

Vegetation monitoring, Tararua Forest Park, New Zealand, 1958-85

DOC RESEARCH & DEVELOPMENT SERIES 212

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Published by
Science & Technical Publishing
Department of Conservation
PO Box 10-420
Wellington, New Zealand

DOC Research & Development Series is a published record of scientific research carried out, or advice given, by Department of Conservation staff or external contractors funded by DOC. It comprises reports and short communications that are peer-reviewed.

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Hardcopy is printed, bound, and distributed at regular intervals. Titles are also listed in our catalogue on the website, refer <http://www.doc.govt.nz> under Publications, then Science and research.

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ISSN 1176-8886

ISBN 0-478-14008-8

This is a client report commissioned by Wellington Conservancy and funded from the Science Advice Fund. It was prepared for publication by Science & Technical Publishing Section; editing and layout by Geoff Gregory. Publication was approved by the Chief Scientist (Research, Development & Improvement Division), Department of Conservation, Wellington, New Zealand.

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CONTENTS

Abstract	5
<hr/>	
1. Introduction	6
<hr/>	
1.1 Objectives	7
1.2 Background	7
1.3 Description of Tararua Forest Park	9
2. Methods	11
<hr/>	
2.1 Cruciform plots	11
2.2 Alpine vegetation transects	12
2.3 Permanent quadrat plots	12
2.4 Plots used in analysis	13
2.5 Ruamahanga and Hutt exclosure plots	13
2.6 Data analysis	13
3. Results	15
<hr/>	
3.1 Characterisation of plot data	15
3.2 Forest structure and changes in species composition	17
3.3 Ruamahanga exclosure plot	19
3.4 Hutt Valley exclosure plots	20
3.5 Mortality and recruitment of <i>Fuchsia excorticata</i> and <i>Weinmannia racemosa</i>	21
3.6 Changes in grassland species composition	21
3.7 Permanent plot power analysis	22
4. Discussion	23
<hr/>	
4.1 Vegetation classification by species composition	23
4.2 Plant community palatability change	24
4.3 Communities most susceptible to browsing	26
4.4 Managing susceptible plant communities	28
4.5 Monitoring outcomes of ungulate management	31
5. Conclusions and recommendations	34
<hr/>	
6. Acknowledgements	36
<hr/>	
7. References	37
<hr/>	
Appendix 1	
<hr/>	
Codes and full names of woody species included in CCA ordination	42

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ABSTRACT

This study uses data from a network of permanent forest, grassland, and alpine scrub monitoring plots that were measured throughout Tararua Forest Park, to the northwest of Wellington, New Zealand, between 1958 and 1985. Objectives were to classify Tararua Forest Park plant communities according to environmental characteristics, then determine trends in the condition of these communities between 1958 and 1985, and review their susceptibility to browsing by red deer, sika deer, and goats. Finally, the study aimed to predict likely biological outcomes of management choices, and suggest appropriate management responses linked to monitoring strategies. Species composition of the plant communities assessed was largely explained by altitude. Four predominant community types were identified based on dominant plant species. Little change in the composition of these four groups between 1958 and 1985 was evident. Communities dominated by kamahi (*Weinmannia racemosa*) and hardwood (*Beilschmiedia tawa*-*Hedycarya arborea*-*Pseudowintera* spp.) are likely to be most vulnerable to ungulate browsing because of the greater occurrence of palatable tree species in these two groups. Communities dominated by silver beech (*Nothofagus menziesii*) and grassland (*Chionochloa* spp.) are likely to be less vulnerable to deer browsing. *Weinmannia racemosa* was unlikely to have been successfully regenerating in 1984, probably because of deer browsing on seedlings. This was despite intensive commercial, government-funded and private deer culling that have lowered deer densities over the preceding three decades. There was also evidence that deer suppressed the regeneration of highly palatable subcanopy hardwood tree species.

Keywords: ungulate browsing, plant palatability, plant succession, permanent plots, kamahi, *Weinmannia racemosa*, silver beech, *Nothofagus menziesii*, hardwood forest, alpine grassland, Tararua Forest Park, New Zealand.

© July 2005, New Zealand Department of Conservation. This paper may be cited as:
Husheer, S.W. 2005: Vegetation monitoring, Tararua Forest Park, New Zealand, 1958–85. *DOC Research & Development Series 212*. Department of Conservation, Wellington. 42 p.

1. Introduction

It is increasingly apparent that, by preventing the growth of tree and shrub seedlings, introduced goats *Capra hircus* and red deer *Cervus elaphus* can alter successional and ecosystem processes in unpredictable and undesirable ways (Wardle 2002). With further research, these influences may prove to be of greater consequence on forest succession than the more intensively managed brushtail possum *Trichosurus vulpecula*, because deer are able to prevent regeneration of whole cohorts of palatable seedlings and saplings whereas possums tend to sustainably harvest foliage of mature trees such as kamahi (*Weinmannia racemosa*) (e.g. Nugent et al. 1997, 2001). Better knowledge of the influence of mammalian herbivory on forest overstories is a critical gap in the understanding of succession in New Zealand forests, which may be why attempts at development of simple rules on the effects of herbivores across landscapes have been largely unsuccessful (Coomes et al. 2003). When several competing mammalian herbivore species are present (such as goats, red deer, possums, and sika deer *Cervus nippon* in Tararua Forest Park), plant-herbivore-herbivore relationships become particularly complex, and multi-species management becomes increasingly important (Forsyth et al. 2000). Therefore, it is also important that any management undertaken is underpinned by robust monitoring systems designed to test how effective management actions such as culling are at meeting conservation outcomes. These concepts are highlighted in the vision of the Habitat Monitoring Strategy (Urlich & Brady 2003) of the Wellington Conservancy of the Department of Conservation (DOC).

New Zealand has a system of permanent 20 m × 20 m forest monitoring plots, that is probably unequalled anywhere in the world. Many of these permanent plots were established by the New Zealand Forest Service (NZFS) in the 1970s and 1980s (Allen 1993). In Tararua Forest Park, the NZFS established a set of cruciform forest and grassland plots through the 1950s and 1960s, followed by 233 quadrat (20 m × 20 m) plots in 1974/75, which were re-measured in 1983/84. This study uses data from measurements of these plots to firstly determine broad changes in Tararua Forest composition between 1958 and 1985. These plots effectively monitor vegetation change during a period of intensive deer and goat culling, undertaken prior to the latest measurement in 1985. Secondly, this report makes recommendations for the establishment of an outcome monitoring system that will show the effects of future ungulate control. The Wellington Conservancy intends to develop an ungulate outcome monitoring project in support of animal control in Tararua Forest Park, according to the Conservancy's monitoring strategy (Urlich & Brady 2003), which was a precursor of this report. That strategy recognised that better information on ecosystem trends was required to more effectively 'maintain, restore and enhance biodiversity'. Of particular concern to Wellington conservation managers is the effect of sika deer on forest regeneration following illegal introductions into the Tararua Range, initially in 1982 (Fraser et al. 2000). Any future monitoring strategy developed to determine the impacts of deer and goats needs to do so in a context of the distribution, abundance and impacts of sika deer, so that proactive management can be implemented if required.

1.1 OBJECTIVES

- Classify and ordinate plant communities with environmental variables, and identify the vegetation sample plot locations within these, using GIS.
- Determine the condition, and trends in the condition, of different plant communities (with respect to palatability) from data collected in the 1984/85 survey, compared with the 1974/75 survey data, and with reference to the findings of the 1958/59 baseline survey.
- Review the susceptibility of different plant community types (including sub-alpine scrublands and alpine grasslands) to impacts from red deer, sika deer, and goats.
- Predict the likely impacts on the different plant community types caused by ungulates under a range of different management regimes, and suggest appropriate management responses.
- Identify the limitations of the historic permanent plot design and evaluate its power to detect trends in the condition of different plant communities from pest impacts, and from management of these pests, and identify monitoring strategies to detect changes in condition of these plant communities.

1.2 BACKGROUND

The negative effects of ungulates on forest regeneration in New Zealand have been apparent for over a century. Soon after the liberation of red deer into New Zealand in 1861 (Logan & Harris 1967) the potential for modification of forest composition was realised (Walsh 1892; Cockayne 1926; Holloway 1950; Riney et al. 1959; Caughley 1983). Cockayne (1908) reported that species that we now know are potentially susceptible to deer browsing, such as *Coprosma tenuifolia* and silver beech (*Nothofagus menziesii*, Table 1),¹ had abundant seedlings and saplings at sites where collapse or dieback had occurred; however, his observations may have been at a stage well before deer populations reached high abundance. Nevertheless, even at this early stage it was realised that introduced ungulates provided novel browsing pressures that New Zealand plants were not adapted to. New Zealand's flora evolved in the absence of mammalian browsers and were likely to have developed defence mechanisms to avoid or tolerate browsing from moas. Moas became extinct through human hunting probably around AD 1400 (Caughley 1989; Holdaway & Jacomb 2000), but their browsing effect on New Zealand plants was in any case probably quite different from that of introduced browsing mammals, particularly ungulates (McGlone & Clarkson 1993), because moa had no teeth and browsed using sight and not smell to identify plants (Atkinson & Greenwood 1989).

Deer in particular, have a complex digestive system capable of coping with plant-defensive strategies such as toxins contained in leaves.

¹ Nomenclature for plants throughout this paper follows Parsons et al. (1995). *Coprosma "taylorae"* is referred to by Eagle (1986) and *Phyllocladus alpinus* by Wilson & Galloway (1993). *Raukawa simplex* is described by Mitchell et al. (1997).

TABLE 1. THE TWENTY-ONE MOST COMMON PLANT SPECIES IN TARARUA FOREST PARK AND RESULTS OF A LITERATURE SEARCH ON THEIR PALATABILITY TO GOATS AND DEER.

These are overstorey species with > 0.5% mean relative abundance in cruciform, forest quadrat, scrub and grassland plots. Based on a consensus of prior results, species were subjectively rated as highly palatable (HP), palatable (P), moderately palatable (MP) and unpalatable (UP).

FULL NAME	COMMON NAME	RATING IN DIFFERENT REFERENCES ^a				CONSENSUS
		HP	P	MP	UP	
<i>Beilschmiedia tawa</i>	tawa			7	4, 16	UP
<i>Carpodetus serratus</i>	putaputaweta	10, 11, 12, 15	2, 6, 14	7, 16		HP
<i>Celmisia spectabilis</i>	cotton daisy			16	13 ^b	UP
<i>Cbionochloa flavescens</i>	broad-leaved snow tussock		13 ^b	16	2	MP
<i>C. pallens</i>	mid-ribbed snow tussock		13 ^b	16	2	MP
<i>Coprosma foetidissima</i>	hupiro		2, 8, 11-13	10, 16		P
<i>Griselinia littoralis</i>	puka	2, 8, 10-13, 15	6, 9, 14, 16			HP
<i>Hedycarya arborea</i>	pigeonwood	10		16		P
<i>Leucopogon fasciculatus</i>	mingimingi				5, 7, 14-16	UP
<i>Melicytus ramiflorus</i>	mahoe	10	2, 3, 7, 8, 13, 14, 16			P
<i>Myrsine divaricata</i>	weeping matipo			8, 10, 14, 16	12	MP
<i>M. salicina</i>	toro			16	13, 14	UP
<i>Notbofagus fusca</i>	red beech			1, 3, 8, 15	16	MP
<i>N. menziesii</i>	silver beech			1, 3, 8, 10, 15	16	MP
<i>Olearia colensoi</i>	tupare				2, 10, 11, 16	UP
<i>Podocarpus hallii</i>	Hall's totara				2, 10, 11, 12, 14, 16	UP
<i>Prumnopitys ferruginea</i>	miro		2	14	10, 11, 12, 16	MP
<i>Pseudowintera axillaris</i>	horopito				2, 13	UP
<i>P. colorata</i>	pepper tree				1, 7-11, 13-16	UP
<i>Raukawa simplex</i>	raukawa		2, 16	11		P
<i>Weinmannia racemosa</i>	kamahi	3, 11, 15	1, 2, 6-8, 14, 16	10, 12		P

^a Sources: 1 McKelvey 1959; 2 Wardle unpubl. 1964; 3 Wallis & James 1972; 4 McKelvey 1973; Knowles & Beveridge 1982; 5 Jane & Pracy 1974; 6 Veblen & Stewart 1982; 7 Allen et al. 1984; 8 Wardle 1984; 9 Stewart & Harrison 1987; 10 Stewart et al. 1987; 11 Nugent & Challies 1988; 12 Stewart & Burrows 1989; 13 Wardle 1991; 14 Nugent et al. 1997; 15 Fraser & Speedy 1997; 16 Forsyth et al. 2003.

