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Deer in the headlights: sleepwalking our way to a native forest catastrophe

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Abstract: A slowly unfolding native forest crisis is currently occurring in Aotearoa | New Zealand (hereafter New Zealand) because the numbers of deer in many native forests have risen rapidly to unsustainable levels. The natural forests of New Zealand are critical for many reasons: as habitat for native biodiversity, for providing resilience to catchments, as vast stores of carbon, and for their cultural values to Māori (indigenous people of New Zealand) and Pākehā (non-Māori people of New Zealand) alike. These critical values are being compromised by high deer numbers. Deer are causing irreversible harm to both old-growth and regenerating native forests through browsing and impacts on forest floor soil and litter. The flow-on effects from these include changes in understorey and eventually canopy composition, reduced habitat quality for fauna, and the retarding of regeneration caused by deer browsing of disturbed areas, such as on landslides, causing reduced water retention and enhanced sediment loss. Introduced mammalian herbivores, especially deer, and climate change stress natural forests. Their combined impacts are likely to be far worse than their effects individually. While the effects of climate change on old-growth forests cannot be directly mitigated, their vulnerability can be reduced by reducing the impacts of feral deer. Regenerating forests can also be protected so that they stand the best chance of being resilient to climate change as they age and mature. Control of deer is complex, especially because of the social values these animals have. Ecologically, the case for reducing deer to effectively zero densities is clear, but this cannot be undertaken in isolation from other values. We propose a national deer strategy and control programme that prioritises native biodiversity and climate change resilience over any other values that deer might have. This will require that in many places, deer numbers are reduced to zero, while in other places deer are managed for their social values. Ignoring the deer problem any longer will result in a catastrophic outcome for our native forests and for all people in New Zealand.

Keywords: biodiversity, climate change, conservation, deer, native forest, pest impact, pest management, resilience, ungulate

New Zealand's extraordinary forest ecosystems

Forests are the natural vegetation cover of most of Aotearoa | New Zealand (hereafter New Zealand), except for alpine zones and small areas of other treeless ecosystems that had limited extent in pre-human times (McGlone 1989; Williams et al. 2007). New Zealand's native forests are unique. Although they have similarities with forests of eastern Australia, southern South America, and, to a lesser extent, New Caledonia and New Guinea, most native plants and animals in New Zealand's forests occur nowhere else (McGlone et al. 2001; Lee & Lee 2015).

About one-third of New Zealand's native forests were cleared accidentally or deliberately with Māori settlement, especially in the drier eastern regions (McWethy et al. 2010). Deliberate deforestation continued with European settlement but extended into wetter regions, resulting in the loss of another third of the original forests (Ewers et al. 2006). New Zealand's native forests are now limited in extent, and many components such as dry lowland forest are almost entirely gone (Walker et al. 2004). Even where not cleared, logging has reduced the stature and, especially, abundance of large trees in many areas of remaining native forests (McKelvey & Nicholls 1957; Carswell et al. 2007).

Before the late 18th century, New Zealand's native forests never experienced mammalian browsing, let alone by ungulates like deer. The argument that deer are moa substitutes and that New Zealand's native forests are simply returning to their earlier browsed state has been conclusively shown to be invalid (Forsyth et al. 2010). The teeth, tongue, and antlers of deer make their browsing far more devastating than the more limited beak snipping of moa. Further, the unprotected nature of much of the New Zealand flora—vulnerable leaves and meristems and general lack of defences such as spines (Greenwood & Atkinson 1977, Lee et al. 2010), generally slow growth rates (McGlone et al. 2022), and limited number of resprouting woody plants (Burrows 1994)—highlights the

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relatively low impact of moa. The heavy foot pressure from hooved animals like deer also contrasts with the low pressure of moa feet (Duncan & Holdaway 1989). New Zealand's native forests are diverse and quite unlike the forests that evolved with mammalian grazers elsewhere and are therefore highly vulnerable to deer.

Humans are the only predator species in New Zealand capable of controlling deer. However, New Zealand's native forests, unlike those of almost any other nation, have minimal extractive economic use as they are no longer exploited for timber (McGlone et al. 2022). The current economic imperatives for control of pest animals like deer are few, aside from amenity values for tourism and recreation. In contrast, some deer have an economic value in and of themselves, now being regarded as game animals under the Game Animal Control Act (2013). Recent research suggests that increasing numbers of people view deer (and other game animals such as tahr) as adding value to the outdoors experience and consider the impact of deer as acceptable (Russell 2014). The only large-scale control of deer funded by past governments was centred on fears of erosion of the high country through canopy collapse (McKelvey 1995). Once that fear (incorrectly) vanished, that control ceased. However, native forests are now being seen as increasingly important for buffering us against the effects of climate change, as native forests reduce the impacts of the more intense storm events we are experiencing (Massey et al. 2025).

There is no doubt in our minds that there is currently a slowly unfolding native forest crisis because of high deer numbers which needs to be addressed now to avert irreversible harm being done to New Zealand's forests. But it is not just a biodiversity issue; deer also have the potential to undermine the capacity of our native forests to buffer downstream human communities from the increasingly severe effects of climate change. In this article, we argue that ignoring the deer problem any longer is to New Zealand's peril. We suggest that New Zealand needs a national deer strategy and population control programme that prioritises the irreplaceable native biodiversity of New Zealand and climate change resilience over any other values that deer might have.

Deer populations since their introduction

Red deer (*Cervus elaphus scoticus*) are the most widespread of the eight deer species in New Zealand (King & Forsyth 2021). Other deer species, notably sika (*Cervus nippon*), fallow (*Dama dama*), and white-tailed (*Odocoileus virginianus*) deer, are more localised in distribution, but their rumen anatomies differ from those of red deer, enabling them to digest plant material that is difficult for red deer to digest (Fraser 1996), hence their impacts on forests are more comparable to goats (*Capra hircus*). Most of the discussion below focuses on red deer, because their ecology is known best, but many of the points made apply more broadly to all feral deer and other ungulates such as goats and tahr (*Hemitragus jemiahicus)* too.

