Saving Our Natural Ecosystem Carbon:

Protecting Aotearoa/New Zealand's massive natural carbon stocks from invasive introduced browsers

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Key points

- Carbon stocks in Aotearoa/New Zealand's natural ecosystems are massive; the above-ground natural vegetation alone stores around 1,450 million tonnes of carbon.
- Direct biomass consumption and methane production by feral introduced herbivores in the natural ecosystems is estimated to be between 2.3 and 4 $MtCO_2e$/year. The mid-point (3.1 $MtCO_2e$/yr) is nearly 3 times the emissions from domestic air travel in 2018[1].
- The mid-point, and upper estimates of the increase in carbon sequestration that could result from significant sustained introduced herbivore control are 8.4 and 17.5 $MtCO_2e$/year, respectively. The mid-point estimate is equivalent to nearly 60% of the 2018 emissions from road transport.
- Between 2002 and 2014 there was a significant (-3.4 $MtCO_2e$/year) decline in the carbon stocks of the largest native forest association (kāmahi-podocarp). The most likely cause of this decline was the impact of introduced herbivore browsing.
- Pest mammal herbivore control is likely to be one of the most significant and cost-effective emissions reductions options.
- Significant and sustained introduced herbivore control to protect and enhance natural ecosystem carbon stocks could make a substantial contribution to achieving the country's goal of carbon neutrality by 2050, and has the potential to allow the country to be carbon positive.

Executive Summary:

1. This research has found that Aotearoa/New Zealand could dramatically reduce its net greenhouse emissions by conducting intensive control of introduced herbivores (deer, goats, pigs, wallabies and possums) in native forest, shrub, and tussock lands.

2. This approach could reduce the country's greenhouse gas emissions by an amount as great as all the 2018 fossil fuel transport (cars, trucks, domestic air-travel, etc.) emissions combined. This makes pest control one of the most important options for greenhouse gas emission reduction available to the country, alongside the many sensible proposals put forward by the Climate Change Commission.

3. This option has not received much consideration, and is largely absent from the Climate Commission's current proposals.

4. Introduced herbivores directly consume natural ecosystem biomass (leaves, branches, buds, etc.) and produce methane that together are estimated to be equivalent to 3.1 million tonnes of CO_2 per year, or 5.6% of Aotearoa/New Zealand's reported 2018 net greenhouse gas emissions.

[1] The comparison with components of Aotearoa/New Zealand's reported 2018 greenhouse gas emissions profile is being used for the purposes of explaining the scale of the impacts of introduced herbivores on the country's natural ecosystems. It is not done to suggest that introduced herbivore control could be used to offset other sources of GHG emissions. Aotearoa/New Zealand must reduce its present sources of emissions, as well as protect and enhance the country's natural carbon stocks.

Even greater is a further 8.4 million tonnes of CO_2 per year resulting from the lost carbon sequestration from plant growth and changes to ecosystem processes caused by the damage from browsing, equivalent to nearly 15% of 2018 net greenhouse gas emissions, or nearly 60% of 2018 road transport emissions.

5. These findings are not surprising as the vast majority of the country's carbon stocks - over 6,500 million tonnes - are found in our natural vegetation and soils. Of this nearly 1,500 million tonnes is stored in the above-ground vegetation of our natural ecosystems. The sheer size of these natural carbon stores means that even very small changes in the condition of these stocks, either positive or negative, can have a massive impact on the country's greenhouse gas emissions profile.

6. The figures in the study are conservative estimates. The potential for increased CO_2 sequestering from pest herbivore control, and thus greenhouse gas reductions, could well be higher.

7. While not part of this study, the costs of this option are likely to be very cost effective for reducing emissions from natural ecosystems and increasing the country's ability to sequester carbon.

8. This pest herbivore control option would have other climate change benefits as well. These co-benefits would enhance both ecosystem and community resilience to future climate change impacts. Healthy leaf litter, humus and soil layers, along with greater seedling and diverse and healthy forest under-story vegetation will increase rain interception and assist moisture retention, helping to reduce peak flood flows, and extend the length of water flow during periods of drought. Maintaining moisture levels will assist in reducing forest floor temperatures, thereby helping reduce forest fire risk, and will have multiple benefits in terms of ecosystem services and species/ecosystem protection.

9. Aotearoa/New Zealand currently has an explosion of numbers of introduced herbivores, following decades of reduced spending on deer and goat control. Department of Conservation (DOC) data suggests that large herbivore populations throughout much of the country now exceed densities not seen since before the venison industry commenced in the 1970s. The result is induced native forest collapse in many areas. If not addressed there is a serious risk that these natural ecosystem carbon stores will go into decline with significant increases in net greenhouse gas emissions.

10. A shift in the wrong direction could potentially dwarf the country's present greenhouse gas emissions profile. This study reveals that the largest native forest type - kāmahi-podocarp forest - that make up 10% of all native forest, underwent a significant decline in stored carbon between 2002 and 2014. The annual loss of carbon from the kāmahi-podocarp forests was equivalent to -3.4 million tonnes of CO_2, or three times the 2018 domestic air-travel emissions, and 80% of the extra annual sequestration that the Climate Change Commission hopes can be generated in the medium term by new native forest plantings.

11. This makes it all the more important that cost effective, and sustained introduced herbivore pest control in Aotearoa/New Zealand's natural ecosystem is a key ingredient of climate policy. This pest control will assist the country to become carbon neutral, and possibly even carbon positive in the next few decades.

Key numbers for understanding the importance of natural ecosystem carbon stores and the threat of introduced herbivores to those carbon stores (numbers in red represent greenhouse gas emissions, in green sequestration):

	Carbon million tonnes (Mt)	Carbon dioxide equivalent (Mt CO_2e)	Percent of reported 2018 net GHG emissions
Above-ground carbon stored in natural vegetation[1]	**1,456**	**5,343**	**9,600%**
Reported net greenhouse gas emissions for 2018[2]	-15.1	-55.5	100%
Direct vegetation consumption plus methane produced by feral introduced herbivores (0.6-1.1 MtC: mid-point 0.85 MtC) [3]	-0.85	-3.1	-5.6%
Annual biomass loss in kamahi-podocarp forests (mid-2000s to mid-2010s) [4]	-0.93	-3.4	-6.1%
Potential extra sequestration from sustained feral introduced herbivore control (0.38-5.2 MtC: mid-point 2.8 MtC) [5]	+2.3	+8.4	+15.1%

[1.] See table 2; [2.] Ministry for the Environment (2019); [3.] See tables 4&5.; [4.] See table 6.; [5.] see table 10.

Recommendations:

1. Significantly increase the sustained and systematic control of introduced herbivores on public, private and iwi land in order to protect and enhance the country's massive stocks of carbon that are stored in our natural ecosystems.
2. Focus on the control of introduced herbivores within the kāmahi-podocarp forest associations (found mainly on the West Coast) that the latest evidence shows are losing significant amounts of stored carbon.
3. Reduce introduced herbivore densities to ensure the recovery of palatable species and ecosystem health.
4. Replace the Recreational Hunting Advisory Council with an Ecological Advisory Council that is mandated to advise on the most effective methods of introduced herbivore control to maintain and restore the long-term ecological health of New Zealand's natural ecosystems.
5. Increase resourcing of introduced herbivore control to levels similar to the very successful Predator Free New Zealand programme, including the development and deployment of new pest control technologies.
6. Carry out a more in-depth analysis of the National Forest Inventory data to better understand the direction of carbon sequestration rates for forest associations that have large components of species that are highly susceptible to introduced browsing.
7. Substantially increase the number of forest plots that are regularly surveyed in the National Forest Inventory to give the Inventory greater statistical power to identify the response to pest control and detect changes in forest carbon stocks.
8. Resource the National Forest Inventory so that a full report on its findings can be released within a year of an inventory cycle's completion – not the 5 to 6-year gap which occurred with the first two cycle reports.
9. Release all such reports to the public on their completion – they should not remain confidential for over a year as happened with the 2019 report.

10. Carry out research on:
    a. the production of methane by Brushtail possums in New Zealand ecosystems to better understand their contribution to the country's methane emissions;
    b. the impact of large introduced herbivore control on soil structure and processes, including forest soil capacity to oxidise methane, and to maintain soil moisture;
    c. the ecosystem response to decreased introduced herbivore density;
    d. more accurate national population size and distribution of introduced herbivores.
11. Increase emphasis on establishing permanent native forests as part of the billion trees programme, particularly given the very short half-life of sequestered carbon post-harvest of exotic plantations.
12. Carry out relatively inexpensive management actions, such as exclusion of domestic stock and low-level wild-animal control to ensure carbon gains occur in natural grasslands and regenerating shrublands that have been previously deforested.
13. Introduce incentives for the retention and growth of native forests on private and iwi land and the control of introduced herbivores, especially where precursor scrublands are potentially the target for exotic afforestation.

Introduction

Aotearoa/New Zealand's contribution to global climate change

Climate change is Nature's reaction to a multitude of human-induced ecological stresses that have increased atmospheric greenhouse gases and therefore global temperatures. To limit the potential severity of climate change there is an urgent need to reduce both emissions of greenhouse gasses and to sequester as much carbon as possible in ways that will reinforce Nature's ability to limit global temperature increases.

A country's emission profile is made up of the difference between its total greenhouse gas production from all sources and the amount of greenhouse gases that are removed from the atmosphere by all mechanisms. In 2018 Aotearoa/New Zealand's gross greenhouse gas (GHG) emissions (Table 1) were equivalent to 79 million tonnes of carbon dioxide (Mt CO_2e). The majority of these emissions came from the burning of fossil fuels such as petrol and coal (45%) and from agriculture (43%), particularly methane emissions from stock such as cows. Countering these gross emissions was a reported 23.4 million tonnes of CO_2 sequestration from measured land-use changes and exotic forestry (LULUCF), giving a net Greenhouse Gas Inventory of 55.5 Mt CO_2e (Ministry for the Environment (MfE); 2019), which represents 15.2 Mt of carbon.

Table 1: Aotearoa/New Zealand's reported 2018 gross greenhouse gas emissions profile by CO_2e and by carbon equivalent. [Adapted from MfE; 2019]

Emissions/sequestration type	Mt CO_2e	Percentage of emissions	Carbon equivalent* Mt
CO_2 (mainly fossil fuels)	35.1	44.5%	9.6
Methane (mainly agriculture)	34.3	43.5%	9.3
Nitrous oxide (mainly agriculture)	7.6	9.6%	2.1
Miscellaneous	1.9	1.5%	0.4
Emissions Total	**78.9**	**100%**	**21.5**
Land-use change and forestry	-23.4		-6.4
Net emissions (emissions – sequestration)	**55.5**		**15.1**

* The atomic weight of Carbon is 12, and Oxygen 16. Therefore, Carbon makes up just under a third (27%) of the weight of a CO_2 molecule.

While Aotearoa/New Zealand has a responsibility under the UN Framework Convention on Climate Change (UNFCCC – see Box 1) to conserve and enhance carbon reservoirs, changes in carbon stocks of our indigenous forest are not presently included in the accounting of emissions or removals under the Kyoto Protocol unless these forests are involved in a land-use change (Kirschbaum et al. 2009). However, in future commitment periods, emissions or removals from these forests and other ecosystems are likely to have to be accounted for as signalled in the UNFCCC 1992.

Box 1.: United Nations Framework Convention on Climate Change (UNFCCC; 1992)

Article 4 COMMITMENTS

4.1(d) Promote sustainable management, and promote and cooperate in the conservation and enhancement, as appropriate, of sinks and reservoirs of all greenhouse gases not controlled by the Montreal Protocol, including biomass, forests and oceans as well as other terrestrial, coastal and marine ecosystems;

Management of natural ecosystems for carbon is relevant not only for New Zealand's reports on carbon sequestration under its obligations to the UNFCCC, but is also relevant for reporting under the **Convention on Biological Diversity (CBD)**. For example, the Aichi Target number 15 is that:

'by 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation, and to combating desertification'.

Where is New Zealand's natural carbon?

The above-ground biomass carbon stocks in all of the country's ecosystems is estimated at 1,860 MtC, with natural ecosystems making up approximately 82% (1456 MtC – Table 2). Around 60% of the above ground biomass is on Public Conservation Land (PCL) administered by the Department of Conservation (DOC). Below the vegetative carbon stocks there is another 4640 MtC stored in our soils (about a third in PCL). This brings the total terrestrial ecosystem carbon store to 6,500 million tonnes of carbon (Ausseil et al. 2014).

Table 2: **Estimated biomass carbon stocks from various ecosystems in Aotearoa/New Zealand. [Adapted from Ausseil et al. 2014].**

	Ecosystem	Area (000ha)	Carbon density (tC/ha)	Estimated total carbon stocks (Mt C)	% of total vegetative biomass
Vegetation: Natural ecosystems	Freshwater wetlands & pakihi	249	20-31	7	0.4%
	Subalpine scrub	478	86	41	2%
	Tussock grassland*	2,583	11-27	57	3%
	Mānuka /kānuka shrubland	1,212	51	61	3%
	Indigenous forest*	6,225	102-224	1290	74%
	Total natural ecosystems	**10,747**		**1,456**	**82.4%**
Vegetation: Managed ecosystems	High-producing grassland	8,765	7	61	3%
	Low-producing grassland	1,658	3	5	0.3%
	Cropland: annual	334	5	2	0.1%
	Perennial: (orchards, vineyards)	192	19	2	0.1%
	Exotic forestry	2,036	124 - 88 (pre-post 1990)	231	13%
	Total managed ecosystems	**13,685**		**404**	**16.5%**
Other		700			0.1%
	All Ecosystems	**24,432**		**1,860**	**100%**
Soil: All ecosystems					
	Soil Carbon	**24,432**		**4,640**	
Vegetation and Soil: All ecosystems		**24,432**		**6,500**	

* The estimated total carbon stocks for tussock grasslands and indigenous forests were calculated from the different per hectare carbon density of the different vegetation types and their relative area size.

For a developed country, our levels of natural ecosystem carbon are high. For example, the United Kingdom is almost the same size as New Zealand, but has a total of around 1800 Mt of carbon stored in all vegetation and soils, with approximately 30% (550 MtC) in high conservation value ecosystems on 20% of the land area (Field et al. 2020), compared to some 1450 MtC in New Zealand's natural vegetation with probably a similar quantity in natural ecosystem soils (Tate et al. 1997).

While most wetlands with mineral soils have relatively low carbon densities (Table 2), those with deep organic (peat) soils have the highest per-hectare densities of carbon storage (Figure 1). However, because of their very limited extent (in part due to extensive historic wetland drainage) they, and the more extensive wetlands with mineral soils, make up only 0.4% of the country's total ecosystem carbon. The next highest per-hectare densities of carbon are indigenous forests and because they cover over around 30% of the land area, they make up nearly 75% of the above-ground natural carbon (Table 2). Shrublands, tussock grasslands and subalpine scrub, together make up a further 8% of our natural carbon stores. Exotic plantation forests contain 13% of our natural carbon, with high and low-producing grasslands as well as croplands, including orchards and vineyards, making up the final 3.5%.

