Effect of one-hit control on the density of possums (*Trichosurus vulpecula*) and their impacts on native forest

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Cover: Kāmahi at Tahunamaere non-treatment site dying as a result of possum browse, 1996. Over the 8 years of the study, 39% of the kāmahi at Tahunamaere died, compared with an overall 8% (range 3–14%) in the seven Matemateaonga sites where possums were aerially poisoned with 1080 in 1996.
Photo: Graham Nugent, Landcare Research.

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ABSTRACT

Introduced possums (*Trichosurus vulpecula*) have unwanted impacts on New Zealand's indigenous forest, but how forests respond to possum control is not well understood. Here, we document how possum populations, and the tree species they feed on, responded over 6–8 years following aerial 1080 poisoning. In each of three areas, possum density and tree condition were monitored at 2–7 poisoned sites and 1–3 unpoisoned sites (each 150–250 ha in area). Trap Catch Indices (TCIs) of possum density were much reduced after poisoning, but usually recovered quickly to near pre-control levels within 6 years. The overall mortality of possum-preferred tree species was about 25% lower at poisoned sites than at unpoisoned sites, but this varied between species. Browse levels initially fell sharply at most sites, but with greater declines at the poisoned sites. Browse pressure then increased again as possum numbers increased to previous levels. Canopy condition (as indexed by % Foliar Cover; FCI) increased at both poisoned and unpoisoned sites, but more so at the poisoned sites. Foliar cover of some the most common and widespread species such as kāmahi (*Weinmannia racemosa*), māhoe (*Melicytus ramiflorus*) and tawa (*Beilschmedia tawa*) continued to increase (i.e. recover from presumed previous defoliation) even when the possum numbers had substantially recovered. Although the interactions between possums and their food supply are complex, possum control does reduce possum browse, and therefore tree defoliation and, ultimately, tree mortality. Importantly, we infer that reducing possum density by 60% will usually be sufficient to protect most of the possum-preferred tree species we studied.

Keywords: possums, *Trichosurus vulpecula*, foliar cover, browse, rate of increase, tree mortality rates, New Zealand

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1. Introduction

The importance of introduced brushtail possums (*Trichosurus vulpecula*) in driving major changes in forest composition in New Zealand was questioned well into the latter half of the 20th century (e.g. Veblen & Stewart 1982). Uncertainty persisted because possum impacts are extremely variable, depending on a range of predisposing factors (Payton 2000). However, the weight of evidence from observational studies now leaves little doubt that possums do contribute to the sometimes catastrophic dieback of vulnerable forest types and can change forest composition through much increased mortality of vulnerable species such as northern rātā (*Metrosideros robusta*; Meads 1976) and Hall's tōtara (*Podocarpus halli*; Bellingham et al. 1999a).

Since about 1960, the view that possums were affecting native forest structure and composition in unacceptable ways prompted large-scale control of possum populations in many areas to either prevent damage or allow the forest to 'recover' (Payton 2000). Initially, the perceived obviousness and urgency of the problem provided sufficient justification for the control. Doing something was seen as better than doing nothing, and it was presumed that reducing possum numbers would reduce browse pressure and at least prevent further damage. However, it is increasingly clear that simply removing or reducing a herbivore may not result in a rapid return to the previous ecosystem state, particularly where the herbivore has been long established (Schmitz & Sinclair 1997; Coomes et al. 2003). For example, species not eaten by the herbivore often increase in abundance (Bellingham et al. 1999b) and can prevent rapid re-establishment of eaten-out species, or any 'recovery' may take decades or centuries or occur only after some major ecological perturbation (Tanentzap et al. 2009).

Thus, although large-scale possum control has been implemented to try to limit or reverse the impacts of possums for four decades in New Zealand, the complexity in the outcomes of possum invasion has made it difficult to be certain that possum control will actually produce the desired mitigation and/or restoration responses. Much early possum control was implemented as a one-off or short-term attempt to avert anticipated catastrophic collapse of, for example, rātā (*Metrosideros* spp.) - kāmahi (*Weinmannia recemosa*) canopies. It is only in the last 10–15 years that most of the possum control aimed at protecting native vegetation has been conducted on an in-perpetuity basis. This is far too short a period to meaningfully measure medium- or long-term changes in forest canopy composition, and only one pair of related studies by Bellingham et al. (1999a, b) has attempted this using a large number of areas. In these studies, they did not detect consistent declines over time in any possum-preferred species other than a major decline in Hall's tōtara, which is an important food of possums (Nugent et al. 2000), and they detected no major differences in composition between areas that had or had not previously received some form of possum control. However, interpretation of these results is difficult, as neither the effectiveness of the control nor its extent were factored into the analysis, and the areas subject to possum control were not selected randomly (they were presumably targeted because the apparent impacts of possums were already severe).