The invasion of deer, especially red deer, has followed a well-documented temporal pattern across the country (Caughley 1970, Forsyth & Caley 2006, Sinclair 2003). This falls into four main phases, with the first two phases, colonisation and peak, showing high coincidence between actual deer numbers and theoretical predictions (Table 1; Fig. 1). Theoretical predictions show an expected oscillation in numbers in a post-peak phase, which we subdivide as a post-peak decline and a current increase in numbers (Table 1; Fig. 1). Rather than the expected oscillation in numbers in the post-peak period that was predicted, we speculate that there was a c. 30-year-long suppression of deer populations by aerial helicopter harvesting (1965 to 1995) which pressed the populations lower and for longer than had it followed an

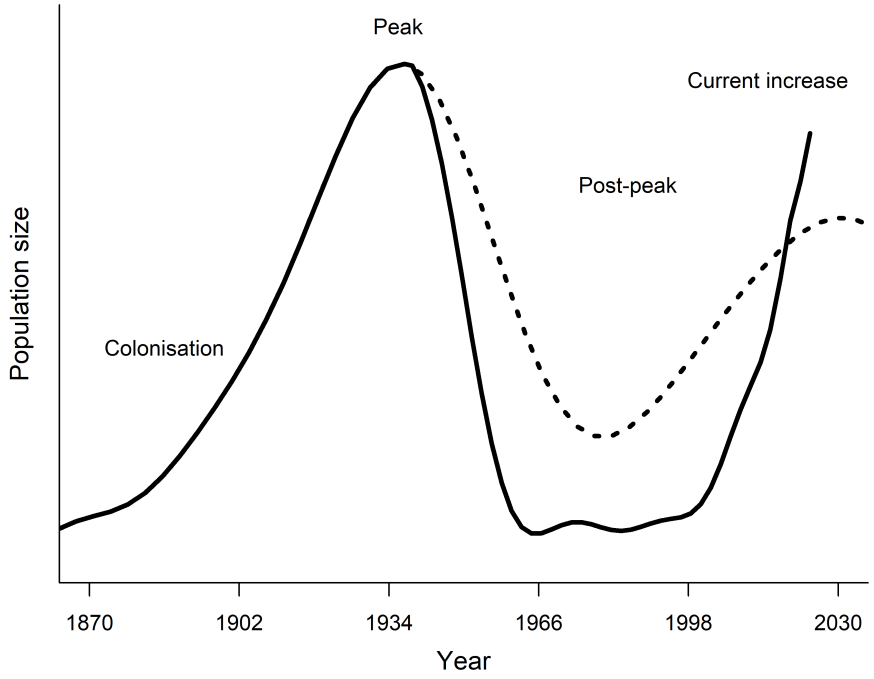


Figure 1. Estimated trends in actual deer population size (black line) and theoretical population size without human intervention (dashed line). Based on Caughley (1970) and Forsyth & Caley (2006) (theoretical population), Nugent & Fraser (2005) (estimated population of red deer up to 2002), Forsyth et al. (2011) (measured pellet densities up to 2010) and an assumed continuation of the increase reported in Forsyth et al. (2011) thereafter.

Table 1. Red deer populations during irruption after introduction to New Zealand, their effects on natural forests, and management regimes during four stages of their population irruption.

Irruption phase	Red deer population	Effects on forests	Management
Colonisation	Introduced to various parts of the North Island, South Island, and Rakiura between 1861 and 1919 (Nugent & Fraser 2005). Exponential increase in population because of high fecundity and early maturation of females, and low mortality rates at all ages, sustained by no limitation in food supply (Nugent & Fraser 2005).	The most palatable species are quickly eliminated or markedly reduced in abundance (Cockayne 1909; Mark & Baylis 1975).	Hunting initially highly controlled (restricted to stags only). Hunting restrictions were lifted in the 1920s and government cullers employed in response to growing concerns about deer impacts on forests and farms (Clarke 1972).
Peak	By the 1940s and early 1950s deer numbers reached very high levels, with declining body weight (Caughley 1970), declining female fecundity, increasing mortality rates (especially of fawns), all caused by diminishing food supply (Nugent & Fraser 2005).	Palatable species largely absent from forests at large spatial scales, tree roots exposed, and litter later reduced (Conway 1949; Jane & Pracy 1974). Obvious deer trails through the forests (e.g. McNeill 2007).	The 1930 Deer Menace Conference in Christchurch saw control switched from the Department of Internal Affairs to the New Zealand Forest Service (NZFS) which continued employing cullers through until the 1970s. Large numbers of deer culled. The size of the deer population appeared little influenced by culling pressure as deer numbers remained high through until the 1960s.
Post-peak	Numbers fall in response to declining food quality and quantity, with local starvation of deer of all ages, coupled with intense aerial helicopter harvesting, followed by more-or-less stable low-density populations (Nugent & Fraser 2005).	Some palatable species increase in abundance, but others remain suppressed (Coomes et al. 2003). Deer densities are sufficient to prevent regeneration of some tree species (Nugent et al. 2001).	Aerial helicopter harvesting. Wild Animal Recovery Operations (WARO) starting in the 1960s, first for meat and then live animals, with deer numbers declining rapidly through into the 1970s and 1980s (Caughley 1983; Challies 1991; Carey 2025).
Current increase	Slow increase from low levels in the late 20th century, both on private and public land, but a more rapid increase in the 2010s/2020s (Moloney et al. 2021).	Mortality rates of palatable tree species exceed their recruitment rates between 2009 and 2020, and vice versa for unpalatable tree forests across all public forests (Hawcroft et al. 2024).	Reduced helicopter hunting and live capture (as deer farms became established), a strong focus on stags in recreational hunting (thus leaving hinds untouched), increasing areas of pine plantations (for timber and carbon), and (potentially) increased biomass of palatable plant material to accelerate deer population increase especially in forests.

expected theoretical population response. As indicators of this pressure on deer populations, culling by helicopters during the late 1960s was often at a rate of 100 and as much as 200 deer per day and, after live capture of deer began, one company caught 7000 deer during 1978–1979 (Drew 2026).