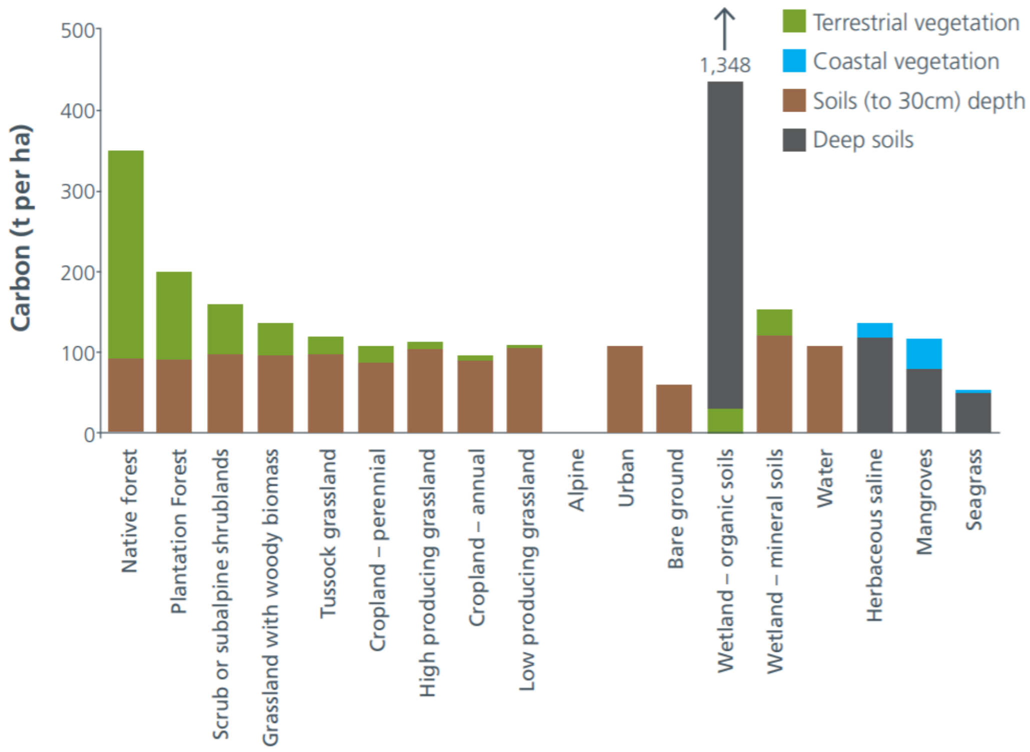


Figure 1: Carbon per hectare currently stored in different Aotearoa/New Zealand ecosystems (including soils to 30cm). Source: Parliamentary Commissioner for the Environment; (PCE 2019)

The sheer size of our natural carbon stores means that even a small change in the condition of these stocks - either positive or negative - could have a significant impact on the country's greenhouse gas emissions profile (Carswell et al. 2012). For example, it would have taken an annual increase in total ecosystem carbon stocks (including soils) of less than 0.2% (one fifth of one percent) to net our 2018 national emissions to zero. But equally, a reduction in our natural carbon stocks of the same small amount would have doubled our net emissions. These numbers highlight the importance that should be attached to the careful management of our natural ecosystems to maintain and enhance their capacity to sequester and store carbon.

In the early 2000s the Department of Conservation (DOC) commissioned a range of research under the Wild Animal Control for Emissions Management (WACEM) programme to consider the potential to manage introduced herbivores to protect and enhance natural carbon stores. This report reviews this valuable benchmark work as well as the wider scientific literature on the size and state of natural carbon stores, and the impact of introduced herbivores on the capacity of those ecosystems to sequester carbon. It will look at new work since the end of the WACEM programme, and attempt to estimate the effect that sustained and comprehensive management of introduced deer, goats, wallabies and possums could have on our net greenhouse gas emissions. It makes recommendations for the future management of our natural ecosystem carbon stores in order to maximize their ability to sequester carbon.

Aotearoa/New Zealand's ecosystems and the impact of introduced mammalian herbivores

Until the arrival of humans, Aotearoa/New Zealand was the largest habitable landmass without land mammals (with the exception of three species of bat). This absence of mammals meant that most of the ecological niches that are dominated by mammals everywhere else were filled by birds, reptiles and invertebrates. Just as New Zealand had a suite of bird predators, including the large Haast eagle, there was also a suite of bird herbivores that included moa, giant flightless geese, and the giant rails, including moho (North Island takahē) and takahē (Tennyson & Martinson; 2006).

Around half the terrestrial bird species consumed some plant foliage, shoots, buds, or flowers. Moa, geese waterfowl and rail species occupied forests, wetlands and grasslands, and parrots and wattlebirds occupied a range of woody vegetation types, feeding on fruits/seeds and foliage/ fruits/nectar, respectively. Other important herbivores were the kererū, hihi, tūī and korimako (Lee et al. 2010).

Moa were the most significant of the herbivores, being physically the largest browsers and grazers within forest and scrubland ecosystems (Lee et al. 2010). Plants evolved structural defenses against the large bird browsers. Adaptations to limit consumption by moa included unusual divaricating form, cryptic colouring and polymorphism where juvenile have very different leaf and branch patterns to the mature plants (Atkinson & Greenwood; 1989).

Following the arrival of Polynesians approximately 800 years ago (Walter 2017), the combination of hunting, modification of mainly lowland ecosystems by fire, and the introduction of two mammalian predators (kiore/Polynesian rat and kurī/dog), led to the rapid extinction of the moa and other large bird herbivores. More extinctions of the keystone bird herbivores followed the arrival of Europeans who carried out further extensive habitat destruction and introduced many more alien species, particularly a range of mammalian predators and large mammalian herbivores (Tennyson & Martinson, 2006; see Box 2).

Prior to the arrival of humans, about 80% of the country was covered in forest (PCE; 2019). Indigenous forests and shrublands currently cover approximately 23% and 10% respectively of the land surface. The current area of woody vegetation represents at least a 70% reduction since the arrival of humans, due to historical fire, forest clearance, and logging (Allen et al.2013).

It has been suggested that the feral mammal herbivore introductions (deer, goats, possums, pigs, wallabies, hares, etc.) from the early 1800s may have filled the ecological niches left vacant by the extinction of large bird herbivores. However, many recent studies have highlighted that moa and ruminants have profoundly different impacts on native forests (See Box 3).

Aotearoa/New Zealand's ecosystems evolved to regenerate rapidly after natural events such as earthquakes and storms in the presence of these avian herbivores. However, those evolutionary mechanisms often fail in the presence of alien herbivores and predators. The rapid growth of riparian shrubs and trees after flooding has been suppressed by deer, the seed rain and nutrient input from birds that saw the Taupo basin reforested after massive volcanic eruptions have been lost to alien predators, and the suppressed seedlings awaiting light from canopy collapse in mountain beech forests have been eliminated by deer, goats, chamois, hares and tahr.

With their large size and sophisticated digestive anatomy, the introduced ruminants such as deer and goats, directly consume virtually all of the foliage of preferred native plants, and within decades to centuries notably change the composition and regeneration patterns of the indigenous ecosystems they occupy. These longer time scale impacts may be considerably more important than the short-term direct effects (Petzler etal. 2010). Recent assessments of the distribution of ungulates established that in 2019/20 they occupied between 80 and 85% of public conservation land (DOC; 2020 a), which was a significant increase since 2012/13. Occupancy (and abundance) were generally lower in national parks (helped by the success of the goat eradication program for Taranaki/Egmont National Park) and in non-woody habitats (Figures 2 & 3).

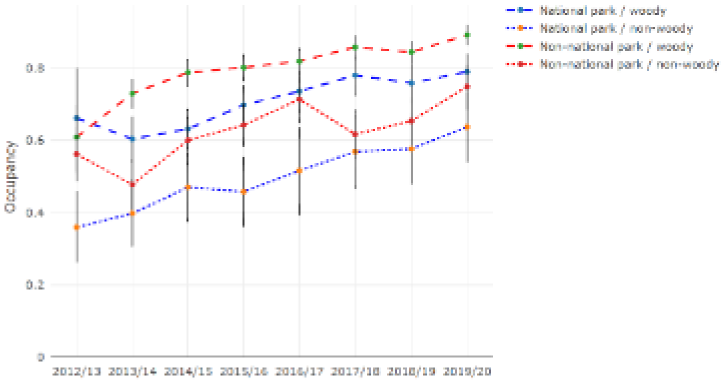


Figure 2: Ungulate occupancy on public conservation land (PCL) between 20012/13 and 2019/20. [Source: DOC; 2020 a]

Figure 3: Distribution of feral deer, goats. Chamois, tahr, wallabies and possums in Aotearoa /New Zealand (from Bellingham et al. 2014; Bengsen et al. 2017; Lantham et al. 2018)

Box 2: Introducing the (large) introduced herbivores:

The 1860s colonial culture gave rise to regional Acclimatisation Societies being set up across Aotearoa/New Zealand. Their motivation was to stock native ecosystems that they perceived as being 'empty'. In 1867 both the Animal Protection Act and the Salmon and Trout Acts were created to protect and encourage the species bought from many other countries to flourish in native forests, tussocklands and waterways.

Deer

Red deer are the most widespread species of deer occupying more than 12 million ha or over 44% of the country. They are selective browsers, concentrating their feeding on plant species they prefer, leading to the local elimination of palatable herbs, shrubs, understorey woody species, and seedlings of larger trees, resulting in an increase of less palatable species. Red deer can also kill trees by bark-stripping. Fallow deer are the second most widespread species of deer but have a very patchy distribution in the North and South Islands occupying around half a million ha. Sika deer are found in the central North Island in the Kaimanawa and Kaweka Ranges, extending to southern Urewera, the Ruahine Range and the southern and western part of Tongariro National Park. There have also been illegal releases in Northland, Taranaki, and the Wellington regions. Rusa deer are east and south-east of Rotorua and are slowly expanding into the forests of the Urewera ranges. White-tailed deer are found across Rakiura/Stewart Island and an area at the head of Lake Wakatipu which includes the lower sections of the Rees River and Dart River valleys. White-tailed deer are also present in safari parks in the South Island (Fraser et al. 2000). The two main populations of wild sambar deer are found in the Manawatu/Wanganui region and the Bay of Plenty. Their range in both these areas is mostly on private land.

Feral goats

Feral goats have similar dietary preferences to deer, but can also eat species poisonous to deer. Their impact is enhanced as they can get by on poorer food so they 'push' the vegetation harder, and their sociable nature means they aggregate in high densities putting severe pressure on favoured habitats. Feral goats occupy around 4 million ha (14 %) (Fraser et al. 2000), of which about 2 million ha is land managed by DOC. Feral goats are present on both the North and South Islands and have been present at various times on 34 offshore islands, but currently occur only on two (Arapawa, and Forsyth Islands). With a reproduction rate that is higher than deer, feral goat populations can rapidly reach very high densities that require substantial control efforts to protect conservation values. The scattered nature of goat distribution allows managers to consider eradication as an option for some mainland populations, as has been achieved on many islands. (Parkes; 1993)

Tahr

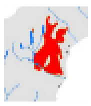
Himalayan tahr and chamois are alpine goat species and can cause significant damage to native grasses and herbs in the alpine habitats, with grazing causing decreases in snow tussock, changes in the composition of grasslands and increases in bare ground. Tahr occupy 425,900 ha in the Southern Alps mostly between the Rakaia River in the north and the Haast River in the south, with some outlying populations (Fraser et al. 2000). Tahr are highly gregarious/social animals and can reach high densities so, in addition to direct grazing damage, large groups of tahr can damage alpine vegetation by trampling. The loss of vegetation cover can result in fine-scale soil erosion.

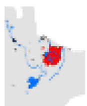
Chamois
Chamois are smaller than tahr, are generally solitary and can also utilise forest ecosystems, where their diet and impacts are similar to those of red deer. While chamois are widespread throughout the South Island high-country occupying nearly 5 million ha, they are absent from parts of Fiordland (Fraser et al. 2000). Chamois in Westland have been shown to have higher concentrations of woody species in their diet, compared to those on the eastern Southern Alps, and are thought to impact kāmahi in the western alps when at high densities. Chamois can also cause damage by trampling areas of vegetation and compactable soils, especially soft mire wetland soils.

Pigs
Because pigs have a single stomach unable to digest cellulose, they favour the most digestible food such as improved pasture grasses, herbs, native tree seedlings, fruit, tubers and bulbs over tussock grasses or the leaves of forest trees. Vegetation usually forms about 70% of feral pigs' diet, and their foraging activity can inhibit recruitment and plant community structure or composition in a range of landscapes from sub-alpine grasslands and to temperate forests. The effect of rooting and seed consumption likely plays a role in altering regeneration processes. Most estimates suggest that pigs now occupy approximately 9.3 million ha (35%) of the land area (McIlroy; 1989, Bengsen et al. 2017). Feral pig populations have an extraordinary capacity for growth as sows are long-lived, can begin breeding from about 7–12 months, and can produce two litters of up to 10 young every 12–15 months (Choquenot et al. 1996).

Possums
Brushtail possums occur throughout the country, with the exception of some alpine areas and parts of South Westland and Fiordland, and cause serious conservation damage. Like many invasive species, possum densities are highest at sites where possums have recently arrived for the first time (Sweetapple et al. 2004). Possums have smaller, simpler, and less efficient digestive anatomy than the larger ruminants. However, possums can make a major impact on overall forest diversity and, through selective browsing, possums reduce diversity and accentuate the strong bias towards unpalatable species (Owen & Norton; 1995).

There were early concerns about possum damage to native forests focused especially on large-scale canopy mortality in southern rata/kamahi forests (Owen & Norton; 1995). There is also strong evidence to link the local reduction and even local extinction of some tree species within forests to the effects of possums such as kōtukutuku in montane Westland, and tutu and northern rātā, in alluvial forests near Wellington (Cowan et al., 1997). A large mistletoe confined to forests and their margins, *Trilepidia adamsii*, has been driven to extinction by possums (Norton; 1991).





Wallabies
There are two species of wallaby on the mainland. Bennett's (also called red-necked) wallabies average between 14 and 20 kg as adults and occupy some 530,000 ha in South Canterbury and north Otago centred on the Hunter Hills. Dama (also called tammar) wallabies are smaller, weighing between 4 to 6 kg and occupy around 200,000 ha centred on the Rotorua Lakes in the Bay of Plenty. Both species occupy a range of lowland, hill, and high-country habitats, including pasture, tussock grassland, shrubland, and forest. Both species are undergoing a rapid expansion of their range and within 50 years could spread across a third of the North and South Islands (Latham et al. 2016).

Impact on native wildlife
Introduced mammalian herbivores can have a considerable impact on native wildlife that rely on the indigenous plant species they selectively eat, or the vegetation structure and regeneration they modify. Goats left on remote offshore islands as food for castaways not only caused rapid and significant impacts on the vegetation, they also directly disrupted seabird colonies. Goats reduced several island plant species to the verge of extinction, with the Kermadec hebe on Raoul Island and *Tecomanthe speciosa* and *Pennantia baylisiana* on Manawa Tāwhi/Great Island being reduced to single plants (Parkes, 1993).

The diets of possum, red deer, and goat have considerable overlap with the diet of kōkako (Leathwick et al., 1983) and deer consume the same tussocks that are preferred by takahē (Rose & Platt; 1987). Feral pigs have been reported to disturb ground nesting birds and eat eggs and chicks including white-capped mollymawk (Flux 2002). Possums can compete with kea for den sites in alpine areas, and also predate a range of native bird eggs and chicks and are known predators of *Powelliphanta* giant land snails (Clout; 2006, Walker; 2003, Moorhouse et al. 2003). Native bird abundance has been found to decline with increasing length of possum habitat occupation (Sweetapple et al. 2004). Chamois-induced changes to high alpine microclimates are thought to impact native grasshopper populations and therefore encourage scree formation (Batcheler; 1967).

Population estimates for these introduced herbivores are reviewed in Appendix 1.

Box 3. Ngā Moa and other bird herbivores versus deer and goats
Recent research has shown that moa diet was richer than the diet of deer and goats, implying that the prehistoric forest understory was more varied and browsed in a different manner to deer and goats. Moa lived at lower densities and bred more slowly than the alien browsers which now occupy New Zealand's ecosystems. The high population densities and intense browsing pressure of introduced mammals such as deer and goats has driven the loss of many understory species which co-evolved with and could therefore survive being eaten by moa (Wood & Wilmhurst; 2019, Lee et al. 2010).

The feet of deer and goats also have a different impact than moa on forest soils. Deer, goat, and tahr foot pressures are 1.8 to nearly 3 times greater than the foot pressures of the various moa species. When moving over soft ground, the ungulate hoof acts like a chisel, and as the toes splay out, the hoof edge shears the soil. In contrast, the moa foot is more flexible, and rolls off the ground causing little or no cutting damage with the edge (Duncan & Holdaway; 1989). At high population densities ungulate hooves disturb soils and have a higher impact on soil compaction than moa.

