NEW ZEALAND JOURNAL OF ECOLOGY

REVIEW

The rise and rise of predator control: a panacea, or a distraction from conservation goals?

John R. Leathwick[1]* and Andrea E. Byrom[2]

[1]Heathcote Valley, Christchurch 8022
[2]Independent Environmental Consultant and Te Pūnaha Matatini, Canterbury, Aotearoa/New Zealand
*Author for correspondence (Email: john.leathwick@gmail.com)

Published online: 17 February 2023

Abstract: We review the recent rise to prominence in Aotearoa New Zealand of predation-focused conservation management, critically assessing the likelihood that this will deliver outcomes consistent with national biodiversity goals. Using a review of literature describing the impacts and control of three groups of introduced mammals (wild ungulates, brushtail possums, and predators), we identify shifts in management emphasis over a century of conservation decision-making in Aotearoa. Predators are now a major focus and wild ungulates are left largely uncontrolled, despite increasing populations and evidence for their negative impacts on a wide range of indigenous species and ecosystems. This imbalance in management effort, which appears to be influenced increasingly by socio-political pressures, is much less likely to deliver outcomes consistent with Aotearoa's biodiversity goals than a systematic approach that addresses a full range of biodiversity threats. Overall, we interpret these shortcomings as reflecting long recognised issues with the governance and leadership of Aotearoa's biodiversity system. Changes are required to provide adequate, stable funding, improve clarity around goals, leadership, responsibilities and accountabilities, strengthen planning and prioritisation of management actions, and coordinate management among various conservation actors. We also argue for (1) a stronger role for ecological sciences through independent research aimed at strengthening the evidence base for management actions, and (2) explicit inclusion of science expertise in conservation policy development and management decision making. While recent extensive, landscape-scale predator control has caught the imagination of many and has undoubtedly delivered some gains for a small subset of indigenous species, it also risks creating a false sense of achievement that diverts attention away from other serious gaps in progress towards achieving national biodiversity goals. We make 12 recommendations to address these shortcomings.

Keywords: biodiversity governance, community engagement, conservation policy, ecosystem restoration, invasive mammal, national goals, pest management, Predator-Free New Zealand, ungulate

Introduction

Two of the most striking changes in biodiversity conservation in Aotearoa New Zealand (hereafter Aotearoa) over the last three decades have been a heightened focus on predation as a threatening process (Towns et al. 2019), and a marked expansion in the number of agencies, communities and individuals engaged in conservation. These changes are currently manifest in Predator Free 2050 (PF2050) (Tompkins 2013; Department of Conservation [DOC] 2020a), a government initiative announced in 2016 to coordinate the efforts of individuals and organisations to eliminate seven introduced mammalian predators—three rat species (ship rat *Rattus rattus*, kiore *R. exulans*, and Norway rat *R. norwegicus*), three mustelids (stoat *Mustela erminea*, ferret *M. furo*, and weasel *M. nivalis*), and brushtail possums (*Trichosurus vulpecula*)—from mainland Aotearoa by 2050.

In this review, we evaluate the ability of predation-focused management to deliver results consistent with Aotearoa's national biodiversity goals (DOC 2020b). We do this via a selective review of the longterm impacts and management history of three groups of mammalian pests: ungulates, brushtail possums, and 'predators'. For the purposes of this review, predators includes omnivorous rodents, whose predation of native species has resulted in them being targeted increasingly by a range of conservation actors; possum impacts are reviewed separately because of their recognised significance both as browsers and predators, and as vectors of bovine tuberculosis (TB). We then assess the outcomes likely to be delivered by predation-focused management compared to an approach that systematically addresses a full range of drivers of biodiversity loss. Finally, we consider the role that improvements in Aotearoa's biodiversity system and strengthening of its links with ecological research might play in supporting progress towards more comprehensive biodiversity conservation.

Aotearoa's biodiversity goals are stated in the national biodiversity strategy Te Mana o Te Taiao, (DOC 2020b), a document produced in part to fulfil our obligations as a

DOI: https://doi.org/10.20417/nzjecol.47.3515

signatory to the Convention on Biological Diversity (DOC n.d.a). Outcomes specified in the strategy include “A full range of indigenous ecosystems are protected and secured for future generations” and “All indigenous species are protected and secure, and none are at risk of extinction due to human activities”. The strategy explicitly recognises the inherent complexity of ecological systems, including “diversity within species (including genetic diversity) between species and of ecosystems”.

Impacts and control of introduced mammals

Humans have a long history of introducing mammals, deliberately or accidently, into Aotearoa’s natural ecosystems (Thomson 1922). Most introductions occurred in the late 1800s, often with disastrous consequences. Non-native mammals that established wild populations include seven deer species, tahr, chamois, goats, pigs, five wallaby species, brushtail possums, hares, rabbits, hedgehogs, cats, three mustelid species, and four rodent species (King & Forsyth 2021); feral sheep, cattle, horses and dogs are less common now, but were locally important in the past (McKelvey 1963). Studying the impacts of these taxa on Aotearoa’s indigenous ecosystems, and their management, has been a preoccupying theme in the *New Zealand Journal of Ecology* (NZJE) and predecessor publications (Perry & McGlone 2021), and mitigating their impacts is now a cornerstone of Aotearoa’s conservation management (Parkes & Murphy 2003; Innes et al. 2019).

Ungulate impacts

Concerns that introduced herbivores were having serious negative impacts on indigenous forests and sub-alpine grasslands were raised from the early 1900s (Perham 1922; Cockayne 1928). Numerous publications in the mid- to late-1900s documented those impacts, peaking in the NZJE around 1990 (Perry & McGlone 2021). Earlier publications focused on the dramatic impacts of red deer during invasion of areas not previously subject to ungulate browsing (Holloway 1950; Mark & Baylis 1975; Clarke 1976). Later studies documented marked reductions in the density and diversity of palatable shrubs in forest understories (Wardle 1984), and modification of both sub-alpine scrub (Wardle 1961) and tussock-grasslands (Rose & Platt 1987). Strong evidence emerged that even low to moderate densities of red deer maintain vegetation changes by disrupting regeneration and survival of palatable species (Nugent et al. 2001), with compensatory increases in abundance of non-palatables (Husheer et al. 2005; Wilson et al. 2006; Tanentzap et al. 2009; Forsyth et al. 2015). Compositional changes in vegetation are likely to be accompanied by significant changes in below ground nutrient cycling and invertebrate community structure (Wardle et al. 2001).

Other deer species (white-tailed, fallow, wapiti, sika, sambar, and rusa) have more localised distributions, although some are expanding through escapes from farms or illegal introductions by hunters, particularly in the northern North Island (Fraser et al. 2000). They have broadly similar diets and impacts to red deer (Forsyth et al. 2002), despite sometimes different habitat preferences. Goats are more widespread, though patchily distributed (Parkes 1993a), and can be highly destructive when congregating in favoured habitats (Atkinson 1964; Wardle 1984); feral populations have been boosted at times by release of animals previously captured for farming, particularly in the 1990s (Parkes 2001). Himalayan tahr occur mainly at higher elevations primarily between the Rakaia and Whitcombe Rivers in the north and the Hunter and Haast Rivers in the south; congregations can cause severe damage (Cruz et al. 2017). By contrast, chamois are widespread throughout the South Island, their selective grazing potentially altering the species composition of both tussock grasslands and forests (Yockney & Hickling 2000).

Feral pigs can be particularly destructive due to their omnivorous diet, which includes indigenous snails (McIlroy 2001) and other invertebrates, and probably lizards (Jolley et al. 2010). They also consume seeds of some tree species (Thomson & Challies 1988), and disturb large areas by rooting (Wilson et al. 2006) with major impacts on ferns, *Astelia* spp., and orchids (N. Singers, Ecological Solutions Ltd, pers. comm.).

Ungulate control

Culling of red deer began in the 1930s with the aim of reducing competition with domestic livestock, preventing soil erosion, and protecting native flora (Caughley 1983). Although cullers employed by the Department of Internal Affairs shot as many as 3 million deer between 1932 and 1954, operations were criticised for targeting a reduction in deer numbers rather than mitigating their impact (Caughley 1983). In 1956 responsibilities for deer culling passed to the New Zealand Forest Service (NZFS), which placed a greater focus on protecting native ecosystems, with operations continuing into the early 1970s. Widespread deer control effectively ceased in 1987 with the formation of DOC (Nugent & Fraser 1993).

This cutback in large-scale deer control was facilitated by substantial reductions in ungulate abundance on public conservation land (PCL) from helicopter-based commercial recovery of game meat for export and live capture of animals for farming. The industry commenced in the 1960s, with an annual harvest of 100 000 animals during its peak in the 1970s (Challies 1991). Deer densities were reduced substantially below levels achieved by government cullers, in open habitats by as much as 90–95% (Challies 1991). Conservation gains were sometimes spectacular, particularly in tussock-grasslands (Rose & Platt 1987); more muted but significant recovery was recorded in forests (Stewart et al. 1987).

Other species harvested for meat included pigs, and small numbers of chamois and tahr, leading to substantial reductions in populations of the latter; protests from hunters resulted in a moratorium on commercial harvesting of tahr in 1983 (Challies 1991). Ten recreational hunting areas were designated in which aerial hunting was prohibited, most of these containing populations of red deer. Further restrictions were applied to wild game animal recovery in 2002 after toxins were detected in some export meat (Parkes & Murphy 2003), and harvesting never returned to 1970s levels, reflecting the economics of wild venison recovery compared to farmed venison. Approximately 10 000 animals were harvested from the wild by commercial operators in 2019 (Stuff 2019).

