Dieback and recruitment of the forest dominants *Nothofagus fusca* and *Libocedrus bidwillii*, central North Island, New Zealand

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ABSTRACT

A field investigation to determine the likely causes and consequences of perceived tree dieback for *Libocedrus bidwillii* and *Nothofagus fusca* forests was undertaken at several localities in the central North Island, New Zealand. Tree health has not noticeably improved at many locations despite extensive pest control, and this could be taken as evidence that other factors are driving tree population imbalances. Here, we use a combination of field observations, previously collected field data, and National Vegetation Survey databank results to assess population imbalances of *L. bidwillii* at Egmont National Park, Hihitahi Forest Sanctuary, and Ruahine Corner, as well as for *N. fusca* in the upper Pohangina Valley. Our findings suggest that both dieback and successful recruitment are patchy or non-existent within each site. (There was almost no evidence for recruitment at Hihitahi or for dieback at Ruahine Corner.) Tree dieback appeared widespread at Hihitahi, varied among patches at Egmont and Pohangina, and was relatively uncommon at Ruahine Corner. Because population imbalances are the sum of both tree death and regeneration, we suggest that a greater emphasis be given to understanding controls on successful tree establishment and recruitment. This is because the mechanisms behind tree mortality are complex and difficult to understand, but also that there has been little emphasis on the equally important process of recruitment. Further research is needed to improve understanding of controls on successful recruitment and regeneration of tree species and, specifically, the role of thickets of *Pseudowintera colorata*, ferns and grasses in regulating tree recruitment. Recommendations for work at each site are given, but the general framework we provide for understanding imbalances in tree populations is thought to apply across a wide range of species and situations.

Keywords: disturbance, forest health, recruitment limitation, *Libocedrus bidwillii,* pahautea, mountain cedar, *Nothofagus fusca,* red beech, tree dieback, North Island, New Zealand

1. Introduction

Advanced dieback and stand collapse of several large populations of *Libocedrus bidwillii* (pahautea or mountain cedar) and *Podocarpus hallii* (Hall's tōtara) as a community of upland conifer forest have been documented from several provinces (e.g. Rogers & Leathwick 1997; Rogers 1997). The consumption of both species by possums is beyond dispute, but whether herbivory leads to increased tree mortality is not resolved. Evidence for a population imbalance would be the critical measure for a significant conservation problem (Allen et al. 2002). Wanganui, Nelson / Marlborough, Canterbury, and Otago Conservancies all conduct possum control programmes in this forest type, but we do not fully understand why the trees are dying, whether there is a population imbalance, and whether irreversible changes in the ecosystem from a long history of pests are compromising our capacity to identify and document ecosystem benefits from pest control for these two species. The canopy collapse and non-replacement of *Nothofagus* forests has also been reported, mainly from high-altitude stands of *N. menziesii* (silver beech) in the Kaimai Range (Jane & Green 1986), for *N. solandri* var. *cliffortioides* (mountain beech) in the Kaweka Range (Hosking & Hutcheson 1988; Allen & Allan 1997), for *N. menziesii* in the Tararua Range (Wardle 1963a, b), and for *N. fusca* (red beech) in the southern Ruahine Range (Rogers & Leathwick 1997). Contingent upon the natural stand collapse of *Nothofagus*, ungulate-modified thickets of unpalatable plants may have smothered the regeneration niche of *Nothofagus* seedlings (e.g. Wardle 1991; Rogers & Leathwick 1997) or seedling growth might have been suppressed by continued ungulate browsing. Possums and ungulates may differentially affect the demographic characteristics of these communities, but their individual or combined roles are imprecisely understood. Against this, DOC invests heavily in pest control programmes aimed at restoring forests to something approaching their pre-invasion composition and structure, but with limited or no success for some forest types, particularly *Libocedrus*-dominated forests, including those of Egmont National Park and Hihitahi Forest Sanctuary. Likewise, the restoration of *N. fusca* forest of the Pohangina catchment, southern Ruahine Range, appears to be impossible following replacement of the original tall forest with *Pseudowintera colorata* scrub (horopito) and *Chionochloa conspicua* tussock grassland.