Effects of introduced invasive herbivores on natural carbon stocks and carbon sequestration.

Freeland (1990) found that for invasive introduced populations of mammalian herbivores, density at carrying capacity was higher for given bodyweights than it was for the same species within their natural range. He argued that herbivores could attain higher densities at carrying capacity outside of their natural range because the food resources they utilised had not evolved defences specific to reducing the impact of those herbivores.

Deer, goats (including tahr and chamois), pigs, wallabies and possums have direct effects on carbon stocks and carbon sequestration through the direct consumption of vegetation biomass (leaves, flowers, fruit and seeds, bark, seedlings, etc.), but the effect of this direct consumption is relatively small compared to the large total biomass stocks in most natural ecosystems. However, there is general recognition that the indirect impacts of herbivores on ecosystem structure and composition through reductions or elimination of preferred species, their preferential consumption of nutritious litter, their impacts on ecosystem microclimates and moisture retention can together have large ecosystem-level impacts on the cycling of nutrients and carbon (Burrows et al. 2008).

As part of the DOC-funded Wild Animal Control for Emissions Management programme, Holdaway et al. (2012) prepared a report on the *"Potential for invasive mammalian herbivore control to result in measurable carbon gains"*. In addition to the known biodiversity benefits of introduced herbivore control, the paper looked at the potential for such control to lead to measurable carbon gains across the whole public conservation estate. While the focus of the DOC-funded work undertaken by Landcare Research was on the conservation estate, the conclusions are applicable to the control of introduced herbivores across all natural ecosystems, regardless of tenure (Carswell et al. 2008).

Direct consumption of vegetation in woody habitats.

The most direct impact of introduced herbivores on climate change is through their consumption of vegetation (mainly leaves, but also buds, flowers, seed and fruit, leaf litter, bark, and the seedlings that would become future forests), and its conversion into animal biomass, CO_2, faeces and urine, and the production of methane (particularly by ruminants).

Direct biomass consumption

The plant species that are palatable to mammals often have rapid growth rates, including new spring growth, thin leaves with high leaf nutrient contents, and have invested little in defence chemistry (e.g. tannins, phenols), or less digestible structural carbohydrates. They often have short lifespans and a low wood density (low carbon). These plants are also often restricted in space (e.g. to more fertile, moist or sunnier sites) or time (e.g. to canopy gaps, or particular seral stages) and therefore are naturally uncommon. Therefore, they often make a comparatively low contribution to total biomass (Nugent et al. 1997).

Among the 16 species that store the most carbon in forests (Table 3), only 3 (kāmahi, pāpāumu/broadleaf, and māhoe) are palatable to deer and goats and 5 (kāmahi, southern and northern rātā, Hall's tōtara, and māhoe) are palatable to possums (Bellingham et al. 2014). Nationally, the group of most possum-preferred species make up approximately 20% of the above ground live tree carbon pool and have a similar growth but significantly more mortality compared with all other species (Beets et al. 2009). The biggest effects of controlling alien browsers on forest carbon storage are likely to be seen in broadleaved forests where a large amount of the vegetation biomass is made up of palatable species.

In podocarp-hardwood forests the direct carbon losses from deer have been measured at around 1% of the annual total foliage production and over 3% for possums, together accounting for 0.055 tC/ha/yr (Nugent et al. 1997). Possums ate around 1 to 2% of annual beech foliage production in montane red-silver beech forests in the Hurunui Valley representing between 0.015 to 0.026 tC/ha/yr. (Sweetaple; 2003). Bark chewing of beech trees was considered to be far more damaging than the relatively small quantities involved would suggest, as the ring barking led to the death of small to moderate-sized branches, particularly where possums were most abundant.

Table 3: The 16 tree species that comprise the largest proportion of total carbon (C) in live stems in forests on public conservation land throughout Aotearoa/New Zealand; values derived from measurement of 542 plots 2009–2013, and species ranked in descending order. *also includes F. cliffortioides and hybrids. (Total C in live stems as tC/ha.) (Adapted from Bellingham et al. 2014)

Species		**Carbon t/ha (2009-13)** Mean ± SEM
Silver beech	*Lophozonia menziesii*	28.03 ± 2.41
Red beech	*Fuscospora fusca*	20.60 ± 2.31
Kāmahi	*Weinmannia racemosa*	15.49 ± 1.30
Black/mountain beech	*Fuscospora solandri**	12.91 ± 1.35
Rimu	*Dacrydium cupressinum*	6.77 ± 1.13
Southern rātā	*Metrosideros umbellata*	6.05 ± 1.19
Tawa	*Beilschmiedia tawa*	5.56 ± 0.92
Hard beech	*Fuscospora truncata*	4.73 ± 1.02
Pāpāumu	*Griselinia littoralis*	1.91 ± 0.38
Miro	*Pectinopitys ferruginea*	1.76 ± 0.31
Mataī	*Prumnopitys taxifolia*	1.65 ± 0.69
Tāwheowheo	*Quintinia serrata*	1.50 ± 0.24
Northern rātā	*Metrosideros robusta*	1.42 ± 0.73
Hall's tōtara	*Podocarpus laetus*	1.34 ± 0.24
Māhoe	*Melicytus ramiflorus*	1.34 ± 0.30
Kahikatea	*Dacrycarpus dacrydioides*	1.13 ± 0.45

The seemingly small short-term effects on carbon stocks from direct consumption by the large browsers hides the important indirect long-term effects on carbon sequestration. For example, in the long-term, death of individual established trees caused by possums becomes more significant if the species are not able to regenerate because of deer or goat browsing of their seedlings. (Coomes et al. 2003; Burrows et al., 2008 Nugent et al. 1997). In regenerating forests, the preferential loss of seedlings of palatable species can alter the course of forest development and reduce potential long-term carbon sequestration by preventing the establishment of high-biomass late-successional species. For example, this has been observed in regenerating kānuka on south Kaipara Spit where fallow deer have prevented the establishment of broadleaved tree species (Smale et al. 1995).

Canopy mortality

The direct browsing of possums or ring-barking by deer (figure 4), goats, and possums can increase the mortality of canopy trees (. Possum browsing has been identified as the main cause of extensive canopy dieback in kāmahi -dominated forests in Te Urewera, and rātā-kāmahi forests in South Westland (Allen et al. 1984; Holdaway et al. 2012).

Figure 4: Trunk of *Pseudopanax colensoi* var. *ternatum* with a dbh of 40cm, ringbarked by deer in lowland forest at Poison Bay, Fiordland. The first deer to invade an area of forest on Secretary Island, Fiordland, showed an almost exclusive preference for bark of this species, killing mature trees in the process. (Photo from Mark & Baylis; 1975).

Investigating significant kāmahi die-back in the Kaitake range of Taranaki/Egmont National Park, Clarkson (1993) noted possums find palatable kāmahi shoots growing from buds on the trunk or branches being targeted as preferred food. The kāmahi die-back was attributed to severe possum browse, with high levels of possum browsing impacting on trees' ability to recover from the damage caused by Cyclone Bola.

In other areas significant shifts in composition and structure of many indigenous ecosystems have been attributed to the combined effects of ruminants and possums. For example, in the southern Ruahine Range where tall ancient forests at low to mid-altitude were replaced in the mid-20th century by short-statured forests in which tree ferns and other non-palatable species are abundant. Such changes resulted in reduced above-ground carbon stocks (Bellingham et al. 2014). Possum damage is also responsible for severe canopy dieback in the mountain cedar forests of the upper Rakaia (figure 5a), which had not experienced pest control for over 30 years (Harding; 2009). In Te Tai Tokerau/Northland the relatively recent arrival of possums (60 years ago) is evidenced by the extent of dying totara and other palatable species (figure 5b). Because possums arrived in most areas of the country decades earlier, this type of tipping point passed decades ago.

Figure 5a: The unique and rare mountain cedar forests in the upper Rakaia Valley collapsing due to possum browsing, in the absence of pest control. (source DOC).

Figure 5b: The forests of Te Tai Tokerau/Northland were one of the last to experience colonisation by possums, starting in the 1960s. Many of the region's forests contain the skeletons of recently lost palatable canopy species, killed by the impact of continuous browsing that weaken the trees to a tipping point where further browsing triggers an irreversible decline, or accelerates dieback caused by insect attack, etc. (photo Dean Wright)

The largest declines in forest carbon are likely to occur where there are high possum numbers in forests dominated by canopy trees that decay quickly (Holdaway et.al 2012), and this has been borne out by the latest national forest carbon stock assessment where the kāmahi-podocarp forest association was the only type of forest to record a significant reduction in carbon store (see below; page 15).

Seedling mortality

The effect of browsing on forest floor seedlings is not only due to deer, goats, and pigs. For example, in beech-podocarp-broadleaved forest near Dunedin, seedling densities in exclosures that excluded both possums and rats were 3.6 times as high as in control plots, and 2.1 times as high as in exclosures that excluded only possums (Wilson et al. 2003), indicating the effects on seedling recruitment of rat seed consumption. Possums have also been found to consume up to around 30% of beech seedfall during beech forest mast events (Sweetapple; 2003). This level of seed consumption by possums could be significant for beech forest recruitment, particularly when combined with the beech seed consumption of rats and mice.

Magnitude of direct Carbon consumption

While the per-animal direct consumption of vegetation is relatively small, when the combined feral populations of large introduced herbivores across all natural habitats are taken into account, their impact takes on national significance. A rough estimate of direct consumption comes to between 2.8 and 5.2 million tonnes of carbon dioxide equivalent per year (Table 4).

Table 4: The direct national annual biomass carbon losses due to introduced herbivore consumption.

Herbivore		**Annual dry matter consumption (kgC/yr)**	**Herbivore population estimate[5]**	**Annual population direct consumption MtC/yr**	**Annual population direct consumption $MtCO_2e/yr$**
Deer		240-490[1]	300,000	0.072 - 0.147	0.26-0.54
Goats		90-180[1]	500,000	0.045 - 0.090	0.16-0.33
Pigs		158-224[2]	300,000	0.047 - 0.067	0.17-0.25
Possums		17-35[3]	30,000,000	0.523 - 1.040	1.92-3.82
Wallabies	Bennett's	65[4]	1,064,400	0.069	0.25
	Dama	28[4]	410,000	0.011	0.04
Total direct biomass C consumption				0.767 - 1.424	2.8-5.23
Total biomass consumption loss (digestibility factor: 70%)				**0.537 to 0.997**	**1.96 to 3.66**

1. From Holdaway et al. (2012); 2. From Dzieciolowski et al. (1990); 3. Range from Cowan (2007); 4. From Latham et al. (2020);
5. For calculations of herbivore population estimates; see Appendix 1.

Between 25 to 30% of the eaten biomass is not metabolised and passes through the animal to become part of the soil carbon pool (Castro Lima et al. 2016). If the upper estimate of 30% that passes through the animal is used, the biomass loss due to direct consumption would be equivalent to between 1.9 and 3.7 million tonnes of CO_2, which is between 3.4 and 7 percent of our reported 2018 net greenhouse gas emissions.

Methane production by introduced herbivores

Methane (CH_4) is a potent greenhouse gas with a 100-year global warming potential at least 25 times that of CO_2 (PCE 2019). Ruminant animals such as deer, goats, tahr, and chamois have specialist fermentation stomachs that produce methane during the digestion. Feral deer are estimated to produce 16.5 kg of methane per animal per year (Hristov; 2011), with individual feral goats producing around 10kg CH_4/yr (Hristov et al. 2013). This is equivalent to around 410kg CO_2e/animal/yr and 250kg CO_2e/animal/yr respectively.

Pigs are not ruminants and their simpler digestive systems produce only a small amounts of methane (1kg methane, or 25kg CO_2e) per animal per year (Crutzen et.al 1986). Based on the population estimates of these larger introduced herbivores (appendix 1), together they produce around 9,800 tonnes of CH_4 a year (table 5).

Wallabies possess a forestomach that supports a cooperative host-microbe association that releases nutrients from plant biomass. Though analogous to rumen fermentation, this results in lower methane emissions (Gagen et al. 2014). Bennett's wallabies produce up to 2.5 litres of methane per day (Madsen & Bertelsen; 2018), which equates to 0.51 kg CH_4/animal/yr. Dama wallabies produce up to 0.8 litres of CH_4 per day[2] (von Engelhardt et al. 1977) which equates to 0.16 kg CH_4/animal/yr.

The even simpler digestive system of possums is generally assumed to produce little or no methane (Holdaway et al. 2012). However, Brushtail possum, and other small arboreal marsupials such as the Greater Glider, nevertheless have extended hindguts where fermentation of plant structural carbohydrates occurs (Foley; 1984). When processing a poorly-fermentable *Eucalyptus* leaf diet, which has high levels of cell wall lignin, the short-chain fatty acids produced by this hindgut fermentation can provide up to 15% of the possum's digestible energy (Foley et al. 1989).

Assuming a similar rate of CH_4 production per body mass as Greater Gliders (Foley; 1987), Brushtail possums with an average weight of 2.5kg would produce around 0.18g CH_4/day or approximately 65g CH_4/yr. While this is not a lot per individual possum, when the total population of approximately 30 million possums is taken into account, it amounts to 1,950 tonnes of methane per year, which in turn is equivalent to some 42,900 tonnes of CO_2. The availability of more easily fermentable food sources in New Zealand ecosystems, where high levels of structural carbohydrates (cellulose, hemicellulose) are replaced by non-structural carbohydrates (starch and sugars), is likely to increase rates of fermentation (Castro Lima et al. 2016), with a potential consequent increase in possum methane production on this side of the Tasman Sea. For the purposes of this analysis the Australian rates of methane produced from poor quality *Eucalyptus* leaf food sources will be used.

Table 5: Estimates of the annual methane emissions from feral introduced herbivores.

Herbivore	Methane emissions per animal kg CH_4/yr ($kgCO_2e$/yr)	Estimated population[7]	Estimated annual CH_4 emissions (tonnes CH_4)	Estimated annual CO_2e emissions
Deer	15 (375)[1]	300,000[6]	4,500	112,500
Goats (including tahr and chamois)	10 (250)[2]	500,000[6]	5,000	125,000
Feral pigs	1 (25)[3]	300,000[7]	300	7,500
Possum	0.065(1.6)[4]	30,000,000[8]	1,950	48,750
Wallabies				
Bennett's	0.51(13)[5]	1,064,400[9]	543	13,570
Dama	0.16(13)[6]	410,000[9]	66	1,650
Total			**12,359**	**308,970**

1. Hristov; (2011); 2. Hristov et al. (2013); 3. Crutzen et.al (1986) and Rivero et al. (2019); 4. Based on assumption that Brushtail possum has a similar rate of CH_4 production per body mass as Greater Gliders (Foley; 1987); see text; 5. Calculated from Madsen & Bertelsen (2012). 6. Calculated from von Engelhardt et al. (1977). 7. For calculations of herbivore population estimates; see Appendix 1.

When the methane production estimates are combined with the population estimates of the large feral introduced herbivores, together they produce around 12,360 tonnes of CH_4, which is the equivalent of around 309,000 tonnes of CO_2 a year (table 5). This is close to one percent of the country's 2018 biogenic methane production, and represents 14% of the annual 90,000 tonne reduction in agricultural methane emissions by 2025 and 6% of the 2035 annual reductions (210,000 tonnes) that are being recommended by the Climate Change Commission (CCC 2021*a*; recommendation 3, p.33).

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[2] The lower range value of methane produced on higher energy/higher protein diet (von Engelhardt et al. 1977).

Combining the direct carbon consumption estimate with the methane production estimate (tables 4 & 5) we get a range for the direct impact of feral introduced herbivore populations of around 2.3 to 4.0 million tonnes of CO_2e per year. To give some context to this estimated range, it is equivalent to 5.6% of Aotearoa/New Zealand's reported 2018 net greenhouse gas emissions and nearly two times greater than the reported loss of between 1.2 to 2.4 Mt CO_2e each year from deforestation (MfE, 2019).

