, stand-level carbon is driven by the dynamics of a few large individual canopy trees (Coomes et al. 2002; Slik et al. 2013). It is possible, therefore, that as long as a few individuals are able to achieve canopy status, carbon stocks may be maintained long term, even with browsing pressure and a concomitant reduction in the presence of palatable species. In other words, in the systems studied, ecosystem function (when indicated solely as carbon) has been maintained in spite of reduced functional diversity and species richness.

### 3.2.3 Are some tree species protective against carbon loss even in the event of their widespread mortality?

We investigated whether decay-resistant southern rātā (*Metrosideros umbellata*) may help protect mature forest against net carbon loss during a widespread mortality event. A large-scale dieback of many canopy tree species began in the Hokitika River catchment in 1950. Many authors suggest that this dieback was caused by possum browsing (e.g. Batcheler 1983; Rose et al. 1992). We found that although the dieback resulted in a significant loss of carbon, it was substantially ameliorated where rātā was previously abundant in the canopy (Mason et al. 2013b). The decay rate of southern rātā is much slower than that of kāmahi, such that even when these trees die, the majority of the carbon is still held in the dead spars and released extremely slowly. This was the only study across the WACEM projects that could be relevant to control of possums.

Implications for short-term management might be that forests dominated by fast-decaying species become prioritised for animal control where there is evidence that these forests suffer mortality caused by browsing animals. Herbivore impacts are highly context-dependent but a recent study has suggested that mortality from possum browse can be predicted as a function of foliar cover and herbivore intake (Holland et al. 2013). Possums are thought to consume between 5.26 kg dry mass/yr (in high quality pasture; Cowan 2007) and c. 80 kgDM/yr (in indigenous forest; Nugent et al. 1997). Therefore, Burrows et al. (2008) estimated total possum consumption to be 0.48–10.3 Mt $CO_2$e across all of New Zealand. Much of this consumption occurs in grassland adjacent to forests and shrubland (sensu Cowan 2007), rather than solely conservation land. Nonetheless, total impacts on carbon are complex given that non-palatable species tend to exhibit compensatory growth when competition from palatable species is reduced (Wardle et al. 2001). Further, Wardle et al. (2001) suggest that biomass (= 0.5 carbon) response to herbivory depends on other habitat characteristics such as nutrient status of the system, carbon availability and macroclimate. This is certainly consistent with the study on ungulate exclusion, discussed above, where no net carbon impact was observed despite the increase in palatable understorey species.

### 3.2.4 Are some impacts of wild animal control more apparent below ground than above?

Existing research suggests that herbivores can play a critical role in mediation of above- and below-ground interactions (Wardle & Bardgett 2004). Herbivore-triggered shifts to less palatable species may result in more recalcitrant litter, and thus a net increase in soil carbon (Wardle & Bardgett 2004). However, it appears as if there is no net increase in soil carbon unless the litter is relatively rapidly incorporated within the mineral soil (Prescott 2010). Again, Wardle et al. (2001) show that feedbacks between plant palatability, litter composition and increases in soil carbon are highly context-dependent.

Through a nationwide study of paired exclosure and control plots in broadleaved-hardwood forests, we examined whether deer impact below-ground carbon stocks either positively (through increased quantities of unpalatable litter) or negatively (through removing carbon present in forest floor litter). When litter, fine woody debris and soil carbon pools were compared between exclosures (where deer had been excluded from broadleaved-hardwood forests) and control plots, we found no difference in carbon stocks (Peltzer et al. in prep.). We have also provided baseline data on carbon stocks in these pools for the Thames Coast Flood Protection Project. If animal control is maintained in this catchment at the multi-decadal timescale, it would be worth reassessing carbon stocks at the same timescale to see if there are any long-term changes.

### 3.2.5 What are the ways that rates of establishment of woody succession could be enhanced through management?

Where non-forest lands are currently grazed, most carbon gain achieved through management for natural regeneration can be claimed as ‘additional’ carbon. This is because livestock grazing can suppress much of the natural regeneration of woody species on such lands. Periodic clearance of the residual regeneration that occurs during periods when livestock may be temporarily absent, or reduced in numbers due to low farm commodity prices, is generally carried out by landowners when farm commodity prices are again high (Trotter et al. 2005). The net result is that at regional spatial scales and multi-decadal time frames, the area of indigenous shrubland—the precursor to indigenous forest regeneration in most areas—has been, on average, static or declining (Barringer et al. 2006). Therefore the baseline sequestration rate against which management must be measured is generally very low and often zero.

In ecological terms, a change in land use from low intensity pastoral agriculture to regenerating indigenous forest represents the addition of another layer of carbon, namely that contained in tree stems. The total increase is significant (c. 7 t $CO_2$e/ha increased to c. 370 t $CO_2$e/ha for mixed indigenous shrubland; Tate et al. 1997) and is undoubtedly the easiest way to increase carbon stocks by a measureable amount. We have shown that mean annual sequestration rates during this type of conversion can be 2.7 ± 1.5 t $CO_2$e/ha and 2.0 ± 1.0 t $CO_2$e/ha for exotic and mānuka shrublands, respectively (Carswell et al. 2009). We have also shown that when these successions proceed to taller forests, these rates of carbon sequestration are maintained. For example, Carswell et al. (2012) showed that a kanuka–red beech succession and a coastal broadleaved succession sequestered c. 2.3 Mg C/ha/year over the first 50 years of succession. Mean above-ground carbon stocks were 148 ± 13 Mg C/ha for kanuka–red beech forests and 145 ± 19 Mg C/ha for tall coastal broadleaved forests after at least 50 years of succession.

Adding seed does not necessarily enhance the rates of woody succession (Burrows et al. 2013; Paul et al. 2013). Addition of seed to three grassland sites in the Marlborough Sounds (kānuka and tauhinu (*Ozothamnus leptophyllus*) seed; Burrows et al. (2013)) and five grassland sites throughout the rest of New Zealand (kānuka seed) resulted in an increase in kānuka seedling establishment at only the Northland site (see Paul et al. (2013) for details). These results are consistent with existing literature suggesting that recruitment of woody species within grasslands depends on the contemporaneous removal of a range of filters, seed availability being only one (Meurk et al. 1989, 2002; Wiser et al. 1997; Norton 2009; Dickie et al. 2010). Other filters include competition (especially from exotic grasses), insufficient or poorly timed rainfall, lack of suitable sites for seedling establishment (especially within pastures with turf-forming ground cover), herbivory, pathogens and also a lack of appropriate mycorrhizal symbionts. We have previously confirmed that a lack of mycorrhizal inoculum can limit mountain beech (*Fuscospora cliffortioides*) establishment within grasslands (Dickie et al. 2012) and shown that the addition of unsterilised O-horizon soil from beneath kānuka can increase the biomass of kānuka seedlings grown in sterilised soil (Davis et al. 2013). However, this growth improvement does not appear to result from increased ectomycorrhizal infection—rather, Davis et al. (2013) hypothesised that the observed increase in biomass resulted from increased arbuscular mycorrhizal infection. Field testing of these results is required.

Burrows et al. (2013) examined whether grazing animals (sheep at high stocking rates) could be used to improve germination success through creation of bare patches that would increase the safe sites for seedling establishment and also reduce the competition from exotic grasses during the early establishment phase of woody seedlings. We found no evidence that grazing animals improved the success of woody species within grasslands, although we did observe increased recruitment of kānuka on bare patches of soil created by other means. We also examined whether browsing animals (goats *Capra hircus*) could assist through reducing competition from exotic grasses, but the goats appeared to preferentially browse the woody species where offered a choice between grasses and shrubs. Again, we found no evidence that a browsing ungulate could improve the establishment success of woody species within grasslands. Although restoration plantings do not replicate a natural succession (Davis & Meurk 2001), planting of nursery-raised stock is the most reliable method for introducing indigenous woody species to grasslands (Davis et al. 2009; Paul et al. 2013) where natural regeneration is not proceeding by itself. We suggest that the carbon stored in planted forest on Tiritiri Matangi Island could be compared with that in the naturally regenerated stands of forest to directly test the effect planting has on carbon sequestration during afforestation for indigenous forest.

We conclude that the single cheapest and most efficient management action for enhancing carbon sequestration is choosing a non-forested site that will rapidly revert to indigenous forest (i.e. with relatively mesic conditions and ample local seed source). Nature should then be supported by preventing fires and removing all grazing animals whether wild or domestic. This will result in measureable and large per-hectare increases in carbon supported by money from carbon revenue, assuming that the global carbon price increases.

A recent economic study suggests that carbon can outcompete grazing of beef or lamb on hill country in the Gisborne District (Funk et al. 2013). Because the land has a relatively low carrying capacity (i.e. can only support a low number of stock units per hectare), the economic projections suggest that revenue generated over 70 years of regeneration could be c. $900 million above that of grazing. However, these projections assume initial, and subsequently increasing, carbon revenue of $15 per tonne of $CO_2$e—this price has not been returned to owners of regenerating indigenous forests for at least 2 years. Currently, the price is hovering around $3 per tonne of $CO_2$e, although we note a proposal to differentiate permanent units from standard Emissions Trading Scheme units from forestry may improve the pricing for owners of permanent regenerating forests.

Again, we acknowledge that exotic shrubland species already play a significant role in kick-starting indigenous successions in many landscapes (e.g., Wilson 1994). Further research is required on whether exotic conifers can also play a role given that they appear able to colonise some sites distant to indigenous forest tree sources. As above, we note that there is currently a lack of evidence for conifer facilitation of tall indigenous tree establishment and there may also be a longer-term trade-off to total carbon that could have been accumulated through a solely indigenous succession (Meurk &Hall 2006). It is likely that conifer success interacts with animal grazing given that domestic stock grazing is currently the major means of wilding control in New Zealand, targeted specifically at preventing conifer establishment in grasslands (Ledgard 2009).