^b Williams (1975) and Wardle (1991) showed that *Celmisia spectabilis*, *Cbionochloa flavescens* and *C. pallens* had low nutrient concentrations in foliage, lowering their palatability to ungulates.

During the expansion of red deer, the time between initial colonisation of an area and deer population crash was typically 25–30 years. In this time there was usually a substantial decline in the abundance of deer-palatable plant species such as *Coprosma grandifolia* and *Griselinia littoralis* (Caughley 1983). By the 1940s and 50s deer had increased to high levels throughout New Zealand's forests and grasslands (Challies 1985; Parkes et al. 1997). This included Tararua Forest Park, where similar eruptive peaks probably occurred prior to the introduction of intensive ground-based deer culling in the 1950s (Holloway et al. 1963). Since that time, wild deer populations have decreased by 75–95% in New Zealand due to a combination of widespread government-funded and commercial deer culling and natural population decline (Nugent et al. 1987; Parkes et al. 1997; Nugent et al. 2001). By the 1980s there were probably c. 250 000 wild deer in New Zealand (Nugent & Fraser 1993), or a density

c. 4 deer/km² in forested country. In many respects, Tararua Forest Park does not differ substantially from many other New Zealand mountain forests. It has extensive beech, *W. racemosa*, and podocarp forests, high relief, open alpine grasslands, and numerous river-flats and open slips. These attributes make commercial harvesting of deer highly efficient in Tararua Forest Park, as elsewhere in New Zealand. Therefore, it is probable that the national deer density estimate of 4 deer/km² would not be substantially different for Tararua Forest Park.

Nugent et al. (2001) provide two different scenarios of what might happen to a highly deer-modified forest if deer and goats were completely removed. After browsing pressure is eliminated, a substantial increase in the abundance of palatable plants could be expected. Alternatively, if deer and goats have allowed slow-growing, long-lived, light-demanding plants with good defences to herbivory (e.g. *Blechnum discolor*, *Podocarpus hallii*, and *Pseudowintera colorata*) to become more competitive and more abundant, then cohorts of these plants will be present for a long time even if deer browse is eliminated. Coomes et al. (2003) further develop this idea of irreversibility or retarded recovery by considering the effects of changes in ecosystem processes and a lack of seed source of highly palatable plants. Some changes following shifts towards unpalatable plants alter conditions sufficiently to affect seedling establishment and growth. Conflicting with this argument of irreversibility is evidence from many sites in forests similar to Tararua Forest of rapid responses by highly light- and nutrient-demanding palatable plant species once deer browsing is removed in exclosure plots (Wardle et al. 2001).

1.3 DESCRIPTION OF TARARUA FOREST PARK

Tararua Forest Park is situated to the northwest of Wellington between approximately 41°00'S and 40°30'S along the northwest-aligned Tararua Range, which is composed almost entirely of uplifted and shattered greywacke, a non-fossiliferous sedimentary sandstone. Rates of rainfall and wind are high (Franklin 1967; New Zealand Meteorological Service 1973); rainfall increases and temperature decreases with altitude. Because of this, soils at low altitude are less developed and more fertile than at high altitude, where podsolisation can occur (Holloway et al. 1963). A strong altitudinal gradient of soil development, fertility, moisture deficit, and temperature exists in Tararua Forest Park.

Along with altitude, disturbance regimes are likely to influence the species composition of vegetation in Tararua Forest Park. William Colenso reported that a large earthquake in 1848 caused considerable slipping and forest collapse. In February 1936, high winds destroyed *N. menziesii* stands over large areas there, particularly on steeper slopes near the tree-line. Disturbance may have also interacted with browsing effects from large populations of deer and goats. Instead of an ensuing pulse of even-aged regeneration of *N. menziesii* or *Olearia colensoi*, browse-tolerant grass and fern communities developed (e.g. *Chionochloa conspicua* and *Hymenophyllum multifidum*) at some sites, probably due to the high levels of deer browsing and trampling occurring at that time. The 1958/59 Tararua Range quantitative vegetation survey showed wide-spread dieback of *O. colensoi*, and identified goats and deer as the chief

agents of damage to plant communities there (Holloway et al. 1963), confirming earlier observations (Kean & Pracy 1948; Zotov 1949; Kean & Davidson unpubl. 1959). However, Wardle (1962) suggested that damage to *O. colensoi* from the bud-destroying moth *Agriophara coricopa* could be responsible. Holloway et al. (1963) recommended increased levels of culling of deer and goats to restore regeneration of *N. menziesii*, sub-alpine shrubland, and alpine grassland vegetation communities, and this led to sustained ground and aerial deer culling in the 1960s, 70s and 80s. Although there is currently no government-funded deer culling in Tararua Forest Park, goat culling continues to the present time.

Aerial deer culling was initiated in Tararua Forest Park in 1975, followed by aerial venison recovery in 1978 and live deer capture operations in 1982. The 12 deer kills per hour achieved in 1975 can be compared with 8 deer kills per hour in 1998 in Kaweka Forest Park, where deer populations were over 10 deer/km² (Husheer & Robertson 2005). A NZFS deer faecal pellet survey in 1982/83 concluded that deer and goat numbers had been restricted to moderate levels by ground and aerial hunting through the 1970s, but that more intensive culling was required to further reduce deer abundance and ensure palatable plant regeneration. In the absence of more recent data on deer and goat populations, estimates of current populations are speculative. It is likely, however, that ungulate populations have not been sufficiently reduced in Tararua Forest Park to allow successful regeneration of palatable species. This has been shown in several other beech forests, even where intensive deer culling has been implemented (e.g. Nugent 1990; Nugent & Challies 1988; Husheer & Frampton in press). It is likely that ungulate populations need to be reduced to near zero before highly palatable species such as *Griselinia littoralis*, *Pseudopanax colensoi*, and *Raukaua simplex* can successfully regenerate.

Nothofagus menziesii is the dominant tree species over large areas in the Tararua Range and, because of high rainfall, is often the species dominant at the tree-line (Wardle 1984). In comparison, on the more northern axial ranges the more drought-tolerant mountain beech (*Nothofagus solandri* var. *cliffortioides*) commonly makes up the tree-line. At the tree-line in Tararua Forest Park, *N. menziesii* is abruptly replaced with shrub species such as *O. colensoi*, *Brachyglottis elaeagnifolia*, *Podocarpus nivalis*, and *Olearia ilicifolia*. Above the sub-alpine shrub zone, Tararua grasslands are dominated by *Chionochloa pallens* and *C. flavescens*, with the former species more common at high altitude or at sites with recent soils, and the latter on well-drained soils (Williams 1975; Wardle 1991). These two low-nutrient status, browse-resistant species have probably become more common since the 1930s peak of deer populations, as potentially competitive and more palatable species, such as *Astelia nervosa*, have been intensively browsed (Wardle 1991). At lower altitudes, *N. menziesii* gives way to red beech (*Nothofagus fusca*) and *W. racemosa*, with some stands containing tawa (*Beilschmiedia tawa*), *Elaeocarpus dentatus*, *Hedycarya arborea*, and *Melicytus ramiflorus*. Full descriptions of the vegetation in Tararua Forest Park can be found in Kean & Pracy (1948), Zotov (1949), Kean & Davidson (unpubl. 1959), Holloway et al. (1963), Wardle (unpubl. 1964), Kibblewhite (1965), Franklin (1967), Nicholls (1974), Moore (1976), Bell et al. (1976), Griffiths (1976), and Wardle (1991).

On a national basis, Tararua forests, shrublands and grasslands are relatively unremarkable. Tree diversity is low. With a small number of exceptions, few species have their limits in the region (e.g. *Elatostema rugosum*, *Mida salicifolia*) and endemism is low (only a few alpine species such as *Hebe evenosa* are endemic), with a few species otherwise restricted to the South Island occurring in the Tararua Range (Sawyer unpubl. 1994). In some instances, South Island species replace species commonly found further north (e.g. *Dracophyllum uniflorum* replaces *D. recurvum*, commonly found in the Central Plateau).

2. Methods

Several protocols have been employed to assess the vegetation of Tararua Forest Park. The first was an assessment for timber harvesting in 1954 by the NZFS using 26 one-acre plots (Franklin 1967). Data from these surveys were not available for this report. In 1958 and 1959, 46 cruciform plots and 36 scrub and grassland plots were established in Tararua Forest Park, based on the protocols described by Holloway & Wendelken (1957). In the summer of 1974/75, a network of 20 m × 20 m permanent forest plots was also established, using a protocol described by Allen (1993), and these were re-measured in 1984/85. The variability in methods (even within the same set of plots in the same survey) and the variety of sampling strategies employed have provided challenges for data analysis, which inevitably has limited the techniques employed.

2.1 CRUCIFORM PLOTS

Between November 1958 and February 1959, 46 cruciform plots, each occupying 0.1 acre (404.69 m²), were subjectively located throughout Tararua Forest Park. Sites that were selected (using aerial photographs) were thought to be generally representative of Tararua montane forests (Holloway et al. 1963), or where animals were thought to be inducing rapid changes in vegetation composition (Brady 1987). Plot establishment methods generally followed the protocol described by Holloway & Wendelken (1957) for forest plots and Wraight (1963) for grasslands. Four 100 link (c. 20 m) × 25 link (c. 5 m) arms were arranged in a cruciform, and all trees > 1 inch (c. 25 mm) within these arms were measured, diameter at breast height (DBH, breast height 135 cm), but were not individually tagged. Twenty milli-acre (114 cm radius) sub-plots were systematically located within the cruciform, and counts of seedlings < 12 in. (c. 30 cm) tall made for each sub-plot. Forty-six of these plots were re-measured between November 1963 and February 1964, and 40 between November 1974 and February 1975.

During the 1984/85 survey, trees in the cruciform plots were tagged for the first time and DBH was measured. In earlier surveys, DBH was estimated within 5 cm classes, and heights of shrubs were estimated within 60 cm height classes. In

the 1974/75 survey, generally only two of four sections were re-measured (Griffiths 1976).

2.2 ALPINE VEGETATION TRANSECTS

Between November 1958 and February 1959, 17 plots were subjectively located in sub-alpine shrublands where *O. colensoi* occurred, and 19 in tussock grasslands. At many of the scrub sites *O. colensoi* had undergone dieback and may not have been the dominant species, so these plots may be better described as grassland plots also. Between November 1963 and February 1964, 16 of these scrub plots were re-measured and another seven plots were established. In March 1985 the 17 scrub plots established in 1958/59 were re-measured along with two of the additional plots established in 1963/64 ($n = 19$). Grassland plots were re-measured in January 1964 and most recently in December 1974 ($n = 18$, with plot 10 not re-measured).

The protocol used to measure these plots was described by Holloway & Wendelken (1957). Briefly, the presence and species of plants occurring at point intercepts (114 cm radius, spaced at 2 link (0.4 m) intervals) were recorded in a similar layout to cruciform plots. Because some plots had two arms of the cruciform measured and other plots all four, plots had total of 50 or 100 intercepts, respectively. By calculating frequency of occurrence for each species relative to the total number of intercepts for all species, plots with different methods and sizes could be compared. This differs from the more commonly used technique of Wraight (1960, 1963), which typically uses 50 rings, 15 cm diameter, spaced at 40 cm intervals for assessing frequency of occurrence of plants (Wiser & Rose 1997).