To date, there are only six published studies that document short-term changes in plant biomass after possum control. Smale et al. (1995) did not detect any increase in canopy cover 5 years after possum control in a West Coast rātā-kāmahi forest. Likewise, Payton et al. (1997) did not detect any increase in foliar cover when possum control was implemented in kauri (*Agathis australis*) forest near Waipoua, Northland, not long after possums had colonised the area—although they did record a decrease in a nearby area where possums were not controlled, from which they inferred that the control had been effective in preventing a similar decline at Waipoua. In both these studies, however, the possum control was implemented to prevent canopy dieback and the trees were still in good health, so there was little scope for recovery.

The four remaining published investigations of short-term vegetation change after possum control all focussed mainly on just one or two highly impacted species and all showed substantial rapid increases in foliar biomass after possum control for those species. Nugent et al. (2002) demonstrated recovery of kohekohe (*Dysoxylum spectabile*) from near universal defoliation to almost full foliage cover within 2 years of possum control. Likewise, Sweetapple et al. (2002) showed the rapid accumulation of mistletoe stem-and-leaf biomass for 3–4 years following aerial poisoning of possums, whereas almost none had accumulated in the years before control (even though the size of the parasitic galls from which the stems emerged indicated that the plants were many years old). Rogers (1997) reported an increase over 2 years in the foliar biomass of Hall's tōtara in an area subject to possum control, but no change in a nearby area where possums were not controlled; for the other conifer monitored, pāhautea (*Libocedrus bidwillii*), there was a smaller increase for the banded trees and those in the area with possum control, but a decline in the area where possums were not controlled. Finally, Urlich & Brady (2005) recorded reduced mortality, increased basal area and higher foliar cover of tree fuchsia over 4–8 years after the initiation of possum control in five areas compared with one area with no possum control.

Pulsed intermittent or periodic control is frequently used to manage possum numbers and impacts in large, remote and rugged areas of native forest (Parkes et al. 2006). This strategy aims to minimise the adverse impacts of possums by reducing their numbers to well below some threshold density at which they are considered to cause unacceptable changes in forest structure and composition, and then repeating the control some years later once possum numbers have increased to close to or above that threshold. The gap of several years between successive control operations distinguishes this strategy from the so-called press or maintenance control strategies that are used in more accessible country where repeat control can be more readily undertaken each year (Parkes et al. 2006).

The period between successive controls has typically been of the order of 5–10 years (Parkes et al. 2006). However, that periodicity has not yet been related in any meaningful way to evidence-based threshold densities of possums at which their impacts are minimal or at least tolerable. As a consequence, there is little evidence to judge whether operations are being repeated too frequently or not frequently enough. This project therefore aimed to characterise those threshold densities and the optimal periodicity of control for the protection of a variety of tree species that are commonly affected by possums. The focus was on the response in canopy cover of existing forest trees, rather than on regeneration responses or native animal abundance.

In 1996, a medium-term study was commenced to determine the rates at which possum populations recovered after one-hit aerial poisoning and the threshold levels at which they again began to have an adverse impact on native forest canopies. The study was replicated in three areas, within each of which possums were controlled at 2–7 ‘treatment’ sites and not controlled at 1–2 ‘non treatment’ experimental control sites. An important design limitation was that the areas designated for treatment were selected by Department of Conservation (DOC) conservancies for conservation purposes, so the experimental treatment (possum control) was not randomly allocated to sites. To minimise the possibility of some selection bias, we endeavoured to select treatment and non-treatment areas that appeared to be similar with respect to the likely nature of the possum impact–density relationships for the indicator species monitored, and we have made extensive use of *in situ* controls in the form of unpalatable species and unbrowsed individuals within species. We report outcomes over the 6–8-year period following possum control.

2. Objective

The aim of this study was to gain a better understanding of how possum populations and forest canopy species responded to a single possum control operation, so that the level of control needed to maintain the desired level of protection of those tree species could be assessed.