### 3.2.6 Do tall-tussock grasslands recover carbon following removal of grazing?

Burrows et al. (2008) concluded that all grasslands are particularly susceptible to browsing by introduced mammals because they provide high quantities of easily accessible food for grazing and browsing animals. Coomes et al. (2006) report strong recovery of alpine grasslands following intensive control of deer and chamois (*Rupicapra rupicapra*) in the 1960s. However, many grassland ecosystems also have a history of agricultural grazing and it has been shown that grassland recovery is strongly influenced by its legacy of degradation (Rose & Platt 1992; Sparrow et al. 2003; Standish et al. 2009). Legacies include management through fire, loss of topsoil, depletion of natural seed sources, change in dominant species pool, and physical changes to the environment. The recovery of previously farmed montane grasslands appears to be primarily driven by the tussock species, overall recovery occurring where the tussocks regenerate and no recovery if tussocks do not increase (e.g. Duncan et al. 2001; Norbury et al. 2002; Mark & Dickinson 2003). Even when tall tussock does successfully regenerate following cessation of grazing, soil carbon does not necessarily increase, even at the multi-decadal timescale (e.g., Basher & Lynn 1996).

We examined recovery of carbon stocks following removal, some decades prior to the study, of grazing herbivores (sheep). We observed highly variable rates of recovery (Holdaway et al. 2014b). On the site where *Chionochloa* or woody shrub recruitment increased there was a significant increase in carbon stocks following grazing removal. However, on the site where there was little recruitment of either *Chionochloa* or woody shrubs following the removal of grazing, carbon stocks continued to decline. Decline on this site and those still subject to grazing was due to loss of the short tussock *Festuca novae-zelandiae*. Animal removal, alone, was not able to halt its decline. Coomes et al. (2003) suggest many reasons why browsing effects on vegetation may not be reversible. These include high susceptibility of palatable species even to very low densities of browsers, niche compensation by non-palatable species, local extinction of seed sources, and compensations between multiple browsing animals and other introduced plant species. Further research is recommended to investigate why woody recruitment successfully occurred on one of the retired sites. Comparison with other sites retired from grazing is also recommended.

Although tussock grasslands contain significantly more carbon (c. 18 t C/ha; Payton & Pearce 2009) than predominantly agricultural grasslands, woody succession is required if increased indigenous carbon is the management goal. Examining successions from abandoned pastures in New Zealand, Standish et al. (2009) concluded that if the pastures are not well connected with remnant vegetation there will be a low probability of native species arrival and a high probability that exotic species will establish instead. This is becoming all too apparent in South Island grasslands where exotic conifers are becoming increasingly dominant (Hunter & Douglas 1984; Allen & Lee 1989). It seems likely that we are now locked in to novel successions/ecosystems in the high country at landscape-scale. We recommend further research on the implications of these novel pathways for longer-term carbon storage.

# 4 Inventory of outputs and key findings

## 4.1 DOC SRU (Investigations 4023, 4024, 4025). 'Synthesise existing C stock information and prepare Wild Animal Control for Emissions Management research programme'

### 4.1.1 Objectives

Cabinet directed the Department to establish the Wild Animal Control for Emissions Management (WACEM) programme to research the potential for controlling wild animal herbivores in indigenous forests to generate forest carbon credits (Cabinet Minute CBC Min (07) 16/1 refers).

- DOC 4023: To prepare a report for formal publication that will synthesise available carbon stock information on indigenous forests, shrublands and grasslands on public conservation land managed by DOC.
- DOC 4024: To prepare a document that synthesises the significant amount of available information and knowledge about the ecology of New Zealand's indigenous vegetation cover and the effects of introduced wild animal herbivores.
- DOC 4025: To propose the programme of fieldwork required for a research programme that would quantify the effects of wild animal control on total carbon stocks in indigenous vegetation.

### 4.1.2 Outputs

Carswell, F.E.; Mason, N.W.H.; Davis, M.R.; Briggs, C.W.; Clinton, P.W.; Green, W.; Standish, R.J.; Allen, R.B.; Burrows, L.E. 2008: Synthesis of carbon stock information regarding conservation land. Landcare Research Contract Report LC0708/071, prepared for the Department of Conservation, Wellington (unpublished).

Burrows, L.E.; Peltzer, D.A.; Bellingham, P.J.; Allen, R.B. 2008: Effects of the control of introduced wild animal herbivores on carbon stocks. Landcare Research Contract Report LC0708/087, prepared for the Department of Conservation, Wellington (unpublished).

Allen, R.B.; Carswell, F.E. 2008: A proposed research plan for the Wild Animal Control for Emissions Management (WACEM) Programme. Landcare Research Contract Report LC0708/176, prepared for the Department of Conservation, Wellington (unpublished).

Holdaway, R.J.; Burrows, L.E.; Carswell, F.E.; Marburg A.E. 2012: The potential for invasive mammalian herbivore control to result in measurable carbon gains *New Zealand Journal of Ecology 36*: 252–264.

### 4.1.3 Overview of each output (abstract from the report/published manuscript)

#### Carswell et al. 2008

Information on current carbon stocks on public conservation land administered by the Department of Conservation (DOC) was synthesised from existing literature. The current carbon stock in all five pools (i.e. above- and below-ground vegetation, litter, coarse woody debris and soil carbon) was estimated to be 2396 Mt (or 8785 Mt $CO_2e$), across c. 8 Mha of New Zealand. About 80% of the existing stock on conservation land is in indigenous forest or forest-shrubland vegetation classes, and about 50% was in the soil pool.

Potential carbon storage on conservation land was predicted from potential indigenous vegetation cover. Predicting the potential carbon stock is difficult because of the absence of empirical data on actual rates of carbon accumulation during the process of succession from forest-shrubland to forest. Our results suggest that an additional c. 190 Mt of carbon (or 698 Mt $CO_2e$) could be stored, largely through the conversion of existing indigenous forest-shrublands to forests and of shrublands and grasslands to forests, over periods from a few decades to over 300 years. Our calculation has assumed that soil carbon remains constant during this conversion and the major increases are therefore in the above- and below-ground biomass pools of carbon, with some additional increases in litter.

We are more confident in our prediction that c. 268 Mt $CO_2e$ could be sequestered through afforestation/reforestation (A/R) of the 400,000 hectares of conservation land estimated to have been ‘non-forest’ as at 31 December 1989. Carbon gains are certain to occur in favourable ‘non-forest’ areas through relatively inexpensive management actions such as exclusion of domestic stock and low-level wild animal control. The potential increase in carbon storage on this subset of conservation land is almost six times larger than the projected 45.5 Mt $CO_2e$ excess of emissions over New Zealand’s target for the first commitment period (2008–2012). Nonetheless, the total potential increase through A/R on conservation land could not be realised over CP1. Further work is required to accurately determine areas of conservation land that are eligible for Kyoto A/R.

While indigenous tree species have lower rates of sequestration compared to some exotic conifer species, they could potentially deliver greater total sequestration in the long term, as well as other desirable biodiversity and landscape outcomes. The long-term ecological, landscape and carbon consequences of proposals to use exotic tree species to sequester carbon, particularly on conservation land, need to be assessed very carefully to avoid undesirable outcomes.

The best ways that conservation land can be managed to help New Zealand to meet its obligations under Article 4.1(d) of the United Nations Framework Convention on Climate Change to conserve and enhance carbon sinks and reservoirs are to (1) minimise losses of carbon from conservation land (e.g. prevention of forest fire) and (2) increase carbon stocks by establishing new forests, particularly through afforestation of grasslands. Unlike gains in carbon from management of existing forests, A/R gains per hectare are large, relatively easy to quantify over short-time periods (5–10 years), and are certain to occur in favourable areas.

The biggest threats that could cause the loss of carbon stored on conservation land are natural disturbances such as vulcanism and fire (particularly given the predicted increase in dryness in eastern areas as the climate changes). There is also a risk that elevated temperatures may push forests in New Zealand into states of net carbon loss if soil respiration is substantially increased.

### Burrows et al. 2008

Very few studies in New Zealand have estimated total C stocks in indigenous vegetation and even fewer have identified any C changes caused by pest animal herbivores. There is an urgent need for primary data that will provide the basis for identifying the interactions between vegetation biomass dynamics, total C stocks and pest herbivore impacts. Total C stocks are not just a function of foliage consumption or tree mortality—they are also a function of decay rate of dead plants, growth rate of the remaining vegetation and, importantly, changes in soil C pools. Therefore, it is incorrect to generally assume that the sometimes obvious changes in above-ground biomass as a result of wild animal herbivory equate to significant changes in total carbon stock.

Effects of pest herbivores or their control in short time frames of the first Commitment Period of the Kyoto Protocol (CP1, 2008–2012) and immediately post-CP1 (assumed in this report to be 2013–2020) are two orders of magnitude smaller than the effects of natural disturbance (e.g. earthquakes) and past human influences (e.g. deforestation). Longer-term effects of pest herbivores or their control on biomass C and total C stocks (and biodiversity) may be profound, especially where vegetation succession is enabled towards more woody vegetation states—but the effects cannot be predicted with confidence.

Three situations are identified where herbivore control is likely to have a substantial effect on total C stocks in both the short- and long-term. Each situation will require additional research to confirm and quantify the C potential that would result (a research plan is being prepared in a companion report). They are:

1. Alpine and subalpine tussock grasslands, which may lose biomass C due to increased herbivory as a result of reduced herbivore control (e.g. through reduced commercial deer harvesting)

2. Reverting shrublands (and mixtures with grasslands), which could become forest more quickly with additional herbivore control

3. Broadleaved-hardwood forests with a high proportion of biomass in palatable tree and small-tree species, in which relatively small increases in total C stocks resulting from herbivore control could be significant overall because of the large area these forests occupy.

#### *General effects of herbivores and their influence on C stocks*

There is very limited quantitative information available about actual or potential total C stocks and C sequestration rates for indigenous grassland, shrubland and forest ecosystems in New Zealand. The longer-term effects on soil C in all systems are unknown. The relationships between invasive species and C dynamics are also only now being investigated internationally. Deer, goats and possums have direct effects on C stocks and sequestration, e.g. consumption of biomass. Any direct consumption effect is relatively small compared with total biomass stocks, and will not be consistent or evenly distributed across all vegetation classes. There is increasing recognition in the international literature that indirect herbivore effects through, for example, the cycling of nutrients or C, have strong forest ecosystem-level impacts via below-ground/above-ground linkages and feedbacks. There may be a time lag in any response to herbivore control, and many vegetation classes will show no short-term (i.e. to 2020) effect, e.g. beech forest, kanuka shrublands. It may take years after a control operation before a biomass response is measurable; i.e. the response may be occurring but will be difficult to quantify in the CP1 and immediately post-CP1 period. Estimation of biomass effects resulting from existing control operations need to be determined against a background of natural vegetation disturbance dynamics and often large biomass fluctuations.