2.3 PERMANENT QUADRAT PLOTS

Between November 1974 and February 1975, 168 permanent 20 m × 20 m plots were representatively established in forests in Tararua Forest Park on 51 transect lines randomly located in water courses and running to the nearest prominent ridge-top or alpine shrub/grassland area. Between November 1984 and March 1985, 168 of these plots were re-measured and an additional 64 plots established on 30 previously established faecal pellet monitoring transects. These additional plots were in areas that were considered to be previously under-sampled, particularly low-altitude seral forest (Brady 1987). Unlike the previously described alpine vegetation transects and cruciform plots, the 20 m × 20 m methodology strictly followed a widely used standard protocol that was also used by the NZFS survey teams (Allen 1993). Data for all measurements were stored on the National Vegetation Surveys databank (Wise et al. 2001). Plots were systematically located (at 200 m intervals from line origin, except for the first plot which was 40 m from origin) on transect lines. In each 20 m × 20 m plot, the species of all trees > 2 cm diameter at breast height (DBH) were identified, tagged and the diameter measured. All saplings (> 135 cm high, < 2 cm DBH) were counted and identified in the 1984/85 survey. Twenty-four circular understorey sub-plots (49 cm radius) were systematically located

according to Allen (1993) within each plot. All established woody seedlings (15–135 cm high) were identified and counted in each of the sub-plots. The presence of small seedlings (< 15 cm high) of vascular plants in sub-plots was also recorded, and their species identified. Altitude at each plot was estimated to the nearest 10 m; slope was estimated with an inclinometer to the nearest 5°; and aspect to the nearest 5° using a compass.

2.4 PLOTS USED IN ANALYSIS

Two broad approaches were used in analysis. Firstly, a comparison of species composition of the most recently measured plots was used to answer questions raised in objective 1 (classify vegetation). Secondly, comparisons of basal area, species composition, and mortality over time in forest and scrub plots were made. Because of discontinuity in plot re-measurement and establishment it was not possible in analysis to achieve a representative balance of plots among surveys.

2.5 RUAMAHANGA AND HUTT EXCLOSURE PLOTS

Data from a paired 20 m × 20 m enclosure plot, established in podocarp- and *W. racemosa*-dominated forests adjacent to Roaring Stag Lodge in the Ruamahanga River in January 1985 and re-measured in March 2004, was available to compare forest regeneration with and without browsing ungulates. Protocols for tree and sapling re-measurement followed Allen (1993). Seedling counts of common woody species in 2004 did not follow the standard protocol and have not been included in the analysis. Saplings were not counted in the 1985 survey, but the existence of foliage was recorded in the 30–200 cm height tier for combined fenced and unfenced plots.

A further set of enclosure plots for which data are available exists at three sites in the Hutt Valley. These plots were established more recently (1996) than the Ruamahunga paired enclosure, and were re-measured in 2002. Plots have one fenced plot and two unfenced plots of varying areas (Axehole 17 m²; Eastern Hutt 76.5 m²; Quoin Ridge 124 m²). Overstorey and seedling counts follow Allen (1993) for these plots.

2.6 DATA ANALYSIS

Detrended Correspondence Analysis (DCA) was used in CANOCO 4.0 (Ter Braak & Smilauer 1998) to summarise species composition of individual plots. This allowed plots to be grouped according to the relative importance of common species. Importance values were calculated using tree stems/ha for forest plots, and the proportion of intercepts where individual species occurred for grassland plots. Only the 21 species with mean relative abundance/occurrence in all plots > 0.5% were included (see Table 1 for a list of these species). Data were standardised by calculating relative abundance and

frequency of occurrence prior to analysis. Default options in CANOCO were used. Multiple regression in SYSTAT 10.0 (SPSS 2000) was then used to explore the relationship between plot DCA axis 1 scores and plot aspect, altitude, slope, and basal area ($P = 0.15$ to include and exclude in an automated forward stepwise procedure). TWINSpan (Hill 1979) was used to further classify sub-groups of the same species by plot matrix that was used in the DCA. Default options and cut levels were used in TWINSpan. Canonical Correspondence Analysis (CCA) was used to test for changes in seedling species composition in cruciform plots between 1958 and 1985. An automatic forward selection procedure was used in CCA to test for the effect of time between surveys (1958, 1963, 1975 or 1985), altitude, aspect and slope on species composition of the 21 most common species. Because permanent plots were repeatedly measured, and spatial variations in temporal changes were expected, plots were treated as blocks in CCA to reduce unexplained variation. Importance values were calculated from seedling density (stems/ha). For overstorey data, relative stem density for each plot (for species with $\geq 2\%$ frequency of occurrence in plots) and for grassland transects, relative frequency of occurrence for each transect (for species with $\geq 2\%$ frequency of occurrence) was used to summarise species composition. Default options were used in CANOCO 4.0. PC-Diam (Hall 1994a) and PC-Ustorey (Hall 1994b) were used to summarise seedling, sapling and tree densities for *W. racemosa* and *N. menziesii* in all surveys, and these results were then compared graphically to determine if any gaps in age-class structure existed among surveys. These programmes were also used to summarise seedling, sapling and tree stem densities from paired exclosure plots. For each plot where *Fuchsia excorticata* and *W. racemosa* were present, annual recruitment and mortality rates (McCune & Cottam 1985) were calculated using PC-Diam (Hall 1994a):

$$\text{Recruitment} = (1 + B_x)^{1/y} - 1,$$

where B_x = proportion of new stems for 10.2 years between 1974/75 and 1984/85 surveys; $y = 10.2$ years

$$\text{Mortality} = 1 - (1 - Q_x)^{1/y},$$

where Q_x = proportion of dead stems for 10.2 years.

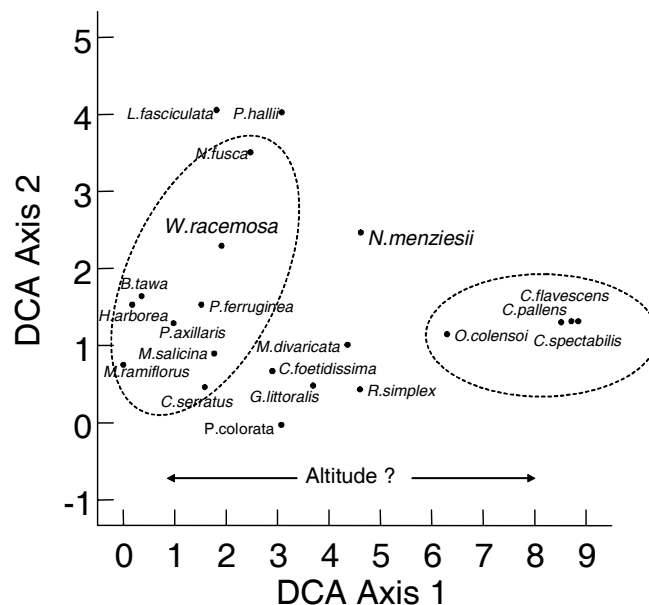
To test for changes over time in species composition of grassland and scrubland, DCA in CANOCO 4.0 (Ter Braak & Smilauer 1998) was used to summarise composition, and differences between surveys were tested using ANOVA in SYSTAT. Power analysis in SYSTAT was employed to determine the number of plots required to be re-measured to detect changes in the tree stem abundance of *W. racemosa* and *N. menziesii* at the 95% level of confidence of type I error, with a type II level of confidence of 80%.

3. Results

3.1 CHARACTERISATION OF PLOT DATA

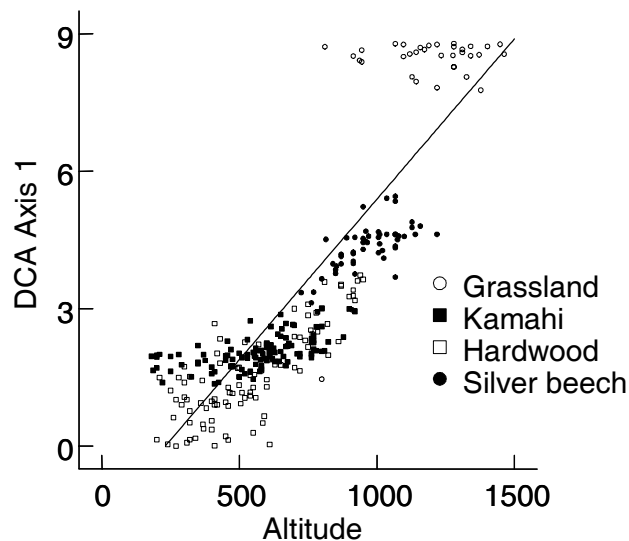
Three clear groups of plots were identified in DCA (Fig. 1). Species occurring at low altitude in *W. racemosa*-dominated forest (e.g. *Carpodetus serratus*, *Coprosma lucida*, *C. robusta*, *Myrsine salicina*) tended to occur in the same plots, while in *N. menziesii*-dominated forest mid-altitude species occurred as associations (e.g. *Cyathodes juniperina*, *Phyllocladus alpinus*, *Podocarpus hallii*). Not surprisingly, species in grassland plots above the tree-line were grouped by DCA into one association. There are two reasons to expect this. Most importantly, only a few species occur commonly in both grassland and forest plots. Secondly, different plot measurement techniques were employed for grassland and forest plots. This would have been a potential concern if overlap in species composition occurred between grassland and forest plots.

Figure 1.
DCA of Tararua Forest Park
vegetation from most
recent re-measurements of
permanent, cruciform
(overstoreys), scrub and
grassland plots.



A linear regression showed that 73% of variation in plant species composition (represented by DCA axis 1) was explained by variation in altitude ($r^2 = 0.729$, $P < 0.001$, Fig. 2). Other environmental variables not related to altitude explained far less variation ($r^2 < 0.05$). If variables closely related to altitude (e.g. temperature, soil podsolisation, and rainfall) were included in future analysis as explanatory variables, even more variation in species composition may be explained. A Twinspan analysis highlighted some weaknesses in the simple classification of plots into three groups, and the exclusion of plots that did not fit into these groups. For example, Twinspan identified a community characterised by *N. fusca*, *Carpodetus serratus*, *Coprosma lucida*, *C. robusta*, and *Myrsine salicina*, but examination of raw data revealed that these plots were not dominated by these species and actually occurred in low abundance.

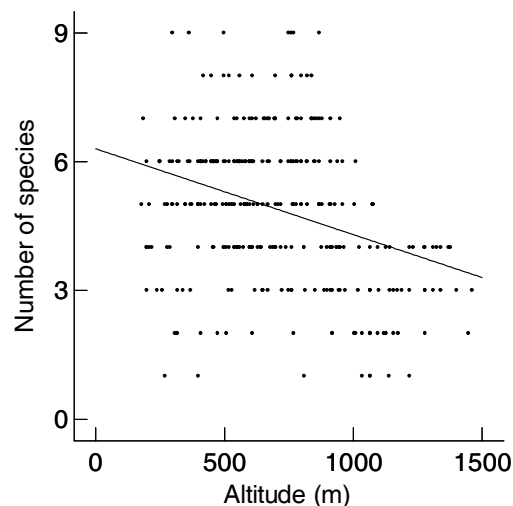
Figure 2. DCA axis 1 plot scores against plot altitude with fitted regression line calculated in linear regression. Importance values were calculated using relative frequency of occurrence.



After further examination of raw data it was confirmed that DCA had identified four groups of plots with dominant indicator species. A group dominated by *W. racemosa* ($\geq 45\%$ relative abundance, mean altitude in m a.s.l. 565 ± 16 , $n = 113$ plots), another group dominated by *N. menziesii* ($\geq 35\%$ relative abundance, altitude 973 ± 15 , $n = 53$ plots), a grassland group (altitude 1181 ± 30 , $n = 35$ plots), and a more complex and diverse group of plots in hardwood forest dominated by *B. tawa*, *Hedycarya arborea*, *Melicytus ramiflorus*, *Myrsine salicina*, *Pseudowintera axillaris* and *P. colorata* (altitude 545 ± 21 , $n = 96$ plots). The remaining 17 stands (altitude 736 ± 51) were intermediate associations of those species dominating the three forest groups. Further data exploration did not reveal any clear groupings of dominant species in this group of plots.