Since 1995, but especially over the last decade, feral deer populations have risen largely unchecked over most of the country (Forsyth et al. 2011; Moloney et al. 2021; MacLeod et al. 2024). New, mainly illegal, liberations have taken fallow and sika deer (and pigs, *Sus scrofa*) to places they had never been before (Nugent et al. 2025). Resident populations of deer (and goats) are growing rapidly over most of the country (as are tahr in the central South Island) (Moloney et al. 2021). We consider that the c. 30-year-long suppression of deer numbers may have allowed enough development of palatable plant biomass in forest understoreys that, when that harvesting pressure was relaxed, the post-peak increase in populations was even greater in response to that resource availability. This could explain anecdotal views that forest understoreys in the 2020s resemble those last seen at peak deer densities in the early 1960s (e.g. in Fiordland; Carey 2025).

While deer have been eradicated from some forests (red deer from Secretary Island [MacDonald et al. 2019], fallow deer from Motu Kaikōura [Nugent & Asher 2021]) and are suppressed in others (Te Tai Tokerau and some parts of Taranaki; Fraser et al. 2003), in most places their populations are uncontrolled other than by recreational hunting (Leathwick & Byrom 2023), which has made no difference to their population trajectories. It is important to acknowledge that historical control was largely achieved by commercial recovery of meat and live animals which is now greatly reduced. In effect, deer (and tahr) control was subsidised by wild animal recovery operations (WARO). While WARO continues, it is at much lower levels than in the past which is why New Zealand conservation managers must now more specifically engage with the issue and actively manage feral deer.

What are deer doing to our forests?

Old-growth forests

The effects of deer in old-growth forests depends on the stage of the population irruption after their introduction, from local elimination of highly palatable species soon after their introduction to widespread reduction of palatable plants and consumption of unpalatable plants when deer are starving at their population peak (Table 1; Fig. 2). Palatable plants removed or severely reduced in old-growth forests by deer include some herb and fern species in forest understoreys (e.g. *Asplenium bulbiferum*), some understorey shrubs (e.g. large-leaved *Coprosma* spp.), and seedlings and saplings of some tree species (e.g. Broadleaf *Griselinia littoralis*) (Jane & Pracy 1974; Stewart & Burrows 1989; Forsyth et al. 2002; Husheer & Frampton 2005). At high animal densities diet breadth can expand to include the widespread and abundant common canopy tree species that are typically avoided such as tawa (*Beilschmiedia tawa*), the beeches (especially silver beech, *Lophozonia menziesii*; Fig. 3), hīnau (*Elaeocarpus dentatus*), and podocarps (Allen et al. 1984; Smale et al. 1998; Carswell et al. 2007). Forests in cool regions and on infertile soils, where growth rates are low, may be less resilient to browsing pressure (Lusk et al. 2021) but landscape-level variation in soils and microclimate not only influence plant communities and the proportion of palatable species within them (Forsyth et al. 2005; Richardson et al. 2008; Bee et al. 2011) but also where deer browse preferentially (Wardle 1967), paradoxically sometimes leaving residual populations of palatable plants surviving in matrices of unpalatable plants (Bee et al. 2009).

Given that deer have only been in New Zealand since 1861, most trees in the canopies of New Zealand forests (podocarps, beech, tawa, hīnau, rātā [*Metrosideros* spp.], etc.) were in the canopy before deer even arrived in New Zealand (Coomes



Figure 2. Red beech (*Fuscospora fusca*)forest lacking palatable plants in the browse layer, Lake Manapouri, Fiordland National Park (photo David Norton, June 2024).



Figure 3. Heavily hedged silver beech regeneration along the edges of a river flat, Turnbull River catchment, South Westland (photo David Norton, January 2026).

et al. 2003; Allen et al. 2023). This is the long-lived capital on which forest understoreys rely for provision of seeds that are critical for continued maintenance of local tree populations and long-distance dispersal of seeds into regenerating forests (Richardson et al. 2009). These large trees are also critical stores of carbon, retaining carbon in the biosphere (Stephenson et al. 2014; Allen et al. 2023).

This capital will not endure indefinitely; over time it will naturally decline and fade out (Bellingham et al. 1999). If those trees aren't replaced because their progeny are eaten out or are antler-rubbed to death (as can be the case for long-lived podocarps like mataī, *Prumnopitys taxifolia*; Giller 2020), the new forests that develop to replace the current ones will be compositionally dissimilar. An argument for complacency could be made that New Zealand's old-growth rain forests have had huge numbers of deer in them before, during the first unconstrained irruption of deer numbers in the 1930s to the 1970s, despite which we still have old-growth forests. This ignores the fact that the canopy trees that survived that first irruption are now 50–100 years older than they were then. Because the capital of old canopy trees will decline naturally, there are no guarantees that the forests will survive repeated unconstrained irruptions of deer and other ungulate species (especially goats, tahr, and pigs).