However, the long-term effect of forest successions and future forest types on biomass C and total C (and biodiversity) may be profound.

### *Likely short-term effects on carbon stocks of maintaining the existing level of wild animal herbivore control operations*

Existing levels of deer, goat and possum control are expected to have little effect on carbon stocks at the national scale. However, C stocks could significantly increase at sites where control enables shrubland succession, or in forests where control is intensive, includes multiple herbivores, palatable species widely dominate, and growth and decomposition are rapid. Deer, goats and possums will continue to expand their distributions. Goats have by far the greatest potential to occupy more land and increase their population size.

### *Likely short-term effects of reducing the existing level of wild animal herbivore control operations in the short-term*

Reducing the existing level of deer, goat and possum control will result in an increase in the abundance of deer, goats and possums at some locations. Goat populations would increase quickly and likely expand their range. Deer numbers could be expected to increase in more remote forest and grassland areas, and to a lesser extent in subalpine grasslands. Possums would increase at some locations. In the short term there would be a small loss of live above ground biomass C in some grassland types, and in forests and shrublands dominated by broadleaved palatable species. The grassland effects could amount to as much as c. 0.715 Mt C lost during ‘CP1’ and 1.43 Mt C lost during the post-CPI period to 2020. The potential forest effect is unknown, but there is potential for a small decrease in live above ground biomass C change (c. 1% over a number of years) in broadleaved-hardwood forests (although it would be difficult to detect and need to assess compensatory growth of trees).

### *Likely short-term effects of implementing new wild animal herbivore control operations*

New control operations of herbivores in subalpine tussock grasslands, some shrublands (e.g. wet heaths), or many existing forest areas are unlikely to influence C stocks, because current control efforts or poor habitat suitability maintain low animal numbers. However, modest above-ground increases in biomass C by new control of goats and deer could be achieved by new control operations that will (1) enable successions from grassland (including pasture) to woody vegetation growing in productive (i.e. moist, fertile) sites, or (2) enhance stalled woody plant successions in some reverting shrublands. Together, these could cover c. 0.22

Mha of public conservation land. It is unknown whether possum control will result in changes in biomass in grasslands, some shrublands, and forest understoreys because their effects are not known or difficult to distinguish from other herbivores.

### *Comparison of potential with New Zealand's commitments under the Kyoto Protocol*

New Zealand has a predicted national shortfall of 21.7 Mt $CO_2$e to meet its CP1 commitment under the Kyoto Protocol (http://www.treasury.govt.nz/government/liabilities/kyoto). The shortfall is approximately 0.25% of the estimated 8,785 Mt$CO_2$e in all five carbon pools on public conservation land (Carswell et al. 2008) (i.e. above- and below-ground vegetation, litter, coarse woody debris and soil carbon). Carswell et al. (2008) recommended: 'the best ways that public conservation land can be managed to help New Zealand to meet its obligations under Article 4.1(d) of the United Nations Framework Convention on Climate Change to conserve and enhance carbon sinks and reservoirs are to: (i) minimise losses of carbon from conservation land (e.g. prevention of forest fire); and (ii) increase carbon stocks by establishing new forests, particularly through afforestation of grasslands. Unlike gains in carbon from management of existing forests, afforestation/reforestation gains per hectare are large, relatively easy to quantify over short time periods (5–10 years), and are certain to occur in favourable areas.' Control of wild animal herbivores will be one important consideration of managing new afforestation/reforestation. However, this review of the effects of the control of introduced wild animal herbivores on carbon stocks has concluded that:

1. There is potential for significant increases in biomass C as a result of wild animal herbivore control on public conservation land owing to the large existing biomass stocks, particularly through some situations where herbivore control is likely to have substantial effect on biomass (and total) C stocks in both the short and long term.
2. It is not currently possible to quantify the potential over the national network of public conservation land.
3. It would not be possible to realise that potential in the short-term as the timescale is too short given the nature of processes involved, the very small change in C pools that will need to be quantified, and because our knowledge is not yet robust enough to assess or optimise total C stock gains from herbivore management. Clearly models for testing scenarios for investigating forest development and herbivore control will contribute to this understanding.

### **Allen & Carswell 2008**

The purpose of this report was to propose a suite of research projects to the Department of Conservation that would quantify the effects of wild animal herbivores on total carbon stocks in indigenous vegetation. The authors of this report were instructed to attempt to ensure that the research will:

- Be developed, where appropriate, on existing wild animal herbivore control projects

- Describe how data from the Land Use and Carbon Analysis System (LUCAS) or other national datasets could be used to analyse the effects of wild animal control at the national scale and, if so, management requirements for these datasets
- Be achievable within the WACEM programme budget, while taking account of any aligned co-funding (e.g. Ecosystem Resilience OBI)
- Assess changes in key biodiversity values for carbon sequestration projects

The research topic area is currently full of speculation, with little empirical data. The proposed projects focus on collecting and analysing new field data, and reporting the results, for a range of study areas where existing knowledge and observations suggest carbon stocks would respond quickly to the effects of wild animal herbivores or their control.

We propose research on one general project, soil carbon, relevant to all other WACEM projects as well as research in three priority ecosystems (comprising five projects – see Table 1 above for a summary of work programme), over the period 2008/09–2011/12:

*Soil carbon:* Soil carbon is an important pool to measure in all sequestration projects. **Project 1** will develop a protocol for sampling soil carbon.

*Broadleaved-hardwood forest:* Changes in indigenous forest carbon pools, as a result of new control of wild animal herbivores, is most likely to occur in broadleaved-hardwood forests. **Project 2** will investigate changes in the soil carbon pool by sampling and analysing samples from many ungulate exclosures that have already been established in broadleaved-hardwood forests. **Project 3** will determine the effect of a recently initiated, ongoing, possum and goat control programme in such forests.

*Mixed successional grassland–shrublands (including kanuka and manuka):* The early phases of woody succession could result in significant carbon sequestration at the national scale (particularly in productive environments) and could generate compliance carbon credits (i.e. under the Kyoto Protocol, or the proposed New Zealand Emissions Trading Scheme). **Project 4** will evaluate the effect of herbivory on cost-effective methods for establishing woody species in productive grasslands. **Project 5** will investigate how carbon sequestration rates in shrublands are influenced by wild animal herbivores during succession to forest.

*Tussock-grasslands:* Indigenous tussock-grasslands are an extensive vegetation type. In tall-tussock (*Chionochloa*) grasslands a relatively high proportion of live biomass can be browsed by ungulates. As a consequence, increases in wild animals with a reduction in commercial hunting pressure, or decreases in sheep numbers through the tenure review process, have the potential to significantly affect carbon stocks. **Project 6** will investigate the influence of sheep removal on carbon sequestration in tall-tussock grasslands – this will indicate the benefits of herbivore management more generally.

The projects were specifically targeted to contribute to a matrix of questions that need to be answered if all facets of carbon management are to be understood. All questions that need to be answered are given in Table 1.

#### Holdaway et al. 2012

Invasive mammalian herbivores (e.g. deer, feral goats and brushtail possums; hereafter 'herbivores') are widespread throughout New Zealand and their control is important for conservation. In addition to known biodiversity benefits, it has recently been suggested that herbivore control could lead to measureable carbon gains when aggregated across a large area of conservation land. However, a significant amount of uncertainty exists regarding the potential effects of herbivore control on carbon, and the practicalities of successfully implementing such projects. This paper provides a general basis for managers and ecologists to design scientifically robust herbivore control projects for carbon gain in New Zealand. Although there are few direct data on changes in carbon sequestration rates following herbivore control, the data that are available suggest that effect sizes are likely to be small in magnitude, variable in direction, and to occur primarily though complex indirect mechanisms. The largest positive effects of herbivore control (carbon sequestration rate of 1–2 Mg C $ha^{-1}$ $year^{-1}$) are likely to occur in localised areas of highly palatable early-successional vegetation and high herbivore densities where control initiates rapid development of woody vegetation. Project location is therefore critical in determining the potential for carbon gain in herbivore control projects. A power analysis reveals that the ability to monitor changes in carbon stock using plot-based methods is limited to effect sizes of > 0.5 Mg C $ha^{-1}$ $year^{-1}$, as smaller effect sizes would require an impractically large number of plots (i.e. >100), and the financial and carbon costs of implementing the control and quantifying the effects are likely to outweigh any potential carbon gains. Although more research is urgently required to quantify potential gains, and the mechanisms that underlie them, our findings suggest that with careful site selection, implementation, and monitoring, control of invasive mammalian herbivores could sometimes provide carbon gains in certain areas of New Zealand's indigenous vegetation.

## 4.2 DOC SRU (Investigation 4040) 'Carbon and conservation management in natural ecosystems'

### 4.2.1 Objective

- To quantify potential and actual carbon gains from conservation management.

### 4.2.2 Outputs

Carswell, F.E.; Burrows, L.E.; Mason, N.W.H. 2009: Above-ground carbon sequestration by early-successional woody vegetation. A preliminary analysis. *Science for Conservation 297*. Department of Conservation, Wellington.

Carswell, F.E.; Burrows, L.E.; Hall, G.M.J.; Mason, N.W.H.; Allen, R.B. 2012: Carbon and plant diversity gain during 200 years of woody succession in lowland New Zealand. *New Zealand Journal of Ecology 36*: 191–202.

Mason, N.W.H.; Carswell, F.E.; Overton, J.McC.; Briggs, C.M.; Hall, G.M.J. 2012: Estimation of current and potential carbon stocks and potential Kyoto-compliant carbon gain on conservation land. *Science for Conservation 317*. Department of Conservation, Wellington.