3. Methods

3.1 STUDY DESIGN

In autumn 1996, we initiated a monitoring programme in the Matemateaonga Range, Wanganui Conservancy, and in the following year established similar programmes in the Richmond Range, Nelson/Marlborough Conservancy, and the Ikawhenua Range, in the former East Coast/Hawke's Bay Conservancy (Fig. 1). The full monitoring programme consisted of an initial survey of possum abundance and canopy condition prior to possum control, followed by three or four resurveys at biennial intervals, all in autumn. Within each of the three areas, we monitored 2–7 treatment sites and 1–2 non-treatment sites. The monitoring of multiple treatment sites within each area (rather than having each site as a fully independent replicate) reflected the desire to explore both between- and within-area variation.

At the treatment sites, possums were controlled by one-hit aerial 1080 poisoning soon after the first survey; there was no further control during the study, and none of the areas had previously been subject to official control. At non-treatment sites, possums were not intentionally controlled. Trends in indices of possum density and their impacts on vegetation in these blocks were assessed using modified variants of the Trap Catch Index (TCI; Warburton 1996) and Foliar Browse Index (FBI; Payton et al. 1999) methods, respectively.

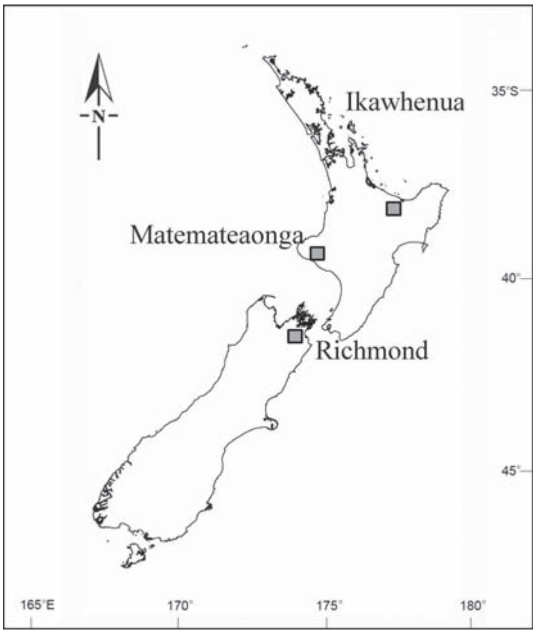
Each study site consisted of a subcatchment of 150–250 ha. These subcatchments were sufficiently separate from each other for the resident possum populations within them to be independent of each other (i.e. the areas were too far apart for there to be any likelihood of overlap in the home ranges of resident possums, which are usually only a few hectares in size; Cowan & Clout 2000). Subcatchments were subjectively selected to represent the full range of variation in forest composition, susceptibility to possum damage, and the current extent of canopy dieback within the total area to be poisoned. The start points for five 400-m-long transects were established randomly along accessible ridgelines or creek beds (depending on area topography) and the transects were directed randomly within the physically accessible areas (i.e. avoiding bluffs). Possum abundance and tree condition were measured biennially along each transect. Forest composition was also assessed along these transects.

The overall study design therefore consisted of three true replicates (areas), within which were nested 2–7 poisoned and 1–3 unpoisoned sites, which also provided information on the variation in impact within areas. The three primary study areas are described below.

The scientific and common names of the most common plant species in each area are listed in Appendix 1, along with a measure of relative abundance (see section 3.4.1)

3.1.1 Matemateaonga Range

As part of a large, multi-stage programme, Wanganui Conservancy, DOC, aerially poisoned possums on the western end of the Matemateaonga Range (12 000 ha) in winter 1996 (Fig. 1A). Seven subcatchments were monitored in the poisoned