With the decline in the commercial venison industry and reduction in hunting pressure, ungulate populations have increased steadily over the last two decades (Forsyth et al. 2011; Moloney et al. 2021). Anecdotal evidence suggests that they are now recognised increasingly as problematic in both agricultural and some urban settings (Stuff 2021, 2023). However, deer control operations by DOC have gradually decreased over the thirteen years since 2008 (Fig. 1) (DOC 2015; 2021), averaging just over 300 000 ha over the five years to 2021. By contrast, DOC treatment to control possums, rats and mustelids averaged over 850 000 ha over the same period. The intensity of deer control is also much lower, as the aircraft delivery of baits containing sodium monofluoroacetate (1080)

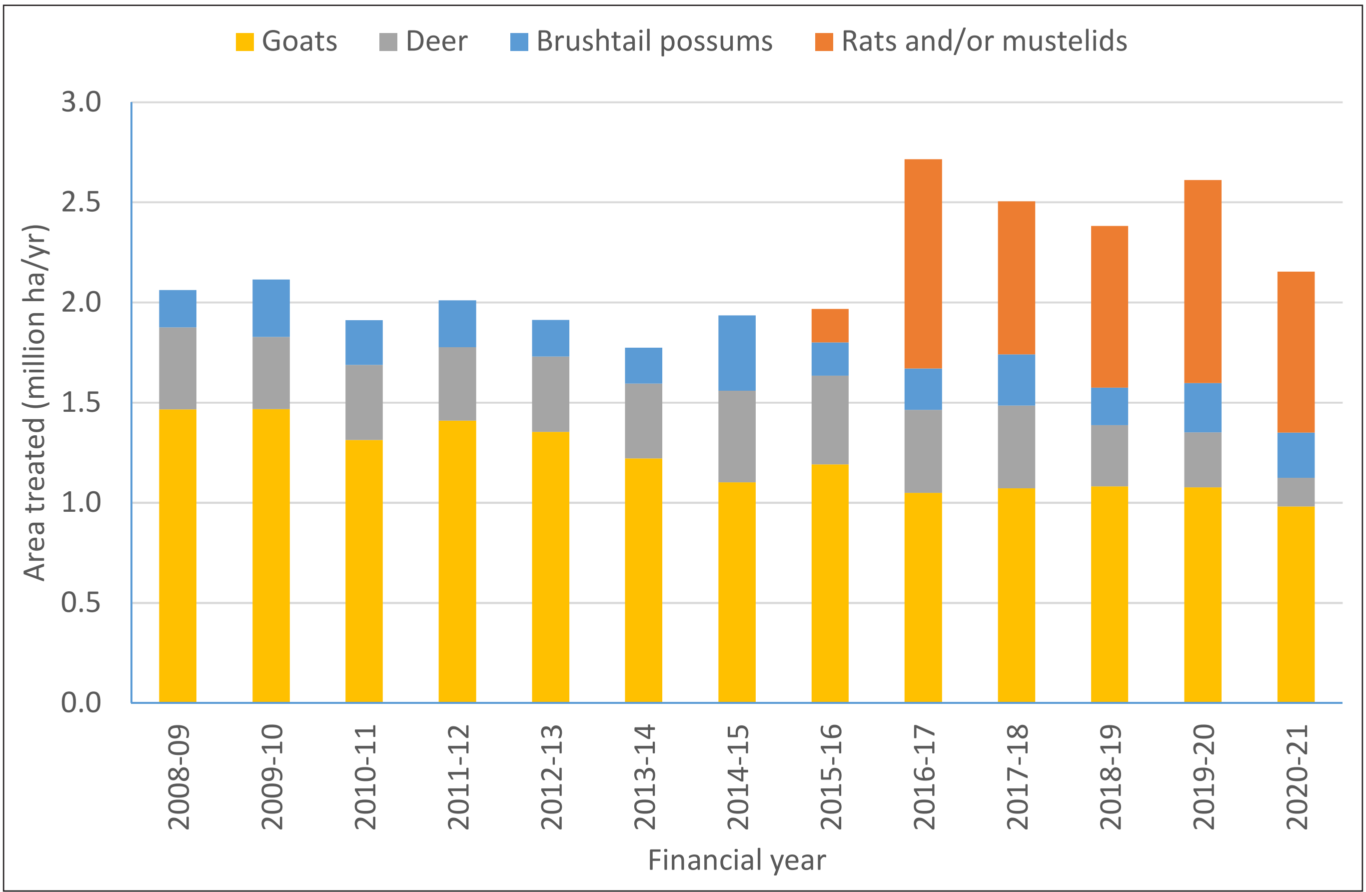

Figure 1. Geographic extent of areas receiving management by DOC for brushtail possums, rats and/or mustelids, deer and goats over thirteen financial years, based on data contained in DOC annual reports (DOC 2013; 2015; 2021).

that is commonly used to control possums, rats and mustelids typically achieves a 90% or higher kill (Byrom et al. 2016), whereas aerial- or ground-hunting operations targeting deer require sustained application to achieve even a moderate level of control, particularly in forests. The area subject to goat control also decreased by more than 30% over this period.

Recent use of deer repellent (Morriss & Yockney 2021) by DOC and OSPRI in aerial 1080 operations targeting possums and/or small predators on PCL, including in National Parks (Driver 2019) and at least one designated forest sanctuary (Environmental Protection Agency 2017) have prompted concerns from conservation groups. On the one hand, use of deer repellent appears to conflict with core conservation legislation (National Parks Act 1980, Conservation Act 1987), and with current deer control policy (DOC 2001). The latter states that "The department's first and over-riding concern is the protection of New Zealand's unique indigenous biodiversity, which takes precedence over the recreational and commercial value of deer as a hunting resource". Conversely, the fact that the toxin loading of 1080 in cereal baits is not optimised for deer gives hunting groups an avenue to raise legitimate welfare concerns highlighting the evolving status of deer and other ungulates within a confusing legislative framework (see below).

Recent limited ungulate control programs have included elimination of illegally introduced sika deer in Northland, continuation of ongoing protection of takahē habitat in the Murchison Mountains and of mountain beech (*Fuscospora cliffortioides*) regeneration in the Kaweka Range (ceased in 2017; Nugent & Speedy 2022), and eradication of red deer from Secretary Island (McDonald et al. 2019).

Some control of ungulates occurs through recreational hunting, although accurate kill data are difficult to obtain. Based on hunter interviews, Kerr & Abell (2014) estimated the kill for 2011–2012 as 135 000 deer, > 230 000 goats, and 132 000 other ungulate species, mostly pigs. Recently a hunter group, the Sika Foundation, were contracted to manage a dense population of sika deer in the Kaimanawa and Kaweka Forest Parks (Nugent & Speedy 2022). While their primary interest is in maintaining "a healthy sika hunting resource" (Sika Foundation n.d.), the contract requires them to "develop a site-specific management programme for deer… to ensure ecosystems are healthy and allow the canopy to regrow" (DOC 2022a). They must also support predator-control and recreation facilities, which will potentially limit resources for deer control. This arrangement appears to risk a significant legislative conflict, given that the two parks together contain four Ecological Areas, identified as representative examples of Aotearoa's indigenous ecosystems (Norton & Overmars 2012); the Conservation Act 1987 specifies that every ecological area "shall so be managed as to protect the value for which it is held". A similar potential conflict occurs in Fiordland National Park where the Fiordland Wapiti Foundation seek to maintain opportunities for trophy hunting while addressing the threats posed to indigenous flora and fauna (Fiordland Wapiti Foundation n.d.).

Brushtail possum impacts and control

Publications describing impacts of possum browsing on indigenous forests peaked in the NZJE in 2000, a decade after those describing ungulate impacts (Perry & McGlone 2021). Some descriptive papers focused on browsing impacts of possums in montane forests dominated by the broadleaved tree species kāmahi (*Weinmannia racemosa*) and rātā (*Metrosideros umbellata*), with spectacular canopy collapse attributed to high population densities of possums that generally occur following invasion of new sites (Pekelharing 1979). Quantitative studies confirmed the ability of possums to induce long-term changes in forest composition through selective feeding (Meads 1976; Veblen & Stewart 1980; Campbell 1990; Allen et al. 1997). Wardle (1984) described selective removal of palatable species by possums in forests with canopies comprising a mix of beech (*Fuscospora* and *Lophozonia*) and broadleaved trees. By contrast, possums have less dramatic canopy impacts in pure beech forests, but they selectively remove a range of palatable species such as *Fuchsia excorticata*, *Pseudopanax* spp., and *Schefflera digitata* from the understorey (Wardle 1984); mistletoes (principally *Peraxilla* spp., *Alepis flavida*) are also highly vulnerable (Sessions et al. 2001). Some accounts describe a synergistic relationship between possum and ungulate impacts on forest composition (McKelvey 1963; Wardle 1984).

Possums were officially declared a pest in 1946 and by 1980 they occupied approximately 90% of Aotearoa, reaching peak numbers in the 1980s (Parliamentary Commissioner for the Environment [PCE] 1994). In the 1960s and early 1970s, small-scale aerial 1080 control by the NZFS was motivated by concerns for browsing impacts on mixed broadleaved (rātā-kāmahi) montane forests (Batcheler 1983) and discovery of their role as vectors for TB (Livingstone et al. 2015). By the late 1970s operations had ceased, reflecting concerns over by-kills of non-target species, uncertainty as to whether possums were the primary cause of forest collapse (Veblen & Stewart 1982), and a shortage of funding for TB-related control (Livingstone et al. 2015). Simultaneously a burgeoning fur industry exported 3.5 million skins annually at its peak in 1979–80 (PCE 2000), most likely with some conservation benefits.

Extensive possum control recommenced in the 1990s after the incidence of TB increased (Livingstone et al. 2015). Initially, TB-related possum control was focussed mainly on farmland and near-farm forests (Warburton & Livingstone 2015), but in 2016 the TB management agency (OSPRI) adopted a goal of eradicating TB everywhere (Nugent et al. 2018). Further impetus for possum control came with recognition in the 1980s of their role as predators of indigenous birds (see below).

Aerial application of 1080 by both OSPRI (and predecessors) and DOC increased substantially from 1992 onwards (PCE 2000; Livingstone et al. 2015). From 2008–2019, OSPRI on average controlled possums over c. 300 000 ha per annum (Environmental Protection Agency 2020), and DOC over a further 225 000 ha (DOC 2015; 2021), although DOC operational coverage varied greatly from year to year depending on the occurrence of mast seeding events in beech forests; regional councils and others on average treated a further 20 000 ha per annum. Notably, TB-related control of brushtail possums is short-term, as the disease can be locally eradicated within 10 years after which control ceases (Nugent et al. 2018). Indigenous plant communities and vulnerable plant species show substantial positive responses to possum control (Holland et al. 2013; Byrom et al. 2016).

Predator impacts and control

The role played by introduced mammalian predators in causing the decline and/or extinction of many of Aotearoa's indigenous species is well documented, particularly for birds (Innes et al. 2010), but also lizards (Reardon 2012) and invertebrates (Bremner et al. 1984; Towns 2008; O'Donnell et al. 2017). In the 1970s and 1980s recognition of the extreme vulnerability to predation of deep endemic bird and reptile taxa resulted in heavy emphasis on the conservation value of predator-free offshore islands (Towns et al. 2012). This led to many previously invaded islands being cleared of predators (and browsers) to create pest-free refuges (Clout & Russell 2006).

Large-scale predation control on the mainland, principally targeting possums, mustelids and rats, began in the late 1990s, focussing on *in situ* conservation of a range of bird species including North Island kōkako and brown kiwi, mōhua, and kākā (Innes et al. 2019). For example, work on kōkako pioneered the use of infrared cameras to identify the role of possums as predators, and demonstrated the feasibility of controlling possums, mustelids and rats to sufficiently low numbers and at large enough spatial scales to enable recovery of a remnant population at Mapara in the central North Island (Innes et al. 1999). These advances prompted DOC's establishment of six 'mainland island' sites, where intensive control of most or all introduced mammals in unfenced areas of up to 6000 ha delivered measurable gains for indigenous taxa (Saunders & Norton 2001).

