

Distribution and demographics of
Fuchsia excorticata
Nelson Lakes National Park

By

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Abstract

The small tree *Fuchsia excorticata* (Onagraceae) is found on disturbed riparian sites throughout New Zealand. Spatial distribution and demographic changes in populations of *F. excorticata* were surveyed in the Rotoiti Nature Recovery Project (RNRP) area, and an associated non-treatment reference area, Nelson Lakes National Park, over the period 2005 to 2007. The RNRP is an intensively managed “mainland island” project in mixed beech (*Nothofagus*) forest, pest animal species especially possums (*Trichosurus vulpecula*), rodents, and mustelids are controlled. The aim of this study was to survey the condition of *F. excorticata* in the managed and reference areas to determine if any differences could be detected between the populations over time. The only significant ($p < 0.05$) difference between the two areas was in sapling abundance, which was higher in the RNRP management area. This was tentatively attributed to pest management; however the concurrence of complex environmental influences, and a lack of management replication, prevented a simple interpretation of the data. There was no significant difference in recruitment, growth, mortality, and tree or seedling abundance between the managed and reference areas. Growth rate, especially of saplings, was significantly greater at lower elevations in the RNRP managed area, however the inverse held for the reference area. Mammalian herbivory was light; the main affect being to restrict seedling growth through hedging. Periodic heavy flooding was responsible for much of the mortality, however many newly disturbed sites were rapidly populated with fuchsia seedlings. The need for rigour in establishing ecologically comparable reference sites for comparative studies was evident. There were inherent difficulties in establishing causal relationships between ecological changes and management practices where management was not replicated geographically.

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Table of Contents

Abstract	i
Acknowledgements	ii
List of tables	v
List of figures	vi
CHAPTER 1 INTRODUCTION AND BACKGROUND	1
Fuchsia excorticata	1
Distribution and ecology	1
Reproductive biology	1
Herbivory	2
Ecological concepts and contexts	4
Riparian ecology	4
Study Area	8
Geography and geology	8
History of land use	10
Vegetation.....	10
Fauna	11
Climate	11
Mainland Islands	12
Rotoiti Nature Recovery Project	14
Overview.....	14
Monitoring and reference sites	14
Department of conservation research permit.....	15
Possum control	15
Rodent control	16
Mustelid control.....	17
Ungulate control.....	17
Anticipated affects resulting from pest management	17
Review of riparian monitoring methodology.....	19
Aims and objectives of the thesis	20
Aims:.....	20
Objectives:.....	21
CHAPTER 2 METHODS	22
Study area	22
Research methodology	23
Survey and monitoring of the natural riparian fuchsia population	26
Experimental design	26
Riparian complexes within the study area	26
Transects – modified greenline method	27
Search strategy.....	29

Mapping of plant locations	31
Biometric measurements	32
Geomorphic and environmental measurements	33
Data reduction	35
Statistical methods.....	35
Germination and seedling survivorship experiment	37
Experimental design synopsis.....	37
Trialling - difficulties and solutions.....	38
Quadrat establishment	38
Seed preparation	40
Seed viability trial.....	41
Seedling herbivory	41
CHAPTER 3 RESULTS.....	42
Survey and monitoring of the natural riparian Fuchsia population	42
Overview.....	42
Abundance and density.....	44
Mean height.....	47
Growth	49
Recruitment	55
Mortality	55
Browse on fuchsia.....	56
Geomorphic and environmental measurements	57
Germination and seedling survivorship experiment	62
Field germination	62
Seed viability trial.....	63
CHAPTER 4 DISCUSSION AND CONCLUSIONS.....	64
Introduction.....	64
Distribution patterns related to management practices.....	65
Abundance.....	65
Distribution patterns related to environmental factors	70
Elevation related patterns	70
Recruitment, growth and mortality	74
Conclusions	76
Critique of methodology, and recommendations	78
Monitoring of patchy riparian populations	78
Timeframes for monitoring temporal change in populations	79
Deer browse – use of riparian exclosure plots.....	79
Selection of project specific indicator species	79
Spatial replication, and the need for ecologically robust reference areas	82
Bibliography.....	83

List of tables

Table 1 Climate data summary. From Land Environments of New Zealand (Leathwick et al, 2003)	11
Table 2 Possum management data 1997 - 2006.	16
Table 3 Rat tracking percentages.	16
Table 4 The number of transects in each stratum	26
Table 5 Riparian Complexes associated with elevation based stratification	27
Table 6 Summary of stream and transect information	30
Table 7 height tier definitions	32
Table 8 Criteria for scoring mammalian browse.	33
Table 9 Summary of quadrat locations	38
Table 10 mean height (millimetres) of Fuchsia excorticata plants in 2005, 2006 and 2007.	49
Table 11 numbers of fuchsia used to monitor growth 2005-2007	50
Table 12 Growth recorded as millimetres change over two years (2005 - 2007).....	52
Table 13 Recruitment data 2005 and 2007, the actual number of new seedlings observed	55
Table 14 Mortality data 2005 to 2006, the actual number of plants observed in 2005 but dead or missing by 2007.	56
Table 15 Mean elevation and profile information for transects surveyed in each stream bed.	57
Table 16 Geomorphic descriptions of transects	58
Table 17. Canopy cover over transects in the two geographic areas.....	58
Table 18 Field germination quadrat result summary	62

List of figures

Figure 1 Study area: Nelson Lakes National Park, South Island, New Zealand.....	9
Figure 2 Rainfall data for St Arnaud 2001 to 2006	12
Figure 3 Lake Head non-treatment reference site, Nelson Lakes National Park	24
Figure 4 Mainland Island - RNRP pest-management site, Nelson Lakes National Park	25
Figure 5. An example of the "Greyline" use to define riparian margins	29
Figure 6 parallel tagging of small seedlings	
Figure 7 tagging using flagging tape	31
Figure 8 horizontal habit of a sapling following flooding	32
Figure 9 browse hedging of seedlings	33
Figure 10 Exclosure cage in situ	40
Figure 11 Temporary plot quadrat marker.....	40
Figure 12 Fuchsia seed - 1 mm grid scale.	41
Figure 13 Total number of <i>Fuchsia excorticata</i> found in each treatment area.	43
Figure 14 Total number of <i>Fuchsia excorticata</i> found in each elevation zone.	43
Figure 15 Mean abundance per 100 m transect in each management area between 2005 and 2007.....	44
Figure 16 log(n+1) transformed seedling abundance for different management areas	46
Figure 17 log (n+1) transformed sapling abundance for different management areas (difference is significant ($p < 0.05$)).	46
Figure 18 log (n+1) transformed tree abundance for different management areas.....	47
Figure 19 Mean height of fuchsia at different elevations in different management areas. Error bars are ± 1 SE.	48
Figure 20 Overall fuchsia growth rate from 2005 - 2007 (mm). Error bars are ± 1 SE.	51

Figure 21 Mean sapling growth between 2005 and 2007 (mm). Error bars are ± 1 SE.....	53
Figure 22 Mean seedling growth between 2005 and 2007 (mm)	54
Figure 23 Mean tree growth between 2005 and 2007 (mm)	54
Figure 24 Defoliation of accessible lower branches suggestive of deer browse. The time of observation (late November) was not consistent with deciduous leaf fall.	56
Figure 25 Heavy deer browse and dieback on a fuchsia tree. The upper branches were accessible to deer using the bank at the rear.	57
Figure 26 Temperature and humidity graph for northern and southern aspects of stream A – recorded at c. 650 m a.s.l. (Lake Head)	60
Figure 27 Temperature and humidity graph for upper and lower elevations	61
Figure 28 seedling of <i>Fuchsia excorticata</i> in lower "PG" stream.....	62
Figure 29 viability trials for germination success of <i>Fuchsia</i> seeds used in the field trials 2007	63
Figure 30 Burial of riparian habitat by scree movement during flooding	73
Figure 31 Stream bank erosion in April 2005	73

Chapter 1 Introduction and background

Fuchsia excorticata

This study focuses on the small tree *Fuchsia excorticata* (J. R. et G. Forst.), Onagraceae (Allan, 1961). Known commonly as kotukutuku or tree-fuchsia, *F. excorticata* is the largest of four New Zealand species of fuchsia, growing up to 12 m though no trees of this stature were found in the study area. It has been found to be palatable to both possums and deer and been the subject of investigation in relation to mammalian browse (Bellingham et al., 1999; Ulrich and Brady, 2005).

Distribution and ecology

F. excorticata is an endemic deciduous tree common throughout the North, South, Stewart and Auckland islands of New Zealand (Allan, 1961; Webb et al., 1988), where it is common in lowland and lower montane forest. It is frequently found on forest margins, in clearings and by streams (Webb et al., 1988). Increase in *F. excorticata* abundance often follows disturbance (Allen and Allan, 1997; Payton et al., 1999). In these forests fuchsia is a secondary coloniser largely limited to riparian margins of side-streams, lakes and tracks. This study was limited to the riparian ecology of *F. excorticata* along streams on talus slope beech forest.

Reproductive biology

Fuchsia excorticata is gynodioecious, thus potentially isolating female plants from pollination and reducing fecundity. The sex of individual trees can be distinguished by the size of flowers and relative length of stamens.

Hermaphrodite flowers are generally larger with long stamens; female flowers have reduced stamens (Sessions, 2001). According to Allan (1961) flowering occurs from June to January, however Leathwick (1984) recorded flowering from spring (September) through autumn (May), and fruiting from December through to late May. In Nelson Lakes flowering and fruiting have been

observed from October through to late May (*pers.obs.*). The later flowering and longer season are probably related to latitude and elevation.

The flowers are bird-pollinated and the fruit bird-dispersed; blackbirds, silvereyes, bellbirds, kereru, starlings have all been seen to feed on fuchsia fruit (Burrows, 1995). The effectiveness of birds as dispersers of fuchsia seeds is exemplified by an observation by Beveridge who saw 178 Fuchsia seedlings germinate from a single blackbird dropping (Beveridge, 1964). Fuchsia fruit is a 10 mm long berry that is dark purple to almost black when ripe. Fruit set was reported between September and February in the Karori Wildlife Sanctuary, Wellington (Moles, 1997), and each fruit contains about 288 ± 85.4 seeds/fruit (Burrows, 1995). Possums are known to eat fuchsia fruit. While typically the seeds are physically undamaged after passing through possum gut, it is not known whether they are still viable (Burrows, 1995). In laboratory trials with regular watering, ambient lighting and no soil, fuchsia seeds began germination in 11-12 days and had reached 96-100% germination within 32 days. They were found to have lower germination success in dark (86%) and in soil (52%) (Burrows, 1995)

Moles (1997) found fuchsia was absent in the seed-rain at Karori despite being well-represented as large adult trees, but it was represented in the seed-bank. The majority of seeds were found to reach the ground under or near the parent tree (Burrows, 1995).

Herbivory

In many locations in New Zealand fuchsia has sustained heavy browsing by introduced brushtail possums and deer and is in population decline (Bellingham et al., 1999; Ulrich and Brady, 2005). The population density of possums varies among different forest types, as does their preferred diet. Fuchsia is a preferentially palatable species in most forest types and consequently is commonly used as a browse indicator (Payton et al., 1999).

It needs to be noted however that there has been considerable variation in the impact of possum browse in different New Zealand locations. Sweetapple and Nugent (1999) found no significant differences in palatability due to fuchsia provenance, and that all provenances tested were palatable. An alternative explanation to account for the variation in impact suggests diet may relate to the presence of other more palatable species, rendering fuchsia relatively less attractive. In the study area broadleaf (*Griselinia littoralis*) and wineberry (*Aristotelia serrata*), both palatable to deer and possums, are found in the streambeds and are subject to mammalian browse (*pers. obs.*).

The long-term impact of possum browsing on fuchsia is well documented. In the DSIR study of vascular flora in the Orongorongo valley near Wellington, New Zealand it was found that "...once common at all altitudes in regenerating forest, especially on slip faces. *Fuchsia* was dying in the study area from 1935 onwards...." (Campbell, 1984). Of 246 fuchsia trees recorded along the margin of a slip face in 1946-47, 5 were unbrowsed, 182 browsed, and 59 dead, by 1984 there was little evidence of any fuchsia. It was noted that while fuchsia was an important component of possum diet in 1946-47, by 1966, it was "entirely absent from the area where Mason's possums had been collected" (Campbell, 1984). This was supported by (Brockie, 1992) stating that in 1946 fuchsia accounted for 32% of possum diet but by 1973 had nearly disappeared. Additionally, it was noted that "Fuchsia seedlings seldom grow more than a metre in height before possums defoliate or smash them..." (Brockie, 1992).

Possums were eradicated from Kapiti Island, New Zealand in 1986 and a longitudinal study of the vegetation response was monitored for 10 years from the start of intensive control (Atkinson, 1992). Twenty Fuchsia trees on the summit ridge marked for browse monitoring in February 1976 had all died by October the same year. A general survey of the island showed only two sites where any fuchsia survived and there were only a few isolated saplings. From 1982 onward the number of seedlings and saplings steadily increased as the density of possums declined.

Ecological concepts and contexts

The concept of community dynamics has undergone considerable revision since the term succession was used by Clements in 1916 (Barbour et al., 1980). Being subject to stochastic flood and avalanche events, changes in high elevation riparian communities are more likely to be allogenic as defined by Tansley (1935) rather than autogenic (Barbour et al., 1980). Following their review of succession Drury and Nisbet (1973) suggested the relay floristic model as a general theory of succession that is relevant to this study. According to the Drury and Nisbet model, initial floristic composition, differential growth rates, survival rates, and colonising ability influence the direction of community dynamics. Applied to a single colonising species such as fuchsia this suggests a need to monitor germination and growth rates, as well as adult survival. More recently the relay floristic model was successfully used to explain successional changes to forest vegetation on landslides in the Fiord Ecological Region, South Western New Zealand, over a 102 year period (Mark et al., 1991).

Riparian ecology

The complexity of environmental influences in riparian habitats is a common focus in many community and population level studies. Naiman and Decamps (1997) discuss the species diversity response to a 'dynamic environment' suggesting that the unusual species diversity they observed was related to variable flood regimes, channel processes, and elevation related climatic differences along fluvial corridors. In a study of invasion by alien plants along fluvial corridors in Central Europe Pysek and Prach (1993) establish a link between species autecology and the frequency of habitat availability, a concept that could be applied to fuchsia colonising riparian margins.

Plant population dynamics in riparian habitats is complex. Major themes occur around flood disturbance regimes in large river systems (Bendix, 1997; Casanova and Brock, 2000; Chapin et al., 2002; Jaekelaeniemi et al., 2005;

Naiman and Decamps, 1997; Naiman et al., 1998; Scott et al., 1997), macro and micro stream geomorphology (Baker, 1989; Bendix and Hupp, 2000; Hupp, 1982; Nagamatsu et al., 2002; Nakamura et al., 1997; Sakai et al., 1999; Scott et al., 1997), and edaphic, seedbank, moisture and light conditions (Casanova and Brock, 2000; Goodson et al., 2002; MacDougall and Kellman, 1992; Parciak, 2002).

Disturbance and flood regimes

In relation to disturbance ecology Jaekelaeniemi et al. (2005) applied meta-population models to localised colonisation and extinctions of *Silene tatarica* in a flood disturbed riparian habitat over a five year study. They found recolonisation of extinct patches was rare and that small patches were at most risk of extinction, though floods could also destroy large populations. They conclude that *S. tatarica* distribution did not behave as predicted for meta-populations at equilibrium; instead the distribution was related to the rate of habitat change. River dynamics both create colonising habitat and cause local extinctions. This paradox was also noted during this study following severe flooding at Nelson Lakes in April 2005.

Stream geomorphology

Two studies on the impact of flood severity and frequency suggest that establishment of riparian tree species is strongly influenced by both. Scott (1997) studied the effect of flow variability on plains cottonwood (*Populus deltoides* subsp. *monilifera*) on the Missouri riverbank. Elevation of germination sites 1.2m above the lower limit of perennial vegetation was seen as being needed for long term survival of seedlings. In a study of riparian species composition in Northern Japan, Nakamura (1997) considered elevation above river bed and the distance from river channel were key determinants in development of the substratum. The soil moisture and organic content increased further from the river channel, while soil particle size diminished, and this was reflected in the relative dominance of different species. Although the study by Nakamura was of a flood plain, similar trends might be expected in other riparian habitats. Nakamura also

determined that frequency of flooding events, leading to a consequent difference in site stability, was a key factor in the dominance of pioneer species. Thus channel and valley characteristics interacted spatially and temporally and this interaction influenced vegetation patterns.

The influence of elevation above river level is also mentioned by Sakai (1999). From an investigation of a riparian forest in Japan the population structure of *Quercus ulmus* was seen to vary with height above flood line. Similar age cohorts were correlated to historical flood levels (Sakai et al., 1999). Sakio (1997) found similar patterns with riparian forest in response to seismically induced land slides. Where stream channels were stable even aged cohorts of *Fraxinus platypoda* were found, the converse was found in areas subject to high frequency flooding (Sakio, 1997). Thus the interaction between large scale seismic disturbance and flood frequency was reflected in local population age structure. The influence of micro-topography on establishment of deciduous trees was the focus of a study in Japan. Ground surface stability, soil moisture, canopy condition and light gaps were correlated to seedling emergence and survivorship. Although no clear general trends, based on spatial distribution, were reported it was noted that seedling survival was species related. The diversity of habitat due to the interaction of environmental parameters was seen as influential in species specific seedling establishment (Sakio, 1997).

Microsites - Edaphic, seedbank, moisture and light conditions

The importance of microsite variation as a determinant of germination success and seedling survivorship was highlighted in two recent studies (Battaglia and Reid, 1993) and (Gray and Spies, 1997). In their experiment Gray and Spies mimicked the effects of logs, different substrate, and understory shade on establishment of seedlings in artificially created gaps of various sizes in an experimental *Pseudotsuga*, *Tsuga* coniferous forest. The results were then compared with survivorship in natural regeneration. Little difference was observed between seedlings on logs compared to forest floor in natural conditions. Understory shade inhibited or facilitated seedling establishment depending on gap size and proportion. Shade provided by

large logs facilitated establishment of *Tsuga*, but the localised importance of sun-fleck was noted. Although the authors indicated that most mortality was due to desiccation, no reference was made in the study to herbivory or any other cause of seedling mortality.

Battaglia and Reid studied the effect of microsite variation on seedling survival rates of *Eucalyptus delegatensis* in both field and glasshouse. Their finding supported the concept of “safe sites” and emphasised that a good site for germination was not necessarily a good site for seedling survival.

Three species-related factors influencing colonization and establishment along riparian corridors are seed size, dormancy and recalcitrance (Casanova and Brock, 2000; Parciak, 2002). Cassonava et al (2000) and Parciak (2002) both stress the issue of desiccation in areas subject to periodic drought as well as flooding is important to survivorship. Desiccation was seen by Parciak to be the predominant cause of seedling mortality especially with small-seeded plants in xeric slope habitats; and fuchsia is a small seeded species. Though large seeds were more likely to be predated than small, the effect of this was offset by the higher growth rate of their seedlings and lower susceptibility to drought. Parciak does however suggest that in riparian habitats plants producing small seeds that are widely dispersed may be at an advantage over those with large seeds, with higher germination rates away from canopies.

Germination rates in colonising species

Germination of *Fuchsia excorticata* seed has been studied under laboratory conditions by (Burrows, 1995). In that investigation Fuchsia seed started germination in 11-12 days and was complete by 29-35 days. Fruit had been collected fresh and seed extracted, cleaned and checked visually (fullness) before sowing in various treatments. Some seed was left in intact fruit and planted to simulate fruit drop. Treatments used either filter paper and tap water (standard) or a soil mix overlaid with vermiculite to maintain surface moisture, were placed in a partially shaded glasshouse with automatic watering but no specific temperature management. Four replicates of 25

seeds were used in each treatment (standard, dark, and soil). Germination rates of 96% and 52% were recorded in standard and soil conditions (Burrows, 1995). Horticultural sources suggested that fuchsia seed be sown fresh or moist-cold stratified for 28 days at 2-5°C (Annon, 2005).

The importance of germination in determining distribution of a colonising species was investigated by Bundard et al. (1997) in a study of the invasive species *Clematis vitalba*. They concluded that “rapid establishment may account for the distribution of *C. vitalba* on recently disturbed, high light sites” (Bundard et al., 1997). The generalisation of this view is shared by Drury and Nisbet (1973) in proposing the influence of differential growth rates on Successional species composition.

In a study of another invasive species *Berberis dawsonii*, McAlpine (2005) compared longevity and germination success of seed stratified *in situ* under intact forest canopy. This was done using replicates of 25 seeds “planted” in nylon mesh bags buried in soil at 44 sites. Seed bags were removed sequentially and the contents planted in laboratory conditions to compare germination rates. Further investigation of *in situ* germination was carried out in 15 replicate plots using 4 random 20x20 cm cells in each, and 50 seeds sown per cell. Sown in November 2001 seedlings were counted after the peak germination period in November 2002 and 2003 to calculate percent germination (McAlpine, 2005). McAlpine found higher germination rates (of the five species tested) in higher light environments and soil moisture, but no differences associated with litter depth.