There are several other drivers for tree population imbalances that are not pest-related, including disturbance (e.g. windthrow, earthquakes, slips, salt scour), stress (e.g. drought), competition, decline in mutualists (e.g. pollinators, mycorrhizal fungi), disease or pathogens, insect outbreaks and age-related declines in tree health (reviewed by Mueller-Dombois 1986, 1987; Manion 1991; Ryan et al. 1997). These factors often interact or occur as part of a complex syndrome rather than causing tree death independently. For example, increasing tree age and stress predispose a stand to pathogens or disease, and can slow recovery following disturbance (which could include animal and insect pests). Although tree mortality rates should be relatively easy to quantify, understanding the factors behind these and disentangling the many causes of tree death is extremely difficult.

Tree population declines can be driven either by unusually high rates of adult tree mortality, or by a lack of sufficient regeneration to sustain a population in a particular area. To date, tree regeneration in natural populations has received less attention than tree mortality for several reasons: regeneration is patchy and varies widely through time, both seed production and fate need to be measured, seedling establishment and recruitment are more difficult to observe than tree death (observer bias), and often greater sampling effort is required to quantify recruitment rates than mortality rates. These are not difficult logistical problems to overcome, and studies that have focused on regeneration indicate that regeneration rates can be used to predict long-term persistence and dominance of different tree species (Clark et al. 1999). In New Zealand, a different suite of pests may reduce recruitment rates from those that may increase mortality rates. For example, mice or rats can consume the majority of seeds produced from a range of native tree species (Beveridge 1964; Williams et al. 2000; W. Ruscoe et al., Landcare Research Lincoln, unpubl. data). Recent work on seed additions in a range of vegetation-types has revealed that many plant species, and even community diversity, are seed-limited (Turnbull et al. 2000; Zobel et al. 2000); this effect would be enhanced through seed reduction by pests. This literature suggests that we need to measure tree recruitment as much as tree mortality to understand tree population imbalances.

At much larger temporal scales, tree populations are rarely, if ever, stable or at an 'equilibrium' (e.g. Wardle 1984; Clark 1996; Camill & Clark 2000). Long-term shifts in the dominance by different tree species can be caused by differential persistence or recovery of species caused by a wide range of factors including climate, dispersal limitation, major disturbances (e.g. glaciation, volcanic eruptions), and interactions with other species (competition and facilitation). For example, *L. bidwillii* has undergone dramatic increases and decreases in abundance across the New Zealand landscape at the scale of several millennia (McGlone et al. 1988); these long-term fluctuations in *L. bidwillii* populations may be linked to volcanic eruptions or other disturbances (e.g. Wells et al. 2001). Similarly, many native podocarps tend to regenerate most strongly following large-scale disturbances such as earthquakes (Wardle 1991; Wells et al. 1998). Evidence from pollen cores throughout New Zealand show that waxing and waning of tree species is the norm rather than unusual. The issue of whether current rates of demographic change for some species are within or beyond the historical rates of change remains unresolved.

Here we address how to gain a better understanding of tree demography or population imbalances, their likely geographic scale and magnitude, and whether ecosystems can recover following pest control or other treatments. Although we focus on observations and specific recommendations for several sites containing *L. bidwillii* or *N. fusca* communities, this work also develops general principles that can be applied to a wide range of tree species and situations. We will apply the principles advanced in Allen et al. (2002) in that forest sustainability issues are essentially population demographic issues, and the timing and nature of any imbalance will strongly influence a restoration vision for each community type.