The Climate Change Commission described this loss from deforestation as "low but non trivial", and has recommended that no further native forest deforestation occurs after 2025. The direct consumption and methane emissions range also encompasses the Commission's estimate of the long-term annual contribution (+4 MtCO_2e/yr) that could be made by new native forests established on steeper, less productive land by 2050 (CCC. 2021*a*).

<u>Browser impact on forest and shrubland soils</u>

Wardle et al. (2001) found that while there were several small but statistically significant effects of browsing on some measures of soil quality, there were also strong and significant effects on species composition of leaf litter, and composition of various litter-dwelling faunal groups. The impact of browsing on soil fauna was correlated with the effects on the leaf species that made up the litter layer, but was not correlated with the effects of browsers on the composition of plant communities. Over a quarter of the locations surveyed showed significant effects of browsers on the soluble organic carbon concentrations of the humus, and, averaged across the 30 locations, concentrations were significantly greater (20%) inside, than outside exclosures.

For many sites, browsing mammals had a significant effect on carbon storage in the humus and litter layers. Despite the strength of these effects, their direction was highly idiosyncratic, with browsers significantly promoting sequestration of carbon in some cases, but having the reverse effect in others. The site-specific effects of introduced browsers reflected the importance of soil texture and moisture on the levels of soil carbon.

Several mechanisms have been proposed for how introduced herbivores can indirectly affect below-ground carbon sequestration - either positively or negatively - by influencing the quantity and quality of the vegetation biomass that is returned to the soil (Peltzer et al., 2010). Replacement of palatable, high-nutrient species with low-nutrient species, plus the preferential consumption of high-nutrient litter from canopy and sub-canopy trees, can lead to reduced litter quality, disrupting the composition of the soil litter and the accumulation of carbon in the soils' mineral layers.

Compaction of soils by large grazing herbivores can alter soil moisture by reducing the porosity and ability for water to infiltrate. This in turn can inhibit the flow of nutrients, slow root or shoot growth, and alter the distribution of soil microbes. In addition to reducing plant growth, declining soil moisture may inhibit microbial biomass and change the soil's bacterial/fungal ratios, which can in turn impact on nutrient cycling and storage (Gass & Binkley; 2011). Compaction that increases soil bulk density and reduces the air-filled spaces within the soil reduces its capacity to hold moisture. This increases the soil's likelihood of becoming water-logged and reduces the soil's capacity to oxidise methane (Fest et al. 2017; see Box 4).

In the Waikato region, Didham et al. (2009) tested the effects of livestock exclusion and mammalian pest control, on leaf-litter invertebrate communities in 30 heavily fragmented forest remnants and larger forest reserves. For key taxa, such as Diplopoda, Isopoda, Coleoptera , Mollusca, Thysanoptera and Formicidae (Hymenoptera), densities were 10- to 100-fold higher in remnants with pest control, particularly where livestock were also excluded. This work was followed up by Denmead et al. (2015) who tested the livestock trampling effects on land snail communities in Waikato forest remnants using simulated trampling under field conditions. They found that even at very low frequency, trampling caused severe changes to land snail communities. The underlying drivers of changes in those communities varied, but were primarily linked to leaf-litter mass although litter and soil moisture contents, and lack of soil compaction, also had significant positive effects on snail density independent of the experimental treatment effects. Denmead et al., concluded that in NZ in particular, the absence of ungulate trampling in the evolutionary history of ecosystems may have made invertebrate communities more vulnerable.

Because organic matter decomposition is predominantly mediated by the soil biota, and is a process that underpins nutrient cycling and the provision of plant nutrients (Carlesso et al. 2019), such major reductions in soil invertebrate communities in the presence of stock and feral introduced herbivores could cause significant changes to soil carbon and nutrient processes.

Reducing the capacity of soils to retain moisture adds to vegetation stress during dry periods and is likely to contribute to greater tree mortality during periods of drought. Kumbaski et al. (2010) found that long term red deer grazing in a Turkish woodland reduced litter mass and caused crucial changes in a range of soil characteristics, including significantly increased compaction, higher soil bulk density, less saturation capacity, lower soil pH, and less organic carbon content.

The forested Haida Gwaii Islands (1 million ha.) off the coast of British Columbia have been isolated from continental North America for around 10,000 years since the last glaciation, and, until 130 years ago, were ungulate free (Gaston et al. 2006). Maillard (2019) studied the soil response to the colonisation and then culling of Sitka black-tailed deer. The study found that the high foot pressure of ungulates induced physical compaction that was significantly higher on islands with over 70 years of deer presence, than on islands never colonised by deer. In addition, soil water content and total phosphorus were significantly lowered after 70 years of deer presence. Trampling of soil by deer also slowed litter decomposition, reducing litter quality and modifying microbial community structure. Most of these effects only became apparent after the long periods of time.

The impact on shrubland soils is similar, with large herbivore grazing being correlated with drier, more compacted, soils and nutrient loss. For example, Bassett et al. (2005) found that soil compaction affected mānuka and cabbage tree seedling root development by increasing soil strength and decreasing oxygen availability. In Colorado, after 16 years of excluding deer in montane mixed grassland–shrub/woodland habitat, the mean increase of soil carbon concentrations in exclosures was +14%, or just under 1% per year. Soil bulk density in the exclosures was 25% less and soil moisture was 15% higher than in grazed areas. In a similar study in remnant grassy woodlands in southern NSW, Spooner et al. (2002) found fenced sites also had less soil surface compaction, significantly higher numbers of tree recruits, and significantly greater cover of native perennial grasses, with less cover of exotic annual species than in the grazed sites.

Box 4: methane oxidation by native forest soils – impact of soil compaction

Soils can reduce methane emissions through the action of methanotrophs, a group of soil bacteria that oxidise methane to use it as a source of energy. Most soils host methanotrophs, but “pristine” Aotearoa/New Zealand beech forest soils have some of the highest rates of methane consumption in the world with the measured rates about 6.5 times higher than rates reported for most Northern Hemisphere forest soils. Nearly half of all the country’s soil methane oxidation occurs in beech forests (Price et al. 2004).

Most of the methane oxidation, which averages -10.5±0.6 kg CH_4/ha/yr, occurs beneath the organic horizon not the organic (litter) layer, with the rate mainly influenced by soil water content, which in turn is determined by the extent of the soil’s air-filled spaces. When the soil airspaces are restricted and/or filled with moisture there is less CH_4 oxidation (Fest et al. 2017). Organic matter in the mineral soil reduces soil bulk density, and increases porosity and the diffusion of gas. The low bulk density allows greater root penetration, which in turn contributes to a greater porosity.

The methane oxidation rates increased through a sequence of naturally regenerating kānuka shrublands, increasing from -1.5kg CH_4/ha/yr in unimproved pasture, to -5.1kg CH_4/ha/yr in 80 year-old kānuka shrublands (Price et al. 2010).

The contribution to greenhouse gas reductions from soil methane oxidation is nationally significant, and has been calculated to be a sink for 147,000 tonnes of CH_4 per year, which is equivalent to around 12% of the 2018 agriculture CH_4 emissions, or nearly 7% of the country’s reported 2018 net greenhouse gas emissions (-3.7MtCO_2e/yr).

Price et al. (2001) suggested that our native forests soils represented an undisturbed low N-input natural forest, that could be considered to be a pre-industrial, pre-agricultural benchmark for temperate forests worldwide. By contrast our pasture soils have very low measured methane oxidation rates that are associated with high mineral-N concentrations and soil compaction due to intensive grazing. The description of "pristine", "pre-industrial, pre-agricultural" aoteroa/New Zealand native forest soils by Price et al., does not take account of 130 years of introduced herbivore grazing, with associated soil compaction.

If large herbivore control can reduce forest soil compaction and increase average methane oxidation by a tenth, this could contribute a further sink of -370,000 tCO_2e/yr.

How do woody ecosystems respond to the removal of introduced herbivore pressure?

Forests

Using fenced plots to exclude deer and/or goats is a very obvious way to observe the impact of large introduced herbivores such as deer and goats on forest carbon sequestration rates and on overall forest carbon storage. Outside the exclosures (which are usually still accessible to possums) there is little in the way of understory shrubs, seedlings or in some cases even ground cover. The non-canopy vegetation that remains is often limited to unpalatable shrubs, ferns, and grasses. Yet inside the exclosures there is usually a mass of shrubs, seedlings, and saplings of palatable canopy species and ground cover such as palatable ferns, grasses, moss, and leaf litter. In some places where there have been drastic reductions of large deer populations by commercial hunters using helicopters, such as in Mt Aspiring National Park during the 1970's and 80's (Mark 1989), there has been significant vegetation recovery (Figure. 6). These stark differences point to the strong impact the introduced herbivores have on species richness and habitat structure within the forest and how their presence is likely to alter forest composition over time.

However, despite these stark biodiversity differences, many of the studies have shown there is not a lot of difference in the amount of biomass between the grazed and ungrazed forest sites. This seemingly contradictory conclusion is because although they are often absent in grazed areas, the palatable forest floor vegetation, seedlings and understory shrubs account for a relatively small amount of the total forest biomass compared with the biomass that has been stored in the forest canopy tree over many decades or centuries. However, as Figure 6 indicates, the prevention of recruitment of canopy species has the potential to have a disproportionate impact on a forest's future biomass.

Figure 6: Beech forest in Mt Aspiring National Park, showing the benefit of introduced deer control from intensive commercial helicopter hunting during the period of the establishment of the wild venison export market and the beginning of the deer farming industry. Left; February 1970. Right; the same spot, February 1999. [Photos courtesy of: Prof, Sir Alan Mark.]. When this site was revisited in 2007 the view from the same photo point was completely obscured (Hunter; 2009. p.265)

A review of numerous exclosure plot studies showed that the transition toward unpalatable species caused by deer and goats is reversible - provided there are still local seed sources available (Stewart et al 1987, Nugent et al. 2001, Urlich et al. 2007). Seedlings of highly palatable, shade-tolerant, broad-leaved trees usually establish within a few years of exclusion, and a dense understorey reminiscent of the pre-browser environment is eventually recreated. Outside of these exclosures, the vegetation is typically made up of only a small number of plant species that browsers will not eat, or those few that are browse-tolerant. Even with suitable conditions, introduced herbivore control may not return the natural diversity of plants to the landscape. Once a forest understorey has been depleted, without almost total elimination of herbivores, it takes only a small amount of browsing to prevent recovery of the most highly edible species.

Because seedlings of favoured species are preferred over litterfall (which provides a large part of the browser diet in a depleted forest), as long as large herbivores remain in that ecosystem, the' tasty' plants will be targeted first because hungry, wide-ranging herbivores will consume any of these seedlings they can find. Complete removal of ungulates, rather than simply reducing their densities, may be required for recovery in heavily browsed forests (Wright et al. 2012).

For the less-palatable, less-preferred species the relationship is more linear, with species affected only at high browser densities, or not at all. The largest impact of introduced herbivores on forest carbon sequestration occurs when ground-based and canopy-based browsing increase disturbance at the same time, as is the case when possums and deer or goats are both uncontrolled in a forest. Then recovery of the natural diversity of tree species is disrupted or prevented (Fraser 2000, Peltzer et al. 2012)

In contrast to many shorter New Zealand studies, a 40-year study of moose exclosure on Isle Royale, Michigan, found tree biomass, tree sequestration rates and litter production (+40tC/ha and +1.45tC/ha/yr and +0.6tC/ha/yr respectively) were all significantly greater in moose exclosures than in browsed plots (McInnes et al. 1992). At the same sites soil nutrient availability and microbial activity, including total carbon and nitrogen, nitrogen mineralisation rates and microbial respiration rates were uniformly higher in the moose exclosures than outside (Paster et al. 1993).

While most plant species adversely affected by possums on Kapiti Island showed a rapid improvement in condition within 2 years of possum eradication (Atkinson, 1992), successful possum control does not always result in recovery, at least as measured by increasing foliage in the canopy. This indicates that for some trees, particularly stressed individuals, partial defoliation triggers an irreversible decline, or accelerates dieback caused by wind or by insect attack. Possums spend a reasonable amount of their time on the ground eating forest floor vegetation, including seedlings of preferred species that can form a reasonable proportion of their diets. Although established possum populations generally appear to have a lighter impact on their preferred food species and the direction of forest successions than deer (Nugent et al. 2001), they can have significant impacts on those ecosystems during the early stages of invasion, as happened recently in Northland (Figure 5b).

Tall forest sequestration rates in the presence of introduced herbivores

It is only since the early 2000s that we have had large-scale studies of changes in forest carbon sequestration rates. One of the first of these was reported in Kirschbaum et al. (2009) in which 180 indigenous forest plots were re-measured with a mean time between measurements of 17.4 years. The average annual change in live tree biomass carbon was found to be 1.98 (±1.24) tC/ha/yr. While the study's sequestration rate frequency distribution peaked strongly between 0–5 tC/ha/yr. (Fig. 6), it is interesting to note that some plots recorded annual carbon sequestration rates of up to 50 tonnes per hectare. The results suggested that the modest (though not statistically significant) average gains in carbon could have been higher were it not that a relatively small number of plots experienced significant mortality of trees that countered the gains from new recruitment and growth of existing stems in the majority of the plots.

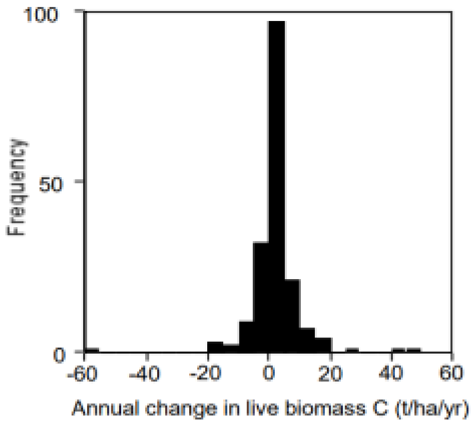


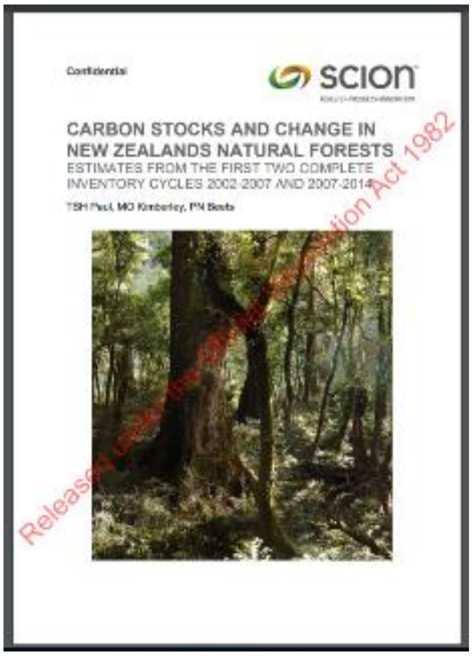
Figure 7: Frequency distribution histogram of annual change in live tree carbon for 180 permanent 20x20m plots located in indigenous forest. Mean = 1.98 (±1.25) tC/ha/yr. Some stands reached sequestration rates of +40 to 50 tC/ha/yr. [From Kirschbaum et al. (2009)]

Weighting the modelled carbon-accumulation rates for the different species by their areal distribution, which is strongly beech-dominated, resulted in a national mean indigenous forest carbon sequestration rate of 1.4 t C/ha/yr (5 t CO_2/ha/1yr) during the first 100 years of stand growth. This sequestration rate declines to 0.95 t C/ha/1yr (3.5 t CO_2/ha/yr) over a 200-year average, and to 0.62 t C/ha/yr (2.3 t CO_2/ha/yr) over 300 years.

For the purposes of our UNFCCC reporting the National Forest Inventory (NFI) established 1,051 permanent plots of pre-1990 natural forest throughout the country in the early 2000's (Figure 8a). Bellingham et al. (2014) reported early results of the re-measurement between 2002–07 and 2009–13 for the plots sited on public conservation land and found that there was a net annual increase in stored carbon of 0.56 metric tonnes per hectare.