Study Area

Geography and geology

The study area was located in Nelson Lakes National Park, at the northern end of the Southern Alps, South Island, New Zealand (see Figure 1). Study sites are situated at two centres; Rotoiti Nature Recovery Area centred at 41°49'S 172°51'E and Lake Head at 41°52'S 172°49'E. The main St Arnaud ridge axis runs in an approximately north-south direction (20°^{true}), and the study streams flow from its western flanks into Lake Rotoiti in the north and

the Travers River in the south. There is an elevation gradient from the ridgeline at c. 1700 m a.s.l. to the lake edge at 620 m a.s.l. All subsequent grid references follow the 7-digit New Zealand Map Grid, Geodic Datum 1949, and elevations are given as metres above mean sea level (m a.s.l.).

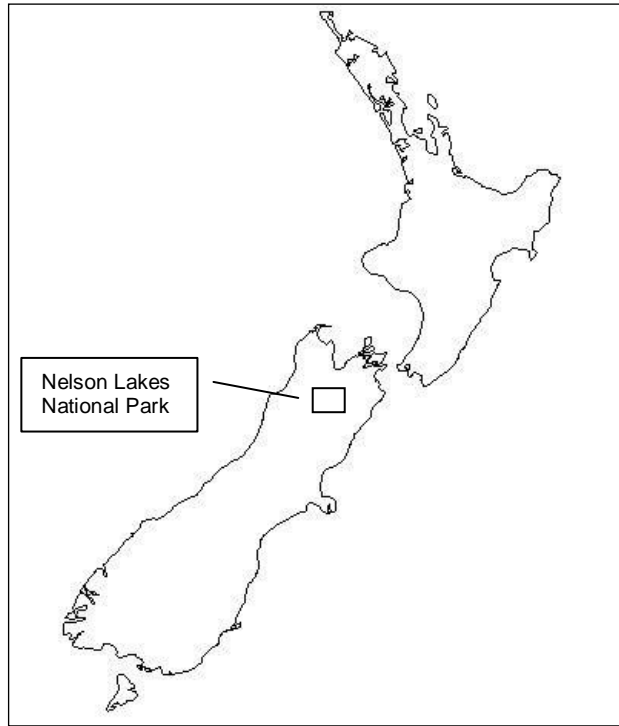


Figure 1 Study area: Nelson Lakes National Park, South Island, New Zealand.

From the ridge top to 800-900 m the talus slopes are moderately steep (c. 30-40°), at this elevation there is an abrupt change to foot slopes and shingle fans (c. 10-20°). At higher elevations stream courses and waterfalls steeply incise these slopes into gorges and ravines. *Fuchsia excorticata* occupies a band between about 1100m, the upper limit of their elevation range, and the lake at 620m.

Parent material for the range is Torlesse greywacke of Triassic or lower Jurassic origin (Suggate R.P et al., 1978). The Travers Valley and Lake Rotoiti were extensively glaciated during the Otira (last) Glacial period. Since 12,000 yr B.P. (^{14}C date) the glaciers have receded (McLea W.L., 1996) leaving a terminal moraine at the northern end of Lake Rotoiti, and glacial outwash terraces and Holocene alluvium in the Travers Valley.

Soils in the study area have formed from recent fluvial deposits or are low-fertility skeletal soils derived from parent rock. At the riparian sample sites exposed soil profiles are zonal Southern and Central Podzols (Gibbs H.S., 1980). They are acidic and vary in depth and proportion of gravels, sand, silt, clay and humus depending on elevation, drainage and slope; better soils are found on the lower slopes. These soils are very slow forming especially at higher elevations. (Gibbs H.S., 1980)

History of land use

The area was used for grazing from 1848 until 1956 when the park was formed (Crouch E., 1981). Since the formation of the National Park it was managed first by the Department of Lands and Survey then from 1985 by the Department of Conservation; both institutions have a conservation and recreational focus.

Vegetation

The dominant vegetation type in montane areas of the Park is mixed beech (Nothofagus). From the upper timberline at about 1350 m a.s.l. to about 1140 m mountain beech (*N. solandri* var *cliffortioides*) forms a nearly pure stand, from this elevation to the lake edge there is mixed silver beech (*N. menziesii*) and red beech (*N. fusca*), with red beech becoming more dominant at lower elevations (Wardle J., 1984). This vegetation has persisted since 8,000 ¹⁴C yr B.P., after the Otira Glacial (Moar N.T., 1971). Beech forest provides canopy cover up to the margins of all streams. During a pilot reconnaissance by the author in 2000, *Fuchsia excorticata* was predominantly found along stream banks on the lower slopes between 600 and 900 m. Isolated plants were found in the upper gorges from 900 to near the headwaters at the timberline at about 1300 m.

Fauna

A range of birds are present in the Park. Of special interest as potential pollinators and dispersers for fuchsia are bellbirds (*Anthornis melanura*), tui (*Prosthemadera novaeseelandiae*) and silvereyes (*Zosterops lateralis*). These species were monitored by RNRP staff using regular 5-minute bird counts (Dawson and Bull, 1975); data trends are presented later.

Climate

The climate in Nelson Lakes National Park is grouped under the central mountains of the South Island Land Environment (Leathwick J. et al., 2003) Data for temperature and solar radiation are summarised in Table 1.

Climatic factor	Mean annual value
Mean annual temperature	7.5°C
Mean winter temperature	-1.6°C
Annual solar radiation	14.2 MJ/m ² /day
Winter solar radiation	4.6 MJ/m ² /day
Annual water deficit	0.9 mm

Table 1 Climate data summary. From Land Environments of New Zealand (Leathwick et al, 2003)

Snowfall on the study area down to lake level is common during winter months (pers. obs.).

Rainfall data from between 2001 and 2006 have been graphed (figure 2). Data was obtained from records in the St Arnaud Area Office, Department of Conservation. The periodic heavy rainfall events (indicated by red circles on figure 2) were seen to cause localised damage to stream beds and banks in the study area.

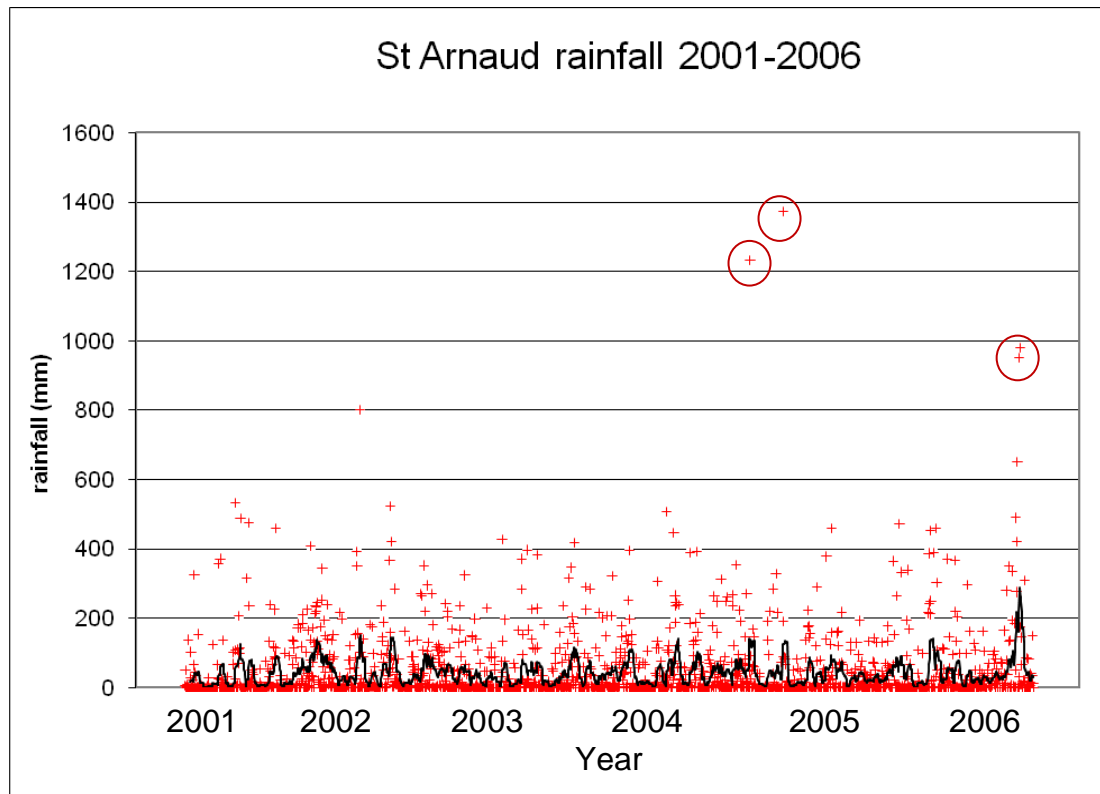


Figure 2 Rainfall data for St Arnaud 2001 to 2006
Flood events of 2005 and 2006 have been circled, the black line shows the moving average.

Mainland Islands

One focus of this study is the use of *Fuchsia excorticata* population monitoring as a tool to measure the ecological impact of a mammalian pest management strategy in montane beech (*Nothofagus*) forest, Nelson Lakes National Park, New Zealand. The Department of Conservation Rotoiti Nature Recovery Programme, one of six Department administered programmes referred to as “Mainland Islands”, is located at the northern end of Nelson Lakes National Park and is the centre of this study.

In 1995-96 the New Zealand Department of Conservation initiated a programme of intensive pest management in six largely forested sites totalling 11,500 hectares with 8,000 hectares set aside as monitoring reference areas (Saunders A., 2000). These projects were influenced by the success of the pest management strategy used in the Marpara Wildlife Management Reserve in the northern King Country, New Zealand. With the aim of arresting the decline in the North Island kokako (*Callaeas cinerea*

wilsoni) population, intensive control of possums, rodents, mustelids and ungulates was undertaken in the late 1980s. The associated research programme targeted both the effectiveness of controls by monitoring pest populations, and the outcome of the pest reduction in terms of recovery of the kokako population. One of the factors contributing to the success of the management strategy was the ecological isolation of the Marpara Reserve. Centred in extensive farmland, the Marpara forest fragment had similar insular properties to the off-shore islands used in species recovery programmes in that reinvasion by pests was more restricted. Though not as effective as the marine “moat” around off-shore islands, a buffer zone of pest management around the reserve was able to significantly reduce the sink effect of migrating pests reinvading the areas depleted through management. In 1990 the concept of “island management mainland style” was coined (Saunders A., 2000), and later abbreviated to “Mainland Islands” when applied to similar styles of management. Saunders (2000) suggests the term “Mainland Restoration” would better reflect the nature of the areas currently being managed, there not being any obvious natural boundary around many. Though this is the case at the site of this study the author will use the term “Mainland Island” as it has since become synonymous with the style of management and, in particular, the projects themselves. The Mainland Island concept has widened to embrace ecosystem restoration as well as single threatened species recovery. The six projects are representative of a range of forest types throughout New Zealand, the Nelson Lakes site being mixed beech forest. In a review of the Mainland Island programme in 2000, Saunders suggested that “any strategic approach to effectively conserving representative New Zealand ecosystems should involve experimental management at sites...”

Rotoiti Nature Recovery Project

Overview

In 1996-97 the Rotoiti Nature Recovery Programme (RNRP), Department of Conservation set up mammalian pest management strategies for the Mainland Island area (NZ grid 5933330N 2497400E) and a reference area on the western slopes of the Travers Valley near Lake Head Hut (NZ grid 5925705N 2495280E). From 1998 the mainland island area was managed to control possums (*Trichosurus vulpecula*), ship rat (*Rattus rattus*), mice (*Mus musculus*), mustelids including stoats (*Mustela erminea*), ferrets (*Mustela furo*), and weasels (*Mustela nivalis*), feral cats, and to a limited extent ungulates (Butler D., 2000). A bait station grid targeting possums and rodents was set up throughout the Mainland Island area. On the talus slopes the gridlines are separated by 100 m and follow a north-south compass bearing approximating the lake shoreline. In the upper elevations the grid spacing increases to 150 m and follows elevation contours. Predator traplines targeting mustelids were set up along all stream courses, invasion fronts along farm and lake margins, and the summit ridge of the St Arnaud Range extending about a kilometre beyond the ends of the Mainland Island itself. The Lake Head reference area, set up to encompass the three southern-most streams that drain into the Lake, was free from any animal management until 1999-2000 when limited mustelid control was introduced along the Travers Valley (Butler D., 2000).

Monitoring and reference sites

The Lake Head reference area, set up to encompass the three southern-most streams that drain into the Lake, was free from any animal management until 1999-2000 when limited mustelid control was introduced along the Travers Valley (Butler D., 2000). A second reference area was established at the southern end of Lake Rotoroa. Both “non-treatment” reference areas were subject to scrutiny by (Brown R. and Norton D.A., 2001), who suggested the vegetation was not comparable to that in the managed area, especially the Rotoroa site, and that further consideration of the reference site locations was warranted. A view shared by the author.

Ecological outcomes of the RNRP pest management were monitored focusing on birds, invertebrates, and vegetation. Bird monitoring followed the standard five-minute bird count protocol (Dawson and Bull, 1975), in addition, single species monitoring of kaka (*Nestor meridionalis*), robins (*Petroica australis*) and kea (*Nestor notabilis*) has been done. Vegetation monitoring was mainly done using a series of 20x20 m vegetation plots in forested areas. None of the vegetation plots encompassed riparian vegetation. Monitoring was carried out in the managed and the reference areas.

Department of conservation research permit

Because the study site was within the National Park a “Low impact, research and collection permit” was obtained to facilitate collection of fruit and planting of seeds. **National Permit Number: NM-17587-RES**

Possum control

Possums were managed using Philproof bait stations spaced at 100 m intervals along the gridlines. In October 1997 bait stations were loaded with Waimate RS 5 cereal pellets containing 1080 toxin at 0.15% w/w/ following pre-feeding with non-toxic bait. From December 1997 the 1080 was replaced with Talon 20WP® containing the anticoagulant brodifacoum for the maintenance programme. From 2000 a cyanide operation using Feratox® was set up along the farm margin and trapping extended to a buffer zone south of the Mainland Island area.

Possum abundance was monitored using the residual trap catch RTC method (National Possum Control Agencies, 2004). This provides an index of abundance rather than absolute abundance data and is used extensively to monitor the effectiveness of possum control programmes in New Zealand. An operational target of 5% RTC (residual trap catch) was set. Autopsy of possums indicated that *Fuchsia excorticata* was among the plants eaten (Butler D., 2003). The 2004 to 2006 RTC figures were below 2% in both the

managed and reference areas, indicating extremely low possum densities during the study period (see table 2).

Year of monitoring	Mainland Island Pest management area	Lake Head Reference area
1997 RTC%	6.5 (0.01)	8.0 (0.03)
1998 RTC%	1.0 (1.1)	8.0 (2.8)
1999 RTC%	1.0 (0.8)	4.7 (1.4)
2000 RTC%	1.0 (0.9)	5.3 (2.8)
2001 RTC%	1.3 (0.8)	6.8 (5.8)
2004-2005 % kill traps	0.2	0.92
2005-2006 % kill traps	0.5	1.75

Table 2 Possum management data 1997 - 2006.

Percentage residual trap catch figures (figures in parentheses indicate ± 1 SE). Condensed from RNRP annual reports 2001, 2005, 2007

Rodent control

The design of Philproof bait stations leaves them open to rodents as well as possums and the bait is toxic to both rats and mice. Rodent abundance was monitored by the use of tracking tunnels and snap traps using standard protocols. Rodent numbers fluctuate widely in beech forest in response to periodic mast flowering of the beech (Wardle J., 1984). This trend was observed in the RNRP data too. It is not known whether these rodents consume fuchsia seeds or fruit under natural conditions.

Date	Mainland Island	Lake Head
1999 February	4	12
2000 February	1	92
2001 February (mast year)	22	77
2004 August	12	47
2005 February	22	60
2006 February	18	43

Table 3 Rat tracking percentages.

Condensed from data in RNRP reports 2001, 2005, 2007

Mustelid control

Mustelid control (predominantly stoats) began during 1998 using Mk VI Fenn traps in tunnels. These were spaced between 50 and 100 m along traplines previously mentioned. Hens' eggs and rabbit meat were used as bait and traps checked and re-baited at regular intervals. Between 2005 and 2006 a total of 141 stoats were killed on the St Arnaud Range (Mainland Island), 54 in Rainbow Valley, 20 on Mt Robert and 12 at Whisky Falls (non-treatment areas did not include the Lake Head reference area). Given the dependence of fuchsia on birds for dispersal, mustelid predation on species such as silvereyes, bellbirds, and kereru has the potential to reduce fuchsia colonisation of new sites.

Ungulate control

Red deer (*Cervus elaphus*) were found in the Cupola basin 1915 (Crouch E., 1981). Deer and chamois were managed through ground shooting above the timberline in 1997-1998 and by aerial hunting between the northern St Arnaud Range and the Arnst River in March 2000 and February 2001. One red deer hind and 3 chamois were shot during 2 ½ hours of flying time. Given the low numbers it was not considered to be justifiable to continue monitoring other than recording sightings and setting up deer exclosure plots. Attempts have been made to involve deer-stalkers in control through recreational hunting.

Anticipated affects resulting from pest management

By April 2005 when this study was set up there had been an 8 year time lag from the start of pest management operations. Bellbird (*Anthornis melanura*) and other pollinator and disperser populations were showing signs of increasing in the Mainland Island area and to a lesser extent in the Lake Head area (Butler D., 2000). It was anticipated that the concurrent reduction in population densities of possums (*Trichosurus vulpecula*) and rodents (*Rattus rattus* and *Mus musculus*) subsequent to intensive control in the treatment area may have improved fruiting and seed production of *F. excorticata* and reduced seedling predation. It was recognised that the

continued presence of occasional red deer (*Cervus elaphus*) in the treatment area may offset any gains to some extent, also that the availability of disturbance-induced germination microsites and the impact of flood events on mortality were unknown. Little was known regarding the natural environmental influences on *F. excorticata* distribution, recruitment or survivorship specific to talus riparian environments in beech forest. It was anticipated that there may be a significant difference in the abundance and structure of populations of palatable species including *F. excorticata* between the Mainland Island and Lake Head areas as a result of the possum control in particular.

Differences in abundance between management areas, and changes in the demographics of the populations have been used as indicators of effectiveness of pest animal management practices (Handford P., 2000). It was proposed that palatable species be studied as part of the outcome monitoring of the RNRP project (Butler D., 2000). Most monitoring of vegetation responses to possum or deer management, including *Fuchsia excorticata*, has focused on adult trees and levels of browse (Campbell D., 1984; Pekelharing C. et al., 1998a; Pekelharing C. et al., 1998b; Smale M.C. et al., 1995; Ulrich S. C. and Brady P. J., 2005). Given the low density of adult fuchsia seen in the study area during the pilot study (Bartholomew R., 2001) and the earlier surveys (Merrifield K., 2000) and (Butler D. and Ure G., 1997), it was decided that this approach would not be appropriate for the Rotoiti Nature Recovery Project monitoring; a longitudinal survey of all size classes was decided on.

Riparian Monitoring

Preliminary estimates of *F. excorticata* abundance and distribution were made by Butler and Ure (1997) in a stream reconnaissance survey, and by Merrifield (2000) in a more comprehensive streambed mapping survey. These surveys indicated an overall low density of adult fuchsia and limited regeneration, with a noticeably higher density in the intensively managed area. In particular Merrifield's stream survey (Merrifield K., 2000) indicated sporadic distribution of *Fuchsia excorticata* with higher densities in upper

gorges(800-1000 m a.s.l.) and foot (600-700 m a.s.l.) slopes and gullies. As individual plants were not tagged in either study they did not allow monitoring of survivorship or growth. They were also limited by small sample size in the two Lake Head streambeds.

A pilot study in 2000-2001 (Bartholomew, 2001) supported these findings, and that in particular fuchsia seedlings were found to be heavily clustered and adult plants sparse. Reconnaissance as part of the pilot study noted that fuchsia in the upper gorges were generally in rock crevices greater than 2 m up cliff sides or waterfalls and largely inaccessible to mammalian browsers.

Review of riparian monitoring methodology

Methods for longitudinal monitoring of riparian vegetation need to take into account the ephemeral nature of many stream banks. Flooding and wind damage to marginal trees erodes banks making definition of the riparian margin difficult in extended studies. The method used for this study was modified from the “woody species regeneration method” (Winward A.H., 2000). For monitoring riparian vegetation Winward (2000) proposed use of the “woody species regeneration method” based on “greenline” sampling. The method involves dividing the area to be surveyed into “riparian complexes”, the equivalent of stratifying the area. In the “woody species regeneration method” each riparian complex was sampled by setting a “greenline”, the margin of green vegetation, along the edge of permanent green vegetation and surveying a 1 m transect either side of this line for a minimum of 110 m on each side of the stream (a total of 220 m of stream bank). The effect of this was to sample a minimum of 0.04 ha per riparian complex.