Although altitude had a strong influence on species composition, there was a much weaker relationship between species richness and altitude. There was a significant influence of plot altitude on the number of common plant species found in plots ($R^2 = 0.119$, $F_{1,312} = 42.053$, $P < 0.001$; Fig. 3), but only 12% of the variation in the number of species present was explained by altitude. The often smaller grassland plot size could explain some of the decrease with altitude.

Figure 3. Relationship between plot altitude and number of common species ($> 0.5\%$ relative abundance) found in the plot, with a line of best fit calculated in linear regression.



Hardwood-dominated stands ($n = 96$) were further divided into three sub-groups:

1. Climax stands, where long-lived canopy species with shade-tolerant seedlings dominate (e.g. *B. tawa* > 10%, *Carpodetus serratus* > 10%, *Griselinia littoralis* > 10%, *Myrsine salicina* > 10% and *M. divaricata* > 1%; $n = 56$).
2. Seral stands with fast-growing, nutrient- and light-demanding tree species more competitive on slips and recently disturbed sites (e.g. *Melicytus ramiflorus* > 10%, *Fuchsia excorticata*, *Coprosma grandifolia* and *Hedycarya arborea* $n = 26$).
3. Stands where slow-growing species unpalatable to deer (e.g. *Podocarpus hallii*, *Prumnopitys ferruginea*, *Pseudowintera axillaris* > 10%, *Pseudowintera colorata* > 10%; $n = 13$) appear to occur in high enough densities to prevent regeneration of faster-growing more palatable species. These slower-growing species may also be better adapted to older, well-developed but low-fertility soils.

Only one stand in the hardwood group, dominated by *W. racemosa* (40% relative importance) could not be classified into these three sub-groups.

3.2 FOREST STRUCTURE AND CHANGES IN SPECIES COMPOSITION

Canonical Correspondence Analysis (CCA) showed a significant effect of site characteristics on understorey woody species composition in 46 cruciform plots between 1958 and 1985, and a significant interaction effect between time and altitude (Table 2). There was no overall significant effect of change in time on vegetation composition. Furthermore, the direction of the interaction between time and altitude in ordination space was very closely aligned to the direction of altitude (Fig. 4). These results show that there was very little change in species composition between 1958 and 1985 and reinforce that altitude was a very important influence on species composition.

Analysis of 20 m × 20 m permanent plot data showed that there was also very little change in the structure of *N. menziesii* and *W. racemosa* populations between 1974 and 1985 (Fig. 5). Examination of *W. racemosa* age structure indicates a gap in regeneration in both the 1974/75 and 1984/85 surveys, with

TABLE 2. RESULTS OF CCA ON WOODY UNDERSTOREY SPECIES COMPOSITION FROM 46 CRUCIFORM PLOTS IN TARARUA FOREST PARK BETWEEN 1958 AND 1985.

The additional variance explained at the time each factor was included (γ conditional) and the significance of the variable at that time (P) are shown.

	γ	P	F
Altitude	0.53	0.005	15.65
Slope	0.12	0.005	3.64
Aspect	0.13	0.005	3.81
Year *Altitude	0.08	0.005	2.35

Figure 4. CCA diagram of woody understorey species composition from 46 cruciform plots in Tararua Forest Park between 1958 and 1985. Significant environmental factors are displayed. Only species with weight in CCA > 10 000 are displayed. Full names for species codes are listed in Appendix 1.

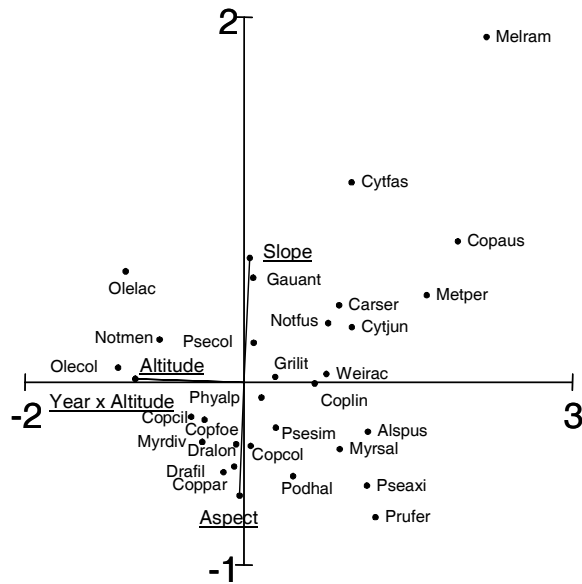
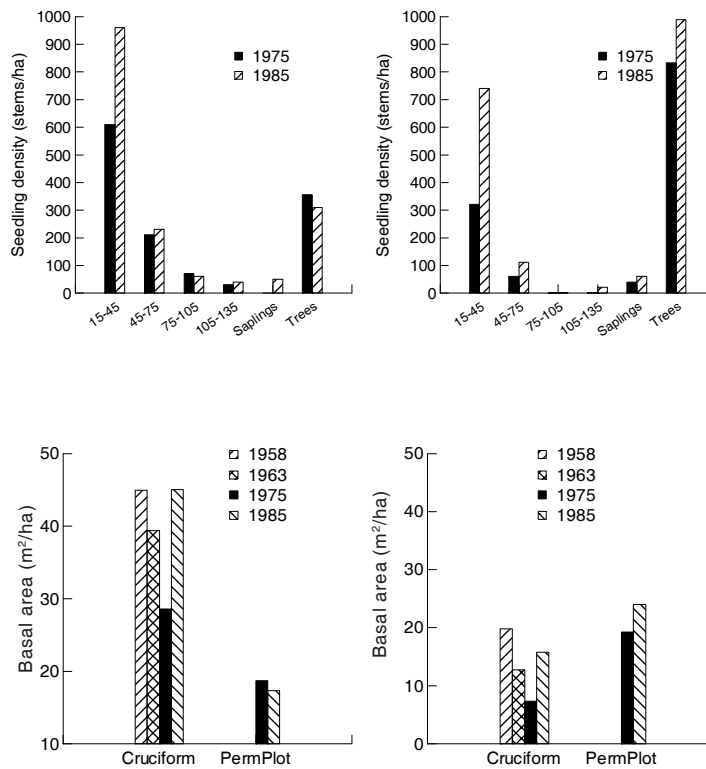
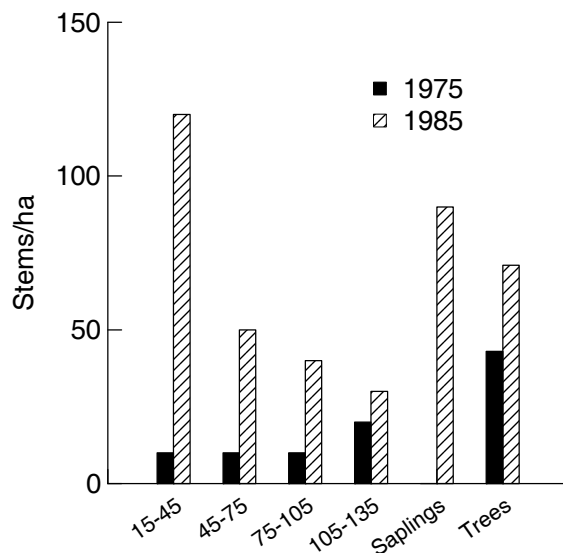


Figure 5. Density of seedlings (four tiers), saplings (> 135 cm high and < 2 cm DBH) and tree stems (> 2 cm DBH) from permanent plot surveys (upper histograms) and basal areas from cruciform and permanent plots (lower histograms) for *N. menziesii* (left) and *W. racemosa* (right).



no seedlings between 75 cm and 135 cm being counted. In contrast, *N. menziesii* populations showed an age class distribution consistent with a successfully regenerating population in 1984/85. Age structure of *B. tawa* populations favoured small seedlings in the 1985 survey, indicating increased regeneration from 1975 (Fig. 6).

Figure 6.
Density of seedlings (four tiers), saplings (> 135 cm high and < 2 cm DBH) and tree stems (> 2 cm DBH) from permanent plot surveys for *B. tawa*.



Mean basal area for *W. racemosa* and *N. menziesii* was consistent between cruciform and permanent plot surveys, showing that most of these stands are likely to be in the late stage of development, with minimal growth, recruitment or mortality (unless high mortality of mature trees is coincidentally balanced with massive recruitment of medium-sized trees). However, there were large differences in figures obtained for cruciform and permanent plot surveys for the same field seasons, indicating that the subjectively located cruciform plots do not accurately represent *W. racemosa* and *N. menziesii* forest structure. Because of the large number of permanent plots established on randomly located lines it can be assumed that this plot system is representative.

3.3 RUAMAHANGA EXCLOSURE PLOT

Results from the Ruamahanga exclosure plot suggest that browsing may influence the tree stem density of *W. racemosa*. In 1985, *W. racemosa* tree stem (> 2 cm DBH) density was similar in the fenced (425 stems/ha) and unfenced plots (400 stems/ha). By 2004, in the fenced plot *W. racemosa* stem density had increased (500 stems/ha) but in the unfenced plot it had declined (325 stems/ha). *Melicytus ramiflorus* followed similar trends, with comparable tree stem density in 1985 between fenced and unfenced plots (18 v. 8 stems/ha), but by 2004 stem density was far higher in the fenced plot than the unfenced plot (700 v. 75 stems/ha). Similarly, trees of *Coprosma grandifolia* were only present in the fenced plot in 2004 (125 stems/ha).

By 2004, density of saplings (> 135 cm high, < 2 cm DBH) was higher for these species in the fenced plot than the unfenced plot (Table 3), along with other deer- and goat-palatable subcanopy hardwoods (*Carpodetus serratus*, *Coprosma lucida*, *Elaeocarpus dentatus*, *Melicytus ramiflorus*) and the unpalatable *Pseudowintera axillaris*.

TABLE 3. SAPLING DENSITIES (plants/ha) IN 2004 IN THE FENCED AND UNFENCED PLOTS RUAMAHANGA RIVER, TARARUA FOREST PARK.

Presence or absence of vegetation (as seedlings, saplings or branches of large trees) of these species was also recorded in a 30-200 cm high tier in 1985 (Allen 1992).

	VEGETATION 1985	FENCED 2004	UNFENCED 2004
<i>Carpodetus serratus</i>	Absent	125	0
<i>Coprosma lucida</i>	Absent	175	0
<i>C. grandifolia</i>	Absent	875	0
<i>Elaeocarpus dentatus</i>	Absent	50	0
<i>Melicytus ramiflorus</i>	Present	1150	0
<i>Pseudowintera axillaris</i>	Present	250	175
<i>P. crassifolius</i>	Absent	25	0
<i>Weinmannia racemosa</i>	Present	25	0

3.4 HUTT VALLEY EXCLOSURE PLOTS

There was no evidence of any consistent difference in seedling abundance between fenced and unfenced plots for the three Hutt Valley exclosures for *Coprosma grandifolia*, *Griselinia littoralis*, *Melicytus ramiflorus*, red and *N. menziesii*, and *W. racemosa* (Table 4). Fast-growing and palatable species such as *C. grandifolia* were not present in any of the seedling sub-plots in 2002. Seedlings of the palatable *G. littoralis* were present at two sites in 2002. Seedlings were more abundant inside one fenced plot in comparison to unfenced controls (Axehole). Reductions in deer densities following aerial possum culling in the 1990s or the short period of exclusion of ungulates is likely to explain the lack of consistent increases in palatable seedlings in fenced plots compared to unfenced plots.

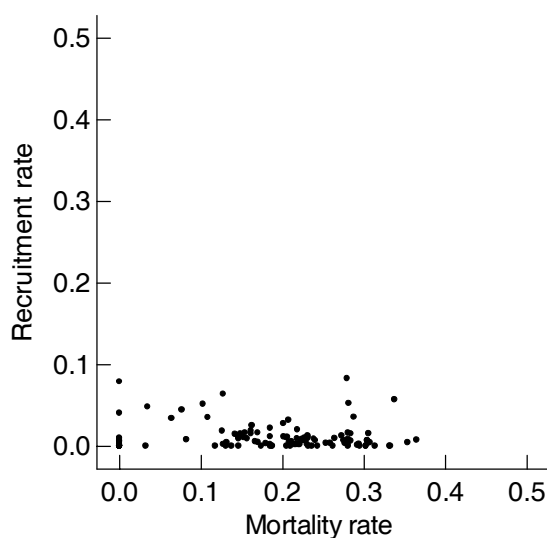
TABLE 4. BASAL AREA AND SEEDLING DENSITIES FOR THREE EXCLOSURE PLOTS IN THE HUTT VALLEY.