The composition of old-growth forests has shifted in the last decade in response to burgeoning deer numbers. Populations of trees that deer browse selectively are in decline nationally and populations of trees that they don't eat are growing in their place (Hawcroft et al. 2024). Deer (and other ungulates) have no regard for land tenure, with impacts just as bad in national parks as they are across other parts of public conservation lands (Hawcroft et al. 2024). Damage on private land is likely to be similar. Coomes et al. (2003) wrote a critique of why burgeoning ungulate numbers could lead to irreversible changes in forests and changes in ecosystem processes by a combination of factors. These included deer and goat browsing suppressing the abundance and biomass of highly palatable plant species (to the extent that these become locally extinct as seed sources) while simultaneously promoting the abundance and biomass of unpalatable or non-native plant species (Nugent et al. 2001), and their browsing and trampling causing shifts in ecosystem function (Wardle et al. 2001; Wardle & Bardgett 2004; Kardol et al. 2014). That critique was written over 20 years ago when deer numbers were far lower than now. Those concerns apply even more now that deer numbers are higher and rising nationally.

Regenerating forests

Deer and other ungulates (especially goats and pigs) are not restricted to old-growth forest or public conservation land. They occur widely across all land types in New Zealand (King & Forsyth 2021) and can be particularly common in regenerating forest across public and private land (e.g. Brennan et al. 1993; Smale et al. 1995; Krull et al. 2013) and in non-native tree plantations (whether established for timber or carbon) (e.g. Nugent 1990). Coomes et al. (2003)

highlighted that browsing by deer can fundamentally alter successional pathways in regenerating forests (Fig. 4). Browsing by deer has fundamentally different effects on ecosystem structure and processes in regenerating forests than in old-growth forests (Wardle & Bardgett 2004; Holdaway et al. 2012). This is because of the differences in resource availability to plants in regenerating forests compared with old-growth forests (i.e. higher light and often greater levels of soil nutrients in regenerating forests). As a result, some plants in regenerating forests have high growth rates, short-lived tissues, high shoot leaf area, poorly defended leaves, and high leaf nutrient contents (e.g. native broom, *Carmichaelia* spp.). These traits make them highly palatable to deer, which can therefore consume a much greater proportion of the biomass in regenerating forests than in old-growth forests (Wardle & Bardgett 2004). Heavily browsed regenerating forests function differently from browsed old-growth forests because of the rapid rates of return of consumed plants to the soil as faeces, which, coupled with high plant litter quality, results in rapid decomposition and low rates of carbon sequestration in soils (Wardle & Bardgett 2004).

Ambitious ecological restoration programmes such as Recloaking Papatūānuku (Pure Advantage 2025) that aim to increase the amount of native forest could be stymied by unchecked populations of deer (Forbes et al. 2020). Deer effects are likely to be large in natural regeneration pathways comprised of many highly palatable plants. For example, post-agricultural regeneration on Banks Peninsula in Canterbury can proceed through unpalatable harakeke (*Phormium tenax*) with abundant palatable trees and shrubs including *G. littoralis* and *Pseudopanax arboreus* establishing within them (Reay & Norton 1999). Both these species are severely reduced in abundance when red deer are common, substantially delaying the rate of forest regeneration and reducing biomass.

More common and widespread post-agricultural natural regeneration pathways are dominated by mānuka (*Leptospermum scoparium*) and kānuka (*Kunzea ericoides* s.l.) which are unpalatable to deer. However, many highly palatable trees and shrubs establish and grow beneath them, some of which overtop and supplant mānuka and kānuka (Esler 1967; Wilson 1994). Ten years after fenced exclosure plots were constructed in regenerating kānuka forests with high numbers of fallow deer, plots had high densities of palatable trees and shrubs (*Coprosma macrocarpa*, *Geniostoma ligustrifolium*, and *Melicytus ramiflorus*) and unfenced plots had few, if any, of these (Smale et al. 1995). In this example, it is unclear whether deer are arresting succession or causing an irreversible shift in the direction and speed of the successional pathway (Coomes



Figure 4. Severely deer impacted regenerating forest, Mount Herbert | Te Ahu Pātiki Scenic Reserve, Banks Peninsula. Both browse and ring-barking by feral deer are having a major impact on regenerating trees (photo Sarah Richardson, February 2025).

et al. 2003) but without litter inputs from rapidly decomposing species such as *M. ramiflorus*, ecosystem processes will be dominated by litter from kānuka alone, likely leading to slower growth rates. Under heavy browse pressure, even unpalatable tree seedlings can fail to establish under kānuka resulting in stalled successions and a loss of tree diversity, structural complexity, and seed sources (Richardson et al. 2014).

Natural regeneration is also critical for reforestation after major disturbance events, such as landslides generated by storms, including extra-tropical cyclones (e.g. the more than 100 000 landslides generated by Cyclone Gabrielle in 2023; Massey et al. 2025), or by earthquakes (e.g. tens of thousands generated by the M_W 7.8 2016 Kaikōura earthquake; Massey et al. 2020). The value of protecting existing forests is highlighted by the reduced areas subject to landslides in catchments with natural forests compared with deforested catchments either in agriculture or subject to clear-cuts in plantation forests (Massey et al. 2025). Many palatable native tree, shrub, and herbaceous species establish and grow on newly generated landslides. Some of the highly palatable trees and shrubs, such as *Carmichaelia* spp. (Yockney & Hickling 2000; Grüner & Norton 2006), have critical roles in landslide succession because they contribute nitrogen, which is otherwise highly limiting on these sites, through their ability to fix atmospheric nitrogen. Their elimination causes significant reduction of biomass and a slowing and alteration of successional trajectories (St. John et al. 2012). High numbers of deer (and goats) in steep catchments where landslides are frequent (e.g. in the Raukūmara, Aorangi, and Remutaka Ranges) may retard the rate of natural regeneration on them such that repeated slope failure occurs when new weather-related or tectonic events occur (Wardle 1967; Campbell 1984; Walker & Shiels 2013), potentially threatening human livelihoods lower in the catchments.