Building on the mainland island concept, in 1995 a community trust established Aotearoa's first fenced ecosanctuary at Karori (now Zealandia), pioneering the use of pest-proof boundary fencing to prevent reinvasion of a 225 ha site cleared of predators by intensive trapping and poisoning (Burns et al. 2012). Fencing for the larger Maungatautari ecosanctuary (3400 ha) in the Waikato was completed in 2005. A further five ring-fenced and seven peninsula-fenced ecosanctuaries have subsequently been established (Innes et al. 2019) and 47 unfenced sites > 25 ha implement multi-species pest mammal control aimed at ecosystem recovery (Innes et al. 2019).

Following the announcement of PF2050 in 2016, DOC began annual reporting of predator management on PCL, with > 1M ha receiving control targeting rodents, stoats and possums in four of the last six financial years (Fig. 1). This includes extensive control of these predators in beech forests following mast seeding events (Elliott & Kemp 2016) when abundant food facilitates a large increase in rodent, and subsequently mustelid, abundance. DOC publications promoting biodiversity management now frequently highlight predation as the predominant threat to Aotearoa's biodiversity. DOC's 2021 annual report describes Predator Free 2050 as "an ambitious goal to make Aotearoa free of the three introduced predators that cause the greatest harm to our native species – possums, mustelids (ferrets, stoats, weasels) and rats – by 2050".

Evaluating current biodiversity conservation

A century of shifting priorities in invasive mammal control

The most dramatic shift in Aotearoa's approach to biodiversity conservation over the last century revolves around which introduced mammalian pests are targeted for control. We highlight three distinct and overlapping phases (Fig. 2).

First, pest control by the Crown commencing in the

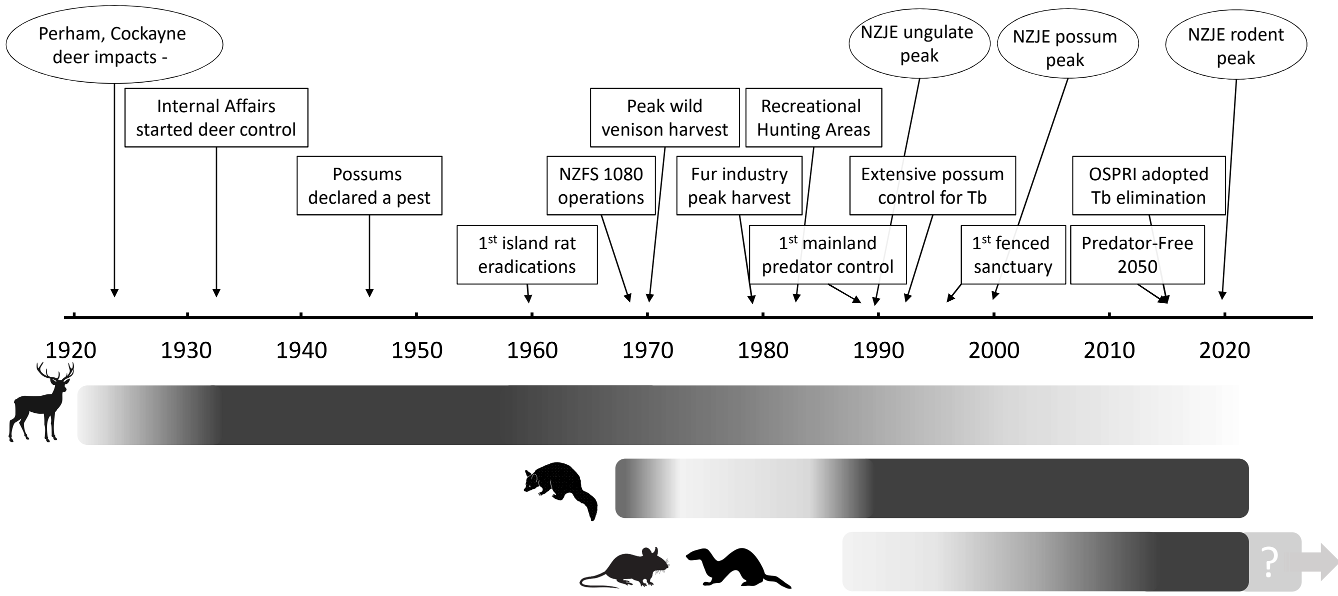

Figure 2. Timeline summarising temporal changes in management of ungulates, brushtail possums and predators by Crown agencies in Aotearoa. Darker shading indicates greater intensity of control operations; key events are listed in text boxes (top bar for deer, middle bar for possums, bottom bar for all predators). Peaks in the number of publications relevant to each group of invasive mammals in NZJE are shown in ellipses.

1930s almost exclusively targeted red deer, but control was progressively scaled back in the 1970s and 1980s following the advent of commercial venison recovery (Challies 1991). Despite a rebound in ungulate populations since the collapse of the venison industry, deer control on PCL has declined over the last decade (DOC 2015, 2021) and use of deer repellent in aerial poisoning operations has increased, even when such operations are carried out in places with the highest levels of statutory protection. Goat operations also declined over the same period (DOC 2015, 2021). These policy swings have been driven almost entirely by lobbying from interest groups; initially, farmers and foresters pressured agencies to remove legal protection and implement deer control in the 1920s (Caughley 1983; Drew 2008); more recently pressure from the hunting lobby has driven the current lacklustre agency response to increasing ungulate populations (Driver 2019; Fig. 2), despite considerable evidence of their negative ecological impacts. Confusion as to the evolving conservation status of ungulates is reflected in legislation and policy. They were first classed as noxious animals, then wild animals, then game animals (while still being wild animals). Most recently, Te Mana o Te Taiao recognises ungulates as "valued introduced species". This evolving status does not align well with current conservation legislation.

Second, operations targeting possums began in the 1970s following recognition of their role both as agents of forest collapse (Batcheler 1983) and as a vector for the spread of TB (Livingstone et al. 2015). Subsequent funding pressures and premature optimism about the apparent success of the TB programme saw a reduction in possum control operations in the early 1980s, resulting in a rapid rebound in both possum abundance and TB prevalence (Livingstone et al. 2015). Renewed commitment to the goal of TB elimination from livestock herds, coupled with recognition of the role of possums as a predator of indigenous taxa (Innes et al. 1999), saw further expansion of possum control operations through the late 1980s and 1990s. Sustained control continues today, but over ever-diminishing areas as TB is progressively eliminated. In the last half-century, possum control arguably has therefore been more coordinated and sustained than ungulate control, in part due to having a clear national goal around TB elimination through a National Pest Management Plan (Livingstone et al. 2015), but also reflecting realisation of their importance as predators as well as browsers. Nonetheless, periods of fluctuating levels of control can be characterised as driven by pressure from interest groups, including hunter opposition to 1080 use, farmer reluctance to fund control in the mid-1970s, subsequent recognition of the economic costs that this incurred, and the inclusion by PF2050 of possums as a predator.

Third, operations targeting rodents and mustelids have increased dramatically in extent since the feasibility of their control on mainland sites was demonstrated in the 1990s (Towns et al. 2019) and since the significance of beech masts in driving predator irruptions became apparent (Barron et al. 2016). Predator control is now the predominant biodiversity conservation action carried out both by DOC (DOC 2021a; Fig. 1) and many other conservation actors including councils, private individuals, community groups, Māori, and philanthropists (Parkes et al. 2017; Towns et al. 2019).

Socio-political influences shape conservation in Aotearoa

A second major shift in Aotearoa's conservation management is the increase of socio-political influences on decision making, particularly in the last few decades. On the one hand, we acknowledge that lobbying can play a positive role: promoting aspirational targets; shaping awareness; and fostering community involvement in conservation. Predator control now involves a wide range of actors including private citizens, community groups, corporate businesses and philanthropists (Parkes et al. 2017; Towns et al. 2019). Further, major advances in predator control have been led by communities, exemplified by fenced ecosanctuaries (Innes et al. 2019), with results shared widely and adopted as best practice (SanctuariesNZ n.d.). Many of these positive outcomes have been driven by individuals and groups who are adept at galvanising and mobilising community action.

However, increased socio-political influence also risks the implementation of inadequate or inappropriate management actions, particularly when promoted by small, vested-interest lobby groups who purport to represent a large proportion of society. For example, individuals or groups within the hunting community have variously mounted vocal publicity campaigns, undertaken legal challenges (High Court Judgement 2021), harassed DOC staff (Stuff 2018), and threatened direct action such as reintroduction of pest species (Towns et al. 2019). While it is difficult to assess the degree to which such activities have driven agency antipathy towards proactive control of deer and feral pig populations, overwhelming evidence for the threats these species pose to indigenous biota indicates a need for much stronger control (informed by density-impact relationships) in areas of high ecological value.

In addition to grassroots interests, well-resourced business and philanthropic investors also shape and influence conservation priorities in Aotearoa, including via political lobbying. While we acknowledge that philanthropic investments can deliver positive biodiversity outcomes, they also have less desirable aspects that have caused concern globally (Holmes 2012). For example, the Tomorrow Accord (Next Foundation n.d.), an agreement between DOC and the Next Foundation, commits government to long-term maintenance of particular large-scale conservation projects initiated by a small group of private investors, exercising a degree of influence that is not, in our view, a robust alternative to systematic evidence-based conservation planning. Further, while public resources are committed to such agreements, critical statutory documents that guide regional conservation management are languishing due to lack of resourcing: as at January 2021, seven of fifteen regional conservation management strategies were out of date, several by more than two decades, and no review was scheduled for five of these (DOC 2021b).

PF2050 has also expanded the influence of business investors by uncritically promoting a shift in language from suppression to eradication of a narrow subset of predators on the mainland (New Zealand Cabinet 2016) without adequate feasibility testing (Parkes et al. 2017; Linklater & Steer 2018; Peltzer et al. 2019), while other damaging predators have been excluded completely from the goal, including feral cats (Rouco et al. 2017), mice (O'Donnell et al. 2017), pigs (Thompson & Challies 1988; Horn et al. 2022), and hedgehogs (Jones et al. 2013). The choice of institutional structure for PF2050 delivery and the decision on which predators should be targeted (New Zealand Cabinet 2016) was driven by influence from interest groups and never underwent consultation with scientists or the public. This has driven a shift in focus to a particular set of management actions (eradication of a subset of mammalian pest species, often in areas that are arguably of low conservation priority) rather than on the desired goals (restoring biodiversity); other drivers of biodiversity loss have been ignored; conservation resources have been diverted from potentially more efficient, multifaceted strategies that could deliver superior outcomes (Innes et al. 2019); and potential risks ignored (Kopf et al. 2017).

Will predation-focused management achieve policy goals?