SITE Plot	AXEHOLE			EASTERN HUTT			QUOIN RIDGE		
	Exclosure	Control 1	Control 2	Exclosure	Control 1	Control 2	Exclosure	Control 1	Control 2
Basal area, 1996 (m²/ha)									
<i>Melicytus ramiflorus</i>	0	0	0	15.8	9.1	31.8	0	0	0
<i>Notofagus fusca</i>	156.2	0	100.0	0	0	0	0	6.9	0
<i>N. menziesii</i>	0	0	0	0	0	0	35.7	8.8	23.8
<i>Weinmannia racemosa</i>	11.8	23.8	17.4	0	5.0	0	0	0	0
Plot total	168.0	23.8	117.4	76.5	62.5	115.2	36.0	16.8	25.3
Seedling densities, 2004 (stems/ha)									
<i>Griselinia littoralis</i>	3310	0	0	0	0	0	660	1990	1330
<i>N. fusca</i>	0	0	0	0	0	0	660	2650	0
<i>N. menziesii</i>	0	0	0	0	0	0	25850	23860	62310
<i>W. racemosa</i>	0	0	0	880	0	0	0	660	0

3.5 MORTALITY AND RECRUITMENT OF *Fuchsia excorticata* AND *Weinmannia racemosa*

Annual mortality and recruitment rates for *Fuchsia excorticata* and *W. racemosa* were calculated for each 20 m × 20 m permanent plot, where these species occurred. Stems of *F. excorticata* only occurred in ten 20 m × 20 m plots, and in these plots annual mortality was consistently higher than annual recruitment (mean mortality ± SE = 0.191 ± 0.095 v. recruitment 0.001 ± 0.001), but this result was not statistically significant (paired $t_9 = 2.008$, $P = 0.076$). For *W. racemosa*, mean annual mortality (0.174 ± 0.108) was sixteen times higher than recruitment (0.011 ± 0.016; paired $t_{122} = 12.063$, $P < 0.001$). Linear regression showed that neither mortality ($P = 0.861$, Fig. 7) or recruitment ($P = 0.910$) were significantly related to plot altitude, and a Pearson correlation showed that plot mortality was not correlated with recruitment (coefficient = -0.009).

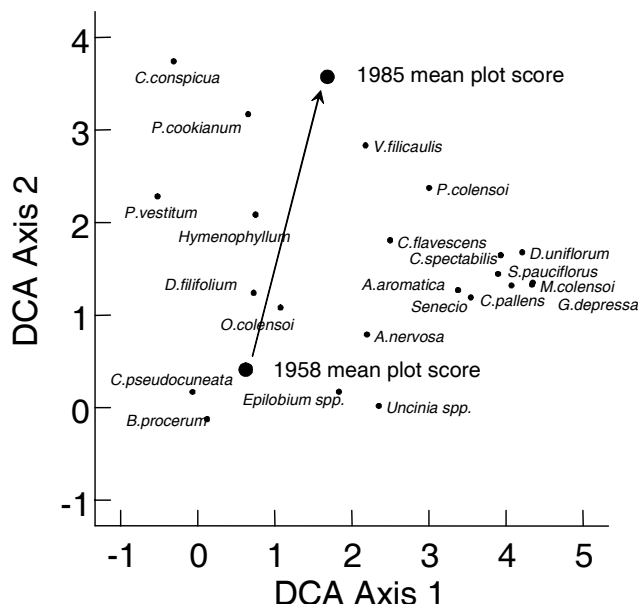
Figure 7.
Relationship between
W. racemosa annual
mortality and recruitment
in permanent plots
between 1974/75 and
1984/85.



3.6 CHANGES IN GRASSLAND SPECIES COMPOSITION

Unlike forest species composition, there were significant changes detected in species composition in grassland/scrub plots ($n = 19$) between 1958 and 1985 (Fig. 8). ANOVA showed that change in mean plot scores between surveys was significant at the 99% level of confidence. DCA axis 1 appears closely associated with environmental characteristics, particularly altitude, while axis 2 appears closely associated with changes in species composition over time. Species such as *Blechnum procerum*, *Coprosma pseudocuneata*, *Epilobium* spp. and *Olearia colensoi* have become less common, and species such as *Chionochloa conspicua*, *Phormium cookianum*, *Poa colensoi*, and *Viola filicaulis* have increased in importance. However, in comparison with the dominant species in these communities (i.e. *Chionochloa flavescens*, *C. pallens*, *Celmisia spectabilis*), species that have changed in occurrence are relatively unimportant. Therefore changes detected reflect relatively minor variation in overall species composition between 1958 and 1985.

Figure 8.
DCA ordination of
frequency of occurrence
from permanent grassland/
scrub plots in Tararua
Forest Park in 1958, 1963
and 1985. Mean plot scores
for 1958 and 1985 are also
displayed.



3.7 PERMANENT PLOT POWER ANALYSIS

Using data from paired *t*-tests between 1975 and 1985 surveys of *W. racemosa* plots, power analysis shows that there are sufficient plots ($n = 121$) to detect a 5% change in *W. racemosa* basal area (n required = 108 at 5% type I error and 20% type II). To detect a 20% difference only 29 plots are required. To detect a 10% change in *W. racemosa* tree stem density, 45 plots are required, but only 11 to detect a 20% difference.

For *N. menziesii*, 19 plots are required to detect a change in stem density of 10% and only seven to detect a difference of 20%. For basal area, 39 plots are required to detect a 10% change and 12 plots for a 20% change.

For *B. tawa*, 20 plots are required to detect a change in stem density of 10% and only six to detect a difference of 20%. *Beilschmiedia tawa* basal area varied substantially between plots and because of this it will only be possible to detect very large changes over time. For instance, fifteen plots are required to detect a 500% change in basal area.

Overall, thirty plots in each of the *N. menziesii*, *W. racemosa*, and *B. tawa* groups would be sufficient to detect reasonably small changes in stand density and basal area. If possible, a similar and sufficient number of *W. racemosa*-dominated plots should be placed in both possum treatment and non-treatment areas so that recruitment and mortality can be compared. Power analysis showed that less than 9 plots are required to confirm that stem mortality exceeds recruitment at the same rate as occurred between 1974 and 1985. More plots would be required if the difference narrowed or if more information on mortality is required at shorter intervals in the future. For *B. tawa*, monitoring objectives should focus on changes in stem density, because of high variability in stem sizes and basal area. Power analysis was not employed on the grassland/scrub plots. Firstly, there is no real obvious single parameter of interest to use. Secondly, if these plots are to be re-measured it will be necessary to measure all that can be found. Some of these plots have not been re-measured for three decades, and it will be difficult to locate them.

4. Discussion

4.1 VEGETATION CLASSIFICATION BY SPECIES COMPOSITION

Altitude explained most of the variation in species composition in Tararua Forest Park vegetation (73% of variation in DCA axis 1 scores was explained by altitude). Other environmental parameters, including time, each explain at most a few percent of variation in species composition. Three groups of plots, each group dominated by a single genus (*Weinmannia*, *Nothofagus*, and *Chionochloa*) were identified. These same three forest types have been found previously (Franklin 1967). These types were divided into groups for analysis and further sampling. This is very useful because it simplifies future sampling procedures and allows clear management objectives to be developed for each grouping. Management objectives and monitoring strategies should consider the different characteristics of each group. A further, more complex, plant community was also identified, composed of hardwood species that often occur at low altitude or disturbed sites. Developing management objectives for this vegetation type will be more challenging and requires a multi-plant species approach because of the greater heterogeneity in composition between plots. Monitoring this vegetation type will consequently be expensive, and is likely to require re-measurement of all 96 plots classified into this group. There was a trend for the hardwood forest type to have a greater number of species occurring in the overstorey, although this was also the case for *W. racemosa*-dominated plots occurring at low altitude. Because there was some altitudinal overlap between plots dominated by *W. racemosa*, *N. menziesii*, and hardwoods, strict altitudinal groupings can not be applied.

The hardwood group was successfully divided into three sub-groups, although the usefulness of this for identifying changes in composition or planning monitoring is very limited. It is likely to be more efficient to treat the hardwood group as a single, more diverse entity. Different stands in this group were dominated by climax species (*B. tawa*, *Hedycarya arborea*), seral species (*Melicytus ramiflorus*), and slow-growing, unpalatable species (*Pseudowintera axillaris* and *P. colorata*). Because of the greater number of species within plots (alpha diversity) and species turnover among plots (beta diversity), this group poses additional re-sampling challenges in comparison to the other three less heterogeneous vegetation types. This means that it would be prudent to sample all plots in this group if rigorous results are required. If managers determine that *B. tawa* forest alone is important, then a reduced number of hardwood plots containing *B. tawa* would be sufficient.

4.2 PLANT COMMUNITY PALATABILITY CHANGE

Prior to plot establishment in 1958 Tararua forest and grassland vegetation had already been modified by browsing and trampling of ungulates. Deer numbers probably peaked in the Tararua Range in the 1930s, with dozens of deer being seen in a hunting day, e.g. Mangatira: 'evening and morning shoot... getting only 17' (Davison unpubl. 1939). Barra (unpubl. 1939) stated that deer numbers increased between 1935 and 1939, and that during that time deer were in the process of eliminating *Pseudopanax arboreus* and *P. colensoi* from forest and sub-alpine shrublands. By the 1940s forest understoreys were more open and seedlings absent from many sites, but deer numbers had also declined, e.g. Hutt Valley: 'deer are not numerous owing to the great shortage of food, and what animals we have destroyed are in very poor condition' (Barra unpubl. 1941a); 'saw one deer', 'practically no sign of deer' (Barra unpubl. 1941b). This was reinforced by Kean & Pracy (1948), who stated that deer reached an irruptive peak in the Ruamahanga Valley in the 1930s, and during that decade palatable species such as *Aristotelia serrata*, *Coprosma grandifolia*, *F. excorticata*, *Griselinia littoralis*, *Melicytus ramiflorus*, *Pseudopanax colensoi* and *W. racemosa* were intensively browsed. By the 1950s these palatable species had largely disappeared from the seedling and sapling tiers in forests, wherever they were accessible to deer. In their place ferns (e.g. *Blechnum discolor*) and a browse-tolerant turf consisting of species such as *Isolepis* spp., *Nertera dichondrifolia*, *Poa breviglumis*, and *Ranunculus reflexus*, had begun to appear (Zotov 1949). It is also likely that there had also been a large reduction in the density of seedlings of other species moderately palatable to deer, such as *Melicytus ramiflorus* and *Griselinia littoralis*. Zotov (1949) and Kean & Pracy (1948) also stated that alpine grassland and scrub vegetation had been browsed and trampled by deer by 1947, with reductions in the abundance of species such as *Astelia nervosa*, *Chionochloa flavescens*, *Olearia arborescens*, and *Polystichum vestitum*. Deer numbers were 'high' on alpine grasslands in 1937 and 1941, but were only 'moderately high' in 1946, 1949, 1955 and 1961 (Pracy unpubl. 1961).

In contrast to the conclusions of Zotov (1949) and other earlier studies, results from data analysed in this study show that there was very little change in vegetation composition in Tararua Forest Park in the two to three decades prior to final plot measurement in 1985. This is consistent with several other permanent plot studies throughout New Zealand (Stewart Island, Bellingham & Allan 2003; Kaimanawa Forest Park, Husheer 2003; Wakatipu Forest, Husheer & Frampton in press), which showed little vegetation change during that time. It appears that throughout New Zealand, plant species palatable to deer were drastically reduced in abundance prior to 1970 (and prior to a period of intensive plot establishment in the 1970s and 1980s). Alpine grasslands were also dramatically affected, but the venison industry removed deer from alpine regions in the 1960s and 1970s allowing a quick vegetation recovery (e.g. Mark 1989). Some ongoing changes in the composition of forest and grassland ground cover may not have been detected, because low-stature vegetation is often omitted from monitoring programmes. For example, increases of *Uncinia* in Tararua Forest Park (Holloway et al. 1963) and Haurangi Forest Park (Wardle 1967) have been attributed to deer, and Husheer et al. (2003) attributed

increases in prostrate herbaceous plants in Kaimanawa Forest Park to ongoing deer browsing.