Figure 8a: The 1051 National Forest Inventory plots in natural forests measured twice (red) or once (blue)during the two measurement cycles (2002-07 and 2009-14). From Paul et al. (2019)

Figure 8b: The final natural forests' carbon stocks change between the first 2 measurement periods analysis (Paul et al. 2019) that was released by the Ministry for the Environment under the Official Information Act, November 2020.



<u>The latest and most comprehensive analysis of the National Forest Inventory plots.</u>

The most recent comprehensive analysis of the full 1,051 permanent plots; *Carbon Stocks and Change in New Zealand's Natural Forests – Estimates from the first two complete inventory cycles 2002-2007 and 2007 – 2014", Paul et al. (2019)* was carried out by the forest research institute (SCION) as a confidential report for the Ministry for the Environment in June 2019. This report was only made public after an Official Information Act request by Forest & Bird in November 2020 (Figure 8b).

The report showed that nationally our natural forests are in carbon "balance" and showed no significant carbon stock changes between the 2002-07 and 2007-14 measurement cycles (Paul et al. 2019).

Dividing the plots into tall forest and regenerating forest, there was an indication (though not statistically significant) that when all the tall forests were combined they had a small decline in carbon stocks averaging -0.3± 1.6 tC/ha. Looking at each forest type separately, only one forest type had a statistically significant change in its carbon stocks. This was a significant decline (-9.0 ± 7.2 tC/ha) and occurred in kāmahi-podocarp forest, a widely distributed forest association. However, the report did not explore the significance of this finding (Paul et al. 2019).

Figure 9: Kāmahi-podocarp forest association, is the most common forest association in New Zealand and covers approximately 800,000 ha, mainly on the West Coast of the South Island. The kāmahi in this forest type are particularly susceptible to possum browsing that can kill individual trees.

Kāmahi-podocarp forest (Figure 9) is the most common of the 20 tall forest associations measured (making up 9.4% of all tall forest plots, and covering approximately 800,000 ha, mainly on the West Coast of the South Island), and had the greatest number of measured NFI plots (n=86). The statistically significant decline represented a loss of 26 million tonnes of CO_2e between the measurement periods, or an annual loss of -3.4 $MtCO_2e$/yr. This level of carbon loss is equivalent to 6% of the country's reported net 2018 greenhouse gas emissions. At March 2021 carbon prices this would be equivalent to a loss worth around $132 million a year.

Worryingly, the three forest types with the next largest declines in carbon stocks (the declines were not statistically significant at the P<0.05 level) were also associations that included kāmahi (Table 6). Between them these three forest types made up another 10% of the national tall forest area, with the number of sample plots ranging from 59 to 21. Between them these three forest associations may have lost around 6.6 million tonnes of CO_2e between measurement periods.

Table 6: The four tall forest types to show the greatest mean loss of carbon between the 2002-07 and 2007-14 measurement periods. The mean carbon loss in the kāmahi-podocarp forest type was statistically significant, while for the other three forest types the mean loss was not statistically significant at the P<0.05 level). [Adapted from Paul et al. (2019)]

Forest type	No# plots	Change in total carbon (tC/ha)	Forest type Area: (000 ha)	Total C loss between 2002-07 and 2007-14 (MtC)	Total CO_2e loss between measurements ($MtCO_2e$)
kāmahi-podocarp	86	**- 9.0** (±7.2)	794	**- 7.15** (-12.9 to -1.4)	**- 26.20** (-47.3 to -5.1)
kāmahi	59	- 2.3 (±6.4)	371	- 0.85 (-3.2 to 1.5)	-3.1 (-11.7 to 5.5)
Silver fern-kāmahi	42	- 2.1 (±6.8)	348	- 0.73 (-3.1 to 1.6)	-2.7 (-11.4 to 5.9)
Hardbeech-kāmahi	21	- 3.3 (±9.8)	68	- 0.22 (-0.9 to 0.4)	-0.8 (-3.3 to 1.5)
Total			**1,581**	**- 8.95**	**-32.8**

While Paul et al. (2019)'s key conclusion was that overall our tall natural forests are in a state of equilibrium, within the report was the important, but little commented on, fact that one of the forest types that is most susceptible to possum browsing (kāmahi-podocarp) is in significant decline. Given that this is the single largest forest type represented in the survey, it is probable that if it had not been in decline, the larger picture could have been that, taken together, our tall forests had gained carbon.

Likewise, the fact that the other forest types that recorded declines (although these were not statistically significant) also contained highly preferred kāmahi or other broadleaved forest associations, was not acknowledged or commented on in the report. Again, this trend of carbon storage decline points to the possibility that if these forests had received sustained control of introduced herbivores over the last two decades, they may have instead recorded an increase in carbon. Windley et al. (2016) found that of the 5 species studied in the Tararua Ranges (kāmahi, toro, rimu, hinau and mahoe), kāmahi had the strongest relationship between foliage nutritional quality and possum browsing. They considered that the seasonal availability of nitrogen in a dominant diet species such as kāmahi could be a major factor determining the likely impact of invasive possum populations.

The way this overall "balance" is represented in the SCION report, and much of the New Zealand scientific literature, has two interesting aspects. The first is that the "balance" finding is in line with a long established ecological trope that mature forests are generally in an equilibrium where primary production is equal to biomass loss through decomposition and respiration (this trope is explored in the Box 5). The SCION report concluded that the presence of an equilibrium state suggests that there have not been major external factors impacting on growth or mortality over recent decades *"or that positive and negative effects from such factors occurred at the same level balancing each other out".* (p.4 Paul et al. 2019)

It is disappointing that the SCION report does not explore which of these two possibilities was at work, despite the report's own evidence that the loss of carbon was occurring in the forests containing significant proportions of kāmahi and other broadleaved species that are highly palatable and susceptible to browsing. We argue that the second assumption appears to have been in play; that the positive carbon sequestration in our tall forests appears to have been 'balanced' out by the loss of carbon caused by the damage done by introduced herbivores.

The second aspect is that the "balance" is presented as a finding that implies the country does not have to be particularly worried about how the management of our natural ecosystems might impact our international climate change commitments.

The Ministry for the Environment kept this report confidential from June 2019 until it was released under the Official Information Act in November 2020. In the intervening one and a half years it seems that little was done to address the significance of the kāmahi-podocarp forest carbon loss. This lack of apparent action may be due to the emphasis on the report's key finding that *overall* our tall forests were in carbon 'balance'. While the Climate Change Commission references the SCION report in its Draft Supporting Evidence for Consultation, this is only to note that "mature natural tall forest store around 920 tCO_2 per hectare" (CCC *b* 2021).

Possibly more concerning than the Ministry for the Environment's unnecessary confidentiality and delay surrounding the public release of the SCION report, or the apparent lack of urgency about the significant carbon loss in the country's most extensive forest type, is the big delay between the end of data collection and the production of the analysis. The last native forest plots were measured in 2014, however, the detailed analysis of the changes in carbon stores only became available at the end of 2020. Given the need to understand and manage the changes in the country's massive native forest carbon stores, this is a disappointingly long lag time.

The country is looking in the rear-view mirror and finding out about worrying changes in the health of native forest carbon stocks nearly a decade after they occurred.

A statistical power analysis carried out by Holdaway et al. (2012) revealed that the ability to monitor changes in carbon stock using plot-based methods is limited to sizes of greater than 0.5 tC/ha/yr, *"…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 gains".*

However, the troubling results of the first two cycles of the natural forest carbon inventory, the millions of hectares of natural forest, and the size of the carbon stocks that are at risk of degradation by introduced herbivores, suggests that the financial and carbon costs of expanding the number of plots would more than repay a more statistically sensitive (powerful) analysis of a greater number of forest types, particularly those sensitive to browsing.

BOX 5: The assumption that old growth/mature forests don't sequester carbon.

The finding by Paul et al. (2019) that New Zealand's tall forests are in overall carbon 'balance' supports previous assumptions and estimates (Kirschbaum et al. 2009, Craswell et al. 2012, Burrows et al. 2018, Holdaway et al. 2014) and coincides with a long-held assumption that old-growth forests are generally carbon-neutral.

However, this assumption is being increasingly challenged (Luyssaert et al. 2008; Keith et al. 2008; Pugh et al. 2019). In Aotearoa/New Zealand, ecosystem disturbance is now seen as having a central role in structuring both conifer-angiosperm and beech forests, and in maintaining forest diversity at the landscape level. These ecosystems are seen as dynamic and regularly subject to natural and human-induced disturbance (Wyse et al. 2018).

Stephenson et al. (2014) reviewed the assumption that after the initial increasing growth period, the growth rate of individual trees declines as the tree size increases. This review conducted a global analysis that directly estimated growth rates from repeated measurements of 673,046 trees belonging to 403 tropical, subtropical, and temperate tree species, spanning every forested continent.

They found that for all continents, aboveground tree growth rates (and hence, rates of carbon gain) for most species increased continuously with tree size. In absolute terms, trees 100 cm in trunk diameter on average add 51.5 kg of aboveground C each year (ranging from 5 kg to 100 kg depending on species). This is nearly three times the rate for trees of the same species at 50 cm in diameter, and is the equivalent to adding an entirely new tree of 10–20 cm in diameter to the forest each year.

Although growth efficiency often declines with increasing tree size, this is compensated for by increases in a tree's total leaf area, which causes the whole-tree carbon accumulation rate to increase. So it is usually tree population dynamics, especially mortality, that contribute to declining productivity at the scale of the forest stand.

It may not only be the forest trees that are accumulating carbon in old growth forests. Between 1979 and 2003, Zhou et al. (2006) measured a total of 230 soil samples in 'old growth' forest (>400yrs old), and showed that soil organic carbon stock in the top 20-cm soil layer increased significantly during that time ($P < 0.0001$), with an average rate of +0.61±0.07 tC/ha/yr. The study suggested that the carbon sequestration in forests' belowground systems supports the establishment of a new, non-equilibrium, carbon budget framework in old-growth forests.

The results of a recent NIWA study (Steinkamp; 2017) indicate a larger net CO_2 sink across the country than has been previously assumed by the National Inventory Report (NIR) (MfE; 2020). The results suggest both stronger photosynthetic and respiratory activity than previously modelled, particularly in the forests of the south-western South Island. For example, Fiordland appears to have taken up between +6.0 and +18.5 MtC each year in 2011–2013. By comparison, previous modelling estimated a range from 0.0 to +0.82 MtC/yr. These larger than expected carbon sinks could be explained by the NIR modelling assuming a steady state ('balance') for the natural forests, when they are actually accumulating carbon in biomass – possibly recovering from past disturbance.

Should we be managing our natural ecosystems using an 'equilibrium-steady state' framing, when they are subject to regular natural disturbance and are currently also depleted relative to their natural (non-mammalian herbivore) state, or would it be better to manage these ecosystems to absorb more carbon on their way to recovering their pre-introduced herbivore state?

Shrubland sequestration rates

Shrublands are generally considered to occur over 2.5 million hectares of Aotearoa/New Zealand either on their own, or mixed with grassland and forest. However, a recent fine scale image analysis of farmland in Northland detected an additional 11.7% and 14.3% woody vegetation cover than the standard LUCAS and LCDB methods respectively (Case & Ryan; 2020). Much of this of was younger regenerating vegetation, which led to an estimate of between a third and three times as much carbon sequestration on farmland landscapes than previously reported.

Many shrubland types are temporary, forming an early stage in the succession to forest. Others are permanent, growing in relatively harsh environments, such as exposed coasts, wetlands, infertile soils, alpine areas, and very dry hill country, where trees fail to prosper. Natural forest succession usually begins with grassland areas reverting to shrubland and later being replaced by species of the mature forest. Grasslands began declining in the early to mid-1980s after farming subsidies were removed. Abandoned agricultural land is usually colonised by shrubland consisting of mānuka and/or kānuka, and/or introduced scrub species such as gorse and broom.

These shrubland species are an important carbon sink (Trotter et al. 2005 Paul et al. 2019). During the first 35–50 years, higher rates of net carbon sequestration can be expected than for indigenous-forest growth (Kirschbaum et al. 2009). Wiser et al. (2011) suggest that 45% (c. 670,000 ha.) of the total pre-1990 shrubland area shows evidence of recruitment of indigenous tree species, including kānuka, and the palatable māhoe, marble leaf, kāmahi, and fivefinger.

The recent SCION review of 134 national plots representing 8 shrubland vegetation types between 2002-07 and 2009-14 (Paul et al. 2019) gives the most reliable national estimate of shrubland sequestration rates. Total carbon in all of the regenerating forest types increased between measurements by a statistically significant average of +4.8 tC/ha, which gave an average sequestration rate of +0.62 ± 0.26 tC/ha/yr. Associations with kānuka and tall shrubland sequestered + 0.87±0.38 tC/ha/yr, while kānuka shrublands with coprosma and mingimingi also showed higher sequestration rates of +1.05 ± 0.74 tC/ha/yr (table 7).

Table 7: Estimates of total carbon stock changes for shrubland vegetation types between 2002-2007 and 2007-2014. Numbers in bold represent statistically significant changes at the P<0.05 level). Adapted from table 9 of Paul et al. (2019).

Shrubland type	**Number of plots**	**Carbon changes between measurement periods (tC/ha)**	**95% Confidence intervals**
Kānuka shrubland with *Coprosma* and prickly mingimingi	24	**+8.1**	**±5.8**
Grey scrub with kānuka	30	**3.3**	**±2.8**
Mānuka shrubland	5	0.0	±3.6
Matagouri shrubland	1	1.0	
Turpentine scrub – *Gaultheria* montane shrubland	9	0.6	±1.4
Gorse shrubland with cabbage trees	5	-9.2	±16.0
Total	**74**	**3.5**	**±2.5**

These national averages are lower than several previous smaller studies that measured carbon gains in a range of regenerating shrublands and suggested national mean sequestration rates for mānuka/kānuka shrubland at about +2.2 ±0.3 tC/ha/yr, with the highest rates measured in cool, moist, high fertility sites. Some kānuka carbon stocks even approached rates modelled for the first 20 years of carbon accumulation for planted pine stands in the same region (Trotter et al. 2005, Kirschbaum et al. 2009), potentially challenging the well-established trope that to tackle climate change through woody carbon sequestration it is best to use fast-growing plantations of exotic forest species (see box 6).

Carswell et al. (2012) noted that the rate of carbon sequestration over the first 50 years (c. +2.3 tC/ha/yr) was the same for the kānuka–red beech succession at Hinewai, Banks Peninsula, as for the coastal broadleaved succession in the outer Marlborough Sounds. They considered that their measured average above-ground carbon stock of 145 ± 19 tC/ha in the coastal broadleaved succession probably represented the upper end of potential carbon stocks for this forest type as a result of extensive wild animal control. This observation about herbivore control may be the clue as to why these and other studies reported higher sequestration rates than the more representative SCION report (Paul et al. 2019).

Regenerating forests have smaller total carbon pools than tall forests, but have high net rates of carbon sequestration and can therefore be considered strong carbon sinks. However, the average increase in live above-ground carbon is greater in tall forest (+1.29 tC/ha/yr) than in regenerating forest (+1.05 tC/ha/yr). Tall forests also have greater losses in carbon from mortality and it is this that offsets the higher gain in carbon from the growing trees (Paul et al. 2019). Constant recruitment is necessary in tall forests to offset mortality losses that replenish the woody debris pool. The much lower level of mortality in regenerating forests is the reason they consistently show higher net gains in carbon. This is also the reason that regenerating forests are potentially very responsive to the control of introduced herbivores.

The largest positive effects of herbivore control (carbon sequestration rate increases of +1-2 tC/ha/yr) are likely to occur in localised low altitude sites with fertile soils and highly palatable early-successional vegetation, with high herbivore densities where control triggers rapid development of woody vegetation (Holdaway et al. 2012; Bellingham et al. 2014). There may be a time lag in any response to herbivore control, and it may take many years after a control operation before a biomass response is measurable. However, the long-term effects of forest succession, on future forest types, on biomass carbon and total carbon, and on biodiversity is likely to be profound (Burrows et al. 2008).