### 4.2.3 Overview of each output (abstract from the report/published manuscript)

#### Carswell et al. 2009

New Zealand's ratification of the Kyoto Protocol offers considerable opportunity for the sale of 'carbon credits' from newly created forests. Given that forests need to have arisen since 1989, shrublands currently represent the biggest potential for Kyoto-compliant carbon (C) gain on conservation land. Little published information currently exists on the potential annual gain of C (sequestration) in shrublands, or whether significant differences occur between different shrubland types. In this study, change in above-ground C stock was measured over an approximate 5-year period and consequent rates of C sequestration calculated for six exotic seral and six mānuka shrubland plots. Mean C stocks were 73 ± 13 t $CO_2$/ha for exotic shrubland plots and 76 ± 33 t $CO_2$/ha for mānuka plots. Mean sequestration rates were 2.7 ± 1.5 t $CO_2$/ha$^{/}$year and 2.0 ± 1.0 t $CO_2$/ha$^{/}$year for exotic and mānuka shrublands respectively. Given the wide range in transition points between shrub and forest stages of succession with respect to C gain estimated in this preliminary study, we recommend more detailed investigation of long-term consequences of C storage during secondary succession. We also note that widely divergent estimates of sequestration rate could be derived depending on the time interval between measurements of stock change. We tested a simplified shrubland measurement methodology and found that predicted C stocks did not differ significantly from those estimated using the standard method.

#### Carswell et al. 2012

Natural regeneration of new forests has significant potential to mitigate greenhouse gas emissions, but how strong is the potential biodiversity co-benefit? We quantified carbon accumulation and biodiversity gain during secondary succession of two New Zealand lowland forests. The rate of carbon sequestration was the same for the kanuka–red beech succession as for the coastal broadleaved succession (c. 2.3 Mg C ha$^{-1}$year$^{-1}$) over the first 50 years of succession. Mean above-ground carbon stocks were 148 ± 13 Mg C/ha for kanuka–red beech forests and 145 ± 19 Mg C/ha for tall coastal broadleaved forests after at least 50 years of succession. Biodiversity gain was investigated through the quantification of 'ecological integrity', which comprises dominance by indigenous species, occupancy of indigenous species or a group of species fulfilling a particular ecological role, and gain in representation of lowland forests within each ecological region. All components of ecological integrity increased with carbon accumulation for both successions. In addition, above-ground carbon stocks were correlated with the Shannon and Simpson diversity indices and species richness for both successions, suggesting that conventional metrics of diversity also show biodiversity gain with above-ground carbon during succession of recently non-forested lands to secondary forest.

#### Mason et al. 2012

There are strong financial incentives for accurate estimation of potential Kyoto-compliant carbon sequestration on conservation lands. Estimation of potential carbon stocks is complicated as it is unclear how the accuracy of estimates should be validated. One way of dealing with this uncertainty is to use several independent methods to estimate potential carbon stocks. Comparison of the estimates produced then provides an indication of uncertainty in predicting Kyoto-compliant carbon gain. The LUCAS vegetation survey plots were used in spatial predictive modelling to estimate current carbon stocks on Conservation land. Three independent methods were used to estimate potential carbon stocks, based on either (1) regression models of potential forest cover using present day forest survey data; (2) spatial models of disturbance-adjusted carbon stock values for LUCAS plots; or (3) a forest dynamics model that explicitly models changes in carbon. Kyoto-compliant lands were identified using the New Zealand Vegetation Cover Map. Conservation lands were estimated to currently contain a total of 2578 Mt of C (9461 Mt $CO_2$e) in vegetation and soil. The different models provided estimates of Kyoto-compliant carbon gain ranging from 63 to 186 Mt Carbon (231–682 Mt $CO_2$e) as a result of land use change from non-forest to forest land. This equates to 3–8 years of New Zealand's total greenhouse emissions, based on estimated levels for 2005. Reasons for the variation in estimates, implications of results and limitations of the methods used are discussed.

## 4.3 DOC WACEM 1 (Investigation 4080) 'Development of a protocol to measure soil carbon in the WACEM programme projects'

### 4.3.1 Objective

- To develop a protocol for sampling soil carbon that quantifies herbivore-induced changes, using paired–plot experiments.

Soil carbon is a major pool, particularly in grassland ecosystems, and it will be vitally important to measure and understand herbivore impacts on soil carbon storage. For all of the WACEM projects there is a need to define a robust soil carbon method for quantifying changes in soil carbon.

### 4.3.2 Output

Stevenson, B.J.; St John, M.G. 2008: WACEM Soil Carbon Sampling Protocol. Landcare Research Contract Report LC0809/077, prepared for the Department of Conservation, Wellington (unpublished).

### 4.3.3 Overview of output (abstract from the report)

Soil carbon is an important pool to measure in all sequestration projects. The objective of the first milestone within this suite of research projects was to develop a protocol for sampling soil carbon that quantifies herbivore-induced changes using paired-plot experiments.

We base sampling design on a randomly located 20 m × 20 m plot as the majority of WACEM projects will use this approach. Four equidistant 'microplot' centre points are marked within each quarter of the 20 m × 20 m plot in a manner comparable with MfE's Land Use and Carbon Analysis System (LUCAS), while making future re-sampling within the vicinity of original samples more consistent and convenient. Around each microplot centre point in a quarter plot, four equidistant locations (along a 1.0 m radius circle) will be sampled for fine woody debris (FWD), litter, fragmented/humic (FH) material and mineral soil to 300 mm depth in 100 mm increments. A standard plot has 16 sample points for organic and mineral soil collection and four for bulk density assessment. Similar samples from each quarter plot (e.g. all litter samples or all 0–100 mm depth mineral soil samples) will be bulked on a whole plot basis.

## 4.4 DOC WACEM 2 (Investigation 4081) 'Determine the effects of red deer on soil carbon in indigenous forests'

### 4.4.1 Objective

- To determine the effects of red deer on soil carbon in indigenous forests by sampling and analysing all carbon stocks in plots within and adjacent to exclosure plots within indigenous broadleaved-hardwood forests.

Wardle et al. (2001) suggest ungulates will change soil carbon storage in forests through modifying soil litter inputs. We know that soil carbon is a large pool in forests. Red deer often modify broadleaved-hardwood understoreys and this modification will influence litter inputs and organic matter decomposition. Changes in soil carbon occur over decadal timescales and existing exclosures provide an experimental design for testing impacts at such timescales. It is necessary to measure all carbon stocks so that any changes in soil carbon can be viewed within a context of ecosystem-level responses to deer exclusion. This project will provide indications of the potential effects of increased deer control on soil carbon in indigenous forests. We expect principles developed will apply to the impact of other wild animal herbivores.

### 4.4.2 Outputs

Kardol, P.; Dickie, I.A.; St. John, M.G.; Husheer, S.W.; Bonner, K.I.; Bellingham, P.J.; Wardle, D.A. 2014. Soil-mediated effects of invasive ungulates on native tree seedlings. *Journal of Ecology* 102: 622-631.

Peltzer, D.A.; Wardle, D.A.; Holdaway, R.J.; St. John, M.G.; Husheer, S.A.; Bellingham, P.J. In preparation for *Biological Letters*.

### 4.4.3 Major results

Some of the strongest evidence for determining the long-term effects of invasive herbivores on ecosystem properties and processes, including forest carbon sequestration, is through the assessment of ungulate exclosures. Exclosures represent a 'best case' scenario whereby animal numbers are maintained at zero density for several decades, and as such, present the strongest potential responses to animal management. Together with paired control (i.e. ungulate-affected) plots, these represent credible evidence for the impacts and response of forest ecosystems to ungulate abundance. We determined the effects of ungulates (deer) in mixed conifer–hardwood forests, using 26 paired c. 400 $m^2$ plots comprised of deer exclosures maintained for > 20 years and unmanipulated control plots. We used paired analyses to determine differences in all major pools of carbon in vegetation and soils to a depth of 30 cm (Fig. 3).