Previous studies have shown that a group of broadleaved, subcanopy, hardwood species are always preferred by red deer and goats, e.g. *Griselinia littoralis*, *Melicytus ramiflorus*, *Pseudopanax arboreus*, *Schefflera digitata* and *W. racemosa* (Forsyth et al. 2002), and previous exclosure plot studies have shown that deer and goat browsing reduces the density of these species as seedlings (Allen et al. 1984; Stewart & Burrows 1989; Smale et al. 1995; Bellingham & Allan 2003). Since the 1970s, permanent plot monitoring has shown that, in some forests where disturbance has occurred, slow-growing plant species have slowly increased in abundance, along with the increase in browse-tolerant grasses and herbs in the forest understorey. Coomes et al. (2003) suggest that a lack of change in New Zealand indigenous forests and alpine grasslands reflects an irreversible impact of deer in the middle of last century. This is consistent with the lack of change in forest composition observed in other New Zealand forests and grasslands (e.g. Husheer & Frampton in press; Husheer 2003). If permanent plots were re-measured soon in Tararua Forest Park, the time frame is sufficient that it is likely that consequential analysis would show a slow increase in species such as *Myrsine divaricata*, *Pseudowintera colorata*, and podocarps, particularly at sites where tree death has occurred.

Of concern is the higher rate of mortality than recruitment for *W. racemosa* in Tararua Forest Park found in this study. It is possible that this imbalance has been translated into reductions in stem density in the two decades since last re-measurement, although a decline in the abundance or occurrence of *W. racemosa* up to 1985 was not detected. It seems likely that deer browsing contributed to reductions in tree stem density of *W. racemosa* between 1975 and 1985, but this conclusion is speculative without further study. *Weinmannia racemosa*, *B. tawa*, and *N. menziesii* all showed increases in the density of small seedlings (15–45 cm high) between 1975 and 1985. This may have been a result of a reduction in browsing intensity following intensive deer culling. An alternative view is that *W. racemosa* regeneration is poor under a closed *W. racemosa* canopy and requires complex succession to take place before it can occur (Payton et al. 1984; Payton 1989). There are indications from other forest surveys that *W. racemosa* regeneration can be suppressed by deer browsing, even with only low deer abundance (e.g. Pureora Forest Park; Copland Valley).

The Tararua dataset contains accurate information on the fates of a large number of trees over several decades and is a nationally valuable resource that should be augmented with paired exclosure plots. It is likely that more exclosure plots were established in the Tararua Range other than the four reported in this study (e.g. Franklin 1967, p. 10), but unfortunately it appears that none of these additional fenced plots have been maintained, and any data previously collected appear to have been lost.

Ulrich & Brady (in press) have shown that, in areas where aerial 1080 operations have been undertaken, declines in foliage density of *F. excorticata* were less than in adjacent areas where no control was undertaken. In some instances there was no decline in foliage density at sites with possum control. This was also reflected in their mortality results, particularly for *F. excorticata* saplings and epicormic growth. It is likely that deer, and to a lesser extent

possums, are partly responsible for a lack of stem recruitment following mortality. In plots used by Ulrich & Brady (unpubl. data) there were also clear signs of *F. excorticata* and *Griselinia littoralis* failing to survive beyond small seedlings. Data from the present study also tend to support a conclusion that *F. excorticata* mortality exceeds recruitment. It would be interesting to know the current fate of the *F. excorticata* stems that occurred in permanent plots (just ten plots from the 1975 and 1985 surveys). More consideration should be given to the effect of ongoing deer browsing on *F. excorticata* when planning future possum control. Is there real benefit if adult trees survive longer and increase in size, if recruitment in the sapling class is suppressed by deer?

In the low-altitude hardwood group there are likely to have been compositional shifts prior to 20 m × 20 m plot establishment in 1974/75. Almost by definition, seral communities undergo rapid compositional shifts. Unfortunately, cruciform plots were subjectively located in 1958 and tended to be concentrated on higher-altitude stands, so these older plots do not provide much information on this group. In the Orongorongo valley, which has a comparable forest type (*Hedycarya arborea*, *Melicytus ramiflorus*, *Prumnopitys ferruginea*), *Metrosideros robusta* declined in importance between 1969 and 1994, presumably because of intense browsing by possums (Bellingham et al. 1999). It is also probable that *M. robusta* and other possum-palatable species such as *F. excorticata* declined in low-altitude Tararua forests following possum populations reaching high densities in the 1940s and 1950s, but this conclusion is speculative.

4.3 COMMUNITIES MOST SUSCEPTIBLE TO BROWSING

Table 1 lists results of a review of the palatability to ungulates of common plant species in Tararua Forest Park. Veblen & Stewart (1982), Wardle (1984), and Forsyth et al. (2003) argued that the palatability and vulnerability of different plant communities is likely to vary on several spatial and temporal scales, reducing the usefulness of simple classifications of plant palatability. The mechanisms affecting spatial and temporal complexities might include gradients of productivity, nutrient availability, soil development, climate, disturbance, and altitude. It is also likely that smaller ungulates such as goats and sika deer are able to browse plant species that are only of moderate or low palatability to red deer (Hanley 1997), although this may vary spatially also. Nevertheless, it is still worthwhile to consider the species composition of the different groups identified earlier and use Table 1 to assess the vulnerability of different plant species found in these groups, particularly the canopy dominants.

4.3.1 Kamahi group

In the kamahi (*W. racemosa*) group, previous studies in similar forests (e.g. Allen et al. 1984; Payton et al. 1984; Bellingham & Allan 2003; Husheer et al. 2003) have shown that red deer are likely to suppress regeneration of palatable species such as *Carpodetus serratus*, *Coprosma foetidissima*, *Griselinia littoralis*, *Hedycarya arborea*, *Melicytus ramiflorus*, and *Raukaua simplex*, as

well as *W. racemosa*. The Ruamahanga exclosure plot reinforces this view, with *W. racemosa*, *Coprosma grandifolia*, and *C. lucida* probably failing to regenerate outside the fenced plot at that site, but with large increases in density inside the fenced plot. In contrast, the three exclosure plots in the Hutt Valley have shown only a limited response after eight years. Rogers & Leathwick (1997) concluded that red deer were instrumental in the almost complete collapse of a similar forest type in the southern Ruahine Range in the 1950s. Although this has not occurred on the same scale in Tararua Forest Park, if goats or sika deer spread beyond their current distributions, there are likely to be even fewer sites where these palatable species regenerate. At some steep sites where deer access is restricted, seedling growth is extremely fast, or hunting pressure is particularly intense, regeneration of these species is still likely.

4.3.2 Silver beech group

For silver beech (*N. menziesii*), previous studies have found that red deer do not usually affect canopy replacement in New Zealand beech forests (e.g. James & Wallis 1969; Wardle 1984; Stewart & Harrison 1987; Stewart et al. 1987; Nugent 1990; but see also Conway 1949; Wardle 1962; Jane & Pracy 1974; Jane 1986), and this is likely to be the case for Tararua Forest Park also (but see Zotov 1949). However, sika deer and goats have the potential to browse more intensively than red deer on seedlings of trees and shrubs (Fraser 1996; Hanley 1997), enabling sika to have higher impacts on vegetation, even in depleted forest habitats (Asada & Ochiai 1996). Over the past two decades there have been at least nine illegal liberations of sika deer into the Tararua region (or dispersal of animals to new ranges from illegal introductions), which poses a potential problem for forest conservation (Fraser et al. 2000). In future, if sika deer displace red deer as has occurred in the Kaimanawa Region (Davidson & Fraser 1991), they may suppress *N. menziesii* seedling regeneration where red deer currently do not, and maintain higher population densities than red deer. If goat populations are allowed to spread and increase in density, *N. menziesii* regeneration may also be impaired. Sika deer and goats also have the potential to suppress regeneration of other species that red deer generally do not affect, such as *Myrsine divaricata*. Fortunately, goats are more sedentary, and disperse more slowly, and culling operations have been more effective in the past than with sika deer. These attributes render goat populations a lesser threat than sika deer.

4.3.3 Grassland/scrub group

Mark (1989) recorded a rapid response in tussock grasslands at some sites in Mt Aspiring National Park following a large reduction in deer densities in the 1970s. Palatable plants such as *Chionochloa* and *Ranunculus* increased in cover within several years, sometimes dramatically at more productive sites where deer had browsed most intensively. Deer abundance has been maintained at low densities in Mt Aspiring National Park for three decades, and in this time palatable species have continued to increase in abundance, while less palatable species such as *Celmisia spectabilis* have declined in importance (Allan Mark, pers. comm.). Tararua grassland plot results did not show such a large response, despite apparently being established at sites of high deer

impact. It could be that deer numbers in the Tararua Forest Park alpine zone have not been lowered to the same extent as in South Island grasslands. Alternatively, deer numbers within the sub-alpine zone could have been lowered by intensive deer culling subsequent to plot establishment, but these sites were not as favoured by deer as initially thought. It is unfortunate that data are not available from the grassland plots from the 1985 re-measurement, because these plots may show more dramatic changes in vegetation composition. This makes re-measurement of the grassland plots a priority, if they can be relocated.

4.3.4 **Hardwood group**

Predictions on the susceptibility to ungulate browsing of plant communities in the hardwood group are more speculative than the other three groups. This group is more complex, has greater species variation between plots and covers a greater range of altitude, even when three sub-groups are considered. Animal impacts are likely to vary with site productivity, which is likely to decrease with increasing altitude, since temperature and soil fertility also decrease with altitude in Tararua Forest Park. Ungulate browsing will probably have least short-term impact in the climax and *Pseudowintera axillaris* sub-groups since the dominant species in those plots (e.g. podocarps, *B. tawa*, and *Myrsine salicina*, *Pseudowintera axillaris*) are generally unpalatable to ungulates. In contrast, seral stands would tend to be dominated by fast-growing, palatable species (e.g. *Aristotelia serrata*, *C. grandifolia* and *F. excorticata*) if ungulate browsing was eliminated. These stands may have undergone recent disturbance, have younger and more fertile soils, and have less shade from large canopy dominants. When fenced exclosure plots have been established at such sites, e.g. Aorangi Forest (Husheer et al. in press), the response of palatable subcanopy hardwoods has often been dramatic, and this is likely to be the case in Tararua Forest Park also.

4.4 MANAGING SUSCEPTIBLE PLANT COMMUNITIES

4.4.1 **Kamaha group**

If deer have continued to suppress regeneration of *W. racemosa* seedlings since 1985, as appears to be the case prior to 1985, then inevitably *W. racemosa* will decline in importance in the canopy as mature trees die and are not replaced. In the absence of possum control, mortality of large *W. racemosa* trees may be accelerated by possum browsing in addition to natural mortality (Payton 1989). In that case *W. racemosa* may decline in importance at many sites in Tararua Forest Park in the next one or two decades. If intensive deer culling is implemented, and red deer are reduced to low densities (< 1 deer/km²), it is likely that *W. racemosa* will successfully regenerate (Nugent et al. 2001). If deer have induced permanent changes in vegetation composition towards species such as *Blechnum discolor*, that are capable of suppressing seedling establishment, regeneration may not occur even if deer browsing is eliminated. Because of the current uncertainty of the regeneration status of *W. racemosa* in Tararua Forest Park, the unclear outcomes of lowered deer abundance, and the high-risk and contentious nature of deer culling, it would be prudent to test the

likely response of *W. racemosa* to reduced deer abundance through a replicated exclosure plot study. Other palatable species that occur in all groups, such as *Carpodetus serratus*, *Griselinia littoralis* and *Raukaua simplex*, are likely to have fewer seedlings through direct browsing by deer. To compare the abundance of seedlings in Tararua Forest Park with other forests throughout New Zealand, 20 m × 20 m plot data were extracted from the National Vegetation Surveys databank (Wiser et al. 2001). Table 5 shows that *Carpodetus serratus*, *Griselinia littoralis*, *Raukaua simplex*, and other palatable species, are at a similar abundance in Tararua Forest Park to other mixed beech-*W. racemosa* forests in New Zealand. Remarkably, the Kaimanawa Region has some of the highest densities of palatable seedlings, despite intensive browsing by high populations of sika deer for several decades (Husheer 2003). This is likely to be because dense thickets of small-leaved shrubs such as *Coprosma* “taylorae”, *C. pseudocuneata*, *C. microcarpa*, and *Myrsine divaricata* have replaced *N. solandri*, *N. menziesii* and *N. fusca* at many sites, and act as a protective screen for palatable seedlings from deer. It is unclear to what extent this will allow regeneration of palatable seedlings in the Kaimanawa Region in the future. There are very few forests in New Zealand that appear to have unimpeded regeneration of palatable seedlings, on a landscape scale. Some large river valley systems in central Westland, where red deer have been nearly completely removed, may be an exception, e.g. Waiho (Phil Knightbridge, pers. comm.). Some catchments in Westland with low numbers of deer, thar, and chamois in the forest also show signs of *W. racemosa* regeneration failure, e.g. Copland paired exclosure plot (Wardle et al. 2001).