Box 6: Exotic forests: quick sequestration, but even quicker emissions?

Exotic plantation forests are often promoted for the rapidity by which they sequester carbon. Current *Pinus radiata* plantations cover around 2 million hectares and hold an estimated 230 Mt of above-ground carbon (13% of the national total; Table 2). Once planted, these forests have high rates of carbon accumulation, but they are usually harvested after 25 - 30 years. At harvest, between 15 and 24% of the trees' biomass (foliage, branches, stumps, etc.) is left on site as residues (Viser; 2018) to decompose within a few years, returning carbon to the atmosphere as CO_2.

Some 53% of the national harvest of logs was exported in 2015, with 96% going to China, South Korea, and India. The carbon stocks in the products manufactured in China from the exported logs are halved in just under two years, in South Korea in just over 12 years, and in India in less than one year (Manley & Evison 2017). The aggregate decay curve for the three countries means that the carbon stocks are halved in just over 2 years. This means that less than 1% of the original exported log biomass remains after 15 years.

When the current decay curves for both domestic and export log products taken from the harvest site are calculated, the minimum value of carbon stocks after harvesting a 'typical' stand, at age 28 years, increases from 54 to 91 tC/ha. This means that for the purposes of removing CO_2 from the atmosphere, pine plantations are 'stuck' at sequestering an average of less than 100 tonnes of carbon per hectare over the timescale of multiple 25-30 year rotations.

Although indigenous forests deliver larger permanent carbon stocks than exotic forests in the medium to long term (Kirschbaum et al. 2009), the rapidity at which exotic forest biomass returns to the atmosphere after harvesting, during processing and through relatively short product half-life, means that 'average' regenerating indigenous forests are likely to achieve carbon stock 'parity' with a typical exotic plantation forest in around 3 exotic forest cycles.

Native tussock grasslands

Excessive grazing is identified as one of the key causes of global grassland degradation and soil carbon loss (Yu et al. 2019). Aotearoa/New Zealand's tussock grasslands, located mainly in the South Island and the central volcanic plateau of the North Island, have evolved in the total absence of mammalian grazing. Prior to the arrival of humans, the extent of the non-alpine tussock grasslands was more restricted, occurring mainly in the drier inland parts of the eastern South Island and areas in close proximity to the Ruapehu and Tongariro volcanoes where natural fire events (lightening and volcanic respectively) held back their natural succession to shrublands and forest.

Māori fires saw a rapid expansion of subalpine tussock grasslands, and after the arrival of European settlers the grasslands rapidly degraded in response to the effects of the novel grazing of livestock (sheep and cattle) and feral mammals (the large herbivores, as well as hares and rabbits), further burning, and invasion by the weed *Hieracium*, all of which significantly reduced biomass carbon (Ausseill et al. 2014, McIntosh et al. 1997).

McIntosh (1997)[3] summarised the decline of above and below-ground biomass along an eastern South Island degradation sequence over 750 years of human occupation and landscape modification. Initially this ecosystem was mountain beech (*Fuscospora cliffortoides*) forest, which became tall tussock grassland, then a short tussock grassland, and has ended up as grazed, degraded *Hieracium* herbfield. In terms of ecosystem carbon, this represented a decline from 173 tC/ha through 32–35 tC/ha and 11 t C/ha to a mere 1–2 tC/ha.

Grazing impacts on tussock grasslands and alpine plants

Using a network of 111 permanent plots in 8 catchments covering the majority of their range, Cruz et al. (2017) studied the long-term impacts of the introduced Himalayan tahr on tussock grasslands. They found a 'highly-vulnerable' relationship with the total vegetation cover declining most rapidly as tahr activity increased from low levels. Tussock height (biomass) declined significantly with increasing tahr impacts. Even at very low tahr activity levels, many plant species other than tussocks were also highly sensitive to tahr browsing. Although the vegetation cover appeared to be recovering from high tahr densities prior to the 1970s, tahr nevertheless continued to fundamentally impact total vegetation cover and tussock height during the 1990–2013 study (Figure 10).

Figure 10: Vegetation in Zora Creek, Westland at two points in time showing the reduction in tussock cover and height (plant biomass) caused by tahr browsing (left image 1999; right image 2012). [Source: DOC & Maanaki Whenua Landcare Research (undated). *The effects of tahr in alpine and subalpine ecosystems*; DOC website.]

[3] quoted in Mark et al. (2013)

Figure 11: Left: Degradation of high country grasslands: St Bathans Range, North Otago. Sheep dispersed widely on the large burnt area to the left of the fence but were concentrated on the small burnt area on its right and have killed most of the tussocks. Right: Low-alpine slim snow tussock (*Chionochloa macra*) grassland, southern Old Man Range, Central Otago, showing fenceline differences in tussock height and cover after several years of retirement and protection from stocking. To the left of the fence *Aciphylla scott-thomsonii* is prominent. [Source: AFM photos, Feb 1959 & Jan 1991; from Mark et al. (2013)]

Relative to the maximum densities recorded for tahr and chamois, the biomass of possums in alpine habitats is low. Possums are more common in shrubland than in grassland, and the mean body weight of alpine adult possums is higher than found elsewhere, suggesting that the populations are limited by den sites rather than by food (Hickling and Forsyth, 2000). The long-term effects of possums on plant and animal communities in alpine habitats are not well known, but are likely to be small relative to the impacts of tahr and chamois.

Natural grassland carbon sequestration - with and without mammal browsers

The near elimination of deer from non-forest areas by helicopter-based hunters during the 1970s resulted in vegetation recovery in many areas (Figure 6). Just over a decade after commercial hunting began in eastern Fiordland, substantial tussock regrowth had occurred, with prolific establishment of snow tussock seedlings, and large herbs increasing in abundance (Fraser, 2000). Deer showed a strong preference for grasslands characterised by the snow tussock *C. pallens* and large-leaved herbs. These occur on fertile soils and showed the most recovery in response to commercial hunting, especially at lower altitudes. Little change occurred in the less-favoured grasslands characterised by *C. crassiuscula* and *C. acicularis* on infertile soils (Rose & Platt 1987).

A comprehensive assessment of three different South Island high-country grassland ecosystems retired from sheep and feral animal grazing for periods of 11–38 years found that most of the variation in total ecosystem carbon and carbon pools arose from site differences rather than the number of years without grazing (Burrows et al. 2012 reported in Mark et al. 2013). The annual increase of carbon through additional sequestration was estimated to have ranged from +0.3 tC/ha/yr at the mixed short-tussock and matagouri site (ungrazed for 20 years) to +0.8 tC/ha/yr at the short tussock with *Hieracium* site (ungrazed for 38 years).

After severe experimental defoliation, Lee et al. (2000) recorded tussock recovery sequestration rates of +0.2 tC/ha/yr in the first 8 years, and 0.4 tC/ha/yr in the following 12 years. McIntosh & Allen (1998) found that after 15 years there was a significant increase (43%, $P<0.01$) in biomass in plots that had excluded sheep and rabbits. Most of the difference in biomass sequestration (+0.29 tC/ha/yr) was from the near doubling of root mass (+0.16 tC/ha/yr) and increases in leafy growth (+0.09 tC/ha/yr), with the remainder (+0.03 tC/ha/yr) from litter (Table 8). Having no sheep or rabbit grazing for 15 years had little effect on soil nutrients or soil carbon. This lack of difference was attributed to indirect effects of grazing, such as soil erosion, continuing within the ungrazed areas long after grazing had ceased (McIntosh & Allen 1998).

Table 8: **Changes over 15 years in the biomass carbon stocks of seasonally dry high country tussock grasslands. Adapted from McIntosh & Allen (1998)**

	Treatment means (tC/ha/yr)		Difference in sequestration (Significance)
	grazed	**ungrazed**	
Herbage/Leafy growth	0.124	0.214	0.09 (NS)
Litter	0.280	0.313	0.03 (NS)
Roots	0.260	0.420	0.16 (P<.05)
Total	**0.663**	**0.949**	**0.29 (P<.01)**

Sheep and rabbit-grazed, ungrazed, ungrazed+fertilized, and ungrazed+irrigated treatments were applied in a replicated experiment on short-tussock grasslands at Luggate, that was sampled annually from 1988 to 2000 (Walker et al. 2003). The study found that grazing reduced the cover of tussocks and certain woody species. It did not decrease the dominance of exotic species, or maintain native species richness at a higher level than in ungrazed vegetation. There was a limited recovery of taller native species with grazing removal alone. However, removing grazing, and providing 12 years of nutrient enrichment, promoted the growth of native tall shrubs and tussocks while not increasing the dominance by exotic species.

Historically these areas would have benefited from ocean nutrients bought inland by burrowing seabirds in their many millions. Succession of short tussock grasslands towards taller native tussock-shrubland communities (with corresponding carbon sequestration) may be achieved by on-going nutrient enrichment in the absence of grazing. The results supported the conclusion of McIntosh and Allen (1998), that grazing removal alone may not be a realistic option for native vegetation rehabilitation in short tussock grasslands in the short and medium term.

Table 9: Reported increases in carbon sequestration rates from introduced herbivore removal from Tussock grassland habitats.

Study	**Lower end sequestration (tC/ha/yr)**	**Upper end sequestration (tC/ha/yr)**
McIntosh & Allen (1998)	-	0.29
Lee et al. (2000)	0.2	0.4
Burrows et al. (2012)	0.3	0.8
Mean	**0.25**	**0.49**

Impact of introduced browsing on grassland soil carbon sequestration

Soils of grasslands also represent a large potential reservoir for storing carbon, but this potential depends on how grasslands are managed for large mammal grazing. Many studies have found both strong positive and negative grazing effects on soil organic carbon, but the reasons for this variation have been poorly understood. McSherry & Ritchie (2013) analysed a sample of 17 studies that compared a grazed sample plot to an ungrazed plot and reported the effects of grazing on soil carbon density, together with soil structure.

They showed that soil texture, rainfall, the way the grass photosynthesised (whether C_3 or C_4 species[4]), grazing intensity, and study duration, as well as soil sampling depth explained a high (85%) portion of the observed variation. Increasing grazing intensity led to a reduction of soil organic carbon by an average 18% in the C_3 species dominated grasslands, such as those we have in Aotearoa/New Zealand, but it led to an increase of soil

[4] See https://en.wikipedia.org/wiki/C4_carbon_fixation

organic carbon by 6-7% on C_4 dominated and mixed grasslands. At sites dominated by C_3 plants with higher clay content soils as well as higher levels of rainfall, removal of grazing had strong positive effects on soil organic carbon levels, with annual changes as large as ±1.5 tC/ha/yr.

Yu et al. (2019) synthesised data from 63 sites in the literature, plus 15 sites in their field study that investigated the dynamics of soil carbon stocks following grazing exclusion in alpine grasslands of the Tibetan Plateau. Soil carbon increased with grazing exclusion at most sites, with average sequestration rates of +1.91 tC/ha/yr to soil depth of 0–30 cm. The rates of change in soil carbon were positively related to increased rainfall. Because of reduced moisture infiltration and root penetration, high levels of soil compaction from grazing animals restricted the recruitment of deep-rooted native species and favoured surface-rooted species such as exotic annuals.

Kauffman et.al. (2004) found that after 9 to 18 years the mean rainfall infiltration rate in dry meadow exclosures was nearly 1,200% greater than in grazed dry meadows. Fine-root biomass was 56% greater in the ungrazed compared to the grazed dry meadows. This significantly higher mass of fine roots in ungrazed communities increased the capacity of streambanks to resist erosion. They calculated that saturated soils of the surface 10 cm in a hectare of ungrazed dry meadow could contain 61,000 litres more water than an equivalent grazed hectare. This estimate did not include the entire soil profile. The increase in soil moisture influenced ecosystem productivity and biogeochemistry, while stabilising soil temperature and stream flows.

From grasslands to scrubland and forests

Cheng et al. (2011) showed that 20 years of stock exclusion on the Loess Plateau in China significantly reduced soil compaction at most sites, which in turn facilitated recovery of forest species.

Several New Zealand studies have investigated the benefit of converting marginal pasture land, mostly on private land, into indigenous forest for enhanced carbon sequestration through afforestation or natural reversion into shrubland (Trotter et al. 2005; Kirschbaum et al. 2009; Case & Ryan. 2020). This conversion would also provide benefits from increased erosion control (particularly in the North Island – see Box 7), enhanced biodiversity and other ecosystem services such as water yield, and less GHG emissions from removal of stock (Ausseil & Dymond 2010).

Increasing carbon storage through the afforestation of non-forest areas of conservation land has also been examined, with an estimate that afforestation, mainly through an increase in the areas of lowland podocarp–broadleaf forest, over many decades could add 461 MtC more than at present (Burrows et al 2013).

Box 7: Soil conveyor belt to the ocean floor

As earth is eroded the sediment in streams and rivers takes much of the soil carbon with it to eventually be deposited on the ocean floor to become ocean carbon sink deposits. In the South Island, erosion is dominated by natural processes in the Southern Alps. Soil erosion and regeneration of soils has been assessed to be approximately in balance. In the North Island, erosion is primarily caused by deforestation in hill country which began occurring during the mid-1800s. Therefore, the regeneration of North Island soils is not necessarily in balance with erosion. Overall, soil erosion results in a net sink of some -3.15 MtC/yr, made up of -0.85 MtC/year for the North Island, and -2.3 MtC/year for the South Island (Kirschbaum et al. 2009).

Opportunities for introduced herbivore/browser removal from natural ecosystems to mitigate climate change impacts

Case and Ryan (2020) used lower-end and higher-end published sequestration rate values for non-public conservation land woody vegetation to estimate the range of total annual carbon sequestration on sheep and beef farmland. As there are very few comprehensive studies of carbon sequestration rates in the absence of introduced animal browsing, we have adopted a similar approach.

To estimate potential annual sequestration rates in the absence of introduced herbivores we have used a similar methodology as Case & Ryan (2020) and grouped together the reported figures of extra carbon sequestration or emissions reductions that might be possible with sustained introduced herbivore control or eradication. Where such figures do not exist, we make the assumption that, in the absence of introduced animal browsing, sequestration rates will be at the higher-end of the published sequestration rates (Table 10).

Table 10: Annual lower-end and upper-end estimates of carbon sequestration and gross greenhouse gas emission savings associated with significant and sustained control of introduced herbivores in natural ecosystems of all tenures in Aotearoa/New Zealand.

Sequestration component	Sequestration rate ($tCO_2e/ha/yr$)	Extent of association (mha)	Potential annual CO_2 sequestration ($MtCO_2e$)
Indigenous tall forest	Lower end -0.9[1] Higher end +0.62[1]	5.19 [4]	- 4.7 + 3.2
Indigenous scrub and shrubland	Lower end +0.48[1] Higher end +2.82[1]	0.51 [4]	+ 0.25 + 1.4
Kānuka forest and tall shrubland	Lower end +1.8[1] Higher end +4.6[1]	0.20 [4]	+ 0.4 + 0.9
Woody vegetation soils	Lower end +0.05[2] Higher end +0.66[2]	8.2 [5]	+ 0.4 + 5.4
Tussock grasslands	Lower end +0.9[3] Higher end +1.8[3]	3.26 [5]	+ 2.9 +5.9
Savings of CH_4 emissions by feral introduced herbivores[6]			+ 0.31
Increased forest soil methane oxidation[7]			+ 0.37
Total	**Lower end** **Upper end**		**- 0.75 $MtCO_2e$** **+ 17.5 $MtCO_2e$**

1. Calculated from Paul et al. (2019) using their mean reported sequestration rates ± the 95% confidence levels; 2. from table 6; Carswell et al. (2008); 3. see table 8, p.28; 4. from Allen et al.(2013); 5. from table 1; Carswell et al. (2008); 6. see table 5; p.15; 7. assumed 10% improvement in methane oxidation capacity (see Box 4, p.17).