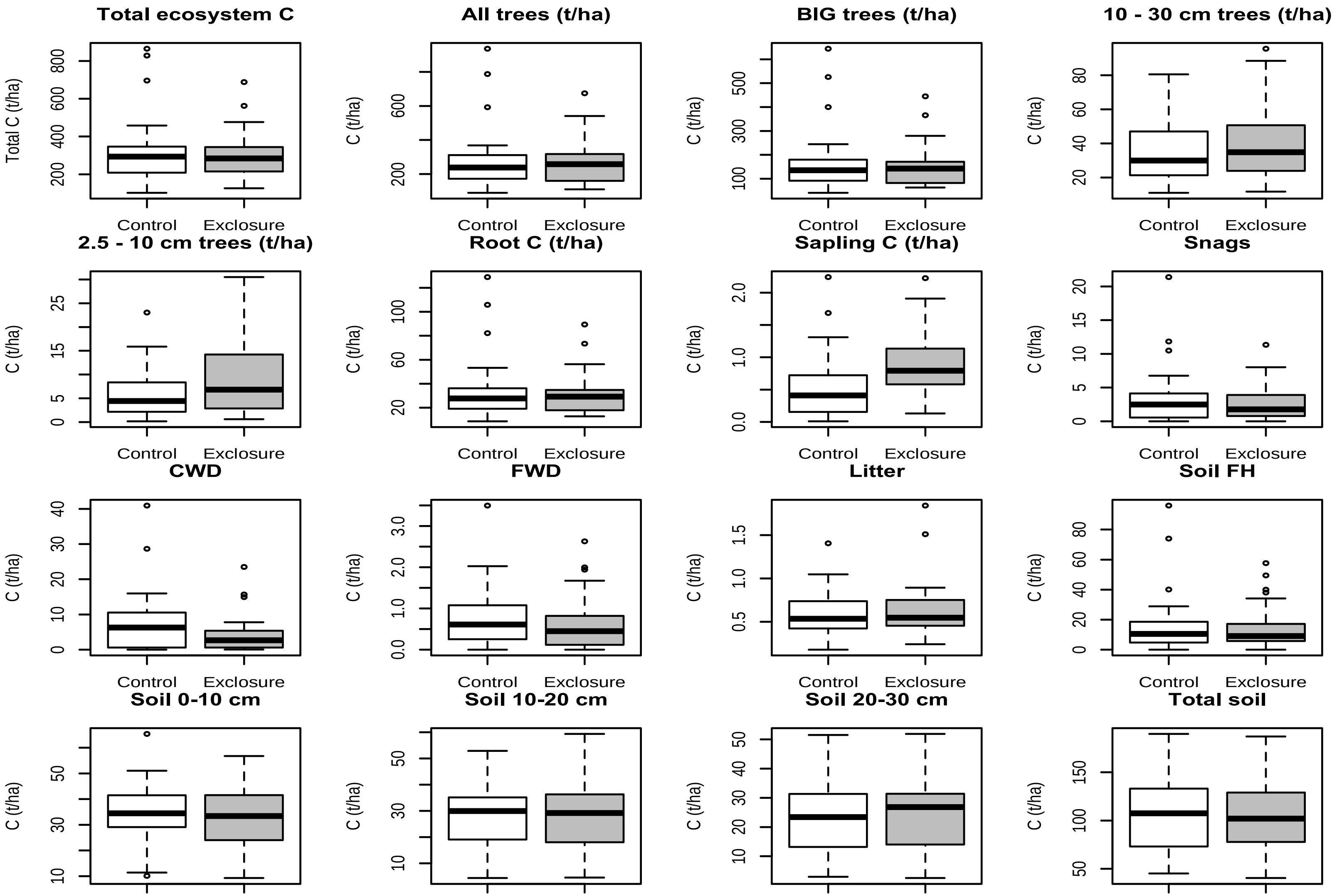

**Figure 3** Carbon (C) response to long-term ungulate exclusion for all pools considered. Greater pools of carbon occurred with ungulate exclusion only for small trees (2.5–10 cm diameter at breast height) and saplings. Coarse woody debris (CWD) was lower with ungulate exclusion.

Invasive ungulates did not consistently affect total ecosystem carbon. This is because the relatively large carbon pools—trees > 20 cm diameter at breast height, litter, and soil—were unresponsive to ungulate exclusion. However, with ungulate exclusion there was significantly more carbon in trees < 10 cm diameter and saplings. This is compatible with the view that ungulates exert their primary influence in the browsing tier, and that these effects are manifested in the regeneration and young tree components of tree populations at the decadal timescale. Although ungulates are having significant effects in this browse tier, this does not result in significant effects for total ecosystem carbon because saplings and small trees typically represent <10% of total forest carbon (Coomes et al. 2002). In addition, our previous work has demonstrated that the responsive pools are comprised of tree species having different functional traits (i.e. higher foliar nutrient contents but lower investment in structural or chemical defences; Mason et al. 2010). In summary, long-term exclusion of invasive ungulates had the expected effects of increasing carbon pools for saplings and small trees, but not for any other pools considered. The implications of these effects will become manifest over the longer term, both through alteration of tree population structure and forest succession, and through filtering of plant functional traits related to ecosystem processes.

## 4.5 DOC WACEM 3 (Investigation 4082) 'Determine the effects of operational-level herbivore control on total carbon stocks in a broadleaved-hardwood forest'

### 4.5.1 Objective

- To undertake an early assessment of the impacts of wild animal control operations on carbon stocks in a broadleaved-hardwood forest.

It is important to understand the potential for DOC's operational herbivore management in forests to increase total carbon stocks and help offset New Zealand's current emissions. We expect relatively rapid live biomass carbon pool responses in broadleaved-hardwood forests found in productive environments. As such we have selected a 25 000 ha study site in the Coromandel Range where DOC has recently initiated intensive, ongoing goat and possum control. We will use a remeasurement of permanent plots in the treated area to assess changes in carbon stocks during CP1-level timescales of the Kyoto Protocol, and over any subsequent period. This project will serve as an important demonstration of how to quantify carbon benefits from operational-level pest control activities. Finally, we will use National Vegetation Survey Databank plots to more widely assess changes in biomass in broadleaved-hardwood forests to establish priority areas for herbivore control.

### 4.5.2 Outputs

Marburg, A.E.; Carswell, F.E.; St. John, M.G.; Holdaway, R.J.; Rose, A.B.; Jacobs, I. 2013: Implications of experimental design on the detection of herbivore impacts on carbon stocks in a broadleaved-hardwood forest. *DOC Research and Development Series 334*. Department of Conservation, Wellington.

Mason N.W.H.; Bellingham, P.J.; Carswell, F.E.; Peltzer, D.A.; Holdaway, R.J.; Allen, R.B. 2013: Wood decay resistance moderates the effects of tree mortality on carbon storage in the indigenous forests of New Zealand. *Forest Ecology and Management 305*: 177–188.

### 4.5.3 Overview of outputs (abstract from the report/published manuscript)

#### Marburg et al. 2013

We investigated whether plots established for the Thames Coast Flood Protection Project (TCFPP) could also be used to estimate the impact of herbivores on carbon storage in warm-temperate secondary forests. To establish baseline whole-ecosystem carbon stocks, in 2008/09 we measured coarse woody debris, forest floor and soil carbon pools of 32 permanent forest plots and eight paired exclosure and control plots (20 × 20 m) that had been established in 2005/06. To estimate total carbon stock at the initiation of intensive herbivore control, we combined these data with stem measurements collected at the time of plot establishment. Total carbon stocks (216.3 ± 9.7 t C/ha) in each of five pools were estimated to be: above-ground live—74.7 ± 5.3 t C/ha; below-ground—18.7 t C/ha; coarse woody debris—10.3 ± 1.9 t C/ha; forest floor (fine woody debris + litter + fermentation-humic horizon)—18.8 ± 1.7 t C/ha; and soil (0–30 cm)—93.9 ± 4.3 t C/ha. To provide guidance for future studies, we also conducted a power analysis investigating a range of plausible effect sizes and background variability. Our results indicate that the current plot design is capable of detecting only extremely large (≥200%) differences in sequestration rates and will be unable to attribute any changes detected to decreased herbivory unless suitable reference plots can be identified. Future studies of carbon storage in forests should be designed as landscape-scale manipulations with appropriate non-treatment plots and will likely require more plots than are required for studies of tree growth or vegetation composition.

#### Mason et al. 2013

The maintenance of carbon (C) storage in indigenous forest is a key component of efforts to manage atmospheric carbon dioxide concentrations. Increased pressures from extreme climatic events and invasive pests and pathogens pose major threats to the future stability of C storage in indigenous forests through elevated canopy-tree mortality. We assessed the potential for interspecific differences in wood decay resistance to moderate decadal-scale net C losses following canopy tree mortality. We recorded tree mortality, growth and recruitment over a period spanning almost 40 years in repeatedly surveyed plots spanning a wide range of mortality rates. We combined these survey data with national data on species-specific wood decay resistance (i.e. retention of wood density) to estimate contemporary C lost through decay of trees that died during our study. We also included C losses from CWD contributed by a major synchronous mortality event before the study period (legacy CWD C loss) for a subset of the plots where CWD C storage measurements were available. C flux from live to dead biomass (1.36 Mg ha$^{-1}$year$^{-1}$, SE 0.16) was the main factor influencing estimated net contemporary changes in C storage, with the largest net contemporary C losses (1.5 Mg ha$^{-1}$year$^{-1}$) observed in plots experiencing high mortality. Estimated net contemporary C loss from tree mortality was reduced when the dominant species had highly decay-resistant wood. The ability to predict contemporary changes in C was significantly improved when a plot-level indicator of CWD decay resistance was included in multiple regressions. Mean legacy CWD C loss was 0.39 Mg ha$^{-1}$ year$^{-1}$, SE 0.16. When legacy losses were incorporated in net C change estimates, both the size of the legacy CWD pool and its interaction with legacy CWD decay resistance explained a significant amount of variation in net C change in multiple regressions. In plots losing large (around 3 Mg ha$^{-1}$ year$^{-1}$) amounts of C from live biomass (or with more than 300 Mg/ha C stored in legacy CWD at the start of the study) wood decay resistance altered the net C balance by as much as 1.11 Mg ha$^{-1}$year$^{-1}$,

which is a considerable effect given that the mean annual C assimilation rate across plots was 1.38 Mg $ha^{-1}year^{-1}$. Thus, our study reveals strong potential for interspecific variation in decay resistance to moderate the impact of canopy tree mortality on C storage in forests. We suggest that research effort on wood decay rates should be prioritised toward areas, such as drought-prone regions of Amazonia, where forests are likely to experience synchronous mortality events more frequently in future.

## 4.6 DOC WACEM 4 (Investigation 4072) 'Early succession – evaluation of cost-effective methods for woody succession establishment in productive grasslands for carbon sequestration'

### 4.6.1 Objective

- To evaluate cost-effective methods for establishing woody species in productive grasslands as a means of sequestering carbon.

Carswell et al. (2008) considered that carbon sequestration benefits on Conservation lands would be greatest from woody successions. Such successions would potentially have the greatest sequestration rates in productive environments. While the planting of seedlings may overcome dispersal limitation and allow for species choices, it is an expensive option poorly justified in a carbon investment sense. This project will instead focus on seed dispersal and seedling establishment on D'Urville Island (in part an MfE site) with and without herbivory by red deer. A replicated experimental design will be used to determine carbon sequestration in the early phases of succession. An additional advantage of this research is its application to Kyoto-compliant sites. In addition, we will remeasure additional established plots in the Marlborough Sounds to assess longer-term rates of carbon sequestration.

### 4.6.2 Outputs

Burrows, L.E.; Carswell, F.E.; Karl, B.J.; Walls, G.J. 2013. Wild Animal Control for Emissions Management – Early succession: evaluation of cost-effective methods for woody succession establishment in grasslands for carbon sequestration. Landcare Research Contract Report LC1519, prepared for the Department of Conservation, Wellington (unpublished).