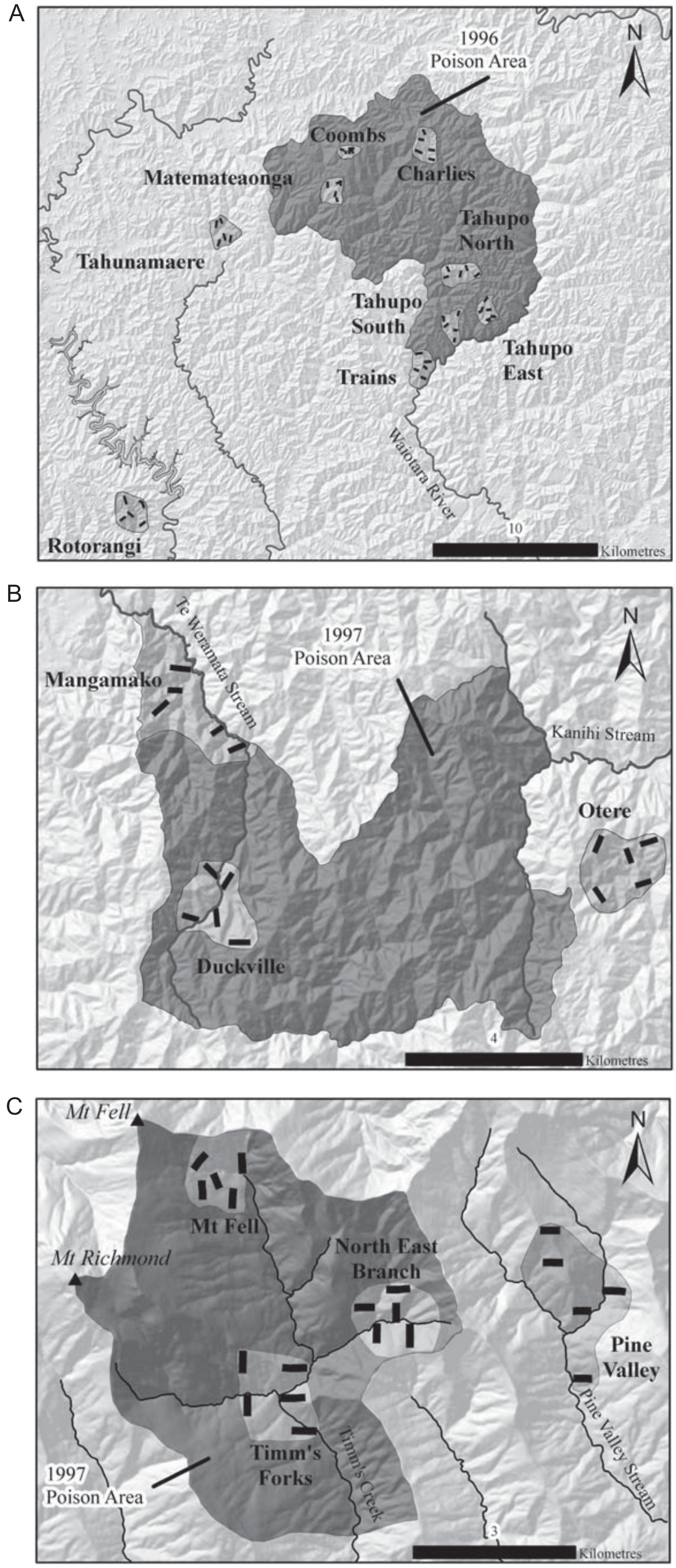


Figure 1. Location of study sites (medium grey) within study areas (light grey), showing locations of traplines and vegetation transects, and the boundaries of the various poisoning operations (dark grey) for A. Matemateaonga Range, B. Ikawhenua Range and C. Richmond Range.

areas, with sites (from north to south) assigned the following names: Charlies (CHA), Coombs (CMB), Matemateaonga Trig (MMO), Tahupo North (THN), Tahupo East (THE), Tahupo South (THS) and Trains (TRN). Two separate unpoisoned subcatchments to the south (Rotorangi; RTR) and southwest (Tahunamaere; THM) were monitored as experimental controls, and a limited amount of data were also gathered in the latter part of the study from a third unpoisoned area, Waitotara (WAI; not shown in Fig. 1A), immediately adjacent to the southern end of the poisoned area.

The Matemateaonga Range comprises soft mudstone (papa) hill country characterised by narrow gorges and ridges, with some flatter ridges to the north. The forest is predominantly mixed broadleaved-conifer forest with tawa (*Beilschmiedia tawa*) or kāmahi forming the main canopy. Kāmahi and northern rātā dominate the ridges, particularly towards the north. Black beech (*Nothofagus solandri* var. *solandri*) occurs on sandstone ridges to the south (Nicholls 1956; Wardle 1991). In the mid-1990s there was a north–south gradient in forest health, with major patches of canopy collapse (presumed by local managers to be possum-induced) in kāmahi-dominated stands in the northern part of the treatment area, but a largely intact canopy in the south. The possum control operation was intended to halt further canopy collapse. Possums were liberated at Wanganui as early as 1868 (Pracy 1974), but it is not known exactly when they first reached the Matemateaonga area. Goats (*Capra hircus*) and feral pigs (*Sus scrofa*) are common throughout the area, and the goats have severely depleted the understorey (Blaschke 1988). Red deer (*Cervus elaphus scoticus*) occur in small numbers at the northern end of the study area.

3.1.2 Ikawhenua Range

The former East Coast/Hawke's Bay Conservancy completed a 6000-ha poison operation in the middle of the Ikawhenua Range, Urewera National Park, in October 1997 (Fig. 1B). The two poisoned sites in this area were Mangamako (MKO) and Duckville (DVL), while the unpoisoned site, Otere (OTR), was located immediately east of the poisoned area.