In their literature review and evaluation of riparian sampling methods Coles-Ritchie *et al* (2004) highlight several specific issues with the method, that mainly relate to observer variability. Variability was considered to be caused by subjective judgment in determining: the location of the “greenline” (thereby sampling different areas); missing small plants (<0.3 m) due to being

obscured by larger plants; high observer variability in describing browse impact; and in assessment of age classes. In summary Coles-Ritchie *et al* (2004) found that the woody species regeneration method was not repeatable, though added, "... we are not aware of any other repeatable methods to assess the impacts of grazing on woody species. This is unfortunate because utilization and recruitment of woody species, such as willows, would seem to be indicators of overgrazing or recovery." In response to this critique and the site factors observed in the pilot study, it was decided to modify the "woody species regeneration method" (Winward A.H., 2000) to mitigate as far as possible the problems raised. These modifications are discussed in chapter 2.

Aims and objectives of the thesis

Aims:

This study aims to determine if there are significant variations in the distribution and dynamics of *Fuchsia excorticata* populations along side-streams of the St Arnaud Range in locations associated with different management practices and environmental factors. It is acknowledged that as replication at the management level was not possible, a causal relationship between population changes and specific management practices could not be established. However trends in fuchsia population dynamics under the different management regimes could be described; and the influence of elevation, herbivory and flooding on these populations could be established.

HYPOTHESES

It was hypothesised that there would be significant differences in abundance, recruitment, growth and mortality between the *F. excorticata* populations at Lake Head and in the Mainland Island. Similarly it was hypothesised that there would be differences related to elevation due to micro-climate and geomorphic variation. Finally it was hypothesised that there would be

temporal variation related to changes in pest animal and avian populations, and the occurrence of disturbance related to stochastic flood events.

Objectives:

The specific objective of this study was to attempt to answer the following research questions.

1. Is there a difference in the density and distribution of *Fuchsia excorticata* populations between the Mainland Island area and the Lake Head area? If so is it changing with time?
2. Is there a difference in the (size-class) structure of *F. excorticata* populations between the two areas?
3. Are there differences in the rates of recruitment, growth and mortality in *F. excorticata* between the two areas?
4. Is germination a significant factor limiting *F. excorticata* recruitment?
5. Is seedling herbivory reducing *F. excorticata* seedling establishment, and if so is there a difference in seedling herbivory between the Mainland Island and the Lake Head areas?
6. Is this method of monitoring riparian *F. excorticata* effective as an indicator of the success of management practices?

Chapter 2 Methods

Study area

The Mainland Island covers the western and north-western slopes of the St Arnaud Range; the Lake Head area is predominantly western slopes. In selecting comparable streams weight was given to the likely influence of differing light regimes and it was decided to use streams dissecting western slopes where possible. Not all streams were fully contained within the park boundaries, the north-western slopes of the Mainland Island area adjoined farmland. Of the six streams within the managed area four were selected as being predominantly on the western slope; Track-Percolator TP (Borlase Stream), Loop-Rata LR, Percolator-Grunt PG, and Grunt-Boundary GB (see map – figure 4). Names relate to the block design grid of the RNRP management area. Not all streams traversed the full elevation range. TP approached farmland at about 700m, LR drained swampland below 700m, PG streambed was dry below 750m, unless in flood, re-emerging via two channels 400-500m from the lake at c. 680m a.s.l., and GB traversed the whole slope but entered the upper gorge at about 800m a.s.l.

The Lake Head area (used as the RNRP experimental control site) is about 4km from the Mainland Island with two large intervening streams. It has five streams within a 4km stretch close to the lake; however three of these only have short lengths through talus slopes before entering gorges and in order to have a total stream length comparable to the Mainland Island all five streams were used; A, B, C, D (Clearwater Stream), and E (see map –figure 3)

The study area appeared visually heterogeneous. In both the managed and reference sites the physiographic factors and canopy status of streambeds varied between streams and along streams. On upper slopes the streambed physiography ranged from gullies through to gorges and ravines, while on mid, foot and toe slopes it varied between channels and gullies. Foot and toe-slope streambeds ranged between full canopy cover (coalescing canopies from opposite banks) and partially closed canopy (light gaps and

incomplete canopy coalescence). On upper slopes in gorges and ravines, canopy cover was generally incomplete; though shade from the adjacent banks and cliffs reduced exposure to direct sunlight. Mid slope streambeds were usually either fully-open (no canopy coalescence) or partially-closed; they were also generally much wider in cross section. There were differences in topography between the Mainland Island site and the Lake Head site. The elevation at which the talus slope changes abruptly reduces from north (1100 m a.s.l.) to south (800 m a.s.l.). The talus slopes were generally steeper in the Lake Head site (average for managed 8° vs. control 15°), and the width of the slope-face narrower (average for Mainland island is 1000-1500m vs. Lake Head 200 – 1200m). Access to all streams was available by way of park and service tracks.

After considering the findings of Butler and Ure (1997), Merrifield (2000), and a pilot survey (Bartholomew R., 2001), it was decided to limit the study to talus slopes and the first accessible gorge (stopping at the first major waterfall greater than 10 m). This area would focus on the highest fuchsia population density, and plants most vulnerable to browse.

Research methodology

Monitoring naturally occurring riparian populations

To answer the first three questions it was decided to sample the spatial and temporal distribution of all size classes of *Fuchsia excorticata* plants along riparian margins of talus streams in 2005, 2006 and 2007. Individual plants were to be tagged, mapped, and biometric and geomorphic measurements recorded each year.

Germination and seedling survivorship experiment

To answer questions four and five an experimental approach using artificial seeding and exclosure plots along the same streams was adopted.

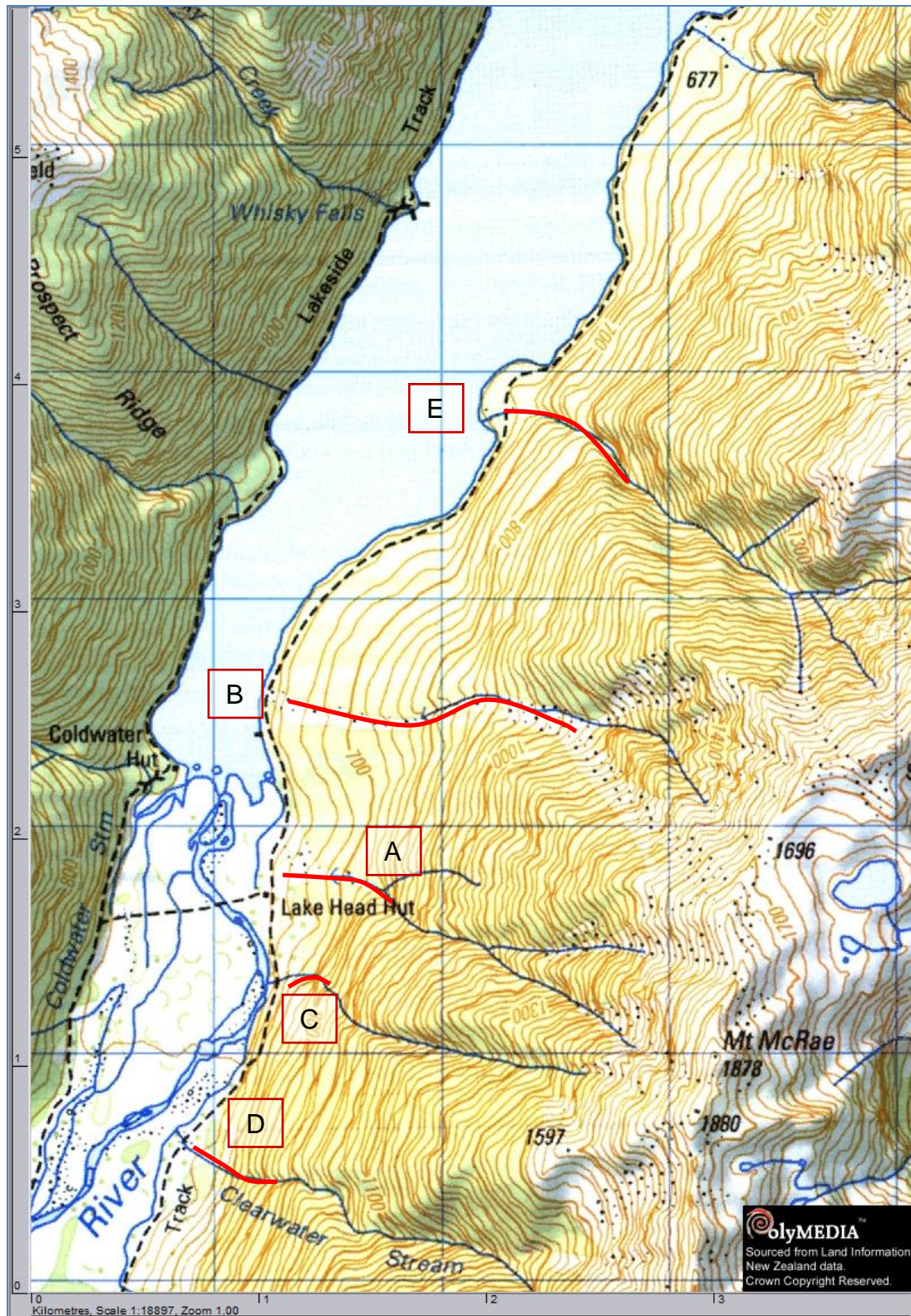


Figure 3 Lake Head non-treatment reference site, Nelson Lakes National Park
Streams surveyed are labelled with letters A through F. Red lines indicate stream sections surveyed.

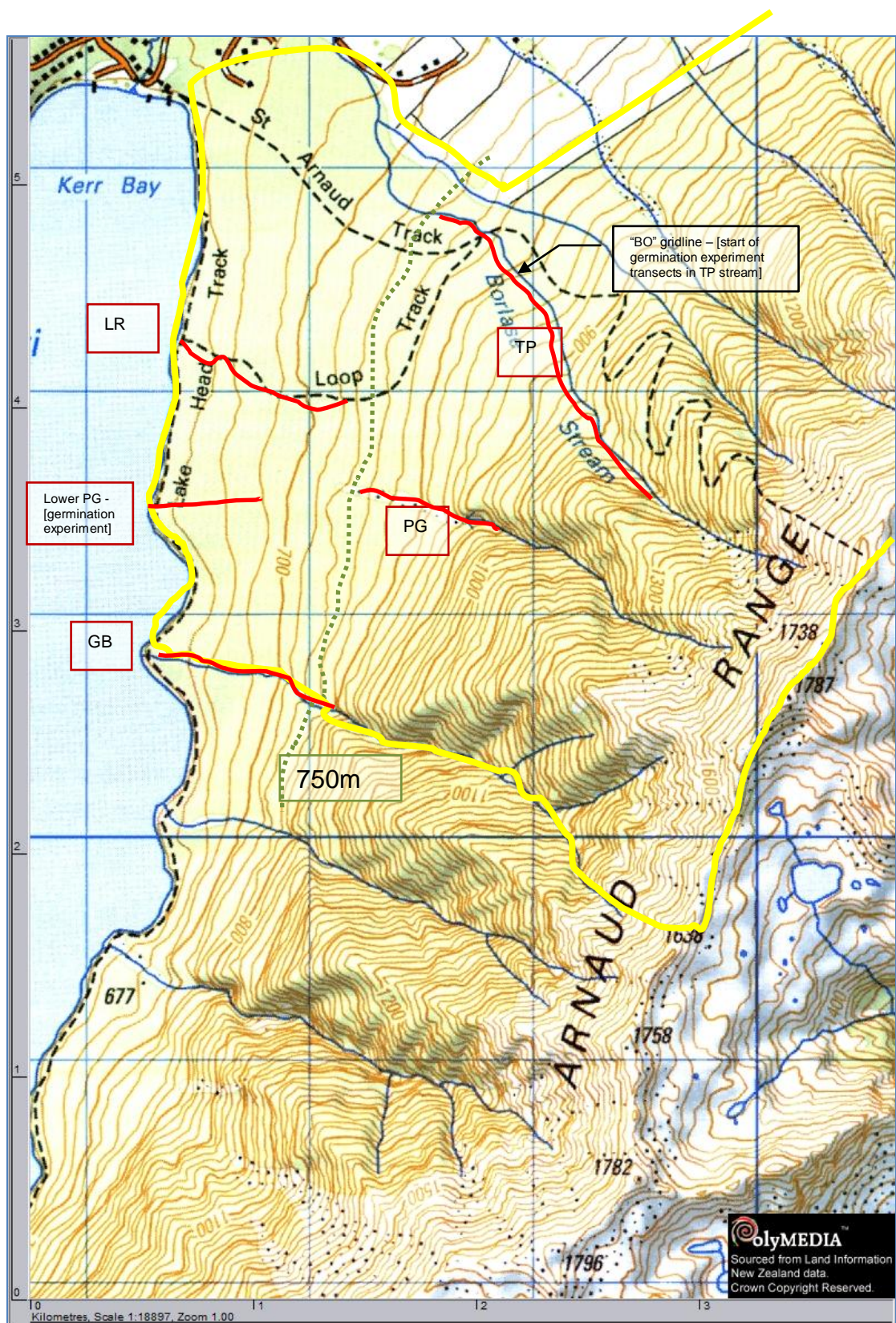


Figure 4 Mainland Island - RNRP pest-management site, Nelson Lakes National Park
Streams surveyed are labelled with two letters from the RNRP names for the spurs. The yellow line is the boundary of the managed area (2005), red lines indicate stream sections surveyed.

Survey and monitoring of the natural riparian fuchsia population

Experimental design

This part of the study compared the spatial distribution and temporal changes in the *Fuchsia excorticata* population along talus slope streams in two areas with different levels of mammalian pest management. It is acknowledged that replication at the scale of management was not carried out.

Because of the heterogeneous environment, the study area was stratified according to elevation and level of pest management. In addition it was decided to record a range of environmental measurements at each sample. Thirty-four, 200x4m, riparian belt-transects were set up along a total of nine streams dissecting the western talus slopes of the northern St Arnaud Range (covering 52-54% of streambed found in the Rotoiti Nature Recovery Programme area). Each transect was measured in March-April of 2005, 2006 and 2007.

		Pest animal management	
		Intensive RNRP pest- management area	Limited Lake Head reference area
Elevation	Upper 750-1100m a.s.l.	9	7
	Lower 620-749m a.s.l.	9	9

Table 4 The number of transects in each stratum

Riparian complexes within the study area

Riparian complexes in the Nelson Lakes study area were based on elevation, reflecting differences in physiography and canopy status. The elevation range up the talus slope is less than 400 m so two strata were considered appropriate. The division was made at 750 m a.s.l. mid way up the talus slope in the Mainland Island area. Upper slope was defined as greater than 750m a.s.l. up to the first inaccessible waterfall (>10 m) where the talus slope meets bedrock. Lower slope was defined as from lake level (620 m a.s.l.), or Travers Riverbed at the Lake Head site (640 m a.s.l.), to 749m a.s.l.

Physiographic elements (channel, gully, gorge or ravine) and canopy status (full, partial, or open) noted in the pilot study were represented as indicated in table 5.

Slope elevation	Lower (620-749 m a.s.l.)			Upper (750-1100 m a.s.l.)		
physiography	Channel	Gully	Gorge/ravine	Channel	Gully	Gorge/ravine
Canopy closure	Full	Full	Full	Full	Full	Full
	Partial	Partial	Partial	Partial	Partial	Partial
	Open	Open	Open	Open	Open	Open

Table 5 Riparian Complexes associated with elevation based stratification

Note: Terms are fully described under measurement methods. Shaded cells indicate presence of the riparian-complex factors, for example at lower elevation, channels had partial or open canopies but never full canopy.

Transects – modified greenline method

Given that the total stream length available for sampling was about 3.5km in the Mainland Island and 2.9km in the Lake Head area it was estimated that nine 100m transects in each stratified area would sample nearly half (52-54%) of the available streambed. These represented all riparian complexes and allow about 100 m spacing between transects. Only seven transects were located above 750m a.s.l. in the Lake Head reference area where the talus slopes enter bedrock ravines at lower elevations. A transect separation of 50-150m was considered adequate to account for geomorphic differences. In the Mainland Island existing 100m transects (Merrifield K., 2000) were used as they were associated with previous data and were easily relocated. These were based on the management grid crossing TP, LR, PG and GB streams (Figure 6) and provide nine samples in each elevation class. In the Lake Head area ten existing transects (Merrifield, 2000) sampled streams A and B, these were used and augmented by a further six sited in streams C, D and E (Figure 5). The first transect in these streams was located 50-60m (randomly) upstream of the track crossing and thereafter randomly 50-60m apart as three of the streams traversed less than 500m of the talus slope.

The shorter inter-transect distances may lead to auto-correlation and results are to be interpreted cautiously. Distances were measured using a Hip-chain® (Topometric Products Limited) along the true right bank (right-hand bank looking downstream) as closely as possible and start and end points marked with pink flagging tape labelled with the authors name, transect-code and date using a UV-stabilised garden marker. Care was taken to retrieve hip-chain from streambeds immediately after use to reduce the likelihood of bird entanglement; which had been a consequence of hip-chain use in earlier projects (Butler *pers comm.*). To reduce error from hip-chain stretching, measure of the location of individual plant along each transect was made using a 50 m measuring tape starting from the transect-identifying marker and measured along the true right of the streambed.

Greyline: The nature of riparian margins in beech forest is such that there is no continuously definable line of “green” vegetation. However the streams are in the most part bounded by definable substrate banks ranging from rocky cliffs to gently stepped silt-sand terraces and terracettes. The tread-riser boundary of the lowest definite terrace, bank or cliff was taken as a substitute for the greenline and will be referred to as the “greyline” as it is defined by a substrate boundary not vegetation (see figure 5). Where this was indefinite the line followed the forest margin marked by the streamside trunks of canopy trees.

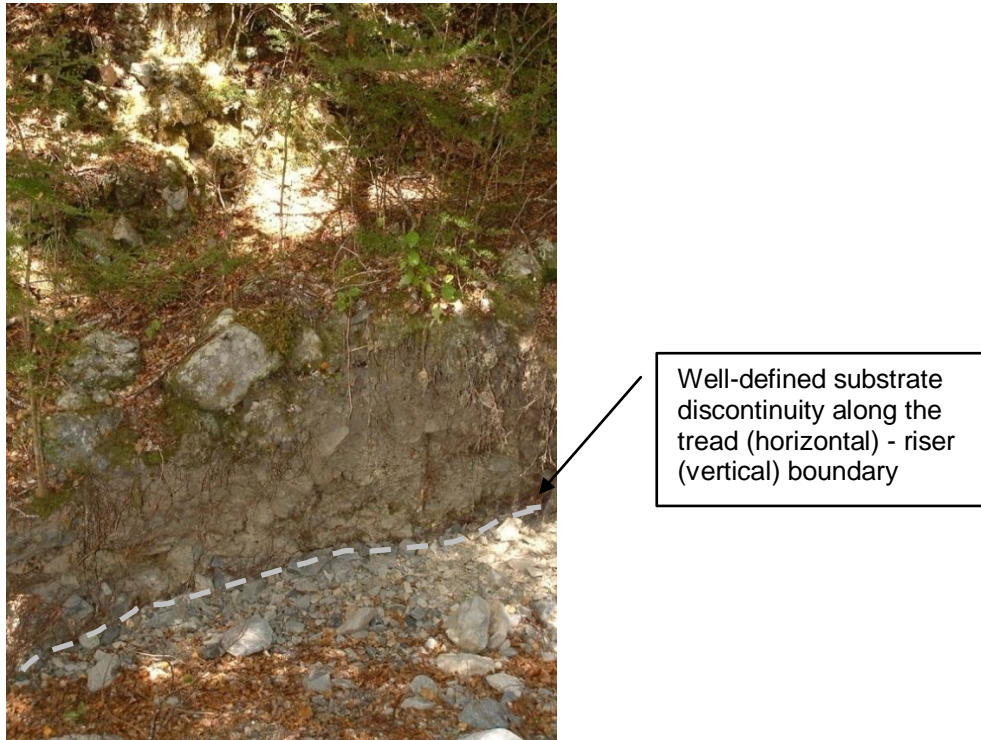


Figure 5. An example of the "Greyline" use to define riparian margins

Belt-transect width: The "woody species regeneration method" used a symmetrical strip approximately 1 m either side of the greenline. For this study the method has been modified to an asymmetric strip, 3 m on the bank-side and 1 m on the streamside of the greyline along both the true right and left banks of the stream. This provided a total belt-transect area of 200x4m (0.08ha) and encompassed all plants found in the pilot except rare (3) mid-streambed specimens, which were recorded separately. Where the bank was a cliff the 3 m distance was estimated to the upper visible reaches of the cliff. No stream surveyed was narrower than 2 m so there was no overlap of true-right and true-left bank transects.