The estimate of a possible upper-end annual sequestration of an extra 17.5 $MtCO_2e$ resulting from the removal of introduced herbivores is between 8 to five times greater than earlier estimate of the direct annual consumption of natural vegetative biomass by introduced herbivores (1.9 – 3.66 MtC/yr; table 4, p12). This difference is not unexpected.

The introduced herbivores prefer to consume the nutritious young foliage, buds, flowers, fruit, and seedlings of the most palatable plants. In any yearly growth cycle this means that they are depriving those plants of the ability to grow and potentially store the season's extra biomass. Also, repeated consumption of a proportion of a plant's foliage (or bark in the case of woody vegetation) can eventually kill the individual branch or the whole plant or tree, which in the case of canopy and sub-canopy species can interrupt the process of forest renewal. The larger herbivores also impact the composition of the litter layer and the soil's physical structure. This means that the presence of these animals and a relatively small annual biomass consumption can have a disproportionally greater impact on an ecosystem's potential to sequester carbon. Where introduced browsing causes severe damage ecosystems can undergo a net loss of stored carbon.

These estimates indicate that in the medium-term a sustained effort to control introduced herbivores in our natural ecosystems would make a very significant contribution to the country's efforts to become carbon neutral. The upper estimates indicate that Aotearoa/New Zealand could even become carbon positive, assuming that existing programs and policies to reduce its net greenhouse gas emissions continue.

The mid-point between the lower- and upper-end estimates (-0.75 $MtCO_2e$/yr and +17.5 $MtCO_2e$/yr respectively) is the sequestration of an extra 8.4 million tonnes of CO_2e annually. The mid-point, and upper estimates are equivalent to 15%, and 31% respectively of the country's reported 2018 net GHG emissions.

Tanentzap et al. (2011) carried out a meta-analysis of the difference in carbon stocks that resulted from large mammalian herbivore exclusion using 108 studies from 52 vegetation types. 106 of these studies were from ecosystems where mammalian herbivores were a natural part of the studied ecosystem, with the two exceptions being from New Zealand. Their review concluded that removing mammalian herbivores across a range of vegetation types can result in changes of terrestrial above- and below-ground carbon stocks ranging from -1.65 tC/ha/yr to +5.77 tC/ha/yr.

If Tanentzap et al.'s results were applied to the approximately 11 million ha of natural ecosystems in Aotearoa/New Zealand they would result in an estimate of between -18 and +63 million tonnes of carbon per year from mammalian herbivore removal. That very rough estimate, plus the scale of the recorded carbon losses (-3.4 $MtCO_2e$) in the kamahi-podocarp associations that make up just 10% of New Zealand's native forest, would indicate that this study's estimates of potential sequestration in the absence of introduced herbivores are in the right 'ball park', if not conservative.

At March 2021 NZ carbon prices the mid-point value of 8.4 $MtCO_2e$ for potential extra sequestration from significant and sustained introduced herbivore control would have a value close to $330m/yr. The economics of this pest control would be well worth the investment.

It may not take long to see improvements in ecosystem health and resilience; with improved soil structure and moisture retention, the recovery of palatable species seedlings, forest floor vegetation, understory shrubs, and stabilisation of forest understory temperature, etc. Short-term priorities should focus on the protection of the existing carbon stores from continued decline, particularly where sustained herbivore control is focused on the forests with substantial components of palatable tree species - such as the live kāmahi component of kāmahi - podocarp forest associations and the collapsing native forests of Te Tai Tokerau/Northland that have recently been colonised by possums.

At a minimum, sustained introduced herbivore control will make a major contribution to preventing further degradation of the existing large natural carbon stores, in some cases catastrophically. Not only will this help meet our international obligations to *maintain* and enhance our existing carbon stores, it will provide insurance against likely future international requirements for nations to fully account for the changes in their natural ecosystem carbon stores.

Co-benefits of increased introduced herbivore control

Global temperature increases are already causing an increase in mean temperatures, with associated increased frequency of extreme weather events, including flooding and droughts that can cause natural vegetation loss both directly and indirectly. In terms of direct effects, flooding can cause landslides resulting in removal of vegetation, while drought stresses vegetation and makes it more susceptible to insect infestation and disease and eventually death. For example, partial forest loss can occur through individual tree mortality which has been seen with dying swathes of taraire in Te Tai Tokerau/Northland and beech forests in the Nelson region during extreme drought events over the past 12 years. In parts of the country that are likely to become drier, we can expect to see reductions in the extent of wetlands.

Indirectly, more frequent wildfires could increase the potential for total forest loss, which would have a significant impact on long-term carbon storage as our native flora is generally not adapted to fire. Warmer temperatures will favour conditions that encourage the spread of disease or pests which could result in dieback or loss of forests or individual tree species. There is also considerable potential for the loss of soil carbon if predicted higher temperatures stimulate higher soil respiration. With the absence of moisture during a drought, trees are much less able to sequester carbon. Increasing drought in eastern parts of the country is predicted to reduce carbon uptake, while sequestration may increase in western regions where temperatures will likely rise along with rainfall (Carswell et al. 2008; Mason et al. 2013; Buswell, 2016).

Removal of, or significant reduction in, introduced herbivore populations in native ecosystems will have many positive impacts beyond improving carbon sequestration and improving biodiversity. Healthy seedling and forest under-story vegetation will reduce forest floor temperatures, increase rain interception and assist moisture retention which in turn will benefit forest wildlife such as kiwi, and also reduce forest fire risk. Likewise, benefits to the litter, humus and soil layers in many ecosystems will increase moisture infiltration and retention rates that will help to reduce peak flood flows during extreme storm events, and extend the length of water flow during drought conditions and recharge aquifers.

As well as the carbon sequestration or biodiversity benefits that are likely to come from removing or substantially reducing introduced herbivore browsing of natural ecosystems, such reductions will remove a source of significant stress from these ecosystems at the very time that the stress from climate change will be increasing. This will provide an extra level of resilience for those natural ecosystems that should improve their ability to respond to climate change pressures and to help mitigate climate change impacts.

Box 8: The centrality of tangata whenua and Te Ao Māori for Aotearoa's ngāhere recovery

> Kei raro i ngā tarutaru, ko ngā tuhinga o ngā tūpuna
> Beneath the herbs and plants are the writings of the ancestors
> Whakataukī, in Waitangi Tribunal, *Kō Aotearoa Tēnei*:

Forest and Bird supports the Climate Commission's call for 'genuine, active and enduring partnership with iwi/Māori' (Climate Change Commission; He Poa a Rangi; 2021 a), and 'Consideration should be given to deeper exploration of the mātauranga relating to the realm of Tāne Mahuta with respect to sustainability, biodiversity, rongoāand traditional practices.' (CCC; 2021b, p.19).

The Commission's evidence affirms that treasured plant and bird varieties act as partners in mātauranga; these taonga species link Indigenous wisdom unique to these islands; healthy forests can help weave Māori relationships with the ngāhere and its lore (CCC; 2021b, chapter 6).

In its report on the flora, fauna and cultural and intellectual property claim (Wai 262), the Waitangi Tribunal emphasized the importance of healthy ecosystems as a matrix for the work of kaitiakitanga and Māori intergenerational wellbeing:

> All parties in this claim shared a concern for the state of the environment and the taonga within it; and all would agree that the survival and health of a species should be the first object of human engagement with it. For kaitiaki, there can be no relationship with taonga if the taonga no longer exist; nor, without the taonga, can the mātauranga survive. (p.340)

Transmission of unique bio-cultural heritage is imperilled though forest collapse. Tui Aroha Warmenhoven compared tribal hikoi through the Raukumara Range over one generation:

> 'Then came the birdsong and it was ear-splitting, but we took it for granted. I would never have believed that 30-odd years later [in 2019], I would go into a dead silence. That was frightening. This huge forest is empty, it's collapsing.' ... During her first hīkoi, she remembers an undergrowth so thick, the group had to cut their way with machetes. 'Jump forward to the hīkoi in 2020, there's nothing. A bit of horopito but mostly barren. It's a deadscape' (Meduna, 2021).

Preventing the advance of a 'deadscape' of biodiversity loss can address climate change, encourage the ongoing transmission of mātauranga Māori and help the Crown honour its Treaty obligations. As Bargh (2019) notes;

> ' In the case of environmental management, climate change and low-emissions transition policies, there is a history of Treaty breaches that have excluded Māori from the protection, restoration and enhancement of natural resources, while at the same time industries that exacerbate climate change have expanded. This suggests that 'vigorous action' is required for the Crown to protect Māori rights and interests…' (p.12)

Supporting iwi rangatiratanga in restoring forest health could form part of the partnership arrangements the Commission calls for. This work could be scaled up to boost both regional employment and climate responsiveness.

Discussion:

New Zealand has seen previous episodes of introduced browser induced natural ecosystem collapse. Many forests and grassland ecosystems have also been taken to the brink of collapse by deer, goat and possum browse, where only directly funded intervention or serendipitous events such as the rise of the venison recovery industry, or the occurrence of bovine TB infestations, has prevented widespread ecosystem loss. DOC data suggests that large herbivore populations throughout much of the country now exceed densities not seen since before the venison industry commenced.

Aotearoa/New Zealand's unique natural ecosystem's that have evolved over tens of millions of years in the absence of mammalian herbivores, and this study has highlighted the particular vulnerability of these ecosystems' carbon stores to the negative impacts of a suite of feral introduced herbivores. It has attempted to highlight how significant those impacts are in the context of the country's present commitment to reduce its greenhouse gas emissions (Table 11).

Table 11: Key numbers for understanding the importance of natural ecosystem carbon stores and the threat of introduced herbivores to those carbon stores (numbers in red represent greenhouse gas emissions, in green represent sequestration):

	Carbon million tonnes (Mt)	Carbon dioxide equivalent (Mt CO_2e)	Percent of reported 2018 net GHG emissions
Above-ground carbon stored in natural vegetation[1]	**1,456**	**5,343**	**9,600%**
Reported net greenhouse gas emissions for 2018[2]	-15.1	-55.5	100%
Direct vegetation consumption plus methane produced by feral introduced herbivores (0.6-1.1 MtC: mid-point 0.85 MtC)[3]	-0.85	-3.1	-5.6%
Annual biomass loss in kamahi-podocarp forests (mid-2000s to mid-2010s)[4]	-0.93	-3.4	-6.1%
Potential extra sequestration from sustained feral introduced herbivore control (0.38-5.2 MtC: mid-point 2.8 MtC)[5]	+2.3	+8.4	+15.1%

1. See table 2; 2. Ministry for the Environment (2019); 3. See tables 4&5.; 4. See table 6.; 5. see table 10.

So why hasn't the government swung in behind a concerted control programme for large introduced herbivores? Part of the answer probably lies in the history of the formation of the Department of Conservation. In 1987 when the Department was formed from an amalgamation of the New Zealand Forest Service, the New Zealand Wildlife Service and the Department of Lands and Survey, none of these agencies had an existing budget or work force focussed on control of introduced herbivores. In the 1950's the Forest Service was a significant agency for such control, but by 1987 this had been lost due to the rise of commercial venison recovery. For 20 years high venison prices made it economic for private operators to control most of these animals to very low levels in much of the country. So by the time the Department of Conservation was formed, no pre-existing budget and staff for wild animal control expertise came with it.

Parsimonious budget allocations through the following decades has seen real demands vastly outstrip financial allocations. Coalition politics also led to the establishment of the Recreational Hunting Advisory Council which has meant that large animal pest control policy has been strongly influenced by hunting advocates and does not have a strong ecological basis.

Another part of the answer may be the disappointment that the WACEM research concluded that, although there would be carbon sequestration benefits from wild animal control, the National Forest Inventory's relatively small number of sampling plots meant that in most situations it lacked the statistical power to quantify those benefits with enough precision to be able to claim carbon credits in the relatively short 5-year monitoring and reporting timeframes.

The difficulty of detecting evidence of carbon stocks changes resulting from pest management does not mean that management is ineffective at local scales, either for the maintenance or enhancement of ecosystem carbon stocks or for protection of biodiversity (Bellingham et al. 2014). Burrows et al (2008) observed that while measured short-term carbon sequestration changes attributable to herbivore control are likely be minor, in the long-term they are likely to be very significant. Peltzer et al. (2010) argued that 5-10 year periods are too short for assessment of changes in carbon stocks as this time period represents only a small part of the long lifespan of forest ecosystems.

The conclusion that the structure of the National Forest Inventory meant it would be hard to claim short-term carbon credit benefits from herbivore control, coincided with a series of debilitating Department of Conservation restructurings in response to funding cuts, and the WACEM work was side-lined. This was an unfortunate outcome, given that Bellingham et al. (2014) reported the findings that between a first measurement period of 2002–2007 and a second in 2009–2013 there was a net increase in stored carbon of 0.56 metric tonnes per hectare per year across public conservation land forests. This equates to 3.1 million tonnes of carbon per year (or 11.3 $mtCO_2e$), which was a significant proportion of the country's 2014 reported net GHG emissions, and worth some \$45m even at the low 2014 carbon prices of \$4 per tonne (Simmons & Young 2016).

In 2007 it was estimated that an extra million hectares of Public Conservation Land was in need of goat control (Arand 2007). A few years later, then-Parliamentary Commissioner for the Environment Dr Jan Wright sated; 'We do not have the luxury of time. Only one eighth of the conservation estate has any pest control at all, and without active management many of our iconic species are in danger of extinction' (PCE, 2011).

By 2016, as part of the DOC2025 process, the Department of Conservation undertook an assessment of the potential management requirements for deer. It resulted in an estimate that in the most likely to be affected ecosystems, deer occupied 6.8 million ha. The impact was judged to be acute (rapidly leading to significant decline) over 3.5 million ha. of warm and mild forest, shrubland, and grassland ecosystems, and chronic (gradually leading to significant decline) over the remaining 3.25 million ha. of cold forest ecosystems. Yet by 2019/20 the total "area under sustained management" for deer on PCL was only 1 million ha.

However, instead of greatly expanding the control of large introduced herbivores following the demise of commercial deer control, there has been little change in the area under annual deer control over the last 2 decades and a reduction in the area under annual goat and possum control by the Department of Conservation on public conservation land (Figure 12). In the same period tahr numbers nearly trebled. (Reddiex 2019).

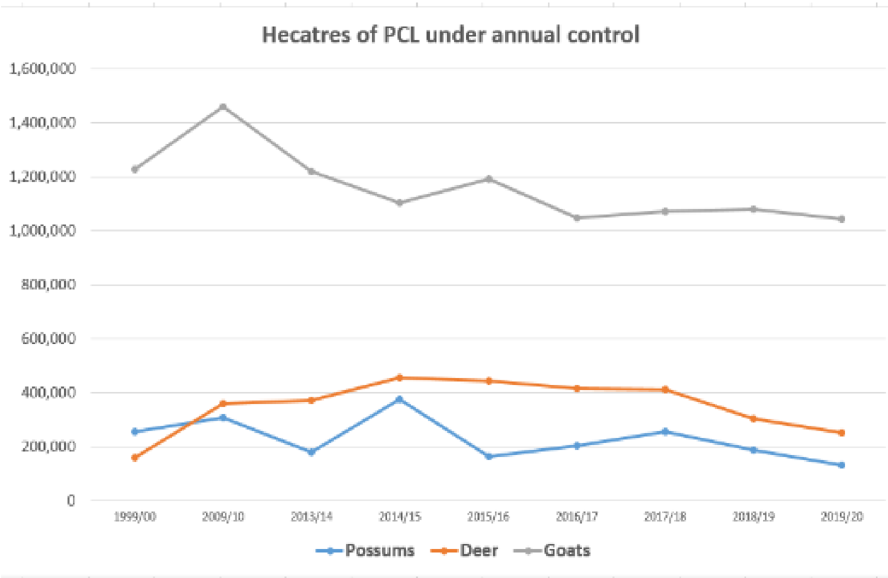


Figure 12: Trends in introduced herbivore control on conservation land. Area under annual pest control (estimated actual). [Source; Vote Conservation, Estimates of appropriations 2000/01, 2010/11 and 2020/21, and Annual Report Year Ended 30 June 2019; DOC (2020 b)]

While the Department's figures for 'areas under sustained management' (AUSMs) show only a modest increase over the same period, we need to be cautious about the veracity of these figures as it appears that AUSMs in various parts of the country have 'come and gone' over the last 2 decades (numerous Vote Conservation, Budget Estimates of Appropriations; DOC; 2020 b).