There are several key caveats concerning the impacts of deer on natural forests today that need to be considered:
(1) There are important differences between different deer species in terms of both where they live and how they feed. For example, red deer are browsers and fallow deer are grazers, but they co-exist over large areas. Differences in rumen morphology among deer species mean that plant species such as mountain beech (*Fuscospora cliffortioides*) can form larger parts of the diet of sika deer than those of red deer, so that the impacts of sika deer on mountain beech seedling densities and growth rates are much more severe than those of red deer (Duncan et al. 2006; Husheer et al. 2006; Bellingham et al. 2016).
(2) The impacts of deer in an area cannot be considered in isolation from those of other ungulates (feral and farmed), or from other herbivores (e.g. hares, rabbits, wallabies) or omnivores (e.g. possums) that might be present.
(3) Deer impacts also depend on management inputs including historical control, current hunting pressure (commercial, recreational, and Department of Conservation cullers), the reasons why people hunt (for trophies or for meat), proximity to population centres, and the location of aerial 1080 operations.
(4) Finally, the spatial configuration of landscapes (e.g. remnant forest patches in a pastoral landscape or continuous tracts of old-growth native forest such as along the axial ranges) and the presence of other habitat elements, especially non-native tree monocultures, will also have an influence on impacts at a particular site. The rapid expansion in areas of non-native conifer plantations from the 1970s onwards has provided more habitat for deer and other ungulates; many of these plantations adjoin old-growth and regenerating natural forests such that movement of deer populations among them is typical (Nugent 1990; Smale et al. 1995).

Deer impacts in an international context

New Zealand is far from unique in its dilemma of rapidly rising numbers of deer in natural forests. In many parts of the world, elimination or extinction of apex predators and a decline in human harvesting has resulted in over-abundant populations of deer in forests in their native ranges (Côté et al. 2004; Carpio et al. 2021; Suzuki 2025). Experiences internationally offer instructive views of the probable consequences for New Zealand's forests of inaction in managing deer. Over-abundant deer have prevented regeneration of some dominant trees at landscape scales in the northern United States and Canada (e.g. Frelich & Lorimer 1985; Alverson et al. 1988), potentially permanently altering their composition and structure (Stromayer & Warren 1997; Suzuki 2025). Deer have depleted and homogenised forest understoreys (e.g., Fukamachi et al. 2023). They have prevented natural regeneration of formerly dominant trees in deforested landscapes (e.g. of *Pinus sylvestris* in Scotland; Scott et al. 2000). In Yellowstone National Park, western United States, over-abundant deer maintained valley floors as deforested ecosystems for decades and only after their top predators, wolves (*Canis lupus*), were reintroduced and deer numbers reduced did willow (*Salix* spp.), alder (*Alnus incana*), and aspen (*Populus* spp.) regeneration take place (Beschta & Ripple 2016), in turn resulting in reinstated trophic cascades involving beavers, with major positive consequences for river hydrology and flood mitigation (Beschta & Ripple 2019).

All these examples demonstrate that native deer can radically alter native forests and the ecosystem services they provide anywhere on the globe if uncontrolled. It is salient that all these deleterious effects have occurred in forests where the resident flora co-evolved with deer browsing, in contrast to New Zealand's forests that have no such history. Finally, many of the concerns about over-abundant deer in northern hemisphere forests are not just about the sustainability of forests; they are as much or more focused on human health and over-abundant deer as part of disease cycles. In particular, the last century has seen a rapid expansion in range and incidence of tick-borne diseases, in part attributable to over-abundant deer (e.g. Lyme disease; Stone et al. 2017). If or when ticks that feed on deer arrive in New Zealand, New Zealand can expect the same serious public health consequences.

Deer impacts are likely to get worse with climate change

The well-documented effects of deer on old-growth forests can be conceptualised through two pathways (Fig. 5): browse of palatable species and mortality of trees from antler thrashing (Giller 2020), and trampling and disturbance of the forest floor (litter and soil). These effects are best highlighted when comparing areas where deer have been excluded (usually through exclosure plots) with adjacent areas where they have access (Wardle et al. 2001; Fig. 6).

These primary impacts then have a range of flow-on effects that impact both native species directly (e.g. through loss of species and the habitat they provide) and the forest ecosystem more broadly (through increased water and sediment yield, drier soils, and alteration of successional trajectories). These,