Search strategy

The search pattern involved two observers (Bartholomew R. and Bartholomew M.). Searchers began at the lower end of the stream bed transect and climbed the transect in 20m sections of the stream one beside the true right bank the other the true left looking for fuchsia, and fuchsia found entailed both stopping, measuring and recording the data. The search was continued to the end of the uppermost transect then repeated on the

reverse journey after exchanging stream sides. During the pilot study in 2000-2001 both observers became proficient at noticing even small seedling fuchsia, and having two observers searching the same section of bank reduced the likelihood of plants being missed. In general one field-day (8-10 hours) was needed to survey each long stream (TP, GB, A, B and E) including time to reach the stream. Where streams were shorter and in close proximity two were surveyed in a day (for example LR and PG were sampled in one day, as were C and D streams). A summary of the number of transects in each stream, and the area sampled is shown in table 6 (below).

Area	Stream and length (m)	Elevation range (m a.s.l.)	Elevation zone	Lower transects	Upper transects	Combined area of belt transects (m ²)
Mainland island	TP 1100m	750-1100	Upper		6	10,400
	LR 625m	620-700	Lower	4		3,600
	PG 730m	750-900	Upper		3	5,600
	GB 1040m	620-750	Lower	5		6,800
Lake Head	A 430m	620-800	Both	2	1	4,000
	B 1450m	620-1100	Both	2	5	10,400
	C 275m	640-750	Lower	1		1,600
	D 365m	640-850	Both	1	1	2,400
	E 455m	620-750	Lower	3		3,600

Table 6 Summary of stream and transect information

Note: streams LR, GB, C and E only flow through the lower slopes from about 750m A.S.L., while streams TP and PG only flow at higher elevations before disappearing into the talus debris (unless in flood). Only streams A, B and D flowed through both elevation strata.

Where a fuchsia was multi-branched or coppiced above ground level the longest branch was measured and tagged, secondary branches were marked using plastic ties without tags. Where below-ground coppicing could not be confirmed individual stems were treated as genets, tagged and a note made in the field record. This was not common unless plants had been flood damaged.

Mapping of plant locations

Individual plants and clusters were mapped in the field and GPS locations recorded. Distances from known field points such as the park track, stream mouth or service track crossings were recorded. Maps are available from the author or St Arnaud Area Office, Department of Conservation.

Photography: Photographic records of tagged plants were made to assist relocation and monitoring of changes in habit. A Fuji FinePix 2800Z digital camera was used at 2 megapixels.

Tags

Plants were tagged using either sequentially stamped aluminium vegetation tags supplied by the Rotoiti Nature Recovery Programme (figure 6), or if unavailable, with “permanent” flagging tape appropriately labelled using garden markers. If flagging tape was substituted, small lengths were used and it was attached discretely so as to reduce the possibility of it attracting deer or possums (figure 7). The vast majority of stems were less than 1.35cm high and had a diameter at breast height (DBH) less than 3cm so nailing tags was not appropriate. Where possible the tags were tied to stems using plastic coated gardeners’ wire-ties, sufficiently loosely to allow for anticipated growth and close to the base of the plant. For small seedlings tags were attached parallel to a nearby companion or peg and a note made in the field record with accompanying photographs and descriptions.



Figure 6 parallel tagging of small seedlings



Figure 7 tagging using flagging tape

Biometric measurements

Height: Height was measured in millimetres from the rooting base to the furthest growing shoot using a 3m builders tape for smaller plants and a 20m DBH tape and Leki® Pole for taller trees. Given the spindly and often prostrate habit of many Fuchsia found in the pilot (especially after flood events, see figure 8) it was decided to define height in terms of stem length to indicate size class and determine growth. Where dieback or coppicing occurred the height was measured to the furthest growing shoot and total length of the apparently dead stem noted.



Figure 8 horizontal habit of a sapling following flooding

Height tiers: Plants were divided into three height tiers related to life stages (see Table 7).

Seedling	Sapling	Tree
0 – 45 cm	46 – 200 cm	>200 cm

Table 7 height tier definitions

Plants exceeding 200 cm were classed as *trees* since they were both generally above deer browse height (Nugent G. et al., 1997) and potentially reproductive; although no flowering or fruiting plants < 250 cm were seen in the study areas during the pilot, some of about 200 cm were found near the start of the track at St Arnaud. Seedlings were defined as 0-45 cm as most newly emergent plants in the pilot between 2000 and 2001 were within this range and it overlaps with tiers used by Department of Conservation staff at St Arnaud (Merrifield, 2000). Saplings were defined as being from 46 cm to 200 cm.

Browse: Foliar browse was scored as insect or mammal, and heavy,



medium or light following incidental information criteria (Allen R., 1992) with the addition of a category of hedging (See figure 9 and table 8). Hedging was defined as browsing across the top layer of the plant, resulting in a uniform plane of upper leaves. Notes on browse type, height or accessibility were made that may indicate whether the herbivore was likely to be deer or possum.

Figure 9 browse hedging of seedlings

Light	Browsing on one or two shoots only
Medium	Browsing on more than one or two shoots, but most not browsed
Heavy	Browsing on most accessible shoots. (or “hedged”)

Table 8 Criteria for scoring mammalian browse.

Geomorphic and environmental measurements

Given the patchy distribution of fuchsia in the study site an attempt was made to characterise microsites based on geomorphology, elevation, stream-profile and canopy-closure.

Geomorphic description: Each site was described at landform and site scale (a radius of not more than 50m). Stream and bank aspect were measured in degrees from true north using a Silva Ranger compass with magnetic declination set to 22° east. Stream and bank slope were measured in degrees from the horizontal using the integrated clinometer. Geomorphic descriptors follow Milne *et al* (1995).

Elevation: Elevation was measured in metres above mean sea level using a Garmin eTrex summit GPS (housing an aneroid altimeter). The altimeter

was calibrated at the lake (620 m a.s.l.) or a known elevation prior to measuring each stream. Measurements were made on the first record of a fuchsia plant and on subsequent surveys, and the mean elevation from these readings used in the data to be analysed.

Canopy closure: The degree of cover by canopy was described as *complete* (total coalescence), *partial* (incomplete, including near-by lightgaps), or *open* (no coalescence). Note was made of any lightgap that could impact on the rooting site.

Stream profile - width, depth, and slope: Width distances were measured using a 20 m DBH tape perpendicular to the bank edge, depth from the rooting site to the deepest section of the streambed was determined using a builder's line and attached level. Slope was measured from the horizontal using the inclinometer in a Silva Ranger compass.

Temperature and relative humidity: Measurements of temperature and humidity fluctuations were made in two streams during April 2007 in order to determine the magnitude of variation between different elevations and stream aspects. Data were collected using two EasyLog® EL-USB-2 dataloggers manufactured by Lascar Electronics. One set of measurements (48 hours) compared elevation affects between Lake Rotoiti at 620 m a.s.l. and the upper Borlase Stream (TP) at 1,100 m a.s.l. The second set of data (72 hours) compared the true right (southern aspect) and true left (northern aspect) banks of stream A.

On April 7, 2007 the two loggers were deployed in the Mainland Island area. One logger was at about 950 m a.s.l. in upper TP (Borlase) stream (see figure 4), the other at the lake margin (620 m a.s.l.) beside LR stream mouth (see figure 9). On April 13, 2007 the two loggers were deployed at about 640 m a.s.l. in stream A (see figure 3) at Lake Head. One logger was set on the northern aspect (true left of stream), the other on the southern (true right bank). Data, were graphed using EasyLog® software over either two or three days.

Data reduction

Data from individually tagged plants were aggregated by transect to provide information on abundance, density, recruitment and mortality.

Abundance – abundance estimates were made by counting tagged plants in each 100 m transect, irrespective of size. Results are reported as plants per 100 m stream-bed.

Density – density was calculated by dividing the abundance by 0.08 ha. (Each belt transect covered 100x4 m on each stream bank, a total of 800 m²)

Recruitment - data were filtered to identify new seedlings not present in previous surveys. These were interpreted as having been recruited into the sample population. Despite diligent searching it was possible that small individuals attributed to recruitment in one year may have been living the previous year but not observed, or that authentic recruits were not observed in any given year and a cautionary approach to interpretation is taken.

Mortality – death of previously tagged plants was determined if the plant was found but had no vegetative growth visible, or the site was positively located but had been destroyed by erosion processes. It is acknowledged that diagnosis of death is problematic in a species that freely coppices and some errors will have been made where a plant devoid of leaves and bark will resprout.

Growth – only plants located in every survey (2005, 2006 and 2007) were used in calculating growth since repeated measures analysis was used.

Statistical methods

Data were analysed using SPSS 14.0 for Windows (SPSS Inc.) statistical software. Preliminary analyses of data checked for agreement with test requirements using Mauchly's Test of Sphericity and the Levene statistic for

homogeneity of variances. Any transformation necessary to meet these requirements was done prior to full analysis. Where conformity requirements are not fully met it is stated in the result. All test statistics were generated at the $p < 0.05$ level.

Spatial and temporal abundance

Spatial comparison of mean fuchsia abundance and density within each year's data was done using one-way ANOVA (univariate analysis) under the General Linear Model. Time series abundance data were analysed using the General Linear Model repeated-measures with treatment and elevation zones as between-subject factors, homogeneity tests run and main effect confidence intervals adjusted using LSD (Least Significant Difference t-test).

Height changes

Within each year's data multivariate analysis of variance under the General Linear Model was to test the significance of differences in mean height between different elevation zones and management areas in individual years. The General Linear Model repeated-measures was used to test the significance of changes in height between years.

Environmental measures

Differences between transects in the measured environmental variables were tested using one-way analysis of variance.

Germination and seedling survivorship experiment

During the 2000-2001 pilot study (Bartholomew R., 2001) it was noted that there were few *Fuchsia excorticata* seedlings. Given that the plant is a known secondary coloniser and might be expected to be more prolific, it was decided to supplement the natural population survey with experimental planting trials in order to study germination rates and subsequent seedling herbivory.

Experimental design synopsis

This part of the study aimed to investigate germination rates and subsequent establishment in-situ along riparian habitat in areas of differing pest animal management. The study area was stratified by level of management, and stream sections containing naturally occurring *Fuchsia* selected within each stratum. It was not considered constructive to place germination plots in stream-bed sections with little shade or subterranean summer stream flow. Thirty-six 5x1 m riparian quadrats were set up along the 7 stream sections selected (Table 9). Environmental factors measured or described at each quadrat were elevation, aspect, percentage canopy cover and shade, stream width, disturbance, and soil texture. Quadrats were set up randomly at 60±10 m intervals along each stream and within each quadrat paired 25x25 cm seed plots and a control plot were established 2 m apart. To detect possible early-seedling herbivory by possums or rats, small enclosure cages were used to cover one plot, one was seeded but without post-germination protection, and a third left as a control for detecting naturally dispersed *fuchsia*. Mature fruit collected in January from *fuchsia* in the same provenance had seed extracted and placed in cool-storage refrigeration at 5°C. Within a week of collection 50 fresh seeds were scattered over each caged plot, 50 over each un-caged, the third control plot being left un-seeded, a total of 3,600 seeds. A sample of 600 seeds was sown the same week in laboratory conditions to check viability on different substrates. Field germination was checked the following April (3 months later).

Management area	Stream	Quadrats		Seeds sown	
Lake Head	A	8	18 quadrats	800	1800
	C	2		200	
	D	3		300	
	E	5		500	
Mainland Island	LR	8	18 quadrats	800	1800
	TP	5		500	
	PG	5		500	

Table 9 Summary of quadrat locations

Trialling - difficulties and solutions

The experimental design was originally trialled in late December 2000 but no Fuchsia seedlings were seen in April 2001. Streams TP, A and B suffered widespread bank collapse subsequent to severe flooding in April 2005, with consequent loss of about half the quadrats. In the same year it was again noted that no seedlings had appeared in the surviving plots and low seed viability was suspected as a cause. To study herbivory of small seedlings it was decided to grow seedlings, from seed collected in January 2005, in the glasshouse for transplant into the field cages in January 2006. Failure of automatic misting in the glass house led to the seedlings dehydrating and eventually dying. New quadrats were set up in December 2006, sown in January 2007, and checked in late April 2007. Given the reduced time frame, seedling herbivory was not able to be studied using seedlings derived from germinated seed in the exclosure cages. However the opportunity for a less rigorous survey of naturally occurring new seedlings was presented in a streambed not previously monitored (referred to as “Lower PG” in figure 4).

Quadrat establishment

Quadrat placement used the same stream start points as the previous survey transects, with the exception of the lower PG stream, which had not been used previously, and TP which was entered at the “BO” grid line crossing at

805 m a.s.l. Measurements in lower PG stream were made relative to the lake edge and in TP stream relative to the “BO” crossing (see figures 3 and 4). Quadrats were placed at random 60 ± 10 m intervals along each stream using a hip-chain along the true right bank. Odd-even random numbers were used to decide whether to place each quadrat on the true right or left of the stream. Quadrats were positioned as close as practicable to the greyline and on the bank-side. Each seed plot was marked with horticultural staples and flagging tape, and each quadrat tagged with flagging tape at the downstream end.

Environmental measurements

Measurements of elevation, stream slope, width and depth followed the same protocols described earlier. Geomorphic and canopy descriptions used the same criteria. Disturbance at the quadrat site was photographed and categorised as stable where there was no evidence of recent movement, bank-fall where substrate was derived from recent movement of soils from the bank above the site, or tree-fall where the quadrat was sited on a root-ball or hollow resulting from the collapse of a standing tree. The soil texture group of substrate was described using the field method in the Landcare Research, Soil Description Handbook (Milne J.D.G. et al., 1995).

Exclosure cages

Thirty-six small steel-mesh cages were designed and built to protect emerging seedlings from browse by rats or possums. They were constructed from 12 mm concrete reinforcing mesh cut to form a right-angle triangular box over a 250 mm square base (see figure 10). Able to be carried flat they were reassembled in the field using horticultural ties and attached to the ground with 100 mm horticultural staples. A triangular section was chosen reduce the accumulation of leaf-fall debris and it proved versatile when placed in less than ideal sites. A 25 mm mesh sheet cut to 250 mm square was used as a temporary “plot marker” for the non-protected and control plots (see figure 11). The diagonal corners of each such plot were marked semi-permanently with horticultural staples and flagging tape. All cages and staples were removed at the end of the study.



Figure 10 Exclosure cage in situ



Figure 11 Temporary plot quadrat marker

Seed preparation

Five ripe (red-black) fruit were taken from each of three fuchsia trees at the start of the Lake Head track at Kerr Bay and 3 roadside fuchsia trees 6 km along the St Arnaud - Blenheim road. Care was taken not to remove >25% of the fruit from any one tree and to collect from a number of trees to allow for genetic mixing while maintaining the same provenance. Fruit was cleaned in 5% chlorine bleach and rinsed in tap water, seed then extracted following maceration. A kitchen sieve immersed in a container of tap water was used to separate fruit debris from seed, the more dense seed settling on the bottom of the container. About 300-400 seeds are found in each fruit and once decanted through paper towels this seed was allowed to dry overnight (see figure 12). Dry seed was counted into samples of 50 using a fine paint brush and placed in small zip-lock packets to facilitate sowing in the field. Seed for use over the following week was stored in a domestic refrigerator until needed.

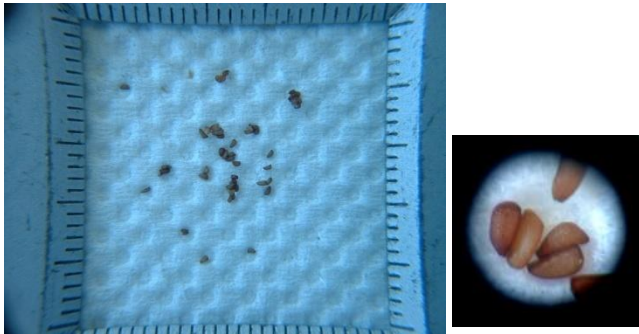


Figure 12 Fuchsia seed - 1 mm grid scale.

Seed viability trial

To determine the viability of seed used in the field trials and germination time, 600 seeds were sown in a mini-propagator into perlite, sand and humus in January 2007. Sterile perlite was used as a control, and sand and humus sterilised (by steam) used as a limited experimental comparison.

Fifty seeds were sown in each cell. A regular watering regime of was standardised by mist spraying twice daily to maintain a moist surface layer. Lighting was indirect ambient sunlight, and daily maximum and minimum temperature recorded.

Seedling herbivory

In December 2005 while establishing quadrats in a secondary stream mouth draining the upper PG stream it was apparent that numerous small seedlings were colonising collapsed bank sides following flood damage in April. It was considered that these could be the subject of monitoring for browse and seedling survivorship and two 3 m clusters of 47 seedlings were photographed, measured and mapped as a trial. In April 2006 they were remeasured, found to be growing vigorously, not suffering mammalian browse, and increasing in abundance. Given the need for a larger sample size to monitoring seedling herbivory and survivorship it was decided to map the lower 300 m of the streambed for resurvey in April 2007.

Chapter 3 Results

Survey and monitoring of the natural riparian Fuchsia population

Overview

In the RNRP intensively managed area 28% of the available north-west facing talus streambed length (6.5km) was sampled, and in the Lake Head RNRP low management area 54% of the 3km streambed available. A total of 253 fuchsia plants were observed and tagged in 34 transects (3.4 km of streambed searched) over the period April 2005 to April 2007 (see figures 13 and 14). Of the 104 plants tagged in 2005, 69 survived through to 2007. A total of 149 plants were recruited and 37 died over the two-year period. The spatial distribution of plants in all areas was patchy, 14 of the 34 transects recording no fuchsia in any height class for 2005 or 2006 while one transect recorded 98 plants (mainly seedlings). Heights of fuchsia observed ranged from 10mm seedlings to 4.2m trees and the proportion of seedling, saplings and adult trees varied between treatments and years.

Statistical analysis

Abundance data were $\log_{(n+1)}$ transformed to meet normality requirements for the Levene statistic test for homogeneity of variances and Mauchly's Test of Sphericity ($p < 0.05$). Mean abundance of fuchsia in the Mainland Island was compared to the Lake Head reference area for each year using univariate analysis of variance ($p < 0.05$), and changes between years were analysed using repeated measures ANOVA. Differences in mean height were compared using univariate analysis, as was growth between 2005 and 2007.

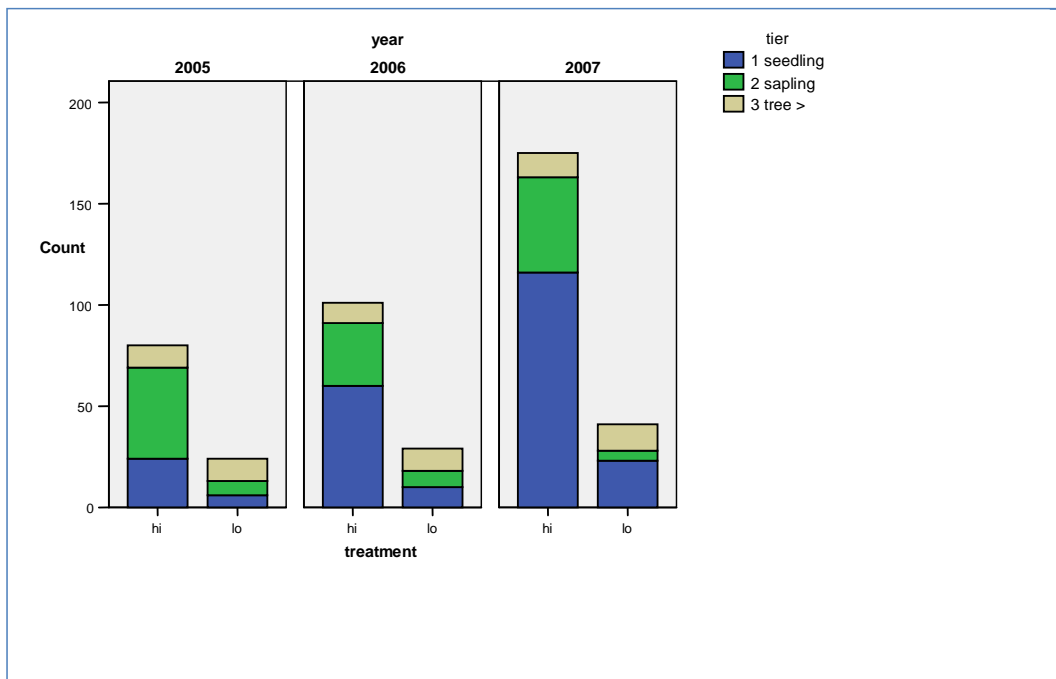


Figure 13 Total number of *Fuchsia excorticata* found in each treatment area.
 Data for upper elevations at Lake Head have been scaled (x 9/7) to compensate fewer transects. Note: For treatment categories, “hi” is the intensively managed Mainland Island area and “lo” the partially managed Lake Head reference area. It is recognised that there is no replication at the management scale and differences in abundance can not be attributed to management practices.