4.4.2 Silver beech group

It is likely that silver beech (*N. menziesii*) is successfully regenerating at most sites in the *N. menziesii* group with current management and deer densities. If deer numbers increase, or if sika deer become dominant, this may change. Husheer (2003) showed that sika deer are able to suppress regeneration of *N. solandri* in the Kaimanawa Region, whereas *N. solandri* is successfully regenerating in other forests where only red deer are present. Palatable subcanopy trees and shrubs occurring in the *N. menziesii* group, such as *Carpodetus serratus*, *Coprosma foetidissima*, *Griselinia littoralis*, *Fuchsia excorticata*, *Pseudopanax colensoi*, and *Raukaua simplex*, are likely to have been prevented from regenerating over large areas in Tararua Forest Park (and elsewhere in New Zealand; Tables 5A, B) by continued browsing by red deer for nearly a century. For regeneration of these species to be assured, deer and goat abundance will need to be maintained at very low levels for at least a decade. Even in the Murchison Mountains, where deer have been intensively culled for over four decades, highly palatable species do not freely regenerate at all sites, so the effort required to lower deer to sufficient levels to restore regeneration should not be underestimated.

4.4.3 Grassland/scrub group

With the recent near demise of the commercial venison recovery industry, there will be increased dependence on recreational hunters for deer control on the Tararua alpine grasslands. This is at a time when hunters as a group appear to be ageing and less active, which may mean an increase in the abundance and

TABLE 5A. SEEDLING STEM DENSITIES FOR SOME SPECIES FROM PERMANENT PLOT SURVEYS IN MIXED BEECH FOREST THROUGHOUT NEW ZEALAND (SEE HUSHEER & FRAMPTON IN PRESS) COMPARED WITH TARARUA PERMANENT PLOT SURVEY RESULTS FROM 1984/85.

PLOTS	<i>n</i>	<i>Aristotelia serrata</i>	<i>Carpodetus serratus</i>	<i>Coprosma lucida</i>	<i>Fuchsia excorticata</i>
Arawhata	43		128 ± 44		
Hurunui North	27		0		
Hurunui South	25		243 ± 243		
Kaipō	10		0		
Maungaorangi	33	0	0		0
Murchison mtns	22	0	0	50 ± 35	0
North Kaimanawa	57	10 ± 10	291 ± 73	87 ± 55	29 ± 22
Rangataua	48	161 ± 94	380 ± 127	46 ± 36	0
Rotoiti	17	0	32 ± 32		0
South Kaimanawa	35	16 ± 16	284 ± 147		0
Waitutu	107		36 ± 13		
Wakatipu	49	11 ± 11	248 ± 226	11 ± 11	0
West Kaimanawa	46	0	216 ± 90	60 ± 39	0
Tararua	234	14 ± 11	154 ± 28	40 ± 11	28 ± 22

(continued opposite)

impacts of red deer on alpine grassland vegetation. This is a situation that should be monitored through the use of a newly developed faecal pellet monitoring system for deer abundance (Forsyth & Veltman, unpubl. data), re-measurement of existing grassland plots, and establishment of paired exclosures at some of these sites. Fast-growing, deer-palatable species are relatively uncommon in the alpine tussock grasslands, but where they are present they may be highly sought after by deer, meaning deer abundance has to be very low for highly palatable plants to survive. Of all the habitats in the Tararua Range this is likely to be the most rewarding for each ungulate control dollar spent. Deer and goats are efficiently hunted in this habitat, both from ground- and aerially-based culling, but without control can reach very high densities. It is possible to reduce animal densities well below 1 per km² with intensive culling, so that palatable plant species such as *Anisotome aromatica*, nutrient rich *Chionochloa* spp., and *Schoenus pauciflorus* can respond (Mark 1989; Williams 1975).

4.4.3 Hardwood group

The response of hardwood communities to various levels of ungulate control is very difficult to predict, other than in general terms. Without reductions in deer densities, plots at recently disturbed, more productive sites are likely to be dominated in the future by slow-growing, unpalatable species such as *Pseudowintera axillaris* and *P. colorata* instead of faster-growing species such as *Aristotelia serrata*, *Fuchsia excorticata* and *Meliclytus ramiflorus*. This could have long-term consequences for forest succession, and even more resilient hardwood communities (e.g. dominated by *B. tawa* and *Myrsine salicina*) may become rarer if a fast-growing, seral community is critical for such forest types to develop. Making specific predictions on the outcomes of management intervention will require very intensive monitoring, probably

TABLE 5B. SEEDLING STEM DENSITIES FOR MORE SPECIES FROM PERMANENT PLOT SURVEYS IN MIXED BEECH FOREST THROUGHOUT NEW ZEALAND (SEE HUSHEER & FRAMPTON IN PRESS) COMPARED WITH TARARUA PERMANENT PLOT SURVEY RESULTS FROM 1984/85.

PLOTS	<i>n</i>	<i>Griselinia littoralis</i>	<i>Notbofagus fusca</i>	<i>Notbofagus menziesii</i>	<i>Notbofagus solandri</i>	<i>Raukawa simplex</i>
Arawhata	43	206 ± 92				938 ± 303
Hurunui North	27	798 ± 482				368 ± 217
Hurunui South	25	199 ± 156				221 ± 159
Kaipō	10	0				0
Maungaorangi	33	151 ± 134	0	603 ± 197	6578 ± 1156	452 ± 98
Murchison Mtns	22	0	0	1732 ± 503	16471 ± 2056	176 ± 152
North Kaimanawa	57	223 ± 148	2529 ± 535	2878 ± 537	58 ± 49	213 ± 104
Rangataua	48	58 ± 47	1300 ± 567	7515 ± 1804	3717 ± 1054	380 ± 114
Rotoiti	17	65 ± 44	975 ± 518	2372 ± 646	10593 ± 3766	97 ± 71
South Kaimanawa	35	284 ± 86	0	0	6597 ± 1187	1894 ± 382
Waitutu	107	21 ± 10				299 ± 48
Wakatipu	49	0	7384 ± 2570	2119 ± 595	12592 ± 5378	101 ± 47
West Kaimanawa	46	72 ± 28	745 ± 320	504 ± 226	1081 ± 441	684 ± 303
Tararua	234	128 ± 24	166 ± 53	1292 ± 296	2 ± 2	185 ± 38

(continued from p. 30)

involving the re-measurement of all 96 plots in this group, along with the establishment of a large number of exclosure plots (e.g. 10–20 sites). Although the hardwood group is clearly important, funding limitations may make it necessary to concentrate on simpler vegetation types in the other groups to obtain clear results from monitoring the outcomes of animal pest control.

4.5 MONITORING OUTCOMES OF UNGULATE MANAGEMENT

Allen et al. (2003) identified nine principles for designing a forest biodiversity monitoring system that should be applied when making decisions on outcome monitoring in Tararua Forest Park. These nine principles are outlined below in a Tararua context.

1. Define the goal of monitoring

The goal of an ungulate outcome monitoring system in Tararua Forest Park is to determine whether deer and goat management have a positive benefit for forest regeneration and succession. Specific objectives should be to show the extent to which the process of canopy regeneration of *N. menziesii* and *W. racemosa* forest types are affected by browsing, to show demographic trends in common palatable species listed in Table 1, and to show changes in overall vegetation composition. Objectives could be developed to monitor the effects of herbivory on ecosystem function. This requires ecological study applied at the population, community, and ecosystem levels, using observational and experimental techniques.

Managers and technical support staff need to develop specific outcome objectives for vegetation condition in Tararua Forest Park before specific

monitoring objectives can be set, although broad aims can be identified with current understanding. For example, a broad aim might be 'to ensure maintenance of a *W. racemosa* canopy, where it currently exists', but this could be broken down into more specific objectives, such as threshold levels of recruitment required to maintain minimum tree stem density. If measurable objectives are set and accurately monitored, management activity can then be better targeted and measured in the longer term. Currently, objectives for management are either absent or too general to be useful for designing monitoring programmes. Unfortunately, once managers attempt to define specific outcome targets they may find they have insufficient information on causes and trends of ecosystem health to set meaningful management objectives, or of being able to set meaningful monitoring objectives to fill this gap in knowledge. In the short term, broad management goals could be set (e.g. maintenance of a *W. racemosa* canopy over the next decade) which would allow some optimisation of monitoring, which would in turn provide more specific management goals in the medium term (e.g. > 100 *W. racemosa* saplings/ha, < 1% annual mortality within the next decade).

2. Do not focus on current perceptions

It is important to use a monitoring system that will not only answer the objectives listed above, but also provide more generalised information on trends in forest health. If possible, managers should not only develop specific objectives for current management, but also make some attempt at predicting likely future management objectives, which would allow more generalised but targeted monitoring systems to be developed.

3. Build on the past

Existing data should be treasured, and used where possible. Tararua Forest Park has an infrastructure of permanent monitoring plots second to none. It has a very large sample size and a long period of re-measurements, particularly if re-measured within the next few years. There is tremendous potential to utilise past information, but plots need to be re-measured within the next few years to ensure they are maintained. If it is decided that plots are not to be re-measured soon, that will be in effect a decision to abandon plots. *Careful consideration should be given before any plots are abandoned.*

4. Make sure data are comparable

Several standardised protocols were followed closely in the Tararua surveys. This allows measurements to be repeated using the same protocols. It is notable that a very high standard of data collection, processing and storage has been maintained throughout all re-measurements. On the negative side, protocols have varied between surveys and in many instances parameters cannot be directly compared. Unfortunately, it is likely that financial constraints will dictate rationalisation of the plot system, which will require some permanent plots to be abandoned. The most widely used plot system consists of 20 m × 20 m permanent plots, for which over one thousand have been re-measured in the past decade in New Zealand, of the several thousand for which records exist (Wiser et al. 2001). This system is well-proven, with several dozen studies based on it having been published in peer-reviewed journals in recent years.

Therefore, the Tararua 20 m × 20 m system should be considered a high priority for re-measurement.

5. Recognise the advantage of repeat measurements

The most efficient system in Tararua Forest Park would be to repeat measurements on existing, established plots, and not to establish additional plots.

6. Account for baselines

Although little evidence of change in Tararua vegetation from permanent plots was found in this study, there is a large amount of anecdotal advice suggesting that deer-, possum- and goat-induced modification took place prior to plot establishment. If possible, other techniques such as aerial photographs (Rogers & Leathwick 1997) or recent pollen core sampling (Wilmhurst 2003) should be employed so that more accurate baselines of vegetation can be considered. Although data from four paired exclosure plots are available, the value of data and re-measurements from adjacent forests where replicated 20 m × 20 m exclosure plots are available (i.e. Rimutaka and Aorangi Forest Parks) should not be underestimated.