Carswell, F.E.; Mason, N.W.H.; Overton, J.McC.; Price, R.; Burrows, L.E.; Allen, R.B. 2014: Restricting new forests to conservation lands severely constrains carbon and biodiversity gains in New Zealand. *Biological Conservation. In Press.*

### 4.6.3 Overview of outputs (abstract from the report/published manuscript)

#### Burrows et al. 2013

Rapid succession to tall forest is one route to increased carbon storage on conservation lands. Establishment of woody species within pasture is thought to be limited because of intense competition from exotic grasses, especially dense sward-forming grasses such as browntop (*Agrostis capillaris*). We investigated (a) the effect of browsing wild animals (goats) on woody species establishment in pasture and (b) the effectiveness of domestic grazing animals (sheep) as a tool to enhance the rate at which woody species establish within pasture. We found that woody shrubs within the experimental enclosure were preferred species for browsing by goats, and the effects of goat browsing impacted on the survival of shrubs. Although there was initial evidence of greater (but still very low) rates of seedling germination on patches of bare ground, we could not correlate these with presence of mob-stocking of sheep at the initiation of the experiment, nor browsing by goats. We conclude that (a) wild animals did not facilitate establishment of woody successional species within pasture and (b) mob-stocking with domestic sheep did not enhance the rate of establishment of woody species within pasture. Instead it appears as if the establishment of woody species within pasture, even when seeds are added, requires contemporaneous control of multiple filters. Favourable conditions likely include adequate seed supply, rainfall at an appropriate time of year, reliable humus moisture for extended periods during germination, creation of favourable microsites for seedling establishment (e.g. ground with less competition or better shelter). Observations at the sites suggest that, even at low densities, persistent browsing of native shrubs and small trees by goats continually depleted potential seed sources. Therefore, browsing wild animals, even at low densities, could have a negative effect on actual and potential carbon sequestration. If the conditions observed in the current study were similar across most early-successional sites, it appears as if control of the wild animals could only enhance rates of establishment of early-successional woody vegetation within grassland if other filters preventing establishment are also removed.

#### Carswell et al. 2014

Increased afforestation of non-productive land could deliver win–win solutions for greenhouse gas mitigation through carbon sequestration and biodiversity gains, referred to here as increased ‘ecological integrity’. We examined the potential trade-offs when selecting non-forested lands in New Zealand for natural forest regeneration to maximise gains in either, or both, carbon and biodiversity. We also examine the effect on potential gains and trade-offs of excluding non-conservation lands from spatial planning for conservation. The most significant per-hectare gains, for both carbon and biodiversity, were those occurring on non-conservation lands because conservation lands are mainly restricted to low-productivity environments where indigenous vegetation is already well represented. By contrast, productive environments, such as alluvial plains, where almost no indigenous vegetation remains, are primarily on non-conservation lands. These lands will need to be included in any reforestation strategy or else the most degraded ecosystems will not be restored. We found that biodiversity suffers a greater trade-off when carbon gain is prioritised than carbon does when biodiversity is prioritised. Trade-offs between carbon and biodiversity were higher on non-conservation lands but decreased with increasing area regenerated. Our study shows that natural regeneration will provide substantial increases in carbon and biodiversity on non-conservation lands compared with conservation lands. This emphasised the need for improved incentives to private land owners if carbon and biodiversity gain from afforestation is to be maximised.

## 4.7 DOC WACEM 5 (Investigation 4073) 'Later succession - determine how carbon sequestration rates are influenced by wild animal herbivores during woody succession to forest'

### 4.7.1 Objective

- To determine how carbon sequestration rates are influenced by wild animal herbivores during woody succession to forest

It is common for relatively unpalatable species (e.g. kānuka and mānuka) to initiate woody successions in productive environments. While mature kānuka stands contain significant carbon stocks, the potential stocks in the long term are driven by factors controlling the establishment and onward growth of mature forest dominants. Te Urewera contains kānuka successions, of varying ages, with the potential for beech, kāmahi and podocarps to replace the kānuka canopies. In this project we will quantify factors controlling the recruitment, growth and mortality of beech, kāmahi and podocarp seedlings during kānuka successions. One key factor to investigate will be the influence of ungulates on seedling performance. We will utilise demographic models based on seedling data, as well as incorporating sapling and tree demographic data from NVS plots, to predict time trajectories for forest composition and carbon sequestration. Trajectory changes in soil and coarse woody debris carbon pools will be based upon sampling these parameters for various stand ages. This research contributes to our understanding of the end-points of Kyoto-compliant successions.

### 4.7.2 Outputs

Richardson, S.J.; Holdaway, R.J.; Carswell, F.E. 2014: Evidence for arrested successional processes after fire in the Waikare River catchment, Te Urewera. *New Zealand Journal of Ecology* 38: 221-229.

Modelling by Robert Holdaway of effects of wild animals (red deer) as ascertained through measurement of seedlings in purpose-built exclosures. This work will be incorporated within the Peltzer et al. manuscript in preparation for publication.

### 4.7.3 Overview of outputs (abstract from the published manuscript or major results)

**Richardson et al. 2014**

Anthropogenic fire has transformed New Zealand's vegetation. Small-scale historical Māori fires in the forests of Te Urewera National Park, North Island, initiated forest successions that were dominated early on by *Kunzea ericoides* (Myrtaceae), and later by *Knightia excelsa* (Proteaceae) and *Weinmannia racemosa* (Cunoniaceae). Previous work in these forests suggested that more recent successions initiated after the arrival of deer in the late 19th century have failed to recover to pre-fire composition and structure. This failure was argued to result from browsing pressure that prevented palatable canopy tree species such as *W. racemosa* establishing at adequate densities. We tested this hypothesis by quantifying changes in forest structure, tree species composition and above-ground tree biomass between 1980 and 2010 using permanent forest plots in three successional communities. The three communities were dominated by (1) *Kunzea ericoides* – Kanuka community, (2) *Knightia excelsa* – Rewarewa community, and (3) *Weinmannia racemosa* – Kamahi community. We show that above-ground biomass increased in all communities between 1980 and 2010. However, tree species composition changed little over the same time except for a more than three-fold increase in tree fern abundance in the Kānuka community where they are now co-dominant with *Kunzea ericoides*. Recruitment of canopy tree species was very low (≤ 0.06% per year) relative to mortality (1.1–3.9% per year) in all three communities. Although above-ground biomass increased in all three communities, the increase in the Kanuka community was low relative to the large increase in basal area brought about by higher abundance of tree ferns that have low stem tissue density. The minimal compositional change over 30 years in these communities and paucity of canopy-tree-species recruitment point to arrested succession. We discuss possible causes and consequences of this in terms of management goals that balance biodiversity alongside ecosystem services such as carbon sequestration.

### Holdaway et al. modelling (to be included in Peltzer et al. in prep. manuscript)

#### *Methods*

Twelve pairs of plots (one fenced, one control) were established in areas of high tree seedling density in the Waikare catchment near the Whakatane River in Te Urewera National Park. Each plot was 5 × 5 m, with a one-metre buffer strip where fenced. Across all 24 plots 3250 seedlings were tagged, targeting (but not exclusively) six of the most structurally significant canopy tree species (rewarewa *Knightia excelsa*; kāmahi *Weinmannia racemosa*; mataī *Prumnopitys taxifolia*; rimu *Dacrydium cupressinum*; kahikatea *Dacrycarpus dacrydioides*; miro *Prumnopitys ferruginea*). These were tagged in Autumn 2011 and remeasured in Autumn 2012 and Autumn 2013. In Autumn 2013 hemispherical photography was also used, to characterise the light environment of each plot.

#### *Results from 2012*

There has been a highly significant ($P < 0.05$) impact of fencing on overall seedling growth (Fig. 4). Comparisons of individual paired plots show the same trend (Fig. 5) although the plots with most light (pair C4, pair C8, pair C10) show evidence of a positive interaction between light and browsing cessation.

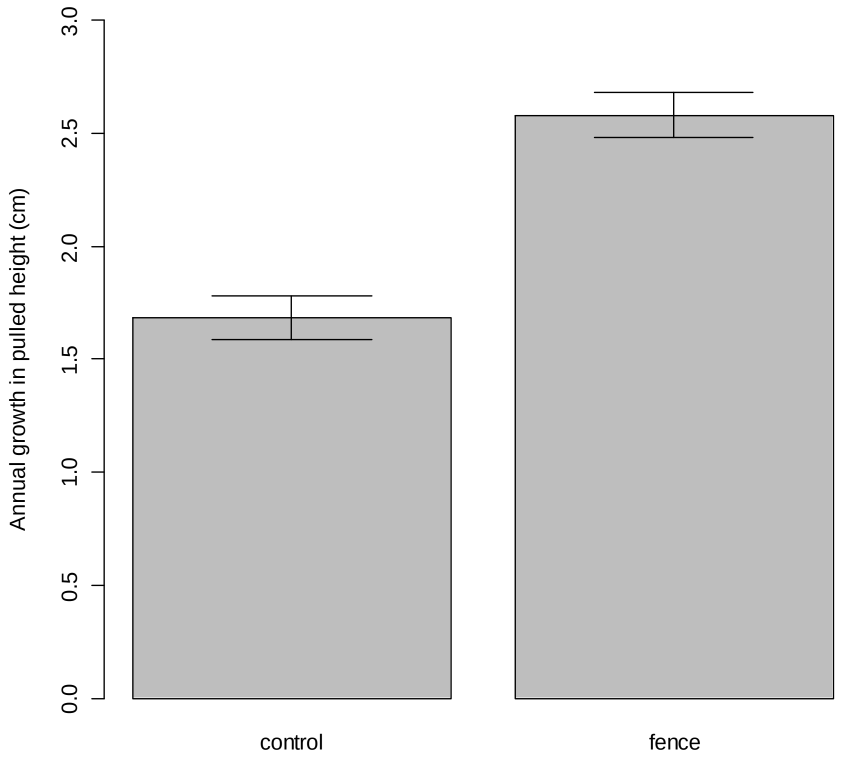

**Figure 4** Comparison of pulled height of canopy tree seedlings in Te Urewera National Park contrasting seedlings within fenced plots with those outside (control). Pulled height is when seedlings are stretched in the direction of the longest leader. (This is to compensate for a drooping growth habit). Error bars represent 1 SEM