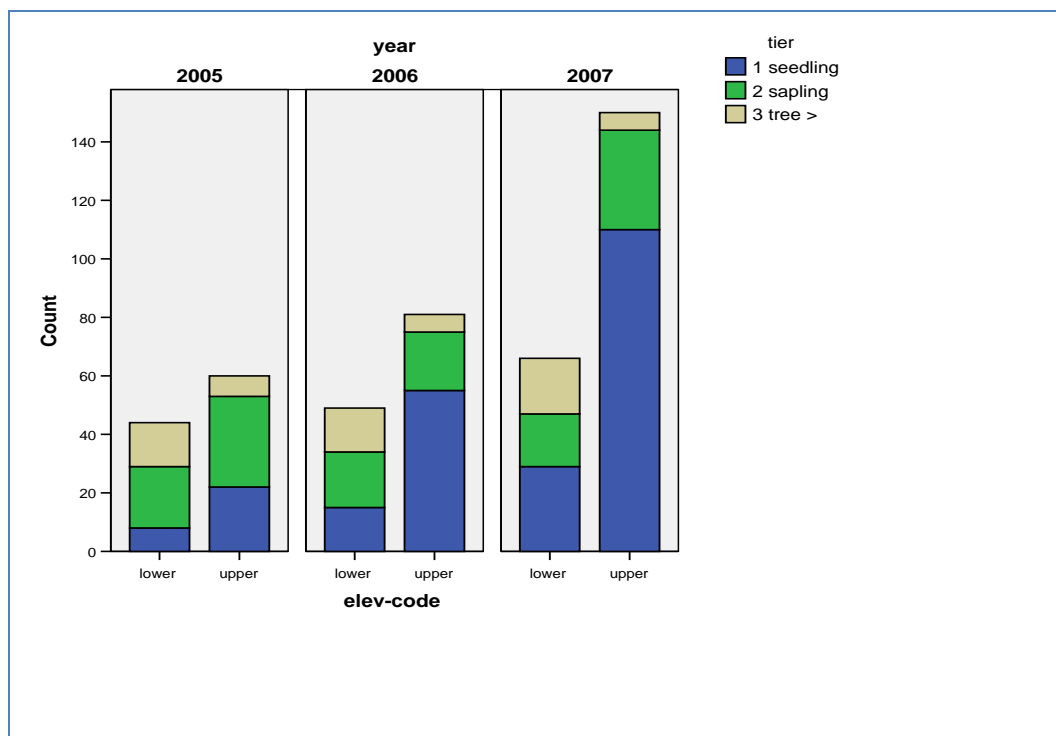


Figure 14 Total number of *Fuchsia excorticata* found in each elevation zone.
 Data for upper elevations at Lake Head have been scaled (x 9/7) to compensate fewer transects.

Abundance and density

There was no significant difference in the $\log_{(n+1)}$ transformed mean abundance (n/100m streambed) between the Mainland Island and Lake Head data ($p < 0.05$). Across the whole study area the mean abundance of *Fuchsia excorticata* increased from 3.09 ± 0.93 (mean ± 1 SE) plants per 100 m streambed (a density of 38.6 ± 11.6 plants per ha) in 2005 to 6.36 ± 2.99 (79.5 ± 37.4 plants per ha) in 2007 ($n = 34$). The 2005-2007 change in the Lake Head area was from 1.56 ± 0.61 to 2.56 ± 1.08 ($n = 16$), and for the Mainland Island from 4.44 ± 1.62 to 9.72 ± 5.50 ($n = 18$). In each year surveyed, abundances showed a consistent spatial pattern in mean fuchsia numbers per transect (n/100m streambed) between the Mainland Island and Lake Head, with more fuchsia consistently observed at the Mainland Island site (figure 15).

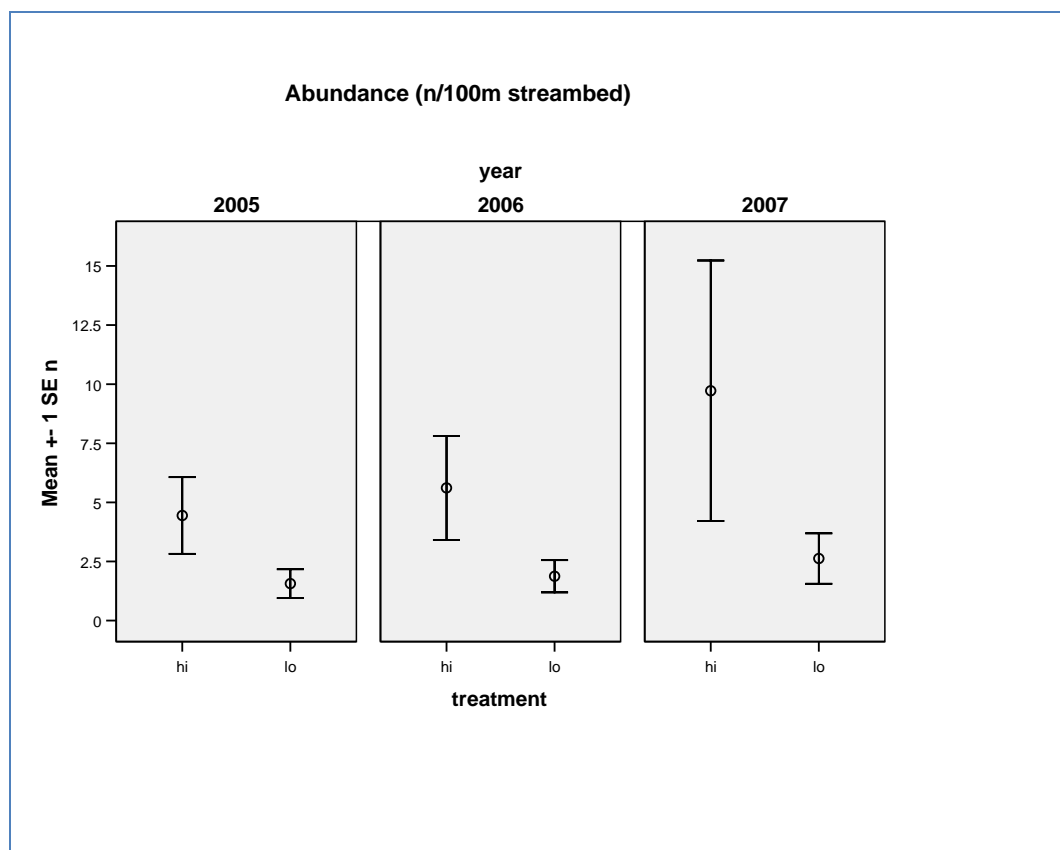


Figure 15 Mean abundance per 100 m transect in each management area between 2005 and 2007.

Note: For treatment categories, “hi” is the intensively managed Mainland Island area and “lo” the partially managed Lake Head reference area. It is recognised that there is no replication at the management scale and differences in abundance can not be attributed to management practices. Error bars are ± 1 SE.

Annual changes in abundance

Total abundance

While total numbers of fuchsia across all 34 transects increased in absolute terms between 2005 and 2007, log(n+1) transformed abundance comparing the Mainland Island and Lake Head data at high and low elevations showed no significant difference in annual changes to total fuchsia abundance; management comparison ($F_{1,30} = 2.56$ $p=0.117$) and elevation ($F_{1,30}=0.81$ $p=0.3756$). However a significant difference was noted in sapling abundance when data were analysed by life-history stage.

SEEDLINGS

When comparing management regimes there was no significant overall difference in the mean log (n+1) transformed seedling abundance between 2005 and 2007 of 0.137 ± 0.118 (mean ± 1 SE) seedlings per 100m streambed ($F_{1,30}=0.118$ $p=0.254$). Similarly no significant difference was found when comparing elevation differences ($F_{1,30}=0.114$ $p=0.738$) (Figure 16)

SAPLINGS

For the period 2005 to 2007, there was a significant difference ($p<0.05$) observed between mean log(n+1) transformed sapling abundance in the managed versus non-managed areas of 0.232 ± 0.100 ($F_{1,30}=5.409$ $p = 0.027$ with elevation as a between subject factor). However there was no significant difference between elevation zones ($F_{1,30}=0.87$ $p=0.771$) Sapling abundance in the Mainland Island area varied between 1.72 ± 0.06 (mean ± 1 SE) in 2006 and 2.61 ± 1.43 in 2007. In the Lake Head area the range was 0.50 ± 0.24 in 2005 and 2006, and 0.31 ± 0.20 in 2007. (Figure 17)

TREES

No significant difference was observed between changes in mean tree abundance in the managed versus non-managed areas, or between years ($p<0.05$). Management comparison ($F_{1,30} = 0.197$ $p=0.661$) and elevation ($F_{1,30}=2.71$ $p=0.110$). Conditions for Levene's test were not met and there were no trees found at upper elevations in the Lake Head area. (Figure 18)

Height-class specific abundance

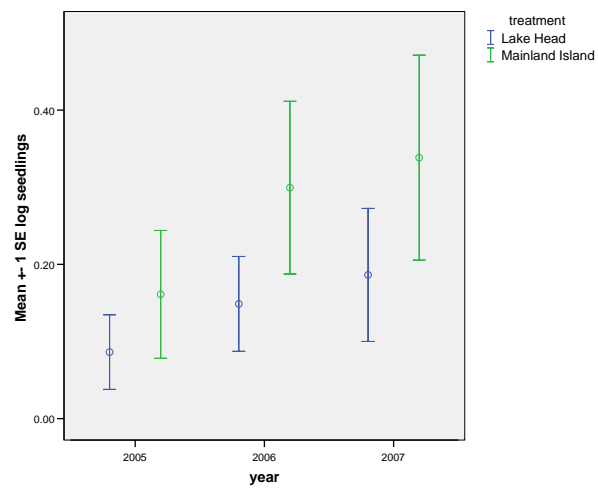


Figure 16 log(n+1) transformed seedling abundance for different management areas

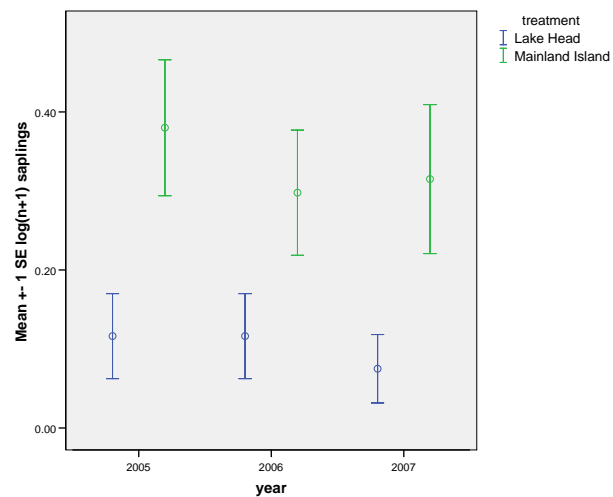


Figure 17 log (n+1) transformed sapling abundance for different management areas (difference is significant ($p < 0.05$)).

It is recognised that there is no replication at the management scale and differences in abundance can not be attributed to management practices.

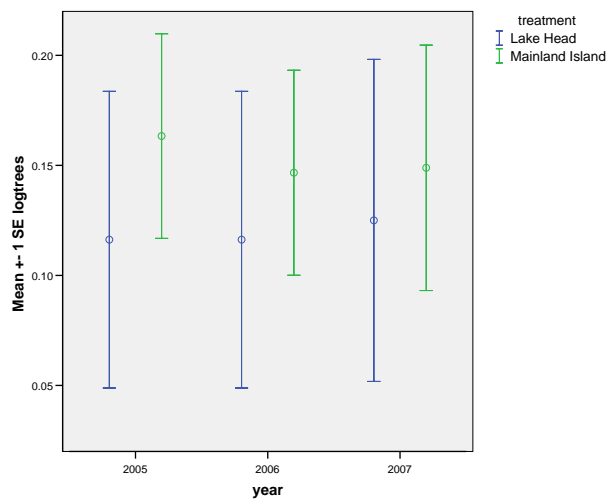


Figure 18 log (n+1) transformed tree abundance for different management areas

Mean height

To give an indication of the stature of Fuchsia populations in different geographic areas, mean heights in each transect were compared over the 2005-2006 and 2006-2007 intervals. There was no significant difference between treatments each year but a difference was evident between the elevation zones. It was noted that the direction of elevation response was inverted between the high and low intensity management. In the Mainland Island site plants were significantly taller ($p < 0.05$) at higher elevation than at lower, while the converse was seen in the Lake Head site. (Figure 19)

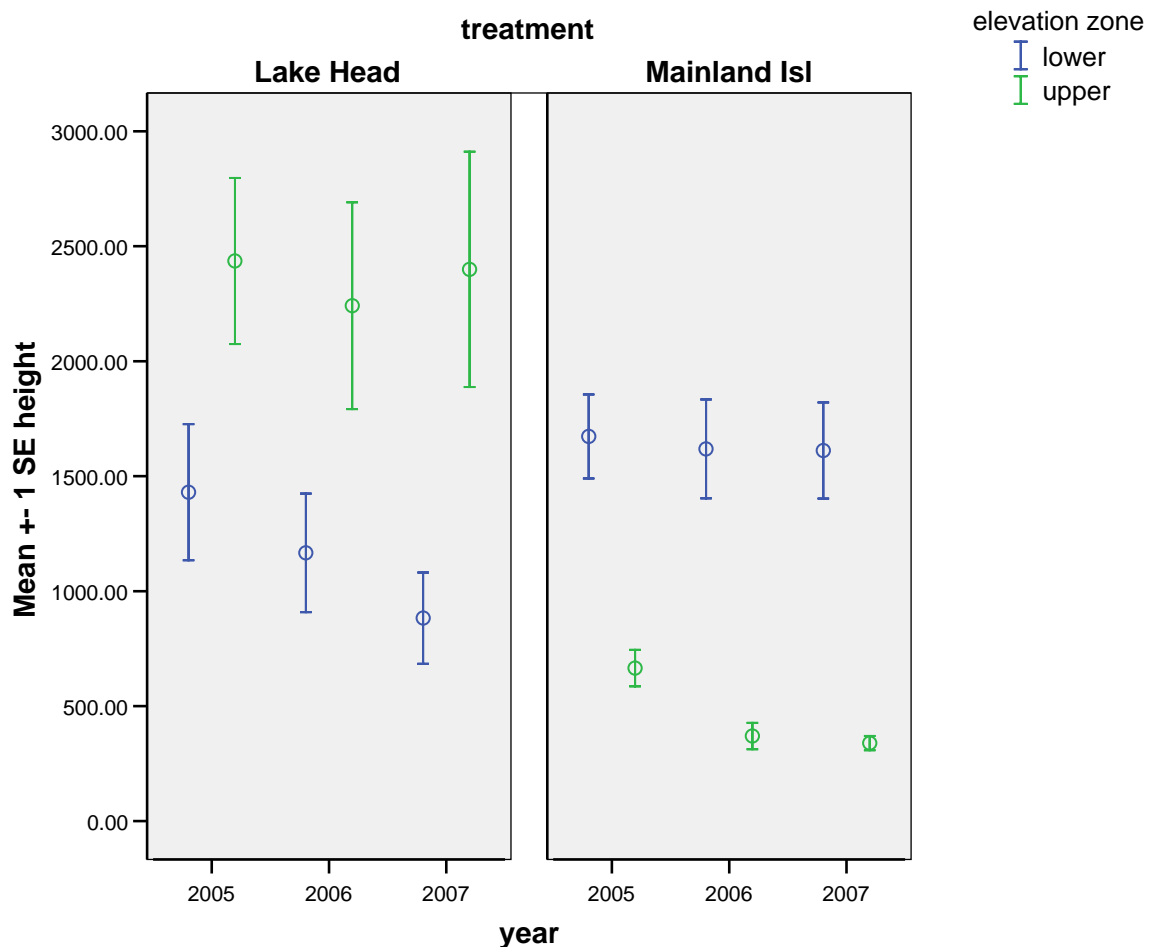


Figure 19 Mean height of fuchsia at different elevations in different management areas. Error bars are ± 1 SE.

Overall there was a trend toward shorter plants at higher elevations (see table 10). However in the Mainland Island site at lower elevations there is a reduction in mean height consistent with increasing recruitment, this was also noticed at higher elevations in the Lake Head area. Overall the change in the mean height difference between 2005 and 2007 is significant 212 ± 62 ($F_{2,65} = 12.42$ $p < 0.001$), and also between 2006 and 2007 134 ± 28 ($p < 0.001$).

	treatment	elevation		Std. Deviation	N
		zone	Mean		
height 2005	Lake Head	lower	1430	1220	17
		upper	2436	956	7
		Total	1723	1222	24
	Mainland island	lower	1734	1056	19
		upper	623	556	26
		Total	1092	970	45
height 2006	Lake Head	lower	1414	1184	17
		upper	2536	1038	7
		Total	1742	1236	24
	Mainland island	lower	2047	1067	19
		upper	669	551	26
		Total	1251	1054	45
height 2007	Lake Head	lower	1604	1206	17
		upper	2716	1229	7
		Total	1928	1293	24
	Mainland island	lower	2332	1036	19
		upper	667	499	26
		Total	1370	1128	45

Table 10 mean height (millimetres) of *Fuchsia excorticata* plants in 2005, 2006 and 2007.

Growth

Height differences between 2005 and 2007 were calculated for each tagged plant and interpreted as growth (mm per 2 years). Of the 104 plants tagged in 2005 only 69 individuals survived through to 2007 and were available for analysis for growth (see table 11). It should be noted that browse or flood induced damage resulted in negative growth for some individual plants. Overall growth data were analysed for mean growth differences using

univariate analysis under the General Linear Model with elevation, management level, and height tier in 2005 as between-subject factors. Height class specific data were analysed using the same tests, with elevation and management level as between subject factors.

Factor	Options	n (=69)
Treatment	Mainland island	45
	Lake Head	24
Elevation	Lower	36
	Upper	33
Height class 2005	Seedlings	19
	Saplings	31
	Trees	19

Table 11 numbers of fuchsia used to monitor growth 2005-2007

Overall growth

Fuchsia growth rate was significantly related to elevation between 2005-2007 ($F_{1,58}=8.88$ $p=0.004$), with the difference in two-year growth between upper and lower elevation being 416 ± 139 mm. No significant difference in growth was observed between levels of management ($F_{1,58}=0.337$ $p=0.564$). When analysed by elevation and management it was interesting to note the trend in sapling growth was for faster growth at low elevations in the Mainland Island, but the reverse was observed at Lake Head (see figure 20).

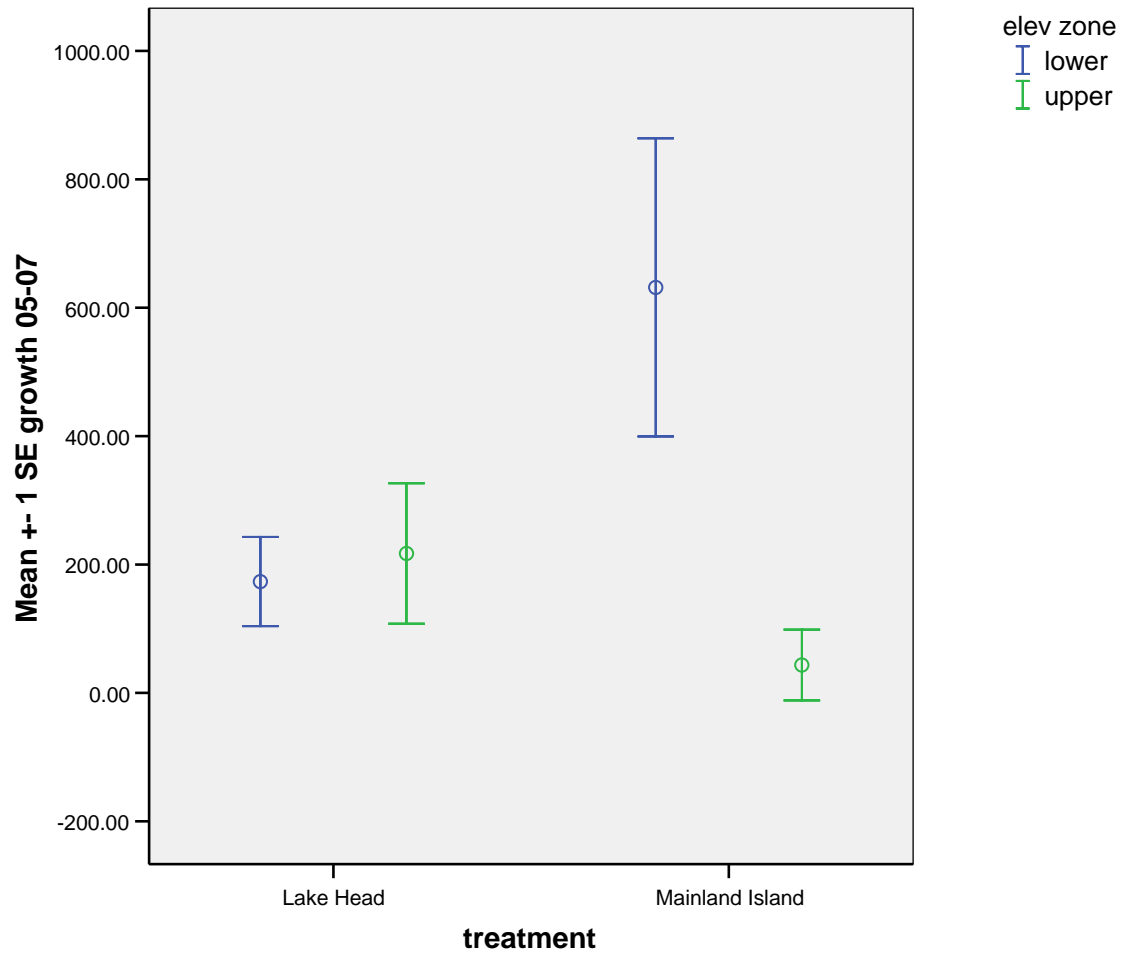


Figure 20 Overall fuchsia growth rate from 2005 - 2007 (mm). Error bars are ± 1 SE.