7. Collect interpretive data

In many instances, monitoring systems that are too specifically targeted overlook important causal mechanisms for changes that may be detected. For example, changes in foliage density in *W. racemosa* might be due to stand development as well as changes in possum browsing (Payton 1989). Without use of permanent quadrat plots, such factors could be overlooked. Allen et al. (2003) suggest that environmental characteristics such as soil fertility should also be measured when plots are established or re-measured. Factors such as local pest abundance and activity, diet, mean temperature, rainfall, soil moisture deficit, nutrient concentrations in soils and foliage, and amounts of light transmitted to the understorey should also be considered if they contribute towards a better understanding of ungulate impacts.

8. Ensure there is a long-term commitment

The best way to ensure that plot systems are useful in the future is to ensure that they are maintained at least once a decade, and re-measured regularly. This will allow managers to demonstrate the effects of management and obtain further funding for culling and monitoring. The Tararua plot system is now approaching a crucial time. Tree tags at many plots with fast-growing trees are likely to have been already overgrown, and seedling sub-plot markers may have disappeared. Plots need to be re-measured within the next 2–3 years to ensure their ongoing viability. For data to be useful in the long-term, quality assurance procedures are essential. Therefore independent audit of re-measurements needs to be undertaken as a standard procedure.

9. Ensure data are not lost

Fortunately, New Zealand now has an excellent databank system (National Vegetation Survey) for permanent plots, the National Vegetation Survey, maintained by Landcare Research with some funding from DOC. This should be used to store Tararua data in the future, as has occurred in the past.

5. Conclusions and recommendations

The existing permanent plot network is of high value, particularly the 20 m × 20 m forest plots, along with any grassland and sub-alpine scrub plots that can be located. This system provides a proven protocol and the sampling strategy used meets the demands of rigorous statistical analysis. It has used a randomised procedure, with plots located at sufficient intervals along lines to ensure that they are statistically independent. Power analysis has shown that for each vegetation grouping there is likely to be sufficient replication to ensure confidence in results.

However, there are some shortcomings in the plot network:

- The most obvious shortcoming is that plots on DOC-managed land have not been maintained or re-measured for two decades. Maintenance of plots (locating and re-marking plots, and re-tagging fast-growing trees) is now urgent, and when undertaken will offer the benefit of giving current data.
- Ideally the Tararua plot system would fit into a national monitoring system. In that way information gained nationally could be used to add statistical and interpretive power to the Tararua dataset (and vice versa). Hopefully, the monitoring component of the Natural Heritage Management System currently under development will overcome this problem.
- More environmental data need to be obtained for individual plot sites. Data on plot locations, canopy cover, light transmitted to the forest floor, soil fertility, and nutrient content of plants can be much more accurately and cheaply obtained now, using modern techniques, than two decades ago. When monitoring is planned, collection of better environmental data should be considered.
- If management is applied to several areas and has a consistent effect compared to paired non-treatment areas, sounder conclusions can be made about its effects. Inferences made from monitoring of unreplicated management procedures are more open to alternative arguments. If possible, plots should be arranged into a series of paired treatment and non-treatment area blocks and a balanced number of plots re-measured in each area. Plots could be selected to represent priority forest types according to specific management objectives. This would only be possible if a sufficient number of comparable treatment and non-treatment areas could be identified.

The recommendations for management made here are based on data that are two decades out of date. Because management of deer is contentious, decision-making needs to be based on more recent data, using a reliable monitoring design. An ideal monitoring programme would include contemporary measurement of sufficient numbers of nationally comparable 20 m × 20 m plots to assess the effects of management on key biodiversity attributes. Plots would be representative of key Tararua forest types, and provide all-embracing information on key threats. Possums and ungulates are key threats to Tararua forests and grasslands. Ideally, replicated possum and ungulate treatments should be used to assess the impacts of these threats. Fenced enclosure plots allow most efficient replication of the effects of ungulates, but replicated possum treatment areas are required to accurately assess the effects of possums, with and without control. Monitoring of the effects of possums and deer should not be considered in isolation, and the recommendations made below should be reconsidered in a context of possum impact monitoring.

Tararua enclosure plots

Establishment of 5–10 enclosure plots (5 m × 5 m to 20 m × 20 m) in each of *W. racemosa*, *N. menziesii*, and grassland habitats is an essential first step to realising the vision of the Wellington Conservancy's habitat monitoring strategy. Because ungulates pose a serious potential threat to Tararua biodiversity, a vegetation plot system monitoring deer impacts should be a priority. Establishment of fifteen 5 × 5 m paired enclosure plots at concurrently measured 20 m × 20 m plots could cost as little as \$20,000. To establish 30 pairs of 20 m × 20 m monitoring plots may cost as much as \$100,000, depending on fence construction design and location.

Tararua permanent plots

There are five main options for re-measurement of the existing plot network in Tararua Forest Park. Each system would ideally include an efficient paired enclosure monitoring plot system. In Kaweka and Kaimanawa *N. solandri* forests, single 10 m × 10 m fenced plots centred within each 20 m × 20 m overstorey plot, has been a successful design for assessing the demographics of canopy dominants (Husheer & Robertson 2005). If knowledge on the effects of less abundant plant species is required (e.g. *Griselinia littoralis*, *F. excorticata*, *W. racemosa* seedlings), a 20 m × 20 m plot size is likely to be needed to ensure that sufficient seedlings are present within plots to provide useful data.

The options are:

- All 350 forest, scrub and grassland plots re-measured, which is likely to cost in the vicinity of \$200,000.
- Re-measurement of all 168 forest and scrub plots established in 1974/75, and any of the 19 alpine grassland plots that can be located. This is likely to cost in the vicinity of \$100,000. Plots could then be re-measured again in one or two decades. This option would provide information on the more complex hardwood group.
- Re-measurement of a random and representative selection of *W. racemosa* and *N. menziesii* forest plots, and all scrub and grassland plots that can be located. Power analysis showed that a minimum sample of 30 plots for each

of the three groups would be optimum. While these plots are being re-measured, deer faecal pellet monitoring along the same transects should be undertaken. This is likely to cost \$50,000.

- A running annual re-measurement of forest plots on 5-10 of the 51 transects established in 1973/74. This is likely to cost between \$20,000 and \$30,000 annually. This option would restrict analysis techniques.
- Permanently abandon all plots. This is likely to be the most expensive option, with a heightened probability of poorly targeted management in the future, followed by a probable need to re-establish a large network of permanent plots when this becomes apparent. Furthermore, the Wellington Conservancy has a responsibility to report on the status of the Land Environment that Tararua Forest Park occupies as part of the Department of Conservation's Natural Heritage Management System. This will be very difficult without a regularly measured permanent plot system. The existing plot network needs to be re-measured within the next few years to ensure plots are properly maintained and not lost.

The third of these options, i.e. re-measurement of 90 permanent plots in Tararua Forest Park (30 each from *W. racemosa*, *N. menziesii*, and grassland groups), together with establishment of five paired 20 m × 20 m exclosures in these three groups, would allow the Wellington Conservancy to make and justify quality decisions on deer and goat management.

Sampling pellet monitoring lines while re-measuring permanent plots is likely to cost only an additional few thousand dollars. The benefits would include a coarse gauge of changes in deer and goat numbers that would be comparable to other areas in New Zealand. For the most efficient use of monitoring resources, selection of plots for re-measurement should also take into account the monitoring needs of possum management.

6. Acknowledgements

I would like to thank the many former New Zealand Forest Service staff who showed great foresight in establishing and re-measuring the Tararua Forest Park plot system. The high quality of fieldwork and data management mean that a very valuable resource has now been inherited. I would also like to thank Steve Urlich for providing the opportunity to produce this report. Steve Urlich, Ian Payton, and Claire Newell provided valuable comments on previous versions of this report.

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Appendix 1

Codes and full names of woody species included in CCA ordination

Sorted in alphabetical order of code.

CODE	SPECIES	CODE	SPECIES
Alspus	<i>Alseuosmia pusilla</i>	Kni exc	<i>Knigbttia excelsa</i>
Ariser	<i>Aristotelia serrata</i>	Leufas	<i>Leucopogon fasciculatus</i>
Beitaw	<i>Beilschmiedia tawa</i>	Mel ram	<i>Meliclytus ramiflorus</i>
Carser	<i>Carpodetus serratus</i>	Melsim	<i>Melicope simplex</i>
Copaus	<i>Coprosma grandifolia</i>	Metdif	<i>Metrosideros diffusa</i>
Copban	<i>Coprosma banksii</i>	Metful	<i>Metrosideros fulgens</i>
Copcil	<i>Coprosma ciliata</i>	Metper	<i>Metrosideros perforata</i>
Copcol	<i>Coprosma colensoi</i>	Metrob	<i>Metrosideros robusta</i>
Copdep	<i>Coprosma depressa</i>	Myraus	<i>Myrsine australis</i>
Copfoe	<i>Coprosma foetidissima</i>	Myrdiv	<i>Myrsine divaricata</i>
Coplin	<i>Coprosma linariifolia</i>	Myrped	<i>Myriophyllum pedunculatum</i>
Copluc	<i>Coprosma lucida</i>	Myrsal	<i>Myrsine salicina</i>
Copmic	<i>Coprosma microcarpa</i>	Neoano	<i>Raukaua anomalus</i>
Coppar	<i>Coprosma parviflora</i>	Neocol	<i>Pseudopanax colensoi</i>
Coppse	<i>Coprosma pseudocuneata</i>	Neoped	<i>Neomyrtus pedunculatus</i>
Coprha	<i>Coprosma rhamnoides</i>	Notfus	<i>Notbofagus fusca</i>
Coprob	<i>Coprosma robusta</i>	Notmen	<i>Notbofagus menziesii</i>
Coprot	<i>Coprosma rotundifolia</i>	Olearb	<i>Olearia arborescens</i>
Coprug	<i>Coprosma rugosa</i>	Olecol	<i>Olearia colensoi</i>
Copsma	<i>Coprosma spp.</i>	Oleili	<i>Olearia ilicifolia</i>
Coptef	<i>Coprosma tenuifolia</i>	Olelac	<i>Olearia lacunosa</i>
Cyasm	<i>Cyathea smithii</i>	Oleran	<i>Olearia rani</i>
Cytfas	<i>Leucopogon fasciculatus</i>	Phyalp	<i>Phyllocladus alpinus</i>
Cytjun	<i>Cyatbodes juniperina</i>	Pitdiv	<i>Pittosporum divaricatum</i>
Daccup	<i>Dacrydium cupressinum</i>	Piteug	<i>Pittosporum eugenioides</i>
Dicsqu	<i>Dicksonia squarrosa</i>	Podhal	<i>Podocarpus hallii</i>
Dracop	<i>Dracophyllum</i>	Podspi	<i>Podocarpus spicatus</i>
Drafil	<i>Dracophyllum filifolium</i>	Podtot	<i>Podocarpus totara</i>
Dralon	<i>Dracophyllum longifolium</i>	Prufer	<i>Prumnopitys ferruginea</i>
Drarec	<i>Dracophyllum recurvum</i>	Prutax	<i>Prumnopitys taxifolia</i>
Eladen	<i>Elaeocarpus dentatus</i>	Psearb	<i>Pseudopanax arboreus</i>
Elahoo	<i>Elaeocarpus bookerianus</i>	Pseaxi	<i>Pseudowintera axillaris</i>
Fucexc	<i>Fuchsia excorticata</i>	Psecol	<i>Pseudowintera colorata</i>
Gauant	<i>Gaultheria antipoda</i>	Psecra	<i>Pseudopanax crassifolius</i>
Gaudep	<i>Gaultheria depressa</i>	Pseedg	<i>Raukaua edgerleyi</i>
Genlig	<i>Geniostoma</i>	Psesim	<i>Raukaua simplex</i>
Grilit	<i>Griselinia littoralis</i>	Ripsca	<i>Ripogonum scandens</i>
Hebeve	<i>Hebe evenosa</i>	Rubcis	<i>Rubus cissoides</i>
Hebsal	<i>Hebe salicina</i>	Rubsch	<i>Rubus schmidelioides</i>
Hebstr	<i>Hebe stricta</i>	Schdig	<i>Schefflera digitata</i>
Hedarb	<i>Hedycarya arborea</i>	Weirac	<i>Weinmannia racemosa</i>