The Ikawhenua Range is moderately steep hill country, with a largely intact mixed conifer–hardwood–beech forest. Tawa and kāmahi dominate much of the canopy, with rewarewa (*Knightia excelsa*) and hīnau (*Elaeocarpus dentatus*) often also present. Northern rātā and rimu (*Dacrydium cupressinum*) are the most common emergents. Red beech (*Nothofagus fusca*) occurs as a canopy dominant on steeper spurs and at higher altitudes (Wardle 1984). Possums were first liberated at Lake Waikaremoana in 1898, and some subsequent liberations were made nearby (Pracy 1974). Red and rusa (*Cervus timorensis*) deer and pigs are present.

3.1.3 Richmond Range

The Nelson/Marlborough Conservancy completed a 2500-ha poison operation in the Timms Creek catchment in Richmond Forest Park in April 1997 (Fig. 1C). This area was selected for control as some possum-preferred species that were widespread as minor elements of this beech-dominated forest were showing evidence of possum impact and localised dieback. The three poisoned sites in this area were designated as Timms Forks (FRK), Mt Fell (MFL) and North East Branch (NEB). Pine Valley (PVA), the catchment immediately northeast of Timms Creek, was chosen as the unpoisoned site.

The Richmond Range has a complex geology, with the study area comprising Marlborough schist. Beech dominates the canopy, with hard (*Nothofagus truncata*) and red beech at lower altitudes intergrading into silver (*N. menziesii*) and finally mountain (*N. cliffortioides*) beech at the treeline (Wardle 1984). Possum-palatable species, including kāmahi, Hall's tōtara, wineberry (*Aristotelia serrata*), haumakaroa (*Raukawa simplex*), māhoe (*Melicytus ramifloris*), lancewood (*Pseudopanax crassifolius*) and red flowering mistletoe (*Peraxilla tetrapetala*), occur occasionally in the canopy and/or commonly in the understorey. Southern rātā (*Metrosideros umbellata*) is present in small patches on steep rocky sites. Possums were not liberated in the Marlborough region until the late 1920s (Pracy 1974). Red deer occur throughout the Richmond Range, and pigs and goats are also present in low numbers.

3.2 TRAP CATCH INDEX OF POSSUM ABUNDANCE

Along each of the five transects within each site, 20 Victor No. 1 traps were set 20 m apart for three fine nights where possible. Traps were set on raised platforms or boards 0.5–0.7 m above ground level to avoid catching ground birds. During all surveys except the first Matemateaonga survey (when heavy leaning-board raised sets were used), the raised sets used were 'Scott Boards', which comprised squares of plywood (15 × 18 cm) with routered holes designed to allow the trap to sit securely on top of the board when it was wedged between three nails hammered into a tree at the desired height. To minimise accidental capture of other small mammals or birds, traps were set so that reasonably heavy pressure was required to trip them.

For the Ikawhenua Range, conservancy staff required an estimate of trap-catch rates for ground-based traps so that the data from our study could be compared with TCI data from other nearby areas in which similar operations were being undertaken. To accommodate this management requirement, a comparison of catch rates using ground- and raised-set (Scott Board) traps was made by alternating five ground and five raised sets along each transect.

In each of the five assessments of trap catch in the Ikawhenua Range, fewer possums were caught overall on the raised sets than on ground sets (Fig. 2). There was considerable variability in the slope of the trend line, mainly reflecting very low catch rates for raised sets in late 1997, immediately after control (Fig. 2B): only five (21%) of the 24 possums captured in that survey were caught on raised sets. For the other four surveys, raised-set TCIs were 54–78% of ground-set TCIs, with an average of 62% for all five surveys combined (Fig. 2F). The slope of the regression forced through the origin was significantly different from 1, showing that catch rates were not equal ($t = 9.63$, df = 69, $P < 0.001$). Therefore, the equation $TCI_{raised} = 0.62 \times TCI_{ground}$ was subsequently used to adjust the trap-catch estimates for the ten ground sets on each line, so that these data could be combined with those from the raised sets. The adjusted Ikawhenua trap-catch data are treated as being directly comparable with the raised-set TCIs of the other two areas.

In the Matemateaonga Range area, surveys were initiated in 1996, with pre-control trapping in February–May, the poison operation in June, and the immediate post-control trapping in July. In the Richmond Range area, surveys were initiated

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