Optimistically, the recent dramatic upsurge in introduced predator control is supported by evidence demonstrating both the threats they pose to indigenous fauna, and the feasibility of predator control at landscape scales (Innes et al. 2010). Increasing predator control since 2000 can be acknowledged as one of Aotearoa's more remarkable conservation initiatives, delivering demonstrable and sometimes spectacular benefits for selected indigenous fauna (Binny et al. 2021; Fea et al. 2021).

On a less positive note, it is difficult to reconcile Aotearoa's current passive approach to ungulate management (and predators not covered by PF2050) with the biodiversity goals stated in Te Mana o Te Taiao and in key pieces of conservation legislation, which together call for protection of a full range of species and ecosystems. The low priority and funding afforded to management of a broader range of mammalian pests, despite their increasing abundance in many areas, ignores considerable evidence demonstrating their serious negative impacts on indigenous flora and fauna.

Fundamentally however, this is more than a debate about the relative value of predator versus ungulate control. There are three reasons why such a strong focus on predation as a threatening process is problematic.

First, treatment of predation as a pressure that can be managed independently from other drivers of biodiversity decline carries a significant risk of over-simplifying the challenges of managing the complex, interacting assemblages of species and environments that we call ecosystems (Christensen et al. 1996; Kopf et al. 2017). Focusing management on just one driver of biodiversity loss not only ignores the effects of other (well-documented) pressures but also increases the likelihood of perverse outcomes such as favouring small subsets of indigenous taxa at the expense of others (Mulder et al. 2009; Fea et al. 2021), or facilitating ecological release of other, potentially more damaging, invasive mammals (Norbury et al. 2013; Peltzer et al. 2019).

To illustrate: evidence from studies in Aotearoa indicates significant potential for ungulates to have cascading impacts on ecosystems by degrading habitats on which a wide range of indigenous flora and fauna depend (Leathwick et al. 1983; Mills et al. 1989, Wardle et al. 2001), an outcome also observed in northern hemisphere temperate forests (Cotê 2004; Newson et al. 2012; Chollet & Martin 2013, Palmer et al. 2015). Many of Aotearoa's montane beech forests have suffered a marked browser-induced reduction in the diversity and density of broadleaved shrub understories (Wardle 1984) with likely flow-on consequences for predator-vulnerable indigenous fauna, including insectivores (e.g. mōhua, bats, lizards) and seed and fruit eaters (e.g. kākāriki). Instead of focusing on a range of threatening processes in beech forests, conservation of these taxa is now attempted almost exclusively through control of a subset of predators, with the impacts of browsers largely ignored.

Second, and related, we already have strong evidence that management regimes that address a wider suite of threatening processes show greater biodiversity gains. Fenced ecosanctuaries, highly valued for their predator free status (Innes et al. 2019), have not been afforded the recognition they deserve for managing a full range of biodiversity threats and pressures (with the exception of mice) (Clarkson 2022). The 'ecosystem management' and systematic monitoring approach taken by ecosanctuaries has also arguably been more successful in building community support for conservation (Burns et al. 2012; Shanahan et al. 2021). Additionally, some (but not all) of the value delivered by fenced ecosanctuaries can be achieved in large unfenced areas (Saunders & Norton 2001; Burns et al. 2012; Byrom et al. 2016; Parkes et al. 2017; Binny et al. 2021), provided that a full range of biodiversity threats are intensively managed. Yet despite this evidence, systematic expansion of, and dedicated investment in, networks of ecosanctuaries and unfenced pest control areas has largely been discarded as a credible policy, even as a stepping stone towards broad-scale elimination of pests.

Third, we found in reviewing the literature that there was much less (published) supporting evidence for the biodiversity benefits of predator control than we expected. That is, current predation management frequently assumes positive biodiversity outcomes, yet fails to test hypotheses (Betts et al. 2021) that would determine the mechanisms behind observed biodiversity responses. Further, experimentally robust assessments of actual outcomes are rare (Allen et al. 2023). For example, we found just one published account of long-term monitoring (defined as > 10 years or three or more 1080 cycles) of a rat-vulnerable bird species following repeated aerial 1080 applications (Robertson et al. 2019); current landscape-scale projects conducted under the umbrella of PF2050 are yet to publish any quantitative assessments of native biodiversity responses to predator eradication. This lack of robust, hypothesis-driven assessment of native species' responses (Betts et al. 2021), or adherence to basic principles of good experimental design (Allen et al. 2023) makes it difficult to critically test assumptions behind predation-focused management, or to quantify the biodiversity gains it delivers relative to cost.

In light of these issues, we suggest that an urgent evaluation is required of the current disconnect between Aotearoa's predominantly predation-focused management and the fact that predation is just one of a complex set of pressures driving the decline of Aotearoa's indigenous biodiversity. Our intent is not to undermine the value of predator control per se, but to argue that achievement of national biodiversity goals will require a systems approach to management that explicitly addresses a full range of threats to indigenous biota and ecosystems. Until then, considerable doubt must be cast on the ability of predation-focused management to achieve Aotearoa's biodiversity policy goals and, by extension, whether limited conservation resources are being deployed in a cost-effective and scientifically-robust manner.

Building a robust biodiversity system

Changes needed at governance level: reforming the system

In our view, the lack of policy and management stability that characterises Aotearoa's approach to pest management – and biodiversity management more generally – is best understood as reflecting a long history of weak governance across our biodiversity system: "the structure that provides methods for maintaining and managing Aotearoa New Zealand's biodiversity on behalf of all New Zealanders" (DOC 2020b). Shortcomings of this system were documented in DOC's review of Aotearoa's previous biodiversity strategy (DOC 2020c) and by others (Green & Clarkson 2006; Brown et al. 2015; Willis 2017; Innes et al. 2019; Clarkson 2022), and have been identified as a problem more broadly across Aotearoa's public service (Chapple 2019). Problems include a lack of strategic leadership and planning, poor coordination across different levels of government, a lack of clear accountability for implementing different goals, scant and unstable funding, key management decisions not being science-informed, and inadequate prioritisation of resources and actions.

Reform of the biodiversity system was identified as a high priority in Te Mana o Te Taiao, which highlighted many aspects requiring attention (DOC 2002b). At a high level, better governance structures are essential to ensure that the Crown gives effect to the Treaty of Waitangi. While we do not discuss the latter in this review, Treaty-centred governance must be an essential component of future decision-making. Other critical components of improved governance include an overhaul of conservation legislation (DOC 2022b) to strengthen explicit legislative protection to indigenous ecosystems and species (underway), and establishment of shared governance mechanisms that bring together a wider diversity of individuals and organisations and facilitate collegial identification of stable priorities for conservation action, both nationally and regionally.

At a practical level, agreed collaborative governance mechanisms are required to (1) ensure that conservation policy and management are aligned with high level biodiversity goals, (2) ensure management actions are consistent with scientific evidence, (3) oversee monitoring that informs independent auditing of progress towards national goals and effectiveness of management actions, (4) coordinate actions among conservation players, and (5) align them towards stated priorities.

There is ample evidence from Aotearoa and globally that such shared, participatory or collaborative governance mechanisms ('environmental governance') deliver better long-term environmental outcomes than state-led governance alone (Scott 2015; Jager et al. 2020). In particular, environmental governance has been shown to mitigate biodiversity loss; buffer against uncertainty over ecological cause-and-effect relationships; and help minimise long-term environmental damage (Lemos & Agrawal 2006). Governance that empowers and supports environmental stewardship of Indigenous peoples and utilises local and Indigenous knowledge (Armitage et al. 2012) is one of the strongest pathways to long-term conservation of biodiversity, particularly when resourced adequately and supported by wider law and policy (Dawson et al. 2021). Of available governance mechanisms, devolution of power to make decisions is likely to deliver the strongest conservation outcomes (Jager et al. 2020), with Indigenous or local governance delivering equally if not more effective environmental protection (Schleicher et al. 2017; Dawson et al. 2021). These findings are particularly notable given Aotearoa's Tiriti o Waitangi context and The Crown's obligations for shared decision-making, further strengthening the case for re-thinking environmental governance in this country.

Most importantly, strengthening governance mechanisms should safeguard against undue influence from minority interests. Private individuals can invest in whatever conservation actions they choose, but they should not necessarily expect to access public funds, nor skew public funding towards priorities of their choice. Similarly, stronger governance can help build social cohesion around conservation, reducing the risks of vested interest groups deliberately undermining local conservation initiatives (e.g. by release of game animals).

Operational changes needed: implementing systematic conservation management

At an operational level, we identify four areas where change is needed. First, we need to robustly quantify the level of investment that would be required to reverse the decline of biodiversity in Aotearoa, as has been done for Australia (Wintle et al. 2019). Despite being explicitly called for in Te Mana o Te Taiao ("The costs and value of restoring indigenous biodiversity have been quantified and are being actively used to inform decision making"; DOC 2020b), to our knowledge such an assessment has not been attempted. Results would clearly identify the magnitude of the work that needs to be done (rather than starting from a scarcity mindset), and would contribute to the implementation of two critical objectives identified in the strategy: "Governance, legislation and funding systems are in place…" and "Biodiversity protection is at the heart of economic activity".

Second, we need to effectively prioritise existing and new investments, recognising that Aotearoa's current conservation investments are insufficient to implement comprehensive management across all surviving indigenous ecosystems. One cost-effective and socially acceptable interim approach would be to revisit Sir Paul Callaghan's original vision (Stuff 2011), i.e. identify a network of high priority sites across the country, both on and off PCL and including large inhabited islands, where all biotic threats would ideally be critically evaluated and comprehensively managed. To mitigate conflicts arising from desired alternative land uses, selection of priority sites should be informed by other values such as tourism and recreation, including hunting. Applying less intensive management between priority sites, where feasible,would help build broader connectivity among sites. Such a network could be progressively expanded if/when additional funds became available. Spatial conservation prioritisation software designed to optimise tradeoffs among conflicting values (Moilanen et al. 2011; Whitehead et al. 2014) would support the design of such a multi-purpose, interconnected network.

Individual sites should be large enough to allow inclusion of sequences of interconnected ecosystems and facilitate efficient management, while collectively these sites should include both representative examples of a full range of Aotearoa's ecosystems, and populations of threatened taxa requiring active management for their recovery or persistence. Additionally, with a strong, time-bound plan for rollout and implementation, active biodiversity restoration could expand outwards between sites (Clarkson et al. 2018; Glen et al. 2013; Parkes et al. 2017). This approach would have the advantage of targeting the highest priority ecosystems first; would foster coordination and collective responsibility across agencies that currently work independently (Willis 2017); support the priorities and values of local communities including mana whenua; and help build social licence (Harper et al. 2020).

DOC has already used an approach incorporating some of these elements to identify a network of representative "ecosystem management units" on PCL, as a framework within which to prioritise management of ecosystems and threatened species (DOC n.d.b; Brown et al. 2015; Parkes et al. 2017). However, while DOC reports annually on the number of these units currently receiving management (DOC 2021), it provides no information on the scope or intensity of management actions prescribed for each site, the proportion of prescribed actions that have been successfully implemented, or the biodiversity gains that have been achieved.