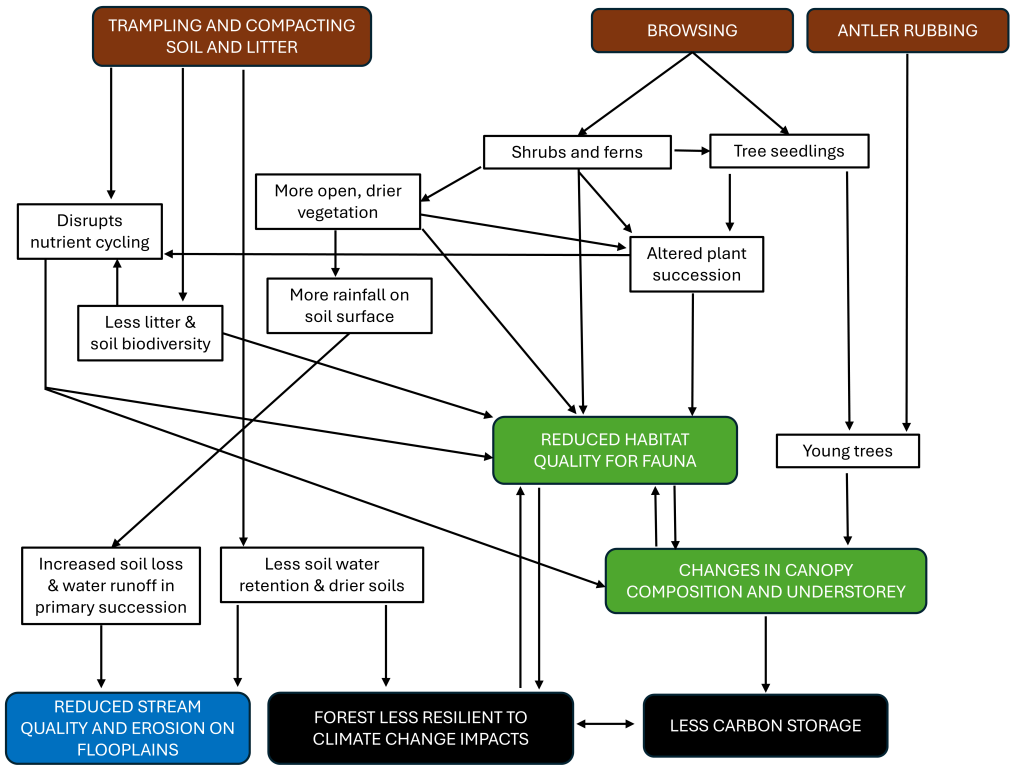


Figure 5. Schema showing how trampling and browsing are the two major pathways through which deer drive change in forest ecosystem composition, structure, and function.



Figure 6. Exclosure plot contrast, Ruahine Ranges (photo Joanna Buswell, January 2010).

in turn, can then have marked impacts on native biodiversity (habitat for fauna and changes to long-term forest composition) and on the role that old-growth forests play in mitigating the impacts of climate change (down-stream impacts). Finally, through all these flow-on effects, deer (and other ungulates) can ultimately reduce the resilience of the forests themselves to the effects of climate change. Under prolonged and intense browsing pressure, forests can release stored carbon back into the atmosphere, further increasing the impacts on biodiversity and downstream human communities.

Human-induced climate change research suggests that New Zealand is likely to experience more extreme weather events, as well as warmer and more variable conditions overall (Ministry for the Environment [MfE] 2025). From a forest perspective, key effects will be more frequent storms associated with both extra-tropical cyclones like Ita (2014) and Gabrielle (2023) (Studholme et al. 2022), and stronger westerly circulation associated with events like the 2025 Sudden Stratospheric Warming over Antarctica. Both are resulting in more frequent intense rainfall events. The other major effect on forests from changing climates will be from more frequent and pronounced summer droughts. New Zealand's native forests have experienced these types of events for millennia and, if healthy, are likely to be resilient to an increasing frequency of such events, at least to some extent.

In the face of ongoing deer (and other ungulate) impacts, the resilience of New Zealand's native forests is, however, likely to be seriously compromised and this will affect the human population of New Zealand because of the flow-on effects highlighted in Figure 5. The two primary consequences of climate change impacts on native forests in the face of ongoing deer pressure will be through (1) changing canopy composition, as existing canopy dominants are unable to regenerate and (2) increased water and sediment loss on new surfaces such as landslides created during severe storm events but prevented from revegetating. Both will lead to long-term loss of habitat for native biodiversity, loss of carbon in forest ecosystems, and loss of the other ecosystem services these forests provide.

Exclosure plots demonstrate that deer result in a reduction in shrub and ground layer vegetation and a loss of the litter layer (Wardle et al. 2001; Mason et al. 2010). Research on livestock impacts in forests also shows us that farmed ungulates such as sheep and cattle impact soil structure through increased compaction (Norton et al. 1995; Denmead et al. 2015). This leads to warmer soils and decreased water infiltration. Warmer soils transpire more and, with less infiltration, can exacerbate drought effects on canopy trees, with eventual tree mortality (Yates et al. 2000). It seems reasonable to expect that deer will have comparable impacts.

The longer-term consequence of these changes is the loss of canopy dominants as they fail to regenerate in the face of both climate change and ungulate pressures. While plant regeneration will occur, it is likely to be of more drought- and browse-tolerant species, many of which are short-statured and will not store large amounts of carbon (e.g. *Leucopogon fasciculatus*) (Laughlin et al. 2020). Aside from the obvious biodiversity and resilience impacts, the other key effect of this will be a net loss of carbon from our old growth forests, carbon that will be released back into the atmosphere as CO_2, further exacerbating the impacts of climate change.

The future of New Zealand's natural forests and all the benefits they provide human society are already in jeopardy because of climate change. Not addressing the high numbers of deer in these forests will only exacerbate this threat.

Killing predators is not enough to safeguard forest ecosystems

Leathwick and Byrom (2023) argued that even if conservation managers in New Zealand are strongly focused on safeguarding vulnerable bird species, they need to control deer to protect forests. We argue, more simply, that conservation managers also need to care about forests for their own sake. While historically there was a major focus in natural forests on deer and possum control, most pest forest management in recent decades has focused on controlling carnivorous predators, especially mustelids and rodents, along with possums (Leathwick & Byrom 2023). This recent focus on carnivorous predator control has been because of the extreme vulnerability of some of our fauna (especially avifauna) to invasive carnivores. Research has clearly shown that without active carnivorous predator control, some New Zealand native birds will quickly go locally and nationally extinct (e.g. Choquenot 2006). The many stories of different species being saved from the brink of extinction are testament to this, but more sobering are the many species that have already been lost due to predator pressure since human settlement (Gill & Martinson 1991).

There is no questioning that conservation managers need to deliver predator control at scale and quickly if New Zealand society is to stop the decline and extinction of threatened fauna. The many predator control projects around the country are helping to achieve this. But while predator control will help sustain native fauna in the short term, conservation managers also need to be addressing the impacts of deer (and other ungulates) on New Zealand's forests because deer impacts are likely to endanger endemic fauna in the longer-term even with predator control (Leathwick & Byrom 2023).