Growth by height class

When analysed by height class in 2005, seedling and tree data did not meet Levene's test of equality of error variances. Means and standard errors are given below (table 12).

Seedling growth 2005-2007		
treatment	Mean	Std. Error
Lake Head	115	47
Mainland island	159	44

Sapling growth 2005-2007

Lake Head	165	236
Mainland island	528	117

Tree growth 2005-2007

Lake Head	238	95
Mainland island	-368	168

Table 12 Growth recorded as millimetres change over two years (2005 - 2007)

Sapling data complied with test assumptions. There was evidence of a significant difference in growth between elevation zones of 605 ± 263 mm over the two year period, faster growth being seen at lower elevation ($F_{1,27}=5.268$ $p=0.030$). (See figures 21-23)

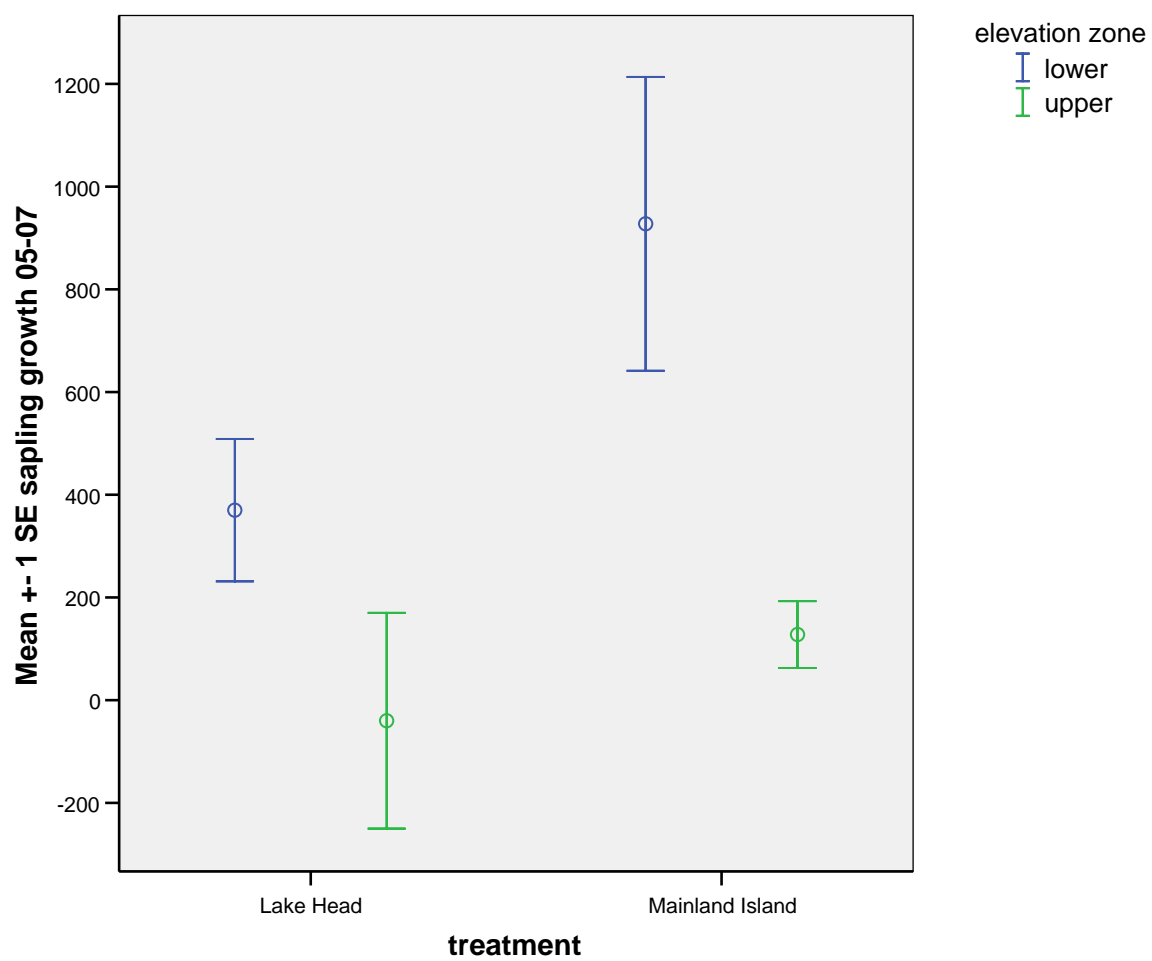


Figure 21 Mean sapling growth between 2005 and 2007 (mm). Error bars are ± 1 SE.

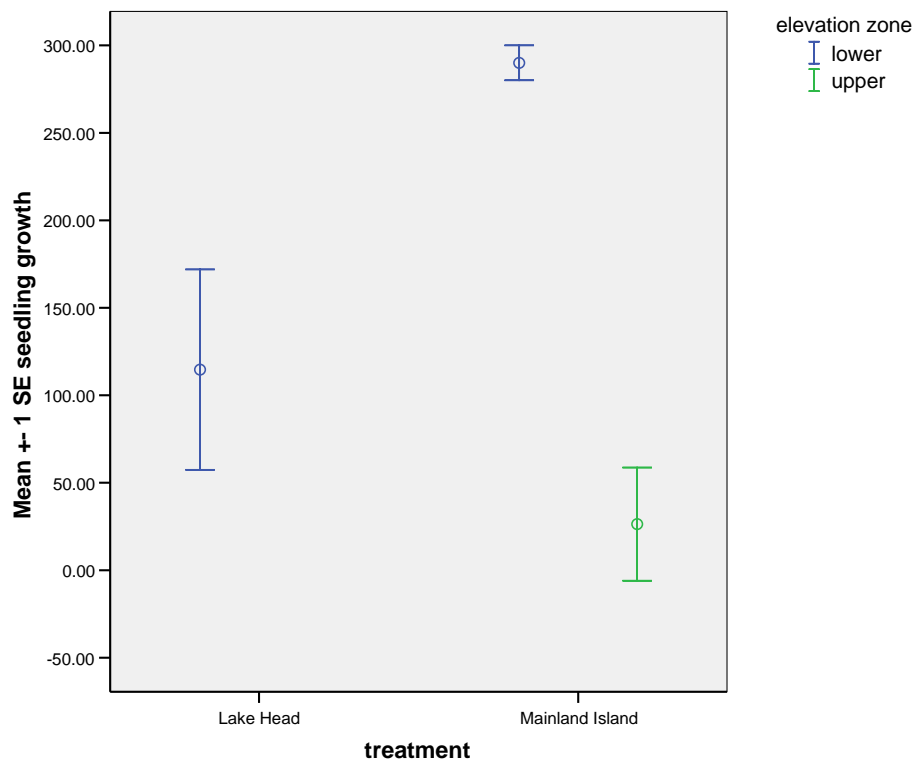


Figure 22 Mean seedling growth between 2005 and 2007 (mm)

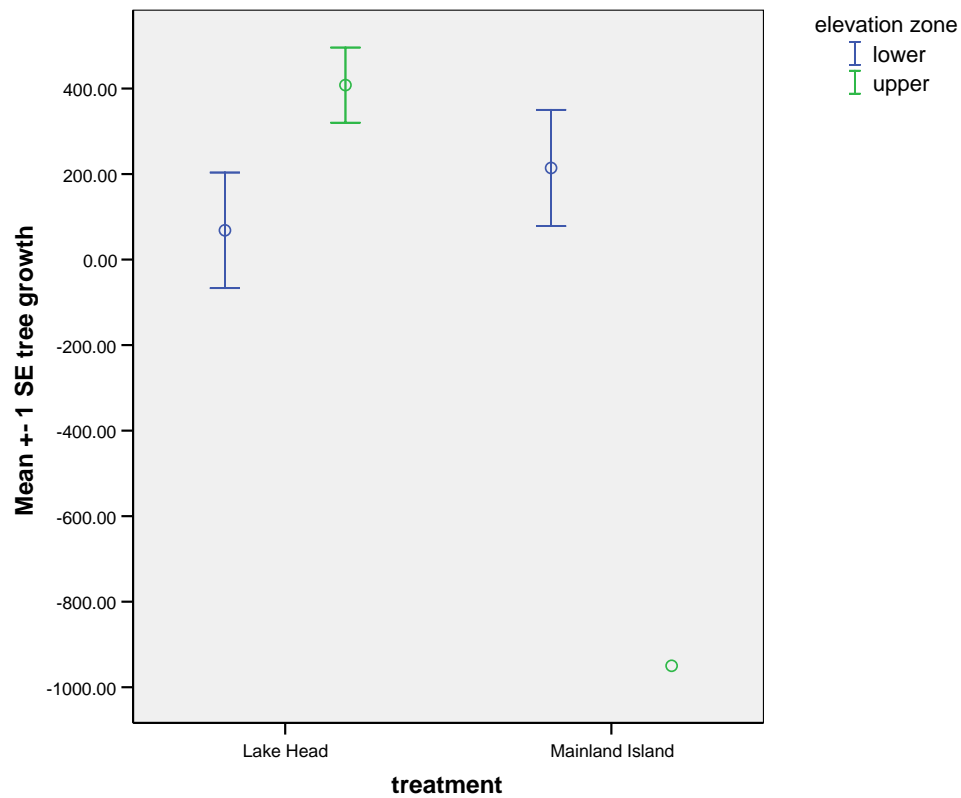


Figure 23 Mean tree growth between 2005 and 2007 (mm)

Recruitment

Seedling recruitment was sporadic and highly localised in both the reference area and the Mainland Island. Across all 34 transects 50 new seedlings were observed in 2006 and an additional 88 found in 2007. Actual count data is summarised for each strata in table 13.

Seedling recruitment 2005-2006		
	TREATMENT	
elevation	Lake Head	Mainland Island
upper	1	42
lower	4	3
TOTAL	5	45
Seedling recruitment 2006-2007		
	TREATMENT	
elevation	Lake Head	Mainland Island
upper	0	71
lower	12	5
TOTAL	12	76

Table 13 Recruitment data 2005 and 2007, the actual number of new seedlings observed

Recruitment was very localised on recently disturbed sites. In the Lake Head area between 2005-2006, 4 of the 5 recruits were found in one cluster in transect A2. In the Mainland Island area 38 of the 45 new seedlings (77%) were found in just two clusters, 14 in transect PG3 and 24 in transect TP6. The pattern in 2006-2007 was similar. All 12 seedlings recruited in the Lake Head area were in one cluster (transect E1), and in the Mainland Island area 67 of the 71 seedlings (94%) were found in a single cluster in transect PG3.

Mortality

Of the 105 plants tagged in 2005, 37 died over the next two-year period, 15 of those in one transect. Three of the deaths were of trees in 2005, eleven were seedlings, and twenty-three were saplings. Of these one seedling and

six saplings from 2005 died in the 2006-2007 year. Twenty-seven were at elevations greater than 800m. (See table 14)

2005-2007

Survey site	Height tier 2005	mortality
Lake Head	All tiers	0
Mainland Island	1 seedling <45cm	11
	2 sapling <200cm	21
	3 tree >200cm	3
	TOTAL	35

Table 14 Mortality data 2005 to 2006, the actual number of plants observed in 2005 but dead or missing by 2007.

Browse on fuchsia

Mammalian browse was noted on nine percent (n=23) of the 253 fuchsia plants seen. The nature of the browse, height above ground and hedging, were suggestive of deer browse rather than possum. Heavy to moderate browse (Figures 24 and 25) was concentrated on trees (n=4) and saplings (n=12) with hedging of saplings (Figure 9) rather than defoliation of single branches. Light browse was confined to trees (n=7) where accessible branches only were browsed. It was noticeable that no seedlings were observed as suffering mammalian browse in any transects. However many did have light insect damage.



Figure 24 Defoliation of accessible lower branches suggestive of deer browse. The time of observation (late November) was not consistent with deciduous leaf fall.

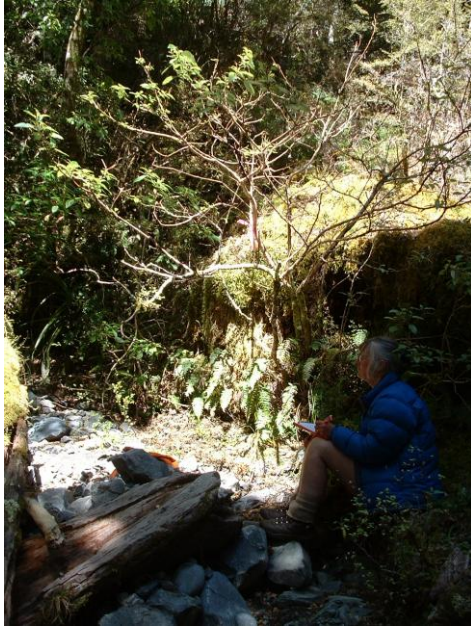


Figure 25 Heavy deer browse and dieback on a fuchsia tree. The upper branches were accessible to deer using the bank at the rear.

Geomorphic and environmental measurements

Elevation and stream profiles:

Elevation ranged between 975 and 620 m a.s.l. across all transects.

Mainland Island transects included more at higher elevations reflecting the higher elevation at which the talus slopes entered the gorges compared to the Lake Head site. Streambeds were comparable in width between the Mainland Island (11.5 m) and Lake Head sites (11 m). However they were about twice as steep in the Lake Head sites (20° vs. 12°) (Table 15).

site	stream	elevation (m a.s.l.)			stream slope			stream width (m)		
		maximum	minimum	mean	maximum	minimum	mean	maximum	minimum	mean
Mainland Island	GB	750	622	697	15	5	8	20	13	17
	LR	685	620	646	15	5	9	5	2	4
	PG	902	820	877	18	0	15	25	10	16
	TP	975	835	915	20	10	16	11	5	9
Lake Head	A	775	665	708	30	5	19	20	4	11
	B	800	685	730	25	12	18	40	20	34
	C	700	625	663	60	30	40	5	2	3
	D	716	660	692	20	12	15	11	5	8
	E	668	654	657	10	5	7	2	2	2

Table 15 Mean elevation and profile information for transects surveyed in each stream bed.

Geomorphic description:

Observations for each geographic area are summarised below in Table 16. Geomorphic descriptors follow (Milne, Clayton, Singleton, & Wilson, 1995). The data reflect the differences in topography between the sites rather than transect variation. In particular, streams “A” and “B” were on predominantly long talus slopes with wide fans at their base; these two constitute the main streams within the formal Lake Head reference site. Streams in the Mainland Island flow across gentle talus slopes and the nature of their banks reflects that, there being more gullies than gorges, and no ravines.

Geomorphic descriptor	% Mainland Island transects	% Lake Head transects
fan	6%	25%
talus slope	17%	19%
gully	67%	25%
gorge	11%	25%
ravine	0%	6%

Table 16 Geomorphic descriptions of transects

Canopy

The degree of stream cover by canopy was described as *complete* (total coalescence), *partial* (incomplete, including lightgaps), or *open* (no coalescence). Given the subjective nature of the descriptions there was a broad level of similarity between the two sites, most transects occurring in open streambeds (Table 17).

Canopy descriptor	% Mainland Island transects	% Lake Head transects
open	44%	56%
semi open	6%	11%
closed	33%	22%

Table 17. Canopy cover over transects in the two geographic areas

Temperature and humidity comparisons by elevation and aspect

The effect of aspect on temperature in stream A was for the bank with a northern aspect to warm more quickly and cool later and more abruptly than the banks with a more southern aspect. Maximum temperatures were 22°C for the northern aspect and 16°C for the southern. (Figure 26)

The influence of elevation on temperature was not as great as expected. On April 8 the mean temperature difference was 0.5°C lower at 950 m a.s.l. than at 620 m a.s.l.; this was accompanied by marked differences in diurnal humidity at the lake side (%RH dropped from 95% to 70% during the day) but not in the upper TP stream where it remained at about 90%. (Figure 27)

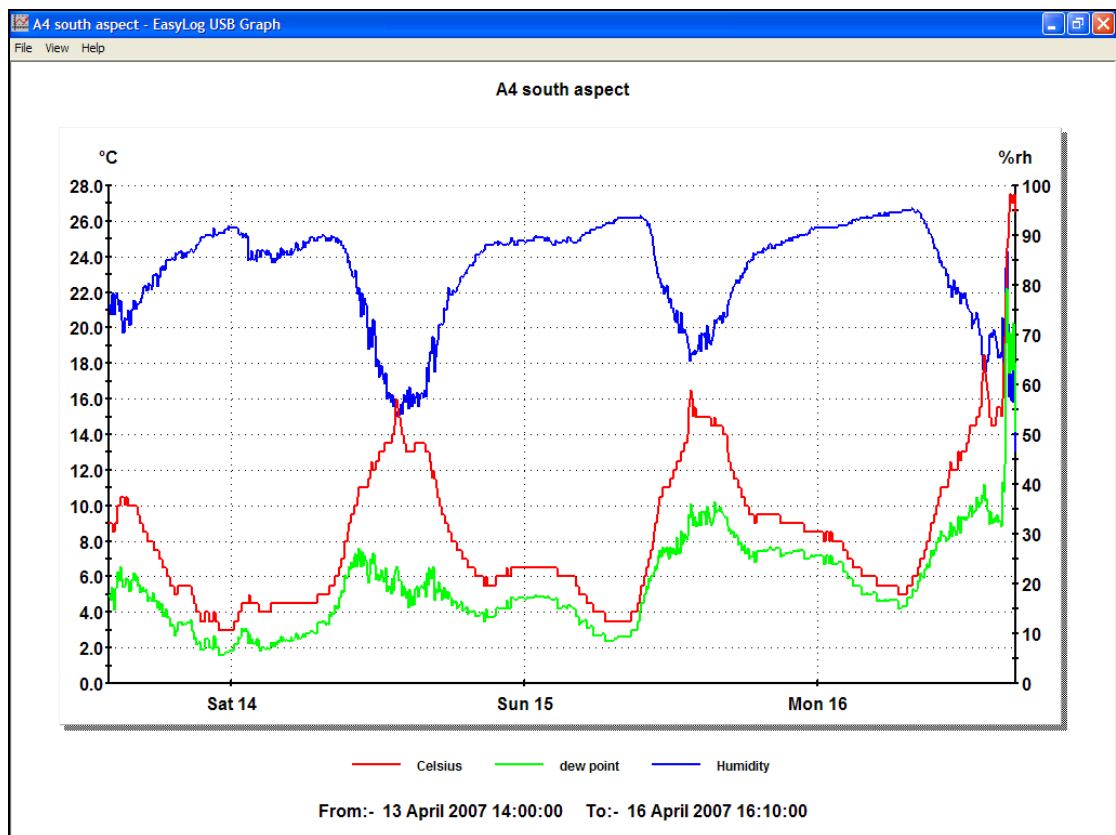
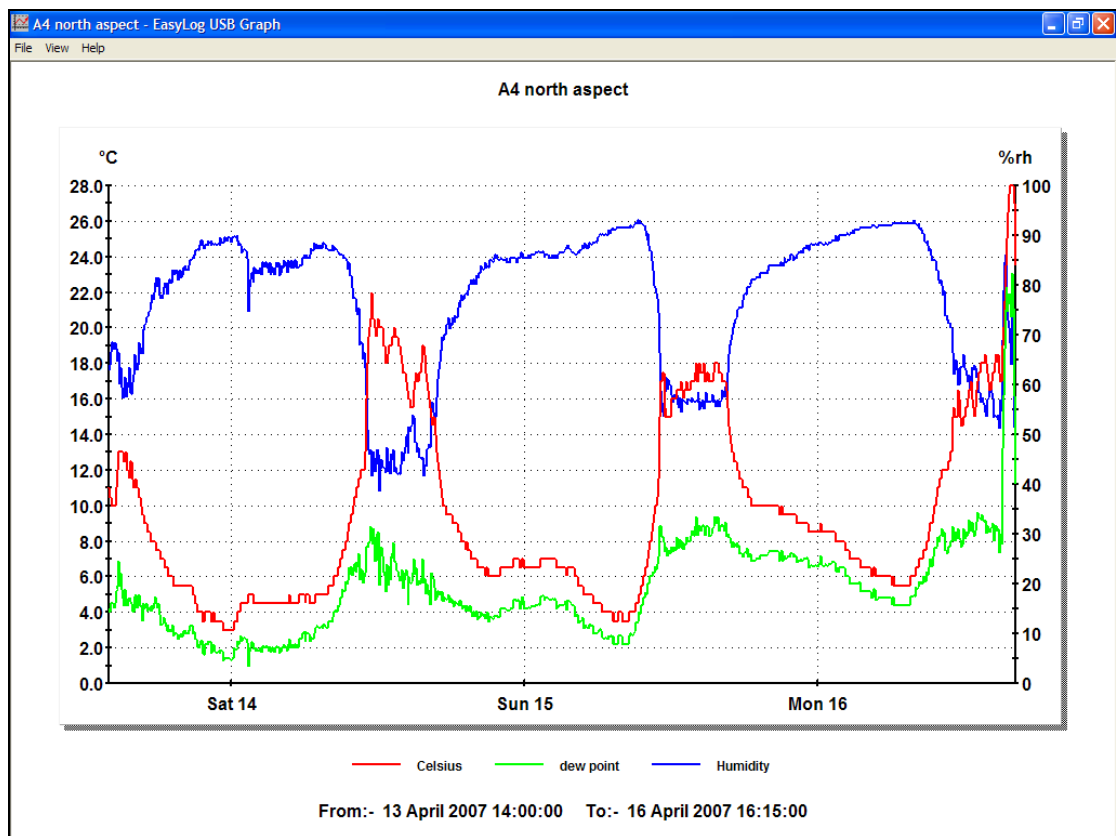