Third, we must find more effective ways to draw together the relative strengths of grassroots conservation done by communities compared to the roles played by top down national vision and leadership. As we note elsewhere, many regional and local communities now play an active role in biodiversity conservation and restoration, these activities often having strong cultural and social benefits in addition to their biodiversity gains. Added advantages would include stronger social support for collectively-agreed and coordinated processes for expansion of management between priority sites, protection of mahinga kai (traditional food-gathering), and enhanced carbon sequestration in regenerating forest. National initiatives that seek to actively support and coordinate these bio-regional restoration activities are one of the more powerful tools we have to mitigate biodiversity loss (Clarkson et al. 2018).

Regional biodiversity priorities have been identified by many councils (e.g. Hawkes Bay Regional Council n.d.), who argue for tighter integration of efforts across a full range of conservation actors (Willis 2017). In practice, on-ground integration amongst councils, mana whenua, DOC, and community groups is highly variable across regions. This highlights an urgent need for the establishment of robust, collegial approaches to national and regional conservation planning that empower and resource communities to collectively contribute to Aotearoa's high level biodiversity goals, as provided for in the draft National Policy Statement on Indigenous Biodiversity (Ministry for the Environment n.d.).

Fourth, we need to systematically monitor outcomes. Despite monitoring being an essential prerequisite for adaptive management (van Dam-Bates et al. 2018; McGlone et al. 2020), investment in measuring biodiversity outcomes in Aotearoa is currently minimal. Tier One of a three-tiered biodiversity monitoring system (DOC n.d.c) was implemented by DOC in 2013, but has been progressively scaled back due to budget and operational constraints; plots are now re-measured every ten years rather than the five years originally planned; and quality control and audit components have largely ceased (DOC 2022c). A Tier Two sampling protocol was designed for intensively managed sites (van Dam-Bates et al. 2018) but has received minimal funding for implementation (\$300 000 year^{-1} from 2018 to 2020); further development of this component is currently paused. Some ad hoc outcome monitoring is undertaken by DOC operations staff, but has decreased over the last decade, and integration and reporting of results is minimal. This leaves assessment of biodiversity gains from DOC's management dependent largely on decadal measurements of a widely spaced network of plots, only a very small proportion of which are located within areas receiving intensive management. Most communities lack the resources and expertise to undertake comprehensive monitoring, so DOC must play a stronger leadership role in providing underpinning logistics, tools and guidance.

The role of ecological science and research

Ecological science needed as robust underpinning evidence

In undertaking this review we became increasingly concerned at the degree to which ecological research has often been a passive responder, focusing on the impacts of narrow groupings of invasive mammals in response to socio-political drivers, rather than offering long-term, whole-system viewpoints. Conversely, where such viewpoints are advanced (Braysher et al. 2012), they are often ignored by conservation managers, politicians, and influential lobby groups. The influence of socio-political pressures on science is clearly evident in recent publications in NZJE, which show a strong shift towards predation-focused studies; a search of keywords associated with papers published since 2000 identified 38 using the keyword ‘predator’, while only fifteen used the keyword ‘browser’ or ‘herbivore’ (and one of these focused almost exclusively on the responses of fauna to predator control).

Although such shifts in research focus are understandable given both the influential role of funding agencies and the high degree of uncertainty in Aotearoa’s science funding system, it is essential that researchers are able to independently challenge and test paradigms driving current conservation management. This has happened historically, e.g. when the primary role of browsers as causal agents for rātā-kāmahi forest decline was questioned and alternative arguments advanced in favour of the importance of natural disturbance (Veblen & Stewart 1982). Subsequent debate (Batcheler 1983) contributed to a more balanced understanding, and the roles of both natural disturbances and the negative impacts of introduced browsers are now generally acknowledged (Ogden et al. 1996).

Ecological research needs to play two main roles to support conservation and biodiversity protection. First, research needs to provide an evidential basis from which to make informed decisions about management interventions such as pest control. Second, scientists should be free to robustly critique assumptions made by policy makers and conservation managers, putting forward alternative perspectives and challenging current paradigms. We suggest four ways in which targeted research could contribute to whole-system, evidence-based biodiversity management.

First, comparative studies could be used to expand beyond a predation-only focus and more explicitly consider the interactive effects of a range of threatening processes and the impact of these on indigenous taxa (Norbury et al. 2013; Binny et al. 2021). Fenced ecosanctuaries and other intensively managed sites, along with mammal-free off-shore islands, already provide valuable opportunities for comparison, given their comprehensive reduction of biodiversity pressures. However, as these are generally biased towards lowland forests (Innes et al. 2019), a different approach would be needed for extensive montane beech forests or dryland ecosystems that currently receive extensive predator control but few if any other interventions. In many mainland locations, hypothesis testing and experimental approaches are needed to assess synergies between predator removal and improvements in habitat quality delivered by concurrent control of ungulates (and a wider suite of predators) (Allen et al. 2023).

Second, experimental manipulations, with supporting ecological models, could be better leveraged to infer impacts of different combinations of pressures on indigenous biodiversity from field observations (Ruscoe et al. 2005; Holland et al. 2013) and to predict responses to management interventions (Tompkins et al. 2013). Large-scale, long-term, and well-funded field experiments, designed to underpin predictive ecological models, are vital components of research needed to support robust biodiversity and ecosystem conservation (Allen et al. 2023). Unless such experiments are done and hypotheses tested, there is a significant risk that key drivers of decline may be overlooked or discounted (Caughley & Gunn 1996; Betts et al. 2021). Currently however, such studies are not viewed as a high priority for research funding (PCE 2020).

Third, as noted above, measuring the biodiversity outcomes delivered by management interventions is essential. At the very least, irrespective of whether the management objective is suppression or eventual eradication of pests, measurement of biodiversity outcomes would support the quantification of density-impact relationships, deepening our understanding of the level of management effort required to deliver a desired biodiversity response (Tanentzap et al. 2009; Forsyth et al. 2013; Norbury et al. 2015).

Finally, whilst there will always be a desire to expand the range of tools in the toolbox for control or eradication of invasive mammal pests, in our view a critical gap is the link to pest ecology: strategic deployment of the right tools in the right sequence to achieve both a target pest density and a desired biodiversity outcome (Parkes 1993b). Where eradication is not yet feasible, sufficient tools must be available to suppress pests to a level that initiates and maintains improvement in the biodiversity condition of different ecosystems. Investment in new tools should thus be strongly linked to density-impact relationships, i.e. to parts of the pest-density spectrum where improved efficiency, reduced costs and adjustments for social licence to operate are needed to achieve a desired biodiversity outcome. Where eradication is feasible, or may become feasible in the future, applying the right tools in the right sequence is vital (Horn et al. 2022). Yet despite significant investment in development of new tools for predators by both DOC (DOC n.d.d) and PF2050 Limited (PF2050 n.d.), such investment is not linked to target densities of predators or to desired biodiversity outcomes (for suppression), nor to strategic sequencing of new tools (for eradication). Such sequencing is critical if eradication is to succeed (Parkes 1993b; Bomford & O’Brien 1995; Horn et al. 2022).

Building ecological research capability and capacity

In general, ecological sciences are strongly single-discipline-focused in Aotearoa: inter- and trans-disciplinary, cross-cutting ecosystem ecology and Mātauranga Māori have been chronically under-funded for decades (Green & Clarkson 2006; PCE 2020; Kukutai et al. 2021). A stronger focus on hypothesis testing and good experimental design are also required to understand the mechanisms behind observed ecological responses (Betts et al. 2021; Allen et al. 2023). As a consequence, the pool of specialist skills needed to advance knowledge on complex ecosystem dynamics, test hypotheses in an experimental framework, measure biodiversity responses to management actions, construct predictive models that can support decision-making, and translate that information into policy is extremely limited. Building such capability and capacity should be an urgent investment priority as part of Te Ara Paerangi, the review of Aotearoa’s research system (Ministry of Business, Innovation and Employment n.d.).

Significant changes will also be required to strengthen agency uptake of science and ensure non-government actors have access to best practice and knowledge if existing (and new) knowledge is to better shape Aotearoa’s conservation policy and management decision making (Linklater & Steer 2018; Selwood et al. 2019). Such uptake requires (1) employment of relevant scientific expertise within regional and national agencies, (2) active inclusion of scientific expertise in evidence-based policy development and decision-making, (3) commitment to maintaining strong links between in-house scientists and expertise housed in universities and Crown Research Institutes, and (4) bridging the gap between knowledge generation and implementation, as noted in Te Ara Paerangi. Planning processes that use spatial conservation planning tools (Moilanen et al. 2009) to integrate expert knowledge, social values and descriptions of biodiversity patterns could offer major improvements in conservation management (Whitehead et al. 2014; Selwood 2019).

Conclusions and recommendations

Although this review was initially prompted by our concerns around imbalances in Aotearoa's management of predation versus other biodiversity pressures, we gradually came to see this imbalance not so much as the primary issue, but as one symptomatic of a broader systemic failure in national biodiversity management. While recent extensive landscape-scale predator control has caught the imagination of many in Aotearoa and has undoubtedly delivered some gains for a small subset of indigenous species, it also risks creating a false sense of achievement that diverts attention from other serious gaps in progress towards reaching national biodiversity goals. Addressing these gaps is a major challenge given the complexity of Aotearoa's biodiversity system with its inadequate funding, multiplicity of players, and current lack of clarity around leadership, directions, roles, responsibilities, and accountabilities. We conclude by identifying 12 areas where in our view change is urgently required:

(1) **Implement system-wide governance** that brings together a full range of actors, centres Te Tiriti o Waitangi, shares decision-making power, facilitates stable conservation management through coordinated, long-term direction setting, and safeguards against undue influence from minority interest groups.

(2) **Quantify the total investment needed to reverse the decline of biodiversity** in Aotearoa to provide a context for future cost-benefit analyses, the prioritisation of current conservation investment, and the exploration of novel mechanisms to secure adequate and stable long-term conservation funding.

(3) **Develop a stronger vision and foster inclusive leadership** that clearly communicates Aotearoa's biodiversity goals to a wide audience, the strategies and tactics that will be used to achieve those goals, and progress made toward them.

(4) Adopt a **systems approach to management that addresses a wider range of biodiversity threats**, rather than emphasising management of one threat to the exclusion of others.

(5) **Critically review the evidential support for, and cost-effectiveness of, the Predator Free 2050 strategy** including the scope of biodiversity pressures it addresses, the appropriateness of the three groups of predators currently targeted, and the degree to which it is likely to deliver outcomes congruent with Aotearoa's high level biodiversity goals.

(6) Use evidence-informed approaches to identify a **network of interconnected mainland sites and offshore islands** that provide **adequate and comprehensive representation** of a full range of Aotearoa's ecosystems and threatened taxa, and undertake **robust feasibility planning** for landscape-scale pest control and/or eradications at those sites.