While introduced herbivore control has 'trodden water' over the last few decades, the opposite is true of introduced predator control. Engagement and resourcing of the Predator Free New Zealand programme has gone from strength to strength. In the last decade successive governments have responded to the serious biodiversity challenge of the (climate change induced) increased frequency of forest seed masting events that result in introduced rat and stoat plagues that devastate native wildlife. Throughout the country individuals, communities, businesses, local and regional governments are also actively involved in projects that protect and restore native wildlife by controlling introduced predators.

Sustained control of introduced herbivores requires a commitment of resources comparable to the very successful predator Free New Zealand programme, including a similar focus on the development and deployment of new technologies, some of which may be borrowed from the Predator Free work.

At the very minimum, introduced herbivore control will help protect existing carbon stores. However, at best it could make a significant contribution to the country's efforts to be carbon neutral, or even carbon positive in the not too distant future.

The Climate Change Commission's advice to Government under the Climate Change Response (Zero Carbon) Amendment Act 2019 provides a singly important opportunity to address the impact of introduced herbivores on Aotearoa/New Zealand's natural ecosystem carbon stores. Such action will help to maintain and enhance the country's significant natural carbon stores. It will also improve ecosystem health and resilience in the face of disturbance by climate change, and provide important co-benefits for communities and the economy by improving upper catchment ecosystems' ability to withstand the expected extremes of weather events.

Summary:

Climate change is Nature's response to a multitude of human-induced ecological stresses that have increased atmospheric greenhouse gases and therefore global temperatures. To limit the potential severity of climate change there is an urgent need to reduce emissions of greenhouse gasses, to protect existing carbon stores and to sequester as much carbon as possible in ways that will reinforce Nature's ability to limit global temperature rise.

Aotearoa/New Zealand's natural land ecosystems - native forests, shrub and tussocklands - are our most important carbon reservoirs and sinks. Plant species evolved here without mammalian herbivores, which means that the stress of introduced animals such as deer, goats, pigs, possums and wallabies has a fundamental impact on their health vigour and resilience, reduces plants' ability to store and sequester carbon. Introduced herbivores can interfere with a range of ecosystem nutrient cycles and can lead to natural ecosystems emitting stored carbon, while also reducing future ecosystem resilience by preventing the regeneration of full plant diversity and density.

The vast majority of our carbon stocks - over 6,500 million tonnes - are found in our natural vegetation and soils. Of this 1,456 million tonnes is stored in the above-ground vegetation of our natural ecosystems. The sheer size of these natural carbon stores means that even a small change in the condition of these stocks, either positive or negative, can have a massive impact on the country's greenhouse gas emissions profile. For example, it would have taken an annual increase in our total ecosystem carbon stocks of less than 0.2% (one fifth of one percent) to clear our reported 2018 national emissions to zero. But equally, a reduction in our natural carbon stocks of the same small percentage would have doubled our net 2018 emissions. To date, avoidance of loss of carbon stocks from natural systems and soils and ensuring recovery of those stocks has not featured highly in climate change policies.

Because all of Aotearoa/New Zealand's natural ecosystems are subject to browsing stress from introduced herbivores, control of these pests has the potential to significantly increase those ecosystems' resilience and capacity to maintain and sequester carbon.

However, between 2002 and 2014 there was a significant decline in the carbon stored in kāmahi-podocarp forest associations. This forest association covers approximately 800,000 ha, and makes up around 10% of all indigenous forest. Kāmahi is one of the key species that stores the most carbon in our forests, but is one of the only species that is susceptible to browsing from possums, deer, goats and chamois.

The decline represented an annual loss of nearly a million tonnes of carbon (equivalent to -3.4Mt CO_2/yr), which was equivalent to 6% of the country's reported net 2018 greenhouse gas emissions; three times the 2018 domestic air-travel emissions; and 80% of the extra annual sequestration that the Climate Change Commission hopes can be generated in the medium term by new native forest plantings.

The next three forest types with the largest declines in carbon stocks (although not statistically significant at the P<.05 level) made up another 14% of all tall forest plots and were all associations that also involved kāmahi.

The main introduced herbivores; deer, goats, pigs, possums, and wallabies, are estimated to directly consume natural ecosystem biomass and produce methane that combined are equivalent to between 2.3 and 4 million tonnes of CO_2 every year (CO_2e). The mid-point of this range is 3.1 million tonnes of CO_2e, which is equivalent to 5.6% of Aotearoa/New Zealand's reported 2018 net greenhouse gas emissions.

However, the indirect impacts on vegetation vigour, seed and fruit production, seedling survival, soil health, and ecosystem processes, including nutrient cycling are far greater. We have calculated a higher-end estimate of the potential improvements in annual CO_2 sequestration that could result from effective control of introduced herbivores.

At the higher-end estimate it might be possible to sequester an extra 17.5 Mt CO_2e/yr, this is equivalent to 31% of the reported 2018 net emissions. The mid-point of the estimated range (8.4 Mt CO_2e/yr) is equivalent

to nearly 15% of the reported net emissions. To help understand the significance of this number, it is equivalent to nearly 60% of the 2018 road transport emissions. At March 2021 carbon prices ($38.90 per tCO_2) this is equivalent to nearly $330m/yr. The economics of implementing this pest control will be well worth the investment.

These numbers highlight the importance that should be attached to the careful management of our natural ecosystems as part of our response to climate change. If we are to make the most of our contribution to the effort to minimise the rise in global temperature, then we must do all we can to maintain and grow our natural ecosystem carbon stocks.

New Zealand has seen previous episodes of introduced browser induced forest collapse. Many forests have also been taken to the brink of collapse by deer, goat and possum browse, where only directly funded intervention or serendipitous events such as the rise of the venison recovery industry, or the occurrence of bovine TB infestations, has prevented widespread forest loss. Department of Conservation data suggests that large herbivore populations throughout much of the country now exceed densities not seen since before the venison industry commenced.

At a minimum, sustained introduced herbivore control will make a major contribution to preventing the existing large natural carbon stores from being further degraded. Not only will this help meet our international obligations to *maintain* and enhance our existing carbon stores, it will also provide an insurance against likely future international requirements for nations to fully account for the changes in their natural ecosystem carbon stores. While natural losses such as storm events will not be avoided, an increase in resilience and retention of regeneration capabilities in dynamic ecosystems will see the quantity and frequency of carbon losses reduced. It will also allow us to live up to our international commitments flowing from the United Nations Convention on Biological Diversity and out national commitments such as the New Zealand Biodiversity Strategy.

The opportunity to improve natural rates of carbon sequestration through such pest control has been largely ignored by land managers and policy makers; the latest example being the Climate Change Commission's January 2021 draft Advice for Consultation and Supporting Evidence for Consultation, where the need to protect and enhance the country's massive existing natural carbon stocks receives little attention.

This has also been reflected in the reductions in spending on deer and goat control over the last few decades. Increased deer, goat, pig, possum and wallaby control not only has the potential to improve natural ecosystem health and carbon sequestration rates, it will also have many co-benefits that will enhance both ecosystem and community resilience to future climate change impacts.

This research shows that without more introduced herbivore control, loss of natural ecosystem carbon stores is likely to significantly increase Aotearoa/New Zealand's net GHG emissions, potentially dwarfing our present greenhouse gas emissions profile. However, greatly improved and sustained introduced herbivore control has the potential to increase carbon sequestration in our natural ecosystems to offset between 3 and 35 percent of the country's current annual net GHG emissions, while also improving ecological and community resilience.

Because we are in a position to actively manage our massive stocks of natural carbon – particularly by controlling introduced herbivores – Aotearoa/New Zealand is in a very good position to become not just carbon neutral, but carbon positive in the next few decades.

Recommendations:

1. Significantly increase the sustained and systematic control of introduced herbivores on public, private and iwi land in order to protect and enhance the country's massive stocks of carbon that are stored in our natural ecosystems.
2. Focus on the control of introduced herbivores within the kāmahi-podocarp forest associations (found mainly on the West Coast) that the latest evidence shows are losing significant amounts of stored carbon.
3. Reduce introduced herbivore densities to ensure the recovery of palatable species and ecosystem health.
4. Replace the Recreational Hunting Advisory Council with an Ecological Advisory Council that is mandated to advise on the most effective methods of introduced herbivore control to maintain and restore the long-term ecological health of New Zealand's natural ecosystems.
5. Increase resourcing of introduced herbivore control to levels similar to the very successful Predator Free New Zealand programme, including the development and deployment of new pest control technologies.
6. Carry out a more in-depth analysis of the National Forest Inventory data to better understand the direction of carbon sequestration rates for forest associations that have large components of species that are highly susceptible to introduced browsing.
7. Substantially increase the number of forest plots that are regularly surveyed in the National Forest Inventory to give the Inventory greater statistical power to identify the response to pest control and detect changes in forest carbon stocks.
8. Resource the National Forest Inventory so that a full report on its findings can be released within a year of an inventory cycle's completion – not the 5 to 6-year gap which occurred with the first two cycle reports.
9. Release all such reports to the public on their completion – they should not remain confidential for over a year as happened with the 2019 report.
10. Carry out research on:
    a. the production of methane by Brushtail possums in New Zealand ecosystems to better understand their contribution to the country's methane emissions;
    b. the impact of large introduced herbivore control on soil structure and processes, including forest soil capacity to oxidise methane, and to maintain soil moisture;
    c. the ecosystem response to decreased introduced herbivore density;
    d. more accurate national population size and distribution of introduced herbivores.
11. Increase emphasis on establishing permanent native forests as part of the billion trees programme, particularly given the very short half-life of sequestered carbon post-harvest of exotic plantations.
12. Carry out relatively inexpensive management actions, such as exclusion of domestic stock and low-level wild-animal control to ensure carbon gains occur in natural grasslands and regenerating shrublands that have been previously deforested.
13. Introduce incentives for the retention and growth of native forests on private and iwi land and the control of introduced herbivores, especially where precursor scrublands are potentially the target for exotic afforestation.

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Appendix 1. Updating introduced herbivore population estimates

A key frustration in doing this piece of work has been that most estimates of introduced herbivore population numbers are now several decades old, with many of them entering the scientific literature in the early 1990s as the country was coming out of two decades of intensive helicopter hunting for feral animals – particularly wild venison for meat export and for live capture for the new deer farming industry.

Below is our attempt to provide more up-to-date estimates of feral herbivore national population numbers and distributions. These are still estimates and we hope that our attempt will prompt some more thorough research to provide more robust numbers.

Table 12. Updating estimates of national population numbers and distribution of introduced herbivores.

Introduced herbivore	New population estimate	Distribution (ha.)
Deer	[1]300,000	[7]12,890,000
Goats (incl. chamois & tahr)	[2,3]500,000	[8]7,000,000
Pig	[4]300,000	[9]9,350,000
Possum	[5]30,000,000	[10]20,000,000
Wallabies (Bennett's & dama)	[6]1,450,000	[11]737,000

1. Based on new populations and expansion of range since the 1990s (see Fraser et al. 2000) and the estimate of "at least 250,000" by Nugent & Fraser (1993); 2. Goat population estimate based on Kirton & Ritchie (1982) and the new populations and expansion of range since the 1990s (see Fraser et al. 2000); 3. Chamois populations based on mean densities from Forsyth & Hickling (1998) and distributions from Fraser et al. (2000), the tahr population based on the management target of 10,000 on public conservation land and an estimate of 5,000 on non-conservation land 4. Feral pig population estimated from Parkes (1993) and from Nugent (1992) combined estimate of recreational and commercial hunting harvest of 113,500 pigs in 1988, plus the new populations and expansion of range since the 1990s (see Fraser et al. 2000); 5. From Warburton et al. (2009); 6. Estimated from Latham et al. (2016); 7. estimated from overlap of distributions of different deer species from Farser et al. (2000); 8. based on estimated 20% of overlap between chamois and tahr distributions with feral goat distribution from Fraser et al. (2000); 9. from Fraser et al. (2000); 10. possum numbers from Warburton et al. (2009); 11. from Latham et al. (2016);

Deer

Nugent & Fraser (1993) estimated total breeding population of wild deer in New Zealand in the late 1980s of probably about 250,000. This most quoted estimate came close on the heels of the two decades of intensive helicopter hunting for wild venison meat export and for live capture for the new deer farming industry. Our estimate of at least 300,000 is 20% greater than Nugent & Fraser's and takes into account the many new populations identified by Fraser et al. (2000) and the significant extension of the range of some deer in regions such as the east coast north of Gisborne.

Even at 300,000 the average deer density would be around 2/km^2, which is considerably lower than the estimated carrying capacity of native forest that probably lies in the range of 15 to 30 red deer/km^2 (Nugent et al. 2001).

Goats

Nugent (1992) estimated a total of 75,600 feral goats were harvested by recreational hunters and commercial ground-based and helicopter-based hunters in 1988. He stated that goats are almost certainly the most abundant wild big-game in New Zealand as, unlike deer, many goat populations are largely unharvested. He considered that the reported harvest was, therefore, probably far less than the annual population increment. The most quoted estimate of feral goat populations is 300,000 made by Parkes (1993). In his 1993 paper, Parkes stated that "the total population is estimated to be at least 300,000, but could be as high as 1,000,000." In 1982, Kirton and Ritchie had estimated a national feral goat herd of 400,000.

Given the increase in new populations and overall range of goats identified by Fraser et al. (2000) we are making an estimate that the feral goat population is at least 400,000.

Chamois

A chamois population estimate of 86,000 is based on the mean densities from Forsyth and Hickling (1998) and the estimated distribution of around 5,000,000 ha (Fraser et.al; 2000).

Tahr

The tahr population on public conservation land is presently being managed down to the 10,000 level agreed to in the 1992 Tahr Management Plan. An estimate of the tahr population on non-conservation land of 5,000 brings the total estimated tahr population of 15,000.

The combined total population estimate for feral goats, chamois and tahr rounds out at 500,000.

Pigs

The most quoted estimate for the national feral pig population is 110,000 initially made by Nugent et al. (1996) based on the estimate of 100,000 (99,267) pigs taken by recreational hunters in 1988 (Nugent 1992). However, Nugent's 1992 paper also surveyed ground and helicopter-based commercial hunters, and based on the percentage of survey returns they harvested an additional 14,243 feral pigs in 1988, bringing the total estimated harvest to 113,510, already bigger than the often quoted estimate. Parkes (1993) thought the national population of feral pigs "may be several hundred thousand as an estimated 100,000 are harvested annually".

Nugent et al. (1996) came to their 110,000 estimate based on the mis-reported 100,000 harvest estimate for 1988, and an assumed exponential rate of population increase of 0.6. Dzieciolowski et al. (1990) estimated that the rate of exponential population increase for feral pigs was 0.9, a rate that McIllroy (2010) considered very high compared with rates of 0.25-0.78 for feral pig populations living in good conditions or from recovery operations in Australia.

Assuming an exponential rate of population increase between those proposed by Nugent et al. (1996) and Dzieciolowski et al. (1990), and factoring in the increase in new populations and overall distribution of feral pigs identified by Fraser et al. (2000), we are estimating that the feral pig population is at least 300,000.

Possums

The possum population estimate is based on the work of Warburton et al. (2009) who estimated 30 million possums in 2008/09.

Wallabies.

Using Latham et al.'s (2016) population densities by habitat type, and the estimated distribution of Bennett's and dama wallabies (532,200 and 205,000 hectares respectively) produces an overall estimate of around 1,450,000 (Bennet's 1,064,400, and dama 410,000). However, the large number of confirmed sightings and animals shot outside these current estimated ranges, suggest they may currently occupy as much as 1,413,500 and 412,600 hectares respectively. If this is the case, then the population estimates could be conservative.

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