Figure 26 Temperature and humidity graph for northern and southern aspects of stream A – recorded at c. 650 m a.s.l. (Lake Head)

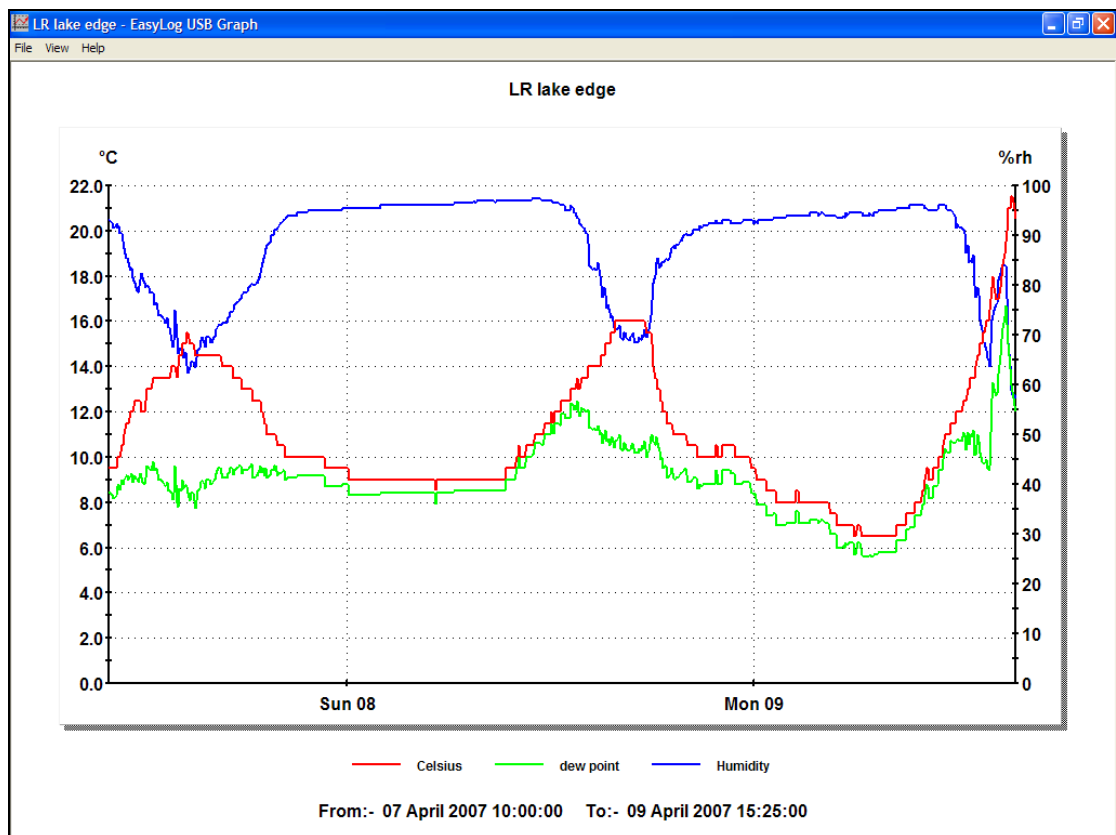
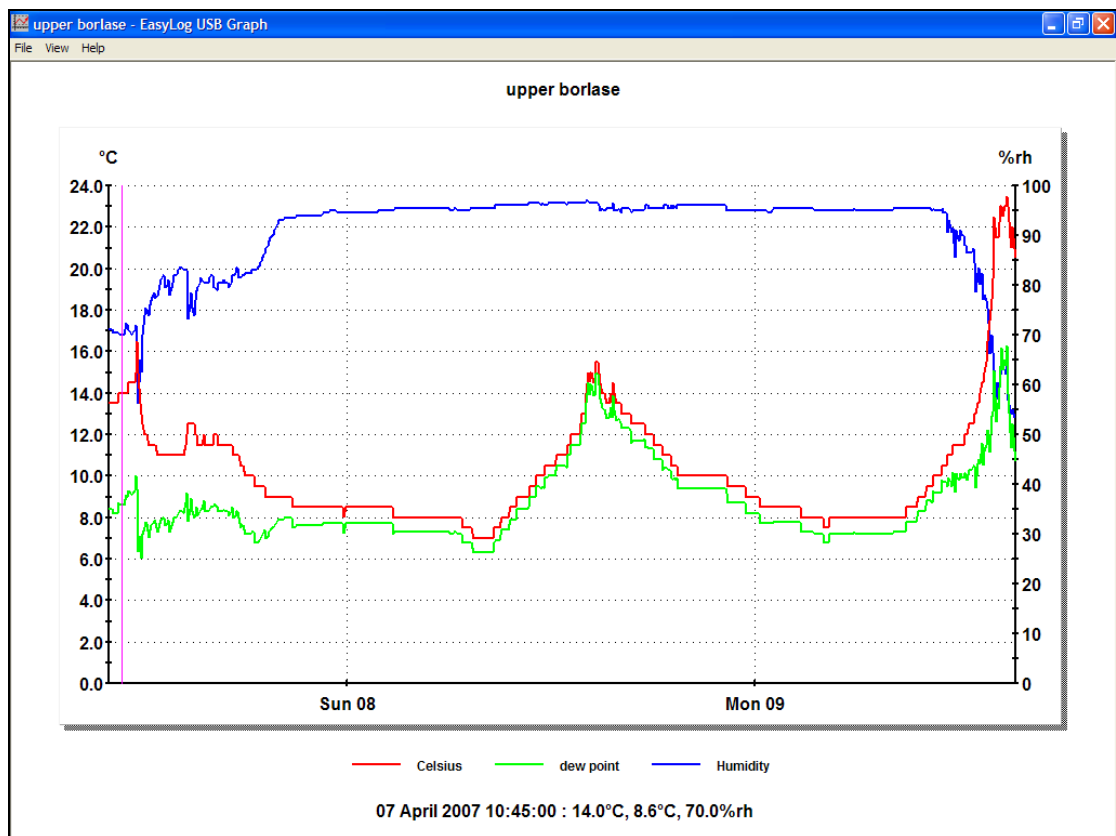


Figure 27 Temperature and humidity graph for upper and lower elevations

Germination and seedling survivorship experiment

Field germination

A total of 61 *Fuchsia* seedlings were found in the 72 plots, 42 of these occurred in one quadrat. Caged plots contained 26 seedlings, non-caged seeded plots 35, and no seedlings were found in any control plots. Only seedlings beyond cotyledon stage that could be positively identified by comparison with seedlings grown in the glasshouse were included (see figure 28). Absolute numbers of seedlings were low and widely distributed, and suggest low overall germination rates. (See table 18)



Figure 28 seedling of *Fuchsia excorticata* in lower "PG" stream

Management area	% plots containing <i>Fuchsia</i> seedlings	Mean % germination
Lake Head	8.3% (n=3/36)	0.28% (n=5/1800)
Mainland Island	22.2% (n=8/36)	3.11% (n=56/1800)
Overall	15.3% (n=11/72)	1.69% (n=61/3600)

Table 18 Field germination quadrat result summary

Data were $\log(n+1)$ transformed to conform to Levene's test of equality of error variances and analysed for means differences using multivariate analysis under the General Linear Model. No significant differences ($p < 0.05$) between management areas were found when comparing the number of plots containing germinated seeds or percent germination. Similarly no significant differences were found between soil texture groups, elevation, or canopy cover. Given the data was not normally distributed,

nonparametric tests were applied. Neither the Mann-Whitney Test nor the two-sample Kolmogorov-Smirnov statistics indicated any significant differences between treatment areas.

Given that so few seeds germinated the second phase of monitoring browse using the exclosure cages was aborted.

Seed viability trial

Overall 52% of the 600 seeds sown germinated over a 10 day period; 73% of those on perlite germinated, compared to 25% on the sand and 6% on humus. Germination in all cells began 13 days after sowing and continued at a declining rate for 10 days. Seeds germinating in humus took about 10 days longer to become visible through the litter and fewer germinated over the 10 day period. (See figure 29)

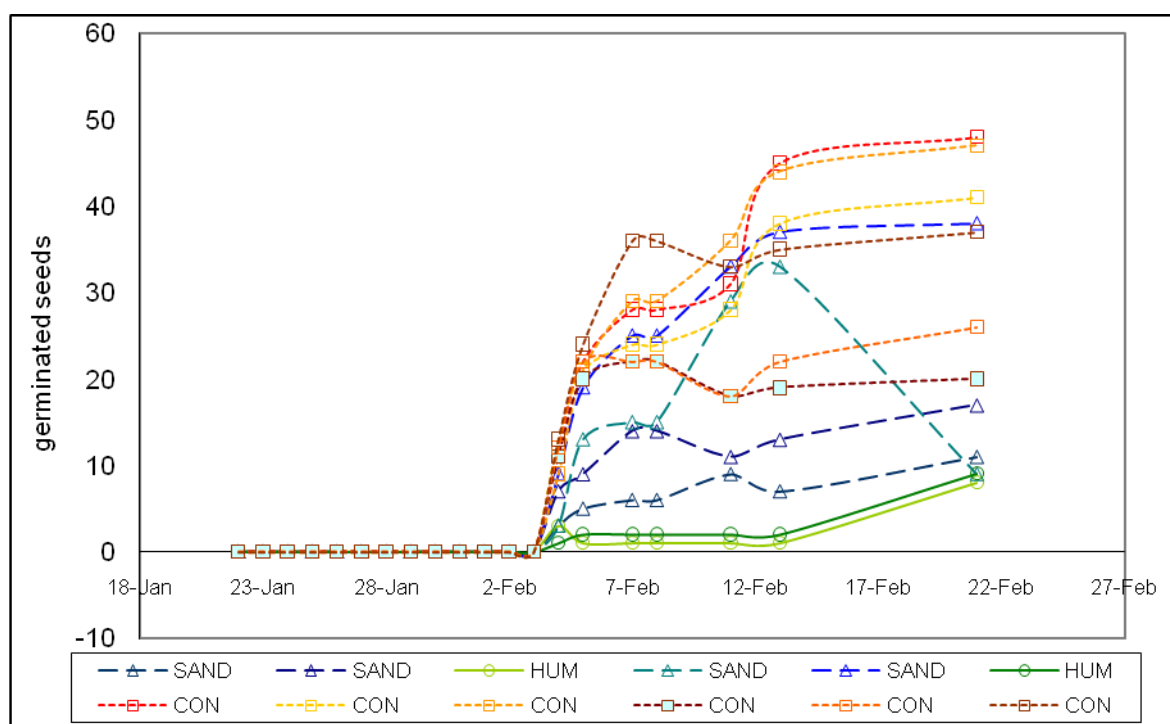


Figure 29 viability trials for germination success of Fuchsia seeds used in the field trials 2007
[50 seeds were sown in each cell, control substrate was perlite (CON), river sand (SAND), and forest humus (HUM) were trialled]

Chapter 4 Discussion and conclusions

Introduction

Overall this study confirmed the impression inferred from Merrifield's earlier data (2000) that fuchsia distribution across the study area was sparse, clustered, and had few adult trees greater than 2 m. The data provide early indications of increasing fuchsia abundance across the whole study area, although not at the level of statistical significance. With the exception of sapling abundance there were no significant differences in any population parameters between the managed and reference areas. The influence of elevation on distribution patterns is complex; different patterns being observed in the managed and the reference areas. There are significant differences in growth rates and mean population heights related to elevation. In commenting on the original hypotheses, the discussion that follows is structured around potential influences affecting the distribution patterns and trends seen in the data.

The ecological significance of distribution patterns is discussed with reference to management and background environmental factors. Management practices are considered in relation to the influence of pest animals, and avian pollination and dispersal vectors on fuchsia distribution in the managed and reference areas. Environmental influences are discussed in relation to elevation, physiognomy, periodic flood events, and microsite factors. Germination, recruitment, and mortality data are reflected on in relation to the paradox of riparian gap dynamics.

Issues of design for monitoring of riparian vegetation responses to pest management are discussed. With no replication at the ecological management scale, no direct causal relationship between population changes and management could be substantiated. Disparity in the physiognomy of the managed and reference areas led to difficulty in drawing defensible conclusions. The decision to use the "woody species

regeneration method” (Winward A.H., 2000) for sampling a sparse, clumped population is critiqued.

Distribution patterns related to management practices

While it was recognised that no direct causal relationship between population changes and management could be substantiated, it was anticipated that fuchsia may have benefited from control of possums, rodents, and mustelids in the Mainland Island area. Studies of *F. excorticata* following possum management operations elsewhere in New Zealand suggested that significant reductions in mortality and herbivory levels were achieved. (Atkinson I.A.E., 1992; Smale M.C. et al., 1995)

Operations targeting management of rodents and mustelids have led to increases in populations of birds known to be pollination and dispersal vectors for fuchsia (Paton B.R. et al., 2007). Related improvements in fuchsia fecundity through better pollination rates, and in seedling recruitment through increased fruit dispersal, were expected.

Abundance

Fuchsia excorticata had a patchy distribution over the whole study area giving rise to high variability in transect data. Clustering, especially of seedling and sapling classes, was marked. Although surveying about a quarter of the available streambed on the talus slopes the number of individual plants was small and presented a heavily-skewed distribution with many transects containing no fuchsia at all; highest densities were found in seedling clusters on recently disturbed sites. Of the 253 plants found in this survey 110 were found in just three 10m sized clusters.

Overall abundance

No statistically significant difference in overall abundance was detected between the managed and reference geographic areas. The mean abundance of *Fuchsia excorticata* across the whole study area of 3.09 ± 0.93

(mean \pm 1 SE) plants per 100m streambed in 2005 had by 2007 increased to 6.36 \pm 2.99. Although below the level of statistical significance the trend in the data suggested an accelerating increase in abundance each year in both geographic areas; and an increasing divergence in fuchsia abundance between the Mainland Island and Lake Head areas. Overall the abundance appeared to rise more rapidly in the Mainland Island, though the findings were suggestive rather than conclusive.

Saplings

The only statistically significant *Fuchsia excorticata* population difference between the managed and reference areas was in sapling abundance. In each of the three surveys a higher abundance of saplings was observed in the Mainland Island. In the Mainland Island the mean number of fuchsia saplings per 100 m varied between 1.72 \pm 0.06 (mean \pm 1SE) in 2006 and 2.61 \pm 1.43 in 2007, while those in the Lake Head area varied between 0.50 \pm 0.24 in 2005 and 2006, and 0.31 \pm 0.20 in 2007. Sapling abundance was significantly greater in areas under intensive pest animal management, though again it could not be confirmed that the level of management was the cause.

Sapling abundance is considered ecologically significant since recruitment into the reproductive population is dependant on establishment and medium-term survival of saplings for long enough to reach reproductive maturity. Given the hedging seen in saplings, ostensibly by deer, it is possible that herbivory may hinder the transition from sapling status to reproductive adult. The majority of saplings observed were found clustered in sites vulnerable to flood events. Few were either above regular flood level or protected by rock outcrops or crevices. As discussed later, bank collapse has been observed to remove large clusters of sapling and seedling sized fuchsia following one flood event. Monitoring the survival and growth of this life history stage is considered by the author to be important for understanding the dynamics of riparian fuchsia.

Seedlings

Increase in the relative abundance and survivorship of seedlings increases the probability of future increase in reproductive plants and accelerated growth in the population. Only six seedlings were found in the Lake Head transects in 2005 compared to twenty-four in the Mainland Island transects. By 2007 the figures had increased to one hundred and sixteen in the Mainland Island and twenty-three in the Lake Head transects. The majority of seedlings were clustered on recently disturbed sites and it may be ecologically significant that there was less bank collapse and tree-fall observed in the Lake Head gorges than in equivalent Mainland Island streams. Although ten fruiting trees were found in Lake Head stream D, only two seedlings were found in the three annual surveys of the same stream. Stream D flowed through a narrow rocky gorge and was not prone to bank erosion during flooding. By contrast in another Lake Head stream (stream A), where one fruiting tree was found, six new seedlings were seen between 2005 and 2007 all located on disturbed sites. Seedling survivorship is often low, therefore recruitment from seedling to sapling stage is demographically significant (McAlpine K., 2005). Over the duration of this study of the twenty-four seedling in Mainland Island transects in 2005 eleven (46%) had died by 2007, the majority following the flood of Easter 2005.

Trees

The abundance of fuchsia trees in the study area was low but stable. In this study "tree" was defined as greater than 200cm high to accommodate all plants found to flower or bear fruit, reflecting the demographic significance of the height tier. Fourteen fuchsia trees were found in the reference area at the start of surveying in 2005 (0.82 per 100m streambed) following widening of the reference search area to incorporate six new transects in three streams not previously surveyed by Merrifield in 2000. Eleven were found in the managed area (0.61 per 100m streambed). Over the duration of the study three saplings grew to tree stature in the managed area transects and two in the reference area. During the same period three trees died in the managed area and none in the reference area. Increase in abundance of fuchsia trees as a result of improved seedling and sapling abundance, and in

the absence of major stochastic flood events, would seem to be a reasonable prognosis.

Possum and ungulate management

It is likely that that impact of possums on fuchsia in both the Mainland Island and the reference area has been low. Whereas decline in possum populations in other NZ areas were often marked following control operations, those in the Rotoiti Nature Recovery Project (RNRP) area were less so. Beech forests typically have low possum densities. In 1997, prior to intensive possum control, the residual trap catch (RTC) data were similar for both the RNRP and reference areas (6.5% and 8% respectively). Within a year the RNRP RTC had dropped to about 1% and remained there until the start of this study in 2005. During the same time interval the Lake Head reference area RTC fluctuated around 6%; not much higher than the level of 5% set as a target for most Department of Conservation possum control operations. From 2005 the RNRP reported possum indices in terms of percentage of kill traps catching possums. Again indices for both the managed and reference areas were low (0.5% and 1.75% respectively), thus in both areas the vegetation had been exposed to broadly similar levels of impact from possums since 1998.

Most fuchsia monitoring following possum control operations has targeted adult trees and focused on foliar browse or browse induced mortality (Ulrich S. C. and Brady P. J., 2005). Fuchsia populations in the study area had few adult plants and foliar browse was localised and generally at low levels. The increase in fuchsia abundance suggested across the study area appeared to be due to increases in seedling recruitment, rather than reduction in adult and sapling mortality resulting from reduced herbivory. Mammalian herbivory to fuchsia in the study area is limited to taller saplings and more accessible branches of a few trees. Where browse has been observed it was generally attributed by the author to deer rather than possums, and has not been seen to lead to death of any plants over the course of three annual surveys.