(7) Develop cost-effective and collegial approaches to the **implementation of comprehensive, intensive management of biodiversity threats** across and between this network of sites.

(8) **Link conservation policy and management actions** much more strongly to **underpinning ecological evidence**, including scientists in policy development, management decision making and the evaluation of outcomes.

(9) Promote initiatives that **support local and regional conservation efforts** both through appropriate transfer of nationally-held technical skills and insights, and facilitating sharing of local and regional insights.

(10) **Clearly state desired biodiversity outcomes for all major conservation projects**, and undertake **systematic and comprehensive monitoring of the indigenous biodiversity responses they achieve** to quantify gains made and determine whether policy goals are being achieved.

(11) Increase **research investment in capability and capacity in complex ecosystem ecology** to robustly underpin evidence-based management and outcome monitoring, and **link investment in new tools and technologies to strategic deployment of the right tools in the right sequence** to achieve both desired target pest densities and biodiversity outcomes.

(12) **Accelerate the planned overhaul of conservation legislation to provide explicit protection** to all indigenous ecosystems and species in a manner consistent with the biodiversity goals stated in Te Mana O Te Taio.

Acknowledgements

The ideas presented here benefitted substantially from discussions with Marie Doole, Rod Hay, John Innes, Graham Nugent, Cynthia Roberts, James Reardon, and Nicholas Singers; their input is gratefully acknowledged. Comments from Bruce Clarkson, John Innes, Grant Norbury, Roger Pech, Bruce Warburton, and two anonymous reviewers substantially improved earlier drafts of the manuscript. Comments from the editor provided significant additional insights.

Data and code availability

There is no data or code associated with this work.

Author contributions

JL wrote an initial draft that was subsequently expanded and revised by JL and AB.

References

Allen RB, Fitzgerald AE, Efford MG 1997. Long-term changes and seasonal patterns in possum (*Trichosurus vulpecula*) leaf diet, Orongorongo Valley, Wellington, New Zealand. New Zealand Journal of Ecology 21: 181–186.

Allen RB, Forsyth DM, MacKenzie DI, Peltzer DA 2023. Robustness of field studies evaluating biodiversity responses to invasive species management in New Zealand. New Zealand Journal of Ecology 47: 3503.

Armitage D, de Loë R, Plummer R 2012. Environmental governance and its implications for conservation practice. Conservation Letters 5: 245–255.

Atkinson IAE 1964. Relations between feral goats and

vegetation in New Zealand. Proceedings of the New Zealand Ecological Society 11: 39–44.
Barron MC, Pech RP, Christie JE, Tait A, Byrom A, Elliot G 2016. Climate change impacts and implications: an integrated assessment in the alpine case study. Synthesis Report: RA2 Alpine Case Study. The beech forests of New Zealand. Climate change impacts and implications for New Zealand to 2100. MBIE contract C01X1225. Wellington, MBIE. 19 p.
Batcheler CL 1983. The possum and rata-kamahi dieback in New Zealand: a review. Pacific Science 37: 415–426.
Betts MG, Hadley AS, Frey DW, Frey SJK, Gannon D, Harris SH, Kim H, Kormann UG, Leimberger K, Moriarty K, Northrup JM, Phalan B, Rousseau JS, Zárrate-Charry D, Stokely TD, Valente JJ, Wolf C 2021. When are hypotheses useful in ecology and evolution? Ecology and Evolution 11: 5762–5776.
Binny RN, Innes J, Fitzgerald N, Pech R, James A, Price R, Gillies C, Byrom AE 2021. Long-term biodiversity trajectories for pest-managed ecological restorations: eradication vs. suppression. Ecological Monographs 91: 1439.
Bomford M, O'Brien P 1995. Eradication or control for vertebrate pests? Wildlife Society Bulletin 23: 249–255.
Braysher M, Buckmaster T, Saunders G, Krebs CJ 2012. Principles underpinning best practice management of the damage due to pests in Australia. In: Timm RM ed. Proceedings of the 25th vertebrrate pest conference. Davis, University of California. Pp. 300–307.
Bremner AG, Butcher CF, Patterson GB 1984. The density of indigenous invertebrates on three islands in Breaksea Sound, Fiordland, in relation to the distribution of introduced mammals. Journal of the Royal Society of New Zealand 14: 379–386.
Brown MA, Stephens RT, Peart R, Fedder B 2015. Vanishing nature: facing New Zealand's biodiversity crisis. Auckland, Environmental Defence Society Incorporated.
Burns B, Innes J, Day T 2012. The use and potential of pest-proof fencing for ecosystem restoration and fauna conservation in New Zealand. In: Somers MJ, Hayward M eds. Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? New York, Springer Science & Business Media. Pp 65–90.
Byrom AE, Innes J, Binny R 2016. A review of biodiversity outcomes from possum-focused pest control in New Zealand. Wildlife Research 43: 228–253.
Campbell DJ 1990. Changes in the structure and composition of a New Zealand lowland forest inhabited by brushtail possums. Pacific Science 44: 277–296.
Caughley G 1983. The deer wars. The story of deer in New Zealand. Auckland, Heinemann. 187 p.
Caughley G, Gunn A 1996. Conservation biology in theory and practice. Massachusetts, Blackwell. 459 p.
Challies CN 1991. Status and future management of the wild animal recovery industry. New Zealand Forestry 36: 10–17.
Chapple S 2019. From mandarin to valet public service? State sector reform and problems of managerialism in the New Zealand public service. Policy Quarterly 15: 49–55.
Chollet S, Martin J-L 2013. Declining woodland birds in North America: should we blame Bambi? Diversity and Distributions 19: 481–483.
Christensen NL, Bartuska AM, Brown JH, Carpenter S, D'Antonio C, Francis R, Franklin JF, MacMahon JA, Noss RF, Parsons DJ, Peterson CH 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological applications 6: 665–691.
Clarke CMH 1976. Eruption, deterioration and decline of the Nelson red deer herd. New Zealand Journal of Forestry Science 5: 235–49.
Clarkson BD 2022. Reversing biodiversity decline in Aotearoa New Zealand. Policy Quarterly 18: 61–70.
Clarkson BD, Kirby CL, Wallace KJ 2018. Restoration targets for biodiversity depleted environments in New Zealand. Report prepared for The Biodiversity Collaborative Group by The Environmental Research Institute, University of Waikato. 23 p.
Clout MN, Russell JC 2006. The eradication of mammals from New Zealand islands. In: Koike F, Clout MN, Kawamichi M, De Poorter M, Iwatsuki K eds. Assessment and control of biological invasion risks. Kyoto/Gland, Shoukadoh Book Sellers and the World Conservation Union (IUCN). Pp 127–141.
Cockayne L 1928. The vegetation of New Zealand. 2nd edn. Liepzig, Verlag W, Engelmann. 456 p.
Cotê SD, Rooney TP, Tremblay J-P, Dussault C, Waller DM 2004. Ecological impacts of deer overabundance. Annual Review of Ecology, Evolution, and Systematics 35: 113–147.
Cruz J, Thomson C, Parkes JP, Gruner I, Forsyth DM 2017. Long-term impacts of an introduced ungulate in native grasslands: Himalayan tahr (*Hemitragus jemlahicus*) in New Zealand's Southern Alps. Biological Invasions 19: 339–349.
Dawson NM, Coolsaet B, Sterling EJ, Loveridge R, Gross-Camp ND, Wongbusarakum S, Sangha KK, Scherl LM, Phuong Phan H, Zafra-Calvo N, Lavey WG, Byakagaba P, Idrobo CJ, Chenet A, Bennett NJ, Mansourian S, Rosado-May FJ 2021. The role of Indigenous peoples and local communities in effective and equitable conservation. Ecology and Society 26(3): 19.
Department of Conservation 2001. Department of Conservation's policy statement on deer control. Wellington, Department of Conservation.
Department of Conservation 2015. Annual report for the year ended 30 June 2015. Wellington, Department of Conservation.
Department of Conservation 2020a. Towards a predator free New Zealand: Predator Free 2050 Strategy. Wellington, Department of Conservation.
Department of Conservation 2020b. Te Mana o Te Taiao – Aotearoa New Zealand Biodiversity Strategy 2020. Wellington, Department of Conservation.
Department of Conservation 2020c. Lessons learnt from the 2000 New Zealand Biodiversity Strategy. Wellington, Department of Conservation.
Department of Conservation 2021a. Annual Report for the year ended 30 June 2021. Wellington, Department of Conservation.
Department of Conservation 2021b. https://www.doc.govt.nz/about-us/our-policies-and-plans/statutory-plans/statutory-planning-status-report/ (Accessed 18 January 2023).
Department of Conservation 2022a. https://www.doc.govt.nz/news/media-releases/2022-media-releases/sika-foundation-turns-sights-onto-ecological-restoration/ (Accessed 18 January 2023).
Department of Conservation 2022b. Partial reviews of the Conservation General Policy and General Policy for

National Parks regarding Te Tiriti o Waitangi / the Treaty of Waitangi Report of the Options Development Group. Wellington, Department of Conservation.

Department of Conservation 2022c. Response to OIA request to DOC September 16th, 2020 (OIAD-2267).

Department of Conservation n.d.a. https://www.doc.govt.nz/about-us/international-agreements/convention-on-biological-diversity/ (Accessed 18 January 2023).

Department of Conservation n.d.b. https://www.doc.govt.nz/about-us/our-role/managing-conservation/natural-heritage-management/identifying-conservation-priorities/setting-priorities/ (Accessed 18 January 2023).

Department of Conservation n.d.c. https://www.doc.govt.nz/our-work/monitoring-and-reporting-system/ (Accessed 18 January 2023).

Department of Conservation n.d.d. https://www.doc.govt.nz/nature/pests-and-threats/predator-free-2050/goal-tactics-and-new-technology/tools-to-market/ (Accessed 18 January 2023).

Drew K 2008. Deer and deer farming - Introduction and impact of deer, Te Ara - the Encyclopedia of New Zealand, http://www.TeAra.govt.nz/en/deer-and-deer-farming/page-1 (Accessed 18 January 2023).

Driver G 2019. DOC spending hundreds of thousands on controversial deer repellent. https://www.wildernessmag.co.nz/doc-spending-hundreds-of-thousands-on-controversial-deer-repellent/ (Accessed 18 January 2023).

Elliott G, Kemp J 2016. Large scale pest control in New Zealand beech forests. Ecological management and restoration 17: 200–209.

Environmental Protection Agency 2017. https://epa.govt.nz/assets/Uploads/Documents/Hazardous-Substances/1080-reports/1080-Operational-Reports/9de5978c35/2016-Manawatu-Wanganui-Hihitahi-Report.pdf (Accessed 18 January 2023).