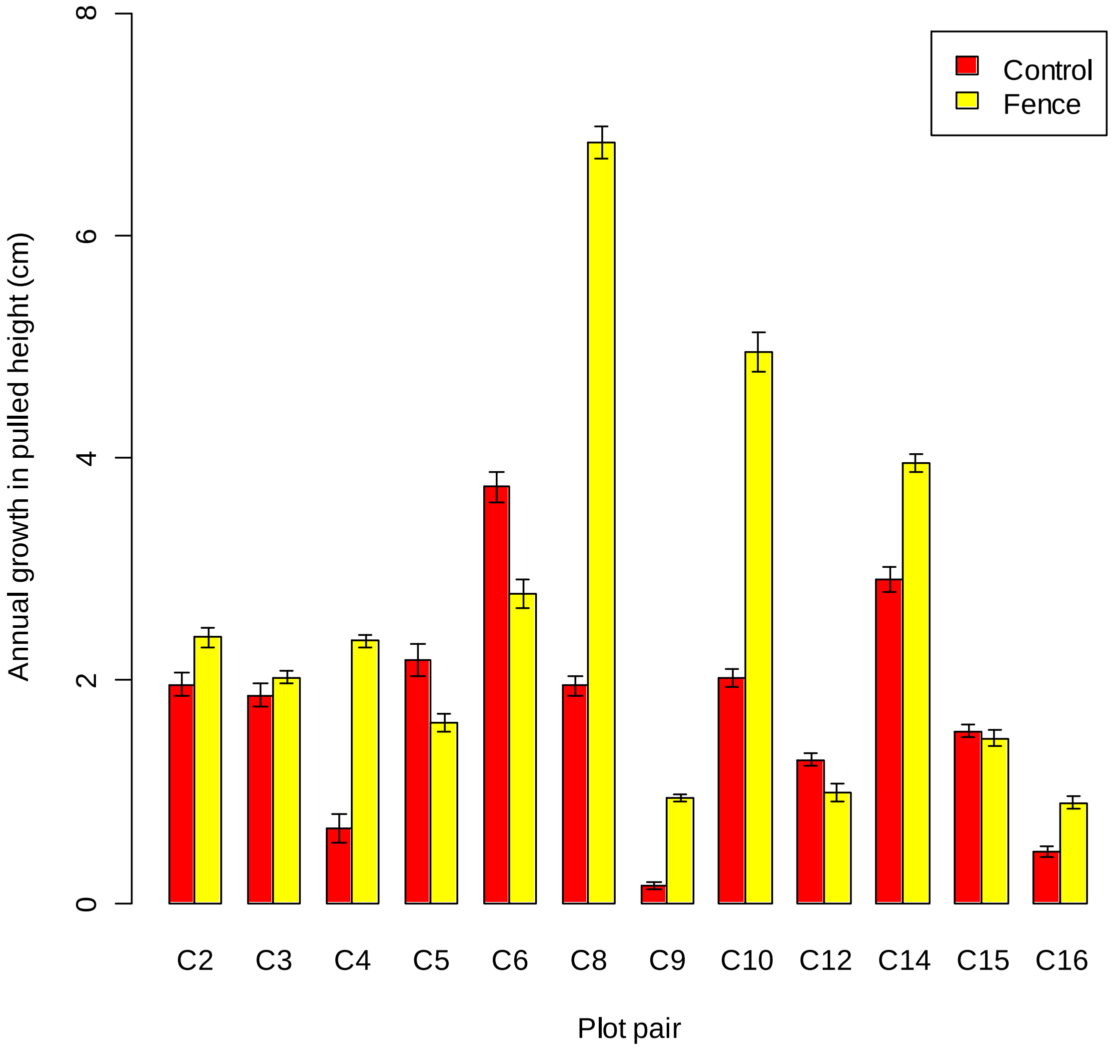

**Figure 5** Comparison of pulled height of canopy tree seedlings in Te Urewera National Park across paired fenced (yellow bars) and non-fenced (red bars) plots.

The effect of fencing has been far greater ($P < 0.05$) on seedling height growth of the highly palatable canopy species, e.g. marbleleaf (CARSER), māhoe (MELRAM) and kāmahi (WEIRAC) (Fig. 6). There has been no effect on the growth of podocarps. All podocarp seedlings showed positive growth in both control and fenced plots.

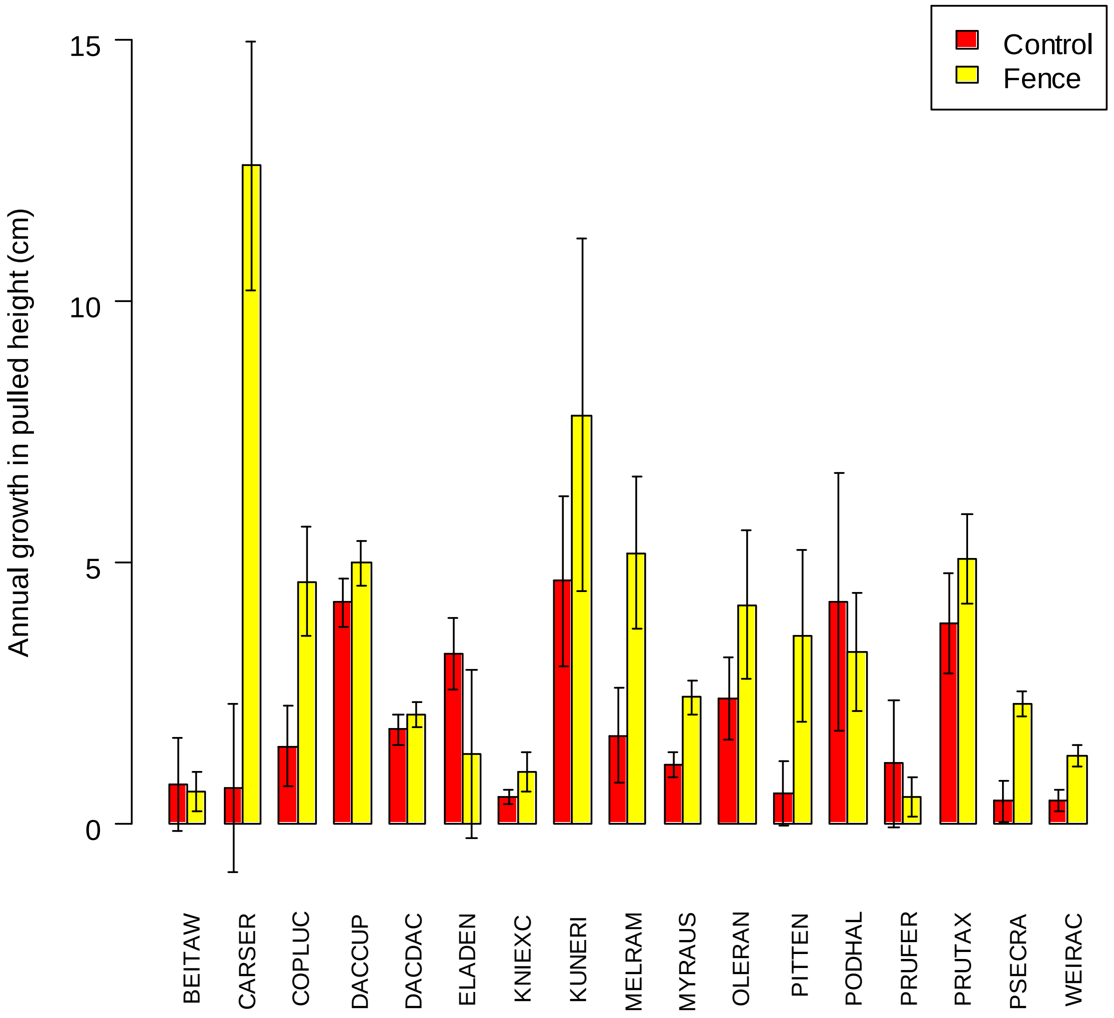

**Figure 6** Pulled height growth of tagged canopy species in Te Urewera National Park, comparing paired fenced (yellow bars) plots with unfenced (red bars) plots.

Mortality rates were also investigated, for all species with at least 50 individuals in total (Fig. 7). There were very high mortality rates for some species (red māpou *Myrsine australis* (MYRAUS) and heketara *Olearia rani* (OLERAN)) in control plots and high mortality for other species in fenced areas (e.g., hinau (ELADEN). Rewarewa (KNIEXC) was particularly ephemeral in both treatments. Podocarp species had a very low mortality rate regardless of treatment.