The implications for native fauna of deer impacts on forests have not been widely discussed, and some are likely to play out over longer-time scales than impacts on palatable plants (Leathwick & Byrom 2023). Healthy New Zealand natural forests provide essential resources for our native fauna, especially the avifauna (Fig. 5). To sustain endemic fauna such as birds, sufficient resources for food and nesting need to be available, as well as low predator numbers. Studies in the native range of deer have shown that increasing abundance of deer results in declines in the density and species richness of forest birds, and some bird species are absent when deer densities are high (deCalesta 1994; McShea & Rappole 2000; Seki et al. 2014). While equivalent studies are needed in New Zealand, our bird communities are highly likely to be affected by dense populations of deer because they eliminate palatable trees and shrubs (Fig. 7), in turn affecting the ability of New Zealand's forests to provide the resources the bird fauna requires such as fruits (Clout & Hay 1989; Burrows 1995), nectar (Castro & Robertson 1997; Roberston et al. 2008), and, as prey, the invertebrates that feed on palatable plants (Moeed & Fitzgerald 1982; Yule & Burns 2020). An open, dry forest understorey and bare forest floor will provide reduced habitat for invertebrates that are also key food resources for many native birds (e.g. McLean et al. 1994).

In the longer term, impeded regeneration of canopy trees is likely to affect some bird populations. For example, antler-thrashing of saplings of podocarp tree species (Cameron 1960) could reduce the abundance of fleshy-fruits that are critical for breeding by native forest bird species (e.g. miro [*Pectinopitys ferruginea*] for kererū [*Hemiphaga novaeseelandiae*]; Lyver et al. 2008). If large canopy trees fail to regenerate, ultimately this will result in a lack of large trees that provide cavities



Figure 7. Deer browsed broadleaf in silver beech forest, Haast Tokoeka Sanctuary, South Westland (photo David Norton, June 2025).

for hole-nesting birds (e.g. Wilson et al. 1998) and roosting sites for bats (e.g. Sedgeley & O'Donnell 1999). Unless land managers address the impacts of deer on New Zealand's forests, the millions of dollars that are spent every year on predator control in forests will be largely wasted as the combined effects of climate change and ungulates on our forests will surely lead to massive loss of fauna anyway.

The social component of deer control

If biodiversity conservation was the only goal, all feral deer and other ungulates in New Zealand would be eliminated for our old-growth and regenerating forests to have the strongest chance of surviving. However, deer management is a complex issue and the control of deer (and other ungulates) needs to be considered within the context of how people perceive these exotic pest animals. There are at least five pertinent social issues within New Zealand:
(1) Deer are seen as an important protein resource, with hunters relying on feral deer (and other ungulates like pigs) to support themselves and their families.
(2) The recreational hunting lobby want good access to a ready supply of deer to meet their needs and are a vocal and well organised lobby group, including through the Game Animal Council (established under the Game Animal Council Act 2013 with a statutory role to provide advice to the Minister for Hunting and Fishing on hunting and the management of game animals [deer, tahr, chamois, and wild pigs]).
(3) Animal rights movements consider that all killing of wild animals is impermissible.
(4) A strong anti-poison, and in particular anti-air-drop poison, lobby are motivated by concerns about the perceived cruelty of poisons (especially 1080), the potential flow-on effects for other parts of our forests (e.g. native birds being killed), and effects on drinking water quality (although water contamination has been shown not to be an issue; Eason et al. 2011).
(5) A small but growing tourism sector based on trophy hunting of deer (and other ungulates).

For these reasons, the hard reality is that elimination is not a socially acceptable option and, even if it were attempted, there would likely be unforeseen consequences that could quickly jeopardise other biodiversity conservation programmes. These social considerations also highlight the critical importance of early stakeholder engagement in developing an approach to better managing deer across New Zealand (see next section).

Rethinking deer management

It seems clear that throughout most New Zealand native forests, deer (and other ungulates such as goats, pigs, and tahr) are at levels that continue to degrade New Zealand's forests (Moloney et al. 2021; Hawcroft et al. 2024). As well as the social issues discussed above, controlling deer in these forests requires consideration of a fundamental ecological issue: Because of the already degraded state of most old-growth forests, it only takes a few deer to maintain forests in this degraded state, and especially to suppress palatable species (Tanentzap et al. 2009). This is ecologically different to the early phases of deer impacts where high deer numbers degraded forests that had never been browsed by ungulates.

The scale of biodiversity decline in New Zealand is large relative to the resources currently available to central and regional government, iwi, and other private landowners to combat it. Native biodiversity is in decline across most public conservation land (MfE 2025) and deer are a key driver of this. While the Department of Conservation (DOC) has been allocated some targeted funding for ungulate control, this is woefully inadequate to address the issue nationally, although there have been some useful local successes (e.g. in eradicating red deer from Secretary Island in 2014, as advocated 23 years earlier [Mark et al. 1991], or the current focus on eliminating deer from Te Tai Tokerau). Except in a few localised situations, it is clear that recreational hunters will never be able to control deer, let alone other ungulates, to levels that sustain biodiversity and support ecosystem resilience in New Zealand (Duncan et al. 2006; Wright et al. 2012; Husheer & Tanentzap 2024). This was recognised more than 50 years ago when the New Zealand Forest Service (NZFS) was running an active culling programme to reduce deer numbers (John Fleming, retired NZFS deer culler, pers. comm.) and the situation has not changed.