Environmental Protection Agency 2020. 1080 use in Aotearoa New Zealand 2019. An annual report on the aerial operations, research, and incidents relating to 1080 use. Environmental Protection Agency, Wellington.

Fea N, Linklater W, Hartley S 2021. Responses of New Zealand forest birds to management of introduced mammals. Conservation Biology 35: 35–49.

Fiordland Wapiti Foundation n.d. https://fwf.net.nz/ (Accessed 18 January 2023).

Forsyth DM, Coomes DA, Nugent G, Hall GMJ 2002. The diet and diet preferences of introduced ungulates (Order: Artiodactyla) in New Zealand. New Zealand Journal of Zoology 29: 333–353.

Forsyth DM, Thomson C, Hartley LJ, MacKenzie DI, Price R, Wright EF, Mortimer JAJ, Nugent G, Wilson L, Livingstone P 2011. Long-term changes in the relative abundances of introduced deer in New Zealand estimated from faecal pellet frequencies. New Zealand Journal of Zoology 38: 237–249.

Forsyth DM, Ramsey DSL, Veltman CJ, Allen RB, Allen WJ, Barker RJ, Jacobson CL, Nicol SJ, Richardson SJ, Todd CR 2013. When deer must die: large uncertainty surrounds changes in deer abundance achieved by helicopter-and ground-based hunting in New Zealand forests. Wildlife Research 40: 447–458.

Forsyth D, Wilson D, Easdale T, Kunstler G, Canham C, Ruscoe W, Wright E, Murphy L, Gormley A, Gaxiola A, Coomes D 2015. Century-scale effects of invasive deer and rodents on the dynamics of forests growing on soil of contrasting fertility. Ecological Monographs 85: 157–180.

Fraser KW, Cone JM, Whitford EJ 2000. A revision of the established ranges and new populations of 11 introduced ungulate species in New Zealand. Journal of the Royal Society of New Zealand 30: 419–437.

Glen AS, Pech RP, Byrom AE 2013. Connectivity and invasive species management: towards an integrated landscape approach. Biological Invasions 15: 2127–2138.

Green WQ, Clarkson B 2006. Turning the tide? A review of the first five years of the New Zealand biodiversity strategy. The Synthesis Report. Wellington, Department of Conservation. 52 p.

Harper GA, Pahor S, Birch D 2020. The Lord Howe Island rodent eradication: Lessons learnt from an inhabited island. In: Woods DM ed. Proceedings 29th Vertebrate Pest Conference, Paper No. 31. Pp. 36–46.

Hawkes Bay Regional Council n.d. https://www.hbrc.govt.nz/environment/biodiversity/biodiversity-strategy/ (Accessed 18 January 2023).

High Court Judgement 2021. CIV-2020-485-324 [2020] NZHC 1669. New Zealand High Court, Wellington.

Holland EP, Pech RP, Ruscoe WA, Parkes JP, Nugent G, Duncan RP 2013. Thresholds in plant–herbivore interactions: predicting plant mortality due to herbivore browse damage. Oecologia 172: 751–766.

Holloway JT 1950. Deer and the forests of western Southland. New Zealand Journal of Forestry 6: 123–137.

Holmes G 2012. Biodiversity for billionaires: Capitalism, conservation and the role of philanthropy in saving/selling nature. Development and Change 43: 185–203.

Horn SR, Sagar RL, Frank VK, Cox FS, Jaques PM, Ware J, Hanley-Nickolls R, Leask EP, MacDonald NL, Kirby-Crowe MS, Le Lievre ME, Broome KG 2022. The next frontier: assessing the feasibility of eradicating mammalian pests from Auckland Island. New Zealand Journal of Ecology 46: 3500.

Hunt E 2020. Every dog has its day. New Scientist, April 2020: Pp. 36–39.

Husheer SW, Hansen QW, Urlich SC 2005. Effects of red deer on tree regeneration and growth in Aorangi Forest, Wairarapa. New Zealand Journal of Ecology 29: 271–277.

Innes J, Hay I, Flux I, Bradfield P, Speed H, Jansen P 1999. Successful recovery of North Island kokako *Callaeas cinerea wilsoni* populations, by adaptive management. Biological Conservation 87: 201–214.

Innes J, Kelly D, Overton JM, Gillies C 2010. Predation and other factors currently limiting New Zealand forest birds. New Zealand Journal of Ecology. 34: 86–114.

Innes J, Fitzgerald N, Binny R, Byrom AE, Pech R, Watts C, Gillies C, Maitland M, Campbell-Hunt C, Burns B 2019. New Zealand ecosanctuaries: types, attributes and outcomes. Journal of the Royal Society of New Zealand 49: 370–393.

Jager NW, Newig J, Challies E, Kochskämper E 2020. Pathways to implementation: Evidence on how participation in environmental governance impacts on environmental outcomes. Journal of Public Administration Research and Theory 30: 383–399.

Jolley DB, Ditchkoff SS, Sparklin BD, Hanson LB, Mitchell MS, Grand JB 2010. Estimate of herpetofauna depredation by a population of wild pigs. Journal of Mammalogy 91: 519–524.

Jones C, Norbury GL, Bell T 2013. Impacts of introduced European hedgehogs on endemic skinks and weta in

tussock grassland. Wildlife Research 40: 36–44.
Kerr GN, Abell W 2014. Big game hunting in New Zealand: per capita effort, harvest and expenditure in 2011–2012. New Zealand Journal of Zoology 41: 124–138.
King CM, Forsyth DM 2021. eds. The handbook of New Zealand mammals. 3rd edn. Canberra, CSIRO Publishing. 576 p.
Kopf RK, Nimmo DG, Humphries P, Baumgartner LJ, Bode M, Bond NR, Byrom AE, Cucherousset J, Keller RP, King AJ, McGinness HM, Moyle PB, Olden J 2017. Confronting the risks of large-scale invasive species control. Nature Ecology & Evolution 1:0172.
Kukutai T, McIntosh T, Boulton A, Durie M, Foster M, Hutchings J, Mark-Shadbolt M, Moewaka Barnes H, Moko-Mead T, Paine S-J, Pitama S, Ruru J 2021. Te Pūtahitanga: A Tiriti-led science- policy approach for Aotearoa New Zealand. Auckland, Ngā Pae o te Māramatanga. 42 p.
Leathwick JR, Hay JR, Fitzgerald AE 1983. The influence of browsing by introduced mammals on the decline of the North Island kokako. New Zealand Journal of Ecology 6: 55–70.
Lemos MC, Argawal A 2006. Environmental governance. Annual Review of Environment and Resources 31:297–325.
Linklater W, Steer J 2018. Predator Free 2050: a flawed conservation policy displaces higher priorities and better, evidence-based alternatives. Conservation Letters 11. e12593.
Livingstone PG, Hancox N, Nugent G, de Lisle GW 2015. Toward eradication: the effect of *Mycobacterium bovis* infection in wildlife on the evolution and future direction of bovine tuberculosis management in New Zealand. New Zealand Veterinary Journal 63: sup1, 4–18.
Macdonald N, Nugent G, Edge K-A, Parkes JP 2019. Eradication of red deer from Secretary Island, New Zealand: changing tactics to achieve success. In: Veitch CR, Clout MN, Martin AR, Russell JC, West CJ eds. Island invasives: scaling up to meet the challenge. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN. Pp. 256–260.
McGlone MS, McNutt K, Richardson SJ, Bellingham PJ, Wright EF 2020. Biodiversity monitoring, ecological integrity, and the design of the New Zealand Biodiversity Assessment Framework. New Zealand Journal of Ecology 44: 3411.
McIlroy JC 2001. Advances in New Zealand mammalogy 1990–2000: Feral pig. Journal of the Royal Society of New Zealand 31: 225–231.
McKelvey PJ 1963. The synecology of the West Taupo indigenous forest. New Zealand Forest Service Bulletin 14. Wellington, New Zealand Forest Service. 127 p.
Mark AF, Baylis GTS 1975. Impact of deer on Secretary Island, Fiordland, New Zealand. Proceedings of the New Zealand Ecological Society 22: 19–24.
Meads MJ 1976. Effects of opossum browsing on northern rata trees in the Orongorongo Valley, Wellington, New Zealand. New Zealand Journal of Zoology 3: 127–139.
Mills JA, Lee WG, Lavers RB 1989. Experimental Investigations of the effects of takahe and deer grazing on *Chionochloa pallens* grassland, Fiordland, New Zealand. Journal of Applied Ecology 26: 397–417.
Ministry for the Environment n.d. https://consult.environment.govt.nz/biodiversity/npsib-exposure-draft/ (Accessed 18 January 2023).
Ministry of Business, Innovation and Employment n.d. https://www.mbie.govt.nz/science-and-technology/science-and-innovation/research-and-data/te-ara-paerangi-future-pathways/ (Accessed 18 January 2023).
Moilanen A, Wilson KA, Possingham HP 2009. Spatial conservation prioritization: Quantitative methods and computational tools. Oxford, Oxford University Press. 304 p.
Moloney PD, Forsyth DM, Ramsey DSL, Perry M, McKay M, Gormley AM, Kappers B, Wright AF 2021. Occupancy and relative abundances of introduced ungulates on New Zealand's public conservation land 2012–2018. New Zealand Journal of Ecology 45: 3437.
Morriss G, Yockney I, Nugent G 2021. High effectiveness of deer-repellent Prodeer 1080 possum bait in the northern South Island high country. Manaaki Whenua Landcare Research contract report prepared for OSPRI. Wellington, OSPRI. 16 p.
Mulder CPH, Grant-Hoffman MN, Towns DR, Bellingham PJ, Wardle DA, Durrett MS, Fukami T, Bonner KI 2009. Direct and indirect effects of rats: does rat eradication restore ecosystem functioning of New Zealand seabird islands? Biological Invasions 11: 1671–1688.
New Zealand Cabinet 2016. Available at https://fyi.org.nz/request/4307/response/14723/attach/3/Spnog0116090814360.pdf (Accessed 18 January 2023).
Newson SE, Johnston A, Renwick AR, Baillie SR, Fuller RJ 2012. Modelling large-scale relationships between changes in woodland deer and bird populations. Journal of Applied Ecology 49: 278–286.
Next Foundation n.d. https://www.nextfoundation.org.nz/wp-content/uploads/2014/03/Tomorrow_accord.pdf (Accessed 18 January 2023).
Norbury GL, Byrom AE, Pech RP, Smith J, Clarke D, Anderson D, Forrester G 2013. Invasive mammals and habitat modification interact to generate unforeseen outcomes for indigenous fauna. Ecological Applications 23: 1707–1721.
Norbury GL, Pech RP, Byrom AE, Innes J 2015. Density-impact functions for terrestrial vertebrate pests and indigenous biota: Guidelines for conservation managers. Biological Conservation 191: 409–420.
Norton DA, Overmars FB 2012. Ecological areas – premier protected natural areas. New Zealand Journal of Ecology 36: 108–120.
Nugent G, Fraser KW 1993. Pests or valued resources? Conflicts in management of deer. New Zealand Journal of Zoology 20: 361–366.
Nugent G, Speedy C 2022. The future of sika deer management in New Zealand: Invasive deer or hunting resource? In: Kaji K, Uno H, Iijima H eds. Sika deer: Life history plasticity and management. Pp 537–549.
Nugent G, Fraser W, Sweetapple P 2001. Top down or bottom up? Comparing the impacts of introduced arboreal possums and 'terrestrial' ruminants on native forests in New Zealand. Biological Conservation 99: 65–79.
Nugent G, Gormley AM, Anderson DP, Crews K 2018. Roll-Back eradication of bovine tuberculosis (TB) from wildlife in New Zealand: concepts, evolving approaches, and progress. Frontiers in Veterinary Science 12: 277.
O'Donnell CFJ, Weston KA, Monks JM 2017. Impacts of introduced mammalian predators on New Zealand's alpine fauna. New Zealand Journal of Ecology 41: 1–22.
Ogden J, Stewart GF, Allen RB. 1996. Ecology of New Zealand *Nothofagus* forests. In Veblen TT, Hill RS, Read J eds.