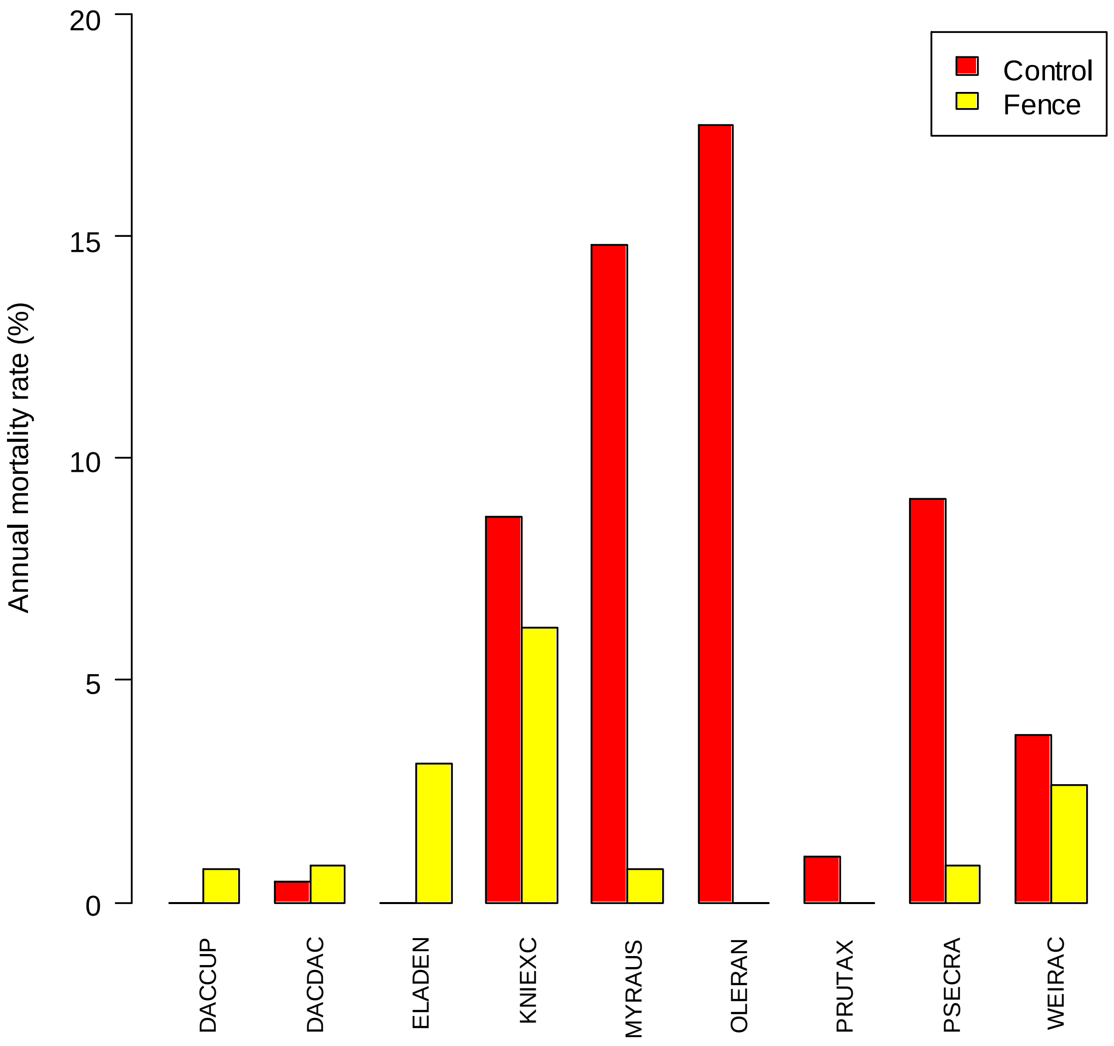

**Figure 7** Mortality of tagged tree seedlings in Te Urewera National Park. All species with at least 50 individuals are included in the figure and seedlings from fenced plots (yellow bars) are contrasted with unfenced control plots (red bars). Species six-letter codes are those used in the National Vegetation Survey Databank.

## 4.8 DOC WACEM 6 (Investigation 4074) 'Tussock and herbivores – evaluation of how tall-tussock grassland is influenced by herbivores'

### 4.8.1 Objective

- To evaluate how tall-tussock grassland carbon stocks are influenced by herbivores.

Burrows et al. (2008) proposed grasslands as a cover type where cessation of control is expected to result in relatively significant declines in carbon stocks. Understanding the influence of ceasing large wild animal herbivore control has proven challenging in grasslands because of a limited ability to contain commercial recovery operations. Therefore, in this project we focus on the converse: what happens when herbivores are reduced? Past studies have shown increasing tall-tussock stature during the commercially led herbivore reductions of the 1970s and 1980s. The impact of this on the relatively large soil carbon pool is unknown. Although we could search for a site where wild animal herbivores (probably tahr) are abundant in grasslands on Conservation land and develop an experiment that involves intensive control – this is likely a risky option, with limited applicability. Instead we propose a study involving the removal of sheep from tall-tussock grasslands. This represents what is happening under the widespread tenure review process. (However, carbon storage in non-woody vegetation cover is currently rather distant from Kyoto Protocol considerations.)

### 4.8.2 Outputs

Carswell, F.E.; Rose, A.; Burrows, L.; Arnst, E.; Drew, K. 2010: Allometric relationships between tussock volume and carbon for *Chionochloa macra* and *C. flavescens*. Landcare Research Contract Report LC0910/166, prepared for the Department of Conservation, Wellington (unpublished).

Holdaway, R.J.; Rose, A.B.; Newell, C.L.; Carswell, F.E. 2014: Demographic drivers of biomass carbon recovery in degraded perennial tussock grassland, with and without domestic grazing. *New Zealand Journal of Ecology* 38: 201-212.

### 4.8.3 Overview of outputs (abstract from the report/published manuscript)

#### Carswell et al. 2010

This study, supported by the DOC R&D programme entitled ‘Wild Animal Control for Emissions Management’ (WACEM), provides allometric relationships between individual total carbon stock and volume of two snow-tussock species (*Chionochloa macra* and *C. flavescens*). Allometric relationships eliminate the need for further destructive harvest of these species which are fundamental components of high-country vegetation. They are also species likely to be adversely affected should introduced wild herbivore numbers increase in montane grasslands. This report provides allometric relationships between volume and tussock carbon for *C. macra* and *C. flavescens*. This information builds on the existing knowledge about tussock carbon accumulation and provides equations that can be used to estimate carbon content of these tussocks in a non-destructive manner.

#### Holdaway et al. 2014b

Many of New Zealand’s natural and induced tussock grasslands are in a degraded low-biomass state due to a combination of fire, overgrazing and weed invasion. The capacity of degraded grasslands to recover biomass is uncertain because legacies of degradation can strongly influence the demographic processes controlling ecosystem recovery. We develop a conceptual framework for understanding biomass carbon (C) flux in degraded perennial grassland based on demographic processes of growth, mortality and recruitment. We apply this framework to 22 years of data from *Chionochloa*- and *Festuca*-dominated perennial tussock grassland in the South Island high country, sampling two grazed sites and two sites retired from domestic grazing. Total biomass C stocks were assessed at site level using 174 temporary plots measured in 1989 and 2011. Long-term demographics were monitored using 24 permanent plots in which 7213 individual tussocks were tagged and monitored every 1–5 years from 1989 to 2011. Although biomass C stocks were generally low (2159 ± 494 kg C ha$^{-1}$), there was a significant increase in total biomass C over the 22-year period at the north-facing retired site of 92 ± 47 kg C ha$^{-1}$ year$^{-1}$. Increases in total biomass C were generally due to recruitment and growth of tall tussock (*Chionochloa* spp.) and/or woody shrubs. Biomass C stock in grazed sites, and the retired south-facing site, declined by up to 65 kg C ha$^{-1}$ year$^{-1}$ or remained constant. Declines were due to dieback of the dominant short tussock (*Festuca novae-zelandiae*) across all treatments, and a lack of recruitment of tall tussock and shrub species. Our results suggest that retirement from grazing was not sufficient to ensure total biomass C gains after 22 years, and highlight the roles of recruitment limitation, shrub establishment and management history in controlling ecosystem recovery.

# 5 Conclusions

Carbon stocks on conservation land could be increased through reforestation (large gains per hectare), advance of existing shrubland successions (moderate gains per hectare) and recovery of existing forests from disturbances including animal browsing (small gains per-hectare over large areas). Wild animal control can play a role in all three types of gain, but we have demonstrated that for existing forests it will be extremely difficult to quantify sequestration that (1) can be attributed to wild animal control and (2) is additional to the sink that already exists. Potential effects of wild animal control will be manifest on carbon stocks at the centennial timescale yet our studies were primarily conducted at the decadal timescale, or shorter. It is our view that the presence of a forest sink in existing forests is a strong endorsement of business-as-usual management of conservation land in forest and that this management should be maintained for the benefit of all New Zealanders.

Successional communities have the greatest potential to generate measureable carbon benefit from wild animal control. Management could include removal of wild animals from broadleaved-hardwood successions where the palatable canopy species of the future need to become established, and supporting other management actions to establish novel woody successions within degraded grasslands. We note that on lands where natural succession does not readily occur, additional, and likely intensive/expensive, management will be required—such as planting for shelter and provision of seed, with associated weeding requirements. These actions should be evaluated alongside the carbon costs of undertaking these management activities. The fastest, and most cost-effective, action to increase carbon sequestration on conservation land is to choose sites close to existing seed sources that are relatively warm (mean annual temperature > 9°C) and receive reasonable rainfall (> 1000 mm per annum) and retire these from grazing, prevent fire and domestic or wild animal incursion as far as possible and to allow natural regeneration to proceed unhindered.

# 6 Recommendations

We indicate above that successional communities offer the greatest potential for increased sequestration through animal control and that the most cost-effective method is to select sites that will naturally regenerate with minimal human intervention (e.g. fencing, pest control, cessation of burning/grazing).

Our recommendations, therefore, primarily concern future research likely to be of most benefit to quantifying the effects of wild animal control on emissions management, or increasing carbon sequestration rate in general. We recommend that:

- The extensive network of nationwide exclosure plots is maintained (i.e. regular inspection and rapid repair where necessary to prevent animal incursions) in order to permit measurement of multi-decadal changes in forest composition and structure and associated carbon
- A case study of carbon sequestration in planted versus naturally regenerating indigenous forest at Tiritiri Matangi (or similar) be conducted
- Further research is initiated on carbon consequences of novel ecosystems (exotic-indigenous mixtures)
- Additional exclosures paired with control plots are placed, and maintained, within successional communities where the trajectory should be towards increasingly palatable species (e.g. the Waikare catchment in Te Urewera National Park)
- Long-term monitoring of sites retired from grazing is continued and new sites included through setting up permanent sample plots. Care should be taken to include measurement of the variables most likely to help/hinder woody succession
- In situ trials be conducted of the potential role for mycorrhizal inoculation to enhance rates of woody succession compared with enhancement through planting of nursery stock

# 7 Acknowledgements

We acknowledge funding from the Department of Conservation for the Wild Animal Control for Emissions Management (WACEM) research programme as a whole, but specifically to compile all outputs and synthesise the information (Investigation No. 4424). Further, we acknowledge helpful discussions with Bill Fleury, Rod Hay and Joseph Arand from the Department. We acknowledge permission from the Ministry for the Environment and the Ministry for Primary Industries to cite reports that we produced under contract to them. We thank the many landowners that allowed access to their lands for this research, especially Noel and Debbie Moleta at Cape Lambert and Mary Richardson and Ralph Powlesland at Manaroa. We also thank the Tūhoe Tuawhenua Trust for their ongoing support in maintaining the animal exclosures in Te Urewera National Park, especially Tom Ata. We thank our many collaborators that contributed significantly across this research programme, especially Alan Rose, Chris Brausch, Geoff Walls, Sean Husheer, Claire Newell, Anna Marburg and Mark St John. We also acknowledge our colleagues that contributed the majority of the primary data, without which there would have been no research programme: Peter Bellingham, Brian Karl, Chris Morse, Rowan Buxton, Kev Drew, Meredith McKay and Elise Arnst.

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