The nature of the leaf-petiole remains and lack of debris on the ground are characteristic of deer browse not possum; fresh deer spore and faeces were noted by the author in three stream beds; and browse was limited to the height red deer could easily reach. Deer impact, monitored by RNRP staff through measuring *Griselinia littoralis* canopy density, had increased consistently from 2002 to 2006; while possum browse on *Raukawa simplex*, considered the most palatable species in the management area, had declined (Paton B.R. et al., 2007). It is significant that red deer are not actively managed in the study area. A single deer could traverse a section stream bed quite quickly if not disturbed, and target easily accessible palatable species, including fuchsia. Thus even low numbers of deer may be capable of inflicting substantial browse damage on the scattered and low number of fuchsia trees and saplings. The impact of either deer or possum browse may become more ecologically significant if the number of adult fuchsia trees increases, or the deer population surges.

Predator management

Predator management and subsequent increase in the silvereye and particularly bellbird populations in the Mainland Island area (Paton B.R. et al., 2007) may have been instrumental in improving fruit dispersal. Increase in seedlings recorded in Mainland Island transects of forty-five in 2005-2006 and seventy-six in the 2006-2007 year, contrasts with the increases of five and twelve in the Lake Head reference area for the same time period. Bellbird indices were significantly higher in the managed area (3.5 per count in 2005 and 4.6 in 2006) compared to the reference area (2.2 per count in 2005 and 2.9 in 2006). Anecdotally the author observed in excess of forty bellbirds in the lower PG stream (not monitored by transects) during the summer of 2006; and mapped six adult fuchsia in flower and fruit and 695 seedlings within the first 300m of the same stream. It should also be noted that limited predator management (stoat trapping) was carried out in the Lake Head area during the course of this study.

Summary of management considerations

Fuchsia abundance appears to be increasing across the whole study area, with the higher rates in the managed area. The size-structure of fuchsia populations at both the Mainland Island and Lake Head sites appeared to be trending toward a higher juvenile to adult ratio. It is suggested that changes in the height-class structure of populations will further accelerate changes to abundance at least at lower elevations. Further study will be needed to confirm this, and the ecological significance of any changes in abundance or growth rates will require monitoring over a longer time-frame.

The ecological impact of possums on fuchsia appears low in both managed and reference areas. Deer browse is causing hedging of saplings in some areas, preventing these plants from reaching reproductive size, but the current scale of the impact is small and localised. Where fuchsia trees are accessible to deer, especially in more secluded parts of the Lake Head area, deer browse is leading to death of some branches and reducing flower and fruit production. Increases reported in bellbird and silvereye populations may be influential in the increasing recruitment seen in both managed and reference areas.

Distribution patterns related to environmental factors

When mean heights of fuchsia were related to elevation there was a curious inversion in the pattern between managed and reference areas. In the managed area taller plants occurred at low elevations, while the converse held for the reference area. Growth rates followed a similar trend, with rates significantly higher at low elevations in the managed area but a suggestion that the reverse occurred in the reference area. Most of the difference was due to sapling growth, with seedling and tree data being inconclusive.

Elevation related patterns

Across the whole study area fuchsia populations at higher elevations were significantly shorter than those at lower elevations. In the Lake Head area, the trend toward shorter plants at higher elevations was consistent with

species approaching the limit of their elevation range, compounded by low abundance of adult plants and more seedlings and saplings in the upper reaches of streams. By contrast in the Mainland Island site the reduction in mean height at lower elevations, may have reflected increasing recruitment in lower regions of the streams rather than the height of adult plants.

Secondly, the sapling growth rate was higher at lower elevations in the Mainland Island, though the reverse was true in the Lake Head area. In the Mainland Island area fuchsia were smaller and grew more slowly at higher elevation than in the lower streambeds. It is not unexpected that stature and growth rate are lower at higher elevations. The influence of lower mean temperatures associated with higher elevations being a likely to reduce growth rates. It was, however, of interest that the same trend was not apparent in the Lake Head data.

temperature & humidity

With respect to increasing elevation, air temperature is influenced by the lowering air pressure at an adiabatic lapse rate of around 0.6°C decrease for every 100 vertical metres. Thus for the 300 m difference in elevation, between the lake and the upper sampling points, a temperature difference of about 2°C was anticipated. The actual difference was less than this, being only 0.5°C lower at 950 m a.s.l. than at 620 m a.s.l. I suggest that this reduced the anticipated influence of elevation on growth rates. There was a marked diurnal difference in humidity at the lake side (%RH dropped from 95% to 70% during the day) that was not evident in the upper TP stream where it remained at about 90%. A corollary to this is the frequent occurrence of orographic cloud at higher elevations which could potentially maintain higher soil moisture levels, though these were not measured.

As expected on the basis of the adiabatic temperature gradient, sapling growth rates at lower elevations in the Mainland Island were higher than at higher elevations. It was unexpected however that the reverse trend should be found at Lake Head. That higher growth rates occurred at higher elevations in the reference area could possibly be due to differences in physiography of the lower streams. There is increased shading and reduced

temperature relating to the deeply incised nature of the southern streams and the proximity of Mt Robert reducing exposure to late afternoon sun at lower elevations. Streams “C”, “D” and “E” have substantial sections under full canopy further reducing the amount of light and the temperature. This difference may have been further exacerbated by the scarcity of fuchsia at higher elevations in the Lake Head area

Disturbance and Flooding

It was noted by the author that, as expected in a secondary coloniser, seedlings and saplings in particular were heavily clustered on recently disturbed sites. Four such 10 m clusters held 47% of the 253 plants found overall. Two clusters in PG stream contained 81 plants, one in stream TP held 29 plants, and in stream E one cluster accounted for 10 of the fuchsia found in the Lake Head area.

Periodic heavy flooding caused localised bank collapse and high mortality in the seedling and sapling classes. Mortality among adult plants was low, especially in the Lake Head area, and there was a steady trend toward annually increasing overall abundance of fuchsia, especially in the Mainland Island area which, while below the level of statistical significance, was encouraging.

At the Lake Head site many plants in streams D and E were adults growing above flood lines in rock crevices. There was no recorded mortality among these. In the Mainland Island site the two main causes of mortality or damage to fuchsia were burial by scree debris (figure 30) and bank collapse (figure 31). In many streams bank collapsed and tree fall was extensive, exacerbated by an extreme flood event in April 2005. In one stream about half of the plants tagged in the 2000 pilot were lost through flood damage. This is consistent with damage through stochastic flooding events being reported as a major influence on riparian vegetation dynamics (Auble G.T. and Scott M.L., 1998; Bendix J. and Hupp C.R., 2000; Casanova M. T. and Brock M. A., 2000; Hupp C. R., 1982; Scott et al., 1997). It is worth noting that several Fuchsia saplings and adults recorded as “dead” in 2006 were relocated in 2007 with epicormic growth from any parts in contact with soil.

Some of these plants had been stripped of bark by sand action in the 2005 flood and appeared as bare uprooted stems.



Figure 31 Stream bank erosion in April 2005



Figure 30 Burial of riparian habitat by scree movement during flooding

As well as causing substantial habitat loss through stream-bank collapse or burial, extreme flood events created localised areas of freshly disturbed soil. Subsequent germination was clustered heavily on these sites. The two most common were silty slumps at the base of new bank-falls, and exposed root-balls following tree-fall across streams. Bank-falls accounted for between 22 and 26 percent of seedling and sapling sites from 2005 to 2007, and exposed root-balls for between 4 and 22 percent. This disturbance cycle of flood induced mortality and germination habitat creation is discussed further in the section on germination, recruitment and mortality.

Recruitment, growth and mortality

Insufficient data were available to fully analyse differences in recruitment, herbivory or mortality; however several patterns were becoming evident. Recruitment and mortality rates showed no significant differences between the managed and reference areas. Germination rates appear very low for fresh seed sown in-situ, and the number of flowering and fruiting adult plants was small. Despite this there was conspicuous but highly localised seedling appearance on recently disturbed sites. Paradoxically, these were the areas of highest mortality during major flood events.

Germination

Germination rates recorded from experimental seeding were less than 2% of the 3600 fresh fuchsia seeds planted in riparian plots. This contrasts with a rate of 52% for 600 seeds planted under laboratory conditions; and with the abundant seedling numbers seen following bank disturbance in proximity to fruiting adults.

The occurrence of germination sites along transects in both managed and reference areas was sporadic and clumped. Where seedlings did occur they were often prolific and, as anticipated with a secondary coloniser, associated with disturbed sites. It was interesting that some fuchsia seedlings were recorded in locations quite distant from the nearest known fruiting adult. In the Lake Head transects enclosed by narrow gorges or ravines, few seedlings or saplings were recorded and those that were found were often isolated.

Recruitment

Despite surveying about quarter of the available talus slope stream bed there were insufficient data to provide reliable estimates for comparing recruitment rates between the managed and reference areas. It was however noticeable that there was a trend of increasing numbers of recruited seedlings each year in both areas. The number of recruits observed was in the order of 1.7 recruits per 100 m of Lake Head stream bed in the year 2005-06, to 9.7 per

100 m of stream bed in the Mainland Island area in the year 2006-07. Most recruitment was clustered on recently disturbed sites following the floods of April 2005.

Mortality

Mortality data were similarly inconclusive although 37 of the 105 plants tagged in 2005 were dead or missing by 2007. It was noticeable that none of the plants tagged in the Lake Head streams in 2005 died over the two years studied, and that 60% of the deaths occurred in just two Mainland Island transects where banks collapsed taking whole clusters of seedlings and saplings with them. Most of the mortality occurred in the 2005-2006 year, only five being lost between the 2006 and 2007 surveys. Despite this there was a trend toward increasing seedling abundance, though their survival in a flood prone riparian habitat is not assured and is likely to be low in the medium term (c. 5 to 10 years).

Most adult fuchsia were found above the 2005 flood level and mortality was low in this height class, the few that died were as a result of substantial bank collapse. The impact of mammalian herbivores seems low in both the managed and reference areas. Deer rather than possums appear to be causing the most damage. None of the trees monitored were killed through deer or possum browse, though some branches accessible to deer were defoliated and remained leafless during subsequent spring surveys. The influence of herbivore management appears to have little effect on adult fuchsia abundance in the short term.

The riparian coloniser paradox - Gap Dynamics

Fuchsia seed is small and the fruit known to be bird dispersed (Burrows C., 1995). Given the mode of dispersal, clumping of seed is likely under perching trees; if this coincides with disturbed substrate and adequate light and moisture conditions germination and subsequent establishment are more probable. It is paradoxical that these conditions are predominantly found in flood prone stream banks which are also the most vulnerable sites. In a forest gap cycle there can be decades between tree-fall events and gap

regeneration proceeds slowly in a stable environment. In riparian systems stochastic flood events can occur at more frequent intervals, and often little time is available for establishment of new seedlings or transition into the reproductive population. Those few seedlings that either grow above the flood-level, or are protected by natural barriers such as rock outcrops, are key to the reproductive stability of the population. It is suggested that future studies of sparsely distributed riparian *F. excorticata* consider applying meta-population theory to demographic changes.

Conclusions

Hypothesis: That there would be significant differences in abundance, recruitment, growth and mortality between the *F. excorticata* populations at Lake Head and in the Mainland Island.

The only statistically significant difference between *F. excorticata* populations in the Mainland Island and the Lake Head references areas was in sapling abundance. The trend in the data was toward an accelerating increase in fuchsia abundance in both areas, the trend being more marked in the Mainland Island area. There were insufficient data to draw any valid conclusions regarding recruitment or mortality, though again there were trends toward increasing seedling recruitment and low adult mortality. Possum damage appeared low but deer browse was evident and could become a problem in numbers rose. The rise in bird populations following predator management, particularly bellbird and silvereyes, could be partially responsible for increases in dispersal and elevated recruitment.

As discussed later, the overriding issue with outcome monitoring of ecological management projects stems from a lack of replication at the scale of management, and no statistically valid conclusions can be reached regarding causal effects of management practices.

Hypothesis: That there would be temporal variation related to changes in pest animal and avian populations, and the occurrence of disturbance related to flood events.

That severe damage to riparian habitat was caused by major flooding in 2005 and 2006 was self evident. The impact on fuchsia mortality in the area was restricted largely to pockets of seedlings and saplings in exposed sites which were either washed away following bank collapse, or buried in mass scree movement seen on the debris fans. Flood related adult mortality was low. The creation of newly exposed silty deposits at the base of collapsed banks and wind-thrown trees with exposed root-balls, provided excellent conditions for germination of fuchsia. Subsequent seedling establishment was markedly clustered on these sites.

The timeframe of the study and lack of management replication precluded drawing any statistically significant conclusions relating to long-term trends resulting from pest animal and avian population changes.

Hypothesis: That there would be differences in fuchsia populations related to elevation.

F. excorticata population stature, mean height, and growth rates are significantly correlated to elevation. In the Mainland Island area plants are taller at higher elevations, and the growth rate lower. The converse is true for the Lake Head reference area. Although seemingly trivial, the influence of small differences in elevation on *F. excorticata* population patterns is considered by the author to be important when selecting future monitoring sites in the area. Changes in physiognomy associated with increasing elevation become marked as the talus slope abruptly reaches the bedrock. The streams become narrower, bound by gorges and ravines, and under coalescing canopy, the higher the elevation. It is more likely to be influences stemming from these physiognomic difference than those of directly elevation related climatic differences that affects fuchsia.

Critique of methodology, and recommendations

Monitoring of patchy riparian populations

Lack of sufficient environmental homogeneity of the sample streams, patchy and clumped distribution of fuchsia, and difficulty in finding comparable non-treatment reference streams all contributed to difficulties encountered during analysis. An indication of fuchsia distribution, population structure and dynamics was able to be achieved from the sampling regime used; however the statistical power of the evidence was compromised by the methods used. Key deficiencies in the riparian survey methodology relate to transect length and placement, handling of variation in physiognomy between streams, and selection of the non-treatment reference site.

The general sampling approach, using the modified “greenline” method (Winward A.H., 2000) combined with tagging, biometric and environmental measurement, was useful in the field; all transects were relocatable, and individual fuchsia plants were reliably recognized for repeated biometric measurements. The thirty-four 100m transects did not appear to adequately represent demography or distribution of an uncommon, widely-dispersed species. Where the distribution of the target species is such that adult plants are few and sporadically and widely dispersed, and seedling and saplings localised and clustered, the transect length and spacing need to reflect this. Having traversed the entire length of each study stream, I was left with a visual impression of disparity between the transect data and the overall representation of actual fuchsia distribution, especially of seedlings. I would suggest separate protocols for monitoring different life stages in future studies. Recording of individual trees along entire stream beds would be possible with the low numbers encountered. Sampling of seedlings and saplings by monitoring cluster size rather than individual plants, combined with longer transects and shorter inter-transect spacing, would yield sufficient data for monitoring demographic changes.

Timeframes for monitoring temporal change in populations

The time lag between the original pest management and different aspects of fuchsia response will vary. Differences in foliar browse may occur rapidly while changes in population structure especially in the sapling to tree transition may be much slower. The ability of transect sampling to detect fine scale changes in flood-prone riparian habitat is poor. It is suggested that resurvey periods be governed by frequency of major flooding, growth rates of the target plant, and any changes to the pest management regime.

Deer browse – use of riparian exclosure plots

In a study comparing red deer and possum diets in mixed podocarp-hardwood forest, (Nugent G. et al., 1997) found that while deer consumed a lower proportion of the total annual foliage production (1.1%) than did possums (3.3%), they were much more targeted. As a consequence of concentrating their browse on palatable species in the shrub tier, where saplings of preferred species were found, the impact of deer on forest structure and composition was seen as more significant in the long term. Nugent (1997) expressed concern that deer were likely to reduce the abundance of palatable species through preventing recruitment from the shrub to the tree tiers more than possums would. The implication of deer in modifying forest communities has been evident from many studies (Allen C. et al., 1997; Mark A.F. et al., 1991). The nature of browse damage seen in the study area (pers. obs.) was consistent with deer feeding. Although no direct sightings were made deer-spoor were seen in the streambeds surveyed on four occasions (pers. obs.). With the currently low priority given to deer management it is suggested that investigation into the damage to fuchsia in riparian beech forest by possums relative to deer would be of value. Construction of riparian deer exclosure plots could be considered.

Selection of project specific indicator species

Use of fuchsia as one target species for monitoring effectiveness of pest management is defensible on the grounds of its sensitivity to mammalian

browse (Payton I.J. et al., 1999). Possum control results in a general reduction in herbivory, though this is likely to be more noticeable in palatable species such as the mistletoes, fuchsia, and wineberry. While not the explicit target of management, fuchsia could benefit from lower possum numbers through reduced herbivory at all life history stages and lower frugivory; there is no evidence, however, to suggest that seed is less viable having passed through a possums digestive system, and possums could be dispersal agents. Foliar browse on adult trees is well implicated in their decline (Atkinson I.A.E., 1992; Pekelharing C. et al., 1998b; Ulrich S. C. and Brady P. J., 2005), and it is possible that some of the hedging reported in this research (pers. obs.) could be attributable to possum browse slowing recruitment from sapling stage to reproductive adults. Measurable reduction in foliar browse on adult fuchsia, seen within a year following possum control in the Tararua Range (Ulrich S. C. and Brady P. J., 2005) and Kapiti Island (Atkinson I.A.E., 1992), would be difficult to measure in the Rotoiti situation where there are few adult trees and they are widely dispersed. Changes in seedling and sapling browse were not measured in the Tararua programme where adult plants were still relatively common in the target area. In the case of Kapiti Island, there were so few adult fuchsia trees that by the conclusion of the possum eradication programme they were individually known and monitored in a complete census (Atkinson I.A.E., 1992).

Predator control programmes have *inter alia* the objective of enhancing breeding success in native bird populations. Stoat control has a direct impact on the population dynamics of kaka and bellbirds, but it also indirectly enhances the pollination and dispersal of many fruiting plants (both indigenous and alien). It needs to be recognised that the lag time between a pest management operation and putatively associated changes in the indigenous flora and fauna may be different and lengthy. Subtle short-term shifts in population structure are difficult to diagnose in complex natural environments.

In the case of fuchsia in the Rotoiti Nature Recovery Programme the proposed links to pest management practices are indirect; there is no specific

intervention for fuchsia itself, such as direct protection or translocation. Increase in populations of known pollinating and dispersing species could indirectly enhance fuchsia fruit set and increase dispersal. Any physiological response in fuchsia fecundity would be related to both reduced browse and improved pollination. Since these factors operate with different lag times, any causal links between them and improved fruit set is likely to be complex. Overlay this complex interaction with natural climatic variation, stochastic flooding and drought, and the design of appropriate and robust monitoring methods is difficult. To monitor changes in fecundity by random sampling in a statistically robust way would require a larger pool of adult fuchsia than has been observed by the author. Given the low number of adult fuchsia in the study area, it is suggested that monitoring of the fuchsia reproductive response to different levels of pest management would be better done using a census of flowering and fruiting levels on all known adults. It is also imperative that there is replication at the management scale to verify any causal relationship.

In the context of monitoring a single species response to pest management programmes, account needs to be taken of all possible interactions between the pests and the proposed species either direct or indirect. Also to the sensitivity of that species to those interactions, the geographic scale of the programme, and of the spatial and age distribution of the proposed species need to be considered. In the case of using fuchsia at Nelson Lakes an argument based on it's sensitivity to pests is justified; however I suggest that there is an issue associated with the population age structure, and its spatial distribution, that causes the author to question the use of fuchsia here as an indicator species of ecosystem recovery. There are insufficient adult plants and, although wide, their distribution is highly clustered.

Spatial replication, and the need for ecologically robust reference areas

Monitoring ecological outcomes resulting from pest management practices is recognised as a critical part of any management programme. (Saunders A., 2000; Smale M.C. and Meurk C.D., 1995). The issue of what to monitor as indicators of improvement in natural systems is related closely to the aims of individual programmes, as is the integrated nature of both pest management practices and their ecosystem outcomes.

To monitor the effects of management practices, survey replication is needed at different scales. Locally there needs to be valid replication between treatment and non-treatment areas, but replicates are also needed at the management scale between projects. Some of the difficulties associated with analysis and interpretation of the data were related to the physiographic disparity between the Lake Head and Mainland Island sites. The two areas were not sufficiently similar to allow an unequivocal matching of transects, especially at higher elevations. Much of the variation in fuchsia demographics could potentially be attributed to differences in physiography or canopy cover, and selection of a more comparable experimental control area would be recommended for future studies. There is little that can be done to mitigate the effects of diverse stream environments except ensure representative sampling through appropriate stratification. The spatial distribution and clustering of fuchsia around a few disturbed sites gave rise to heavily skewed transect data. Many transects contained no fuchsia at all, most had low numbers, and several contained large numbers of juveniles creating a bimodal distribution. Use of analytical methods better suited to this configuration of data is recommended.

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