The ecology and biogeography of *Nothofagus* forests. New Haven, Yale University Press. Pp. 25–82.
Palmer G, Stephens P, Ward A 2015 Nationwide trophic cascades: changes in avian community structure driven by ungulates. Scientific Reports 5: 15601.
Parkes JP 1993a. Feral goats: designing solutions for designer pests. New Zealand Journal of Ecology 17: 71–83.
Parkes JP 1993b. The ecological dynamics of pest-resource-people systems. New Zealand Journal of Zoology 20: 223–230.
Parkes JP 2001. Advances in New Zealand mammalogy 1990-2000: Feral livestock. Journal of The Royal Society of New Zealand 31: 233–241.
Parkes JP, Murphy E 2003. Management of introduced mammals in New Zealand. New Zealand Journal of Zoology 30: 335–359.
Parkes JP, Nugent G, Forsyth DM, Byrom AE, Pech RP, Warburton B, Choquenot D 2017. Past, present and two potential futures for managing New Zealand's mammalian pests. New Zealand Journal of Ecology 41: 151–161.
Parliamentary Commissioner for the Environment 1994. Possum management in New Zealand. Wellington, Parliamentary Commissioner for the Environment.
Parliamentary Commissioner for the Environment 2000. Possum control. Background Note 2000/2. Wellington, Parliamentary Commissioner for the Environment.
Parliamentary Commissioner for the Environment 2020. https://www.pce.parliament.nz/publications/environmental-research-funding-review (Accessed 18 January 2023).
Pekelharing C 1979. Fluctuation in opossum populations along the north bank of the Taramakau catchment and its effect on the forest canopy. New Zealand Journal of Forestry Science 9: 212–224.
Peltzer DA, Bellingham PJ, Dickie IA, Houliston G, Hulme PE, Lyver PO'B, McGlone MS, Richardson SJ, Wood J 2019. Scale and complexity implications of making New Zealand predator-free by 2050. Journal of the Royal Society of New Zealand 49: 412–439.
Perham AN 1922. Deer in New Zealand. New Zealand Parliament. Appendix to Journal of House of Representatives C–3A. 12 p.
Perry GLW, McGlone MS 2021. Networks and themes in the publications of the New Zealand Ecological Society over the last six decades. New Zealand Journal of Ecology 45: 3438.
PF2050 undated. https://pf2050.co.nz/products-to-projects/ (Accessed 18 January 2023).
Reardon JT, Whitemore N, Holmes KM, Judd LM, Hutcheon AD, Norbury G, Mackenzie DI 2012. Predator control allows critically endangered lizards to recover on mainland New Zealand. New Zealand Journal of Ecology 36: 141–150.
Robertson HA, Guillotel J, Lawson T, Sutton N 2019. Landscape-scale applications of 1080 pesticide benefit North Island brown kiwi (*Apteryx mantelli*) and New Zealand fantail (*Rhipidura fuliginosa*) in Tongariro Forest, New Zealand. Notornis 66: 1–15.
Rose A, Platt KH 1987. Recovery of Northern Fiordland alpine grassland after reduction in the deer population. New Zealand Journal of Ecology 10: 23–33.
Ruscoe WA, Elkington JS, Choquenot D, Allen R. 2005. Predation of beech seed by mice: effects of numerical and functional responses. Journal of Animal Ecology 74: 1005–1019.
Rouco C, Torre R, Martin-Collado D, Byrom A 2017. New Zealand shouldn't ignore feral cats. BioScience 67(8): 68.
SanctuariesNZ n.d. https://www.sanctuariesnz.org/ (Accessed 18 January 2023).
Saunders A, Norton DA 2001. Ecological restoration at mainland islands in New Zealand. Biological Conservation 99: 109–119.
Schleicher J, Peres CA, Amano T et al. 2017. Conservation performance of different conservation governance regimes in the Peruvian Amazon. Scientific Reports 7: 11318.
Scott T 2015. Does collaboration make any difference? Linking collaborative governance to environmental outcomes. Journal of Policy Analysis and Management 34: 537–566.
Selwood K, Wintle B, Kujala H 2019. Collaborative conservation planning: Quantifying the contribution of expert engagement to identify spatial conservation priorities. Conservation Letters. e12673.
Sessions LA, Rance C, Grant A, Kelly D 2001. Possum (*Trichosurus vulpecula*) control benefits native beech mistletoes (Loranthaceae). New Zealand Journal of Ecology 25: 27–33.
Shanahan D, Maseyk F, Johnston G, Doole M 2021. Social and ecological outcomes from community-led conservation. Collaborative research project prepared for Predator Free 2050 Ltd. The Catalyst Group Contract Report No. 2021/160. Wellington, The Catalyst Group. 64 p.
Sika Foundation n.d. https://sikafoundation.co.nz/ (Accessed 18 January 2023).
Stewart GH, Wardle J, Burrows LE 1987. Forest understorey changes after reduction in deer numbers, Northern Fiordland, New Zealand. New Zealand Journal of Ecology 10: 35–42.
Stuff 2011. https://www.stuff.co.nz/environment/6112401/Pest-free-sanctuaries-could-avert-disaster (Accessed 18 January 2023).
Stuff 2018. https://www.stuff.co.nz/northland/108762762/doc-staff-increasingly-threatened-and-abused-over-1080 (Accessed 18 January 2023).
Stuff 2019. https://www.stuff.co.nz/business/farming/110689471/wild-venison-helicopter-hunting-supplies-upmarket-european-and-us-restaurants (Accessed 18 January 2023).
Stuff 2021. https://www.stuff.co.nz/environment/climate-news/126790256/forestmunching-pests-colonised-a-third-more-conservation-land-in-the-past-eight-years (Accessed 18 January 2023).
Stuff 2023. https://www.newsroom.co.nz/deer-devils-of-the-deep-south (Accessed 18 January 2023).
Tanentzap AJ, Burrows LE, Lee WG, Nugent G, Maxwell JM, Coomes DA 2009. Landscape level vegetation recovery from herbivory: progress after four decades of invasive red deer control. Journal of Applied Ecology 46: 1064–1072.
Thomson GM 1922. The naturalisation of animals and plants in New Zealand. Cambridge, Cambridge University Press. 607 p.
Thomson C, Challies CN 1988. Diet of feral pigs in the podocarp-tawa forests of the Urewera Ranges. New Zealand Journal of Ecology 11: 73–78.
Tompkins DM, Byrom AE, Pech RP 2013. Predicted responses of invasive mammal communities to climate-related changes in mast frequency in forest ecosystems. Ecological Applications 23: 1075–1085.
Towns, D.R. 2008. Eradications as reverse invasions: lessons

from Pacific rat (*Rattus exulans*) removals on New Zealand islands. Biological Invasions 11: 1719–1733.
Towns DR, Bellingham PJ, Mulder CPH, Lyver PO'B 2012. A research strategy for biodiversity conservation on New Zealand's offshore islands. New Zealand Journal of Ecology 36: 1–20.
Towns DR, Daugherty CH, Broome K, Timmins S, Clout M 2019. The thirty-year conservation revolution in New Zealand: an introduction. Journal of the Royal Society of New Zealand 49: 243–258.
van Dam-Bates P, Gansell O, Robertson B 2018. Using balanced acceptance sampling as a master sample for environmental surveys. Methods in Ecology & Evolution 9: 1718–1726.
Veblen TT, Stewart GH 1980. Comparison of forest structure and regeneration on Bench and Stewart Islands, New Zealand. New Zealand Journal of Ecology 3: 50–68.
Veblen TT, Stewart GH 1982. The effects of introduced wild animals on New Zealand forests. Annals of the Association of American Geographers 72: 372–397.
Warburton B, Livingstone P 2015. Managing and eradicating wildlife tuberculosis in New Zealand. New Zealand Veterinary Journal 63(sup1): 77–88.
Wardle DA, Barker GM, Yeates GW, Bonner KI, Ghani A 2001. Introduced browsing mammals in New Zealand natural forests: aboveground and belowground consequences and ecosystem properties. Ecological Monographs 71: 587–614.
Wardle JA 1984. The New Zealand beeches: Ecology, utilization and management. Wellington, New Zealand Forest Service. 447 p.
Wardle P 1961. The effect of deer on subalpine forest and scrub. Proceedings of the New Zealand Ecological Society 8: 52–54.
Whitehead A, Kujala H, Ives C, Gordon A, Lentini P, Nicholson E, Wintle B, Raymond C 2014. Integrating biological and social values when prioritizing places for biodiversity conservation. Conservation Biology 28: 992–1003.
Willis G 2017. Addressing New Zealand's Biodiversity Challenge. A Regional Council thinkpiece on the future of biodiversity management in New Zealand. Pukekohe, Enfocus. 88 p.
Wilson D, Ruscoe W, Burrows L, McElrea L 2006. An experimental study of the impacts of understorey forest vegetation and herbivory by red deer and rodents on seedling establishment and species composition in Waitutu Forest, New Zealand. New Zealand Journal of Ecology 30: 191–207.
Wintle BA, Cadenhead NCR, Morgain RA, Legge SM, Bekessy SA, Cantele M, Possingham HP, Watson JEM, Maron M, Keith DA, Garnett ST, Woinarski JCZ, Lindenmayer DB 2019. Spending to save: What will it cost to halt Australia's extinction crisis? Conservation Letters 12: e12682.
Yockney IJ, Hickling GJ 2000. Distribution and diet of chamois (*Rupicapra rupicapra*) in Westland forests, South Island, New Zealand. New Zealand Journal of Ecology 24: 31–38.

Received: 27 October 2022; accepted: 24 January 2023
Handling editor: Matt McGlone