1.1 CONCERNS OF FOREST MANAGERS

As mentioned above, Wanganui, Nelson / Marlborough, Canterbury, and Otago Conservancies all target *L. bidwillii* and *P. hallii* forest for possum control operations, either as one of many forest types covered in large control operations (e.g. Egmont National Park), or as the sole focus (e.g. Hihitahi Forest Sanctuary). Although the palatability of *L. bidwillii* and *P. hallii* to possums is beyond dispute, quantifying their differential responses to possum control needs refinement, partly because browse of their small leaves is difficult to detect from the ground (Rogers 1997). Of greater uncertainty is why the trees have died or are dying (natural processes and/or pests), whether there is, indeed, a population-threatening imbalance between mortality and recruitment of juveniles, and whether there are irreversible changes in the ecosystem from a long history of pests (Coomes et al. 2003). Moreover, *L. bidwillii* recruitment may depend on landscape-scale disturbances such as landslide, ash showers, fire, and windthrow (Veblen & Stewart 1982a). All these uncertainties compromise the ability of forest managers to identify and document ecosystem benefits from pest control in these upland conifer forests (Allen et al. 2002).

The sustainability of *N. fusca* in parts of the Pohangina catchment of the southern Ruahine Range is of equal concern to managers (W. Fleury, Wanganui Conservancy, pers. comm.), albeit as an issue concerning ungulate modification of regeneration niches. DOC encourages deer harvesting by commercial and recreational hunters as a means of containing deer impacts on forests. DOC and many forest user groups have expressed concern about the viability of high-altitude cohort stands of *N. solandri* var. *cliffortioides* following dieback episodes in the Kaweka Range, and DOC is examining whether recruitment of *N. solandri* is enhanced by experimentally reducing densities of sika deer (*Cervus nippon*).

The diets and related understory modifications by ungulates and possums are well known, but the impact of these animals on the maintenance of key canopy trees remains a debated topic (Veblen & Stewart 1982b; Rose et al. 1992). It has received the greatest research attention in the forests of Westland, but less attention in the region of greatest stand collapse and transformation, the southern Ruahine Range (Rogers & Leathwick 1997). In view of this lack of research consensus, this scoping study builds on a pragmatic approach, adopted by Allen et al. (2002), which examines what forest managers can do about maintaining key tree species in central North Island forests.

1.2 OBJECTIVES

- Evaluate the evidence for a population imbalance of *L. bidwillii* and *N. fusca* within each of the localities of concern, using a combination of site visits and permanent plot data.

- Assess the reasons for any population imbalance and consider the consequences for tree populations and ecosystems. This will be based on the field visits and wider literature.

- Recommend research required to better establish the timing and nature of any population imbalances as well as any consequences.

2. Significance of canopy dieback

2.1 DIEBACK AND DISPLACEMENT OF *Libocedrus bidwillii* AND *Podocarpus hallii*

Dieback and stand collapse of several populations of both *L. bidwillii* and *Podocarpus hallii* (Hall's tōtara) as a community of upland conifer forest have been documented using a combination of ground-based scoring of canopy foliar condition, and comparing canopy tree density in historical (1939–50) and modern (mid 1990s) sets of aerial photos (e.g. Rogers & Leathwick 1997; Rogers 1997; Smale et al. 1997). However, the spatial extent of stand dieback or collapse varies widely. At Hihitahi, the density of dead-standing stems of *L. bidwillii* exceeds that of healthy stems across most tree size classes, whereas in similar old-growth stands at adjacent Ohutu Ridge and Ruahine Corner the opposite situation occurs (in Rogers 1997: appendices 8, 9, and 11). Seedlings and saplings are virtually absent at Hihitahi, whereas they are abundant at Ohutu Ridge and Ruahine Corner. The palatability of the overall communities to possums and ungulates at the two areas also differs: palatable shrub and tree species dominate (or did dominate) at Hihitahi, whereas unpalatable *Phyllocladus alpinus* and *Halocarpus bidwillii* co-dominate with *L. bidwillii* at Ruahine Corner. In addition, the interpretation of *L. bidwillii* stand structures suggests that landscape-scale disturbance stimulates regeneration at Hihitahi, whereas tree fall–gap stimulated regeneration is the norm at Ruahine Corner. For *P. hallii*, dead-standing stems consistently outnumber live stems at Hihitahi and Ohutu Ridge (Rogers 1997: appendices 8 and 9), but this species is absent at Ruahine Corner.