There is no simple solution to the deer dilemma, but it is clear that what New Zealand is currently doing is not succeeding and New Zealand is facing a native forest catastrophe because of skyrocketing deer numbers. We believe an urgent rethink of our approach to deer management is required and suggest that this involves at least three key elements:
(1) Developing a national deer strategy for New Zealand.
(2) Securing enduring funding (government and private sector) to implement this strategy.
(3) Undertaking the management work that is required to implement this national deer strategy in a professional and coordinated manner.

We suggest that the core of a national deer strategy should be categorisation and prioritisation of New Zealand's native forests (and related ecosystems) based on their biodiversity and resilience values. Much of this work has already been done through the recognition of ecological areas (Norton & Overmars 2012) and through DOCs prioritisation of ecological management units (DOC 2023). In addition, native forests that directly support resilience of downstream human communities need to be prioritised.

Based on this prioritisation, some areas will require deer to be managed to very low or zero densities to safeguard ecological or resilience values. In other areas, hunting and community groups might be tasked with managing deer numbers at densities that can continue to provide recreational and food opportunities, while also minimising ecological impacts. Economically profitable operations (e.g. WARO) yielded some improvement in ecological integrity (e.g. of alpine grasslands in Fiordland National Park; Rose & Platt 1987) but this was seldom the primary goal. Moreover, as the last few decades have shown, as soon as those operations become unprofitable, deer numbers increase (Challies 1991) and ecological integrity declines. Future management models may be akin to the Recreational Hunting Areas the NZFS had in the past (Nugent 1988), or wapiti (*Cervus elaphus nelsoni*) and sika management as Herds of Special Interest. Commercial WARO operators will also play a key role in many areas, along with a range of other innovative solutions.

Funding may come through some form of carbon plus biodiversity credits (Chartres 2021), but equally, central government in New Zealand needs to recognise that healthy native forests provide enduring benefits for catchment resilience in the face of climate change. These benefits from native forests will accrue to both local communities and New Zealand in toto and are an investment in the future of New Zealand.

Development of a national deer strategy needs to be based on discussions among all stakeholders interested in New Zealand's natural forests and deer. Stakeholders must include Māori and local communities alongside conservation organisations, scientists (research institutes, DOC, universities), recreational and commercial hunting groups, the farming, horticulture, and plantation forestry sectors, pest management entities (Zero Invasive Predators etc.) and government agencies (Ministry for Primary Industries, Ministry for Cities, Environment, Regions and Transport, Department of Conservation, Climate Change Commission, and regional and territorial authorities). These discussions need to be positioned within broader discussions on how best to build resilience in New Zealand to the increasingly severe impacts of climate change. Critically, development of a national deer strategy needs to be informed by the best economic, social, and environmental evidence available.

Developing and effectively implementing a national deer strategy will need to address issues that previous multi-stakeholder efforts have been unable to agree on (DOC1998). Those issues included: (1) whether changes in forests induced by deer and other ungulates are unacceptable, (2) whether deer should be viewed as a pest or resource, (3) whether recreational hunting is an effective means of deer control, (4) whether management of hunting should be DOC's responsibility, and (5) under what circumstances, if any, deer should be controlled by poisons (Griffiths 1998). Failure to resolve these difficult issues in 1998 and the years immediately afterwards, when deer numbers were comparatively low, resulted in an inconsistent approach to deer management (Figgins & Holland 2012). Addressing them is even more important now that deer numbers are rising rapidly and causing potentially irreversible damage to New Zealand's natural forests.

While this paper has primarily addressed the issues around deer and their impacts on forests, other feral ungulates, especially goats, pigs, and tahr, also need to be considered. After red deer, pigs are the second most widespread ungulate in New Zealand and have several additional impacts on top of herbivory, especially through soil turnover and predation (e.g. Parkes et al. 2015). Goats are the third most widespread ungulate and have substantially greater impacts on forests than red deer because of their rumen physiology and the way they feed (Parkes 2021). Tahr are becoming increasingly common in forests in South Westland and appear to have similar impacts to goats in these areas (Yockney & Hickling 2000).

In many parts of New Zealand several ungulates are present (e.g. red deer, goats, and pigs in the Richmond Ranges in the northern South Island, and red deer and tahr in the forests of South Westland) and their collective impact is likely even more catastrophic for the long-term resilience of these forests than that of deer alone (Fig. 8). A key issue that will need to be tackled in developing a national deer strategy is whether this should be broadened to include all ungulates and, if so, how this is done. As ecologists, it makes sense to us that this should occur.



Figure 8. Silver beech forest lacking any understorey vegetation in the presence of red deer and tahr browsing, Landsborough Valley, South Westland (photo David Norton, February 2026).

Conclusions

Our natural forests are critical to New Zealand for so many reasons; as habitat for native biodiversity, providing resilience to catchments, as vast stores of carbon, and for their cultural values to Māori and Pākehā alike. Climate change and introduced animal species, especially deer, stress New Zealand's natural forests. While both are likely to compromise the values that these forests provide New Zealand society, their combined impacts are likely to be far worse. New Zealand society cannot directly mitigate the effects of climate change on old-growth forests, at least in the short-term, as climate change impacts are already locked in by the atmospheric warming that has already occurred, but the vulnerability of old-growth forests to the effects of short- and medium-term climate change can be reduced by reducing the impacts of feral deer. Regenerating forests can also be protected from feral deer so that they stand the best chance of being resilient to climate change as they age and mature.

Control of deer is, however, complex, especially because of the social values these animals have. Ecologically, the case for reducing deer to effectively zero densities is clear, but this cannot be undertaken in isolation from other values. We therefore propose that some native forests are managed to effectively zero deer densities, while others are managed to allow low but non-zero deer densities for both social and ecological values. We believe that there is an urgent need for development of a national deer strategy coupled with the securing of enduring funding as the foundation of averting the looming native forest catastrophe in New Zealand.

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