Aerial photographs of Egmont National Park, Hihitahi, Ohutu Ridge, and the southern Ruahine Range show a dramatic, but spatially variable increase in the density of both dead-standing *L. bidwillii* and *P. hallii* stems and their density in the last 40–50 years (Rogers 1997). A few dead-standing *L. bidwillii* are evident in the earlier air photos. How long they remain standing is poorly known; preliminary data show some dead stems still stand nearly two centuries after death (G. Rogers and J. Palmer unpublished data).

2.2 DIEBACK AND DISPLACEMENT OF *Nothofagus* SPECIES

Canopy collapse and non-replacement of beech (*Nothofagus*) forest have been reported mainly from high-altitude cohort stands of *N. menziesii* in the Kaimai Range (Jane & Green 1983, 1986), *N. solandri* var. *cliffortioides* in the Kaweka Range (Allen & Allan 1997), *N. menziesii* in the Tararua Range (Wardle 1962), and areas of *N. fusca* in the southern Ruahine Range (Rogers & Leathwick 1997). Although, historical (1946) and modern aerial photos of 6400 ha of *N. fusca*-dominated forest types in the southern Ruahine Range reveal mostly low levels of canopy thinning, some upper montane stands show virtually complete

stand collapse and replacement with *C. conspicua*, *P. colorata*, or invasive subalpine scrub (Rogers & Leathwick 1997: table 3). While much of the low levels of canopy thinning can be attributed to loss of other possum-palatable trees such as *Weinmannia racemosa* (kāmahi) and *L. bidwillii*, complete loss of *N. fusca* and *N. solandri* var. *cliffortioides* stands is another matter. The hypothesis is that, contingent upon the natural stand senescence and collapse of beech, ungulate-modified thickets of unpalatable plants may have smothered the regeneration niche of *Nothofagus* seedlings, which is typically stumps and fallen stems (June & Ogden 1975; Wardle 1991; Rogers & Leathwick 1997; Coomes et al. 2003).

2.3 DEMOGRAPHIC CONSEQUENCES OF DIEBACK

While dieback events in beech forests have been recognised as a natural phenomenon for some time (Silvester & Watt 1983), the involvement of pest herbivores in the dieback of other species and in the structural, compositional, and functional consequences to affected stands, is imperfectly understood. Showing there is a valid concern about a tree species means demonstrating that there is an undesirable shift in the demographic processes regulating or limiting population size and viability (Allen et al. 2002). The critical life stages that affect population viability include flowering and pollination, seed production, seed dispersal, and seedling establishment.

Cowan & Waddington (1990) showed that possums prematurely reduce reproductive output of *Elaeocarpus dentatus* by reducing tree vigour and by browsing flowers. Nectivorous birds as pollinators of Loranthaceous mistletoes are now so reduced that these hemiparasites may have unsustainably low seed output (Robertson et al. 1999). Extinction of pigeon (*Hemiphaga novaeseelandiae*) as a primary disperser of large fleshy-fruited seed has compromised the potential expansion of several woody species after the elimination of goats (*Capra hircus*) on Great Island in the Three Kings group (Cameron et al. 1987). Atkinson & Campbell (1999) show that on northern nearshore islands, present forest composition differs significantly from that of forests before introduced rats (*Rattus* spp.) began consuming seeds and seedlings.

Seedling establishment is another critical life stage influencing tree demographics. In terms of stand replacement, the requirements for tree establishment are not well known but probably relate to the availability of suitable seedling microsites, the size of canopy openings formed by disturbance, and the availability of seeds about the time of disturbance (Allen et al. 2002). In general, large disturbances are required for conifer regeneration (Stewart 2002), as shown by *Libocedrus* (Clayton-Greene 1977; Norton 1983; Veblen & Stewart 1982a), but *P. hallii* may have greater shade tolerance and can also be found with 'gap-phase' regeneration structures (Smale & Kimberley 1993; Rogers 1997; Allen et al. 2002). Variations in the life history attributes of species means that they exploit disturbances and establishment microsites in different ways, leading to multiple species stands with spatial and temporal variation in disturbance. Different introduced mammals impact upon seedlings

and their microsites in different ways, producing potentially different successional pathways (Nugent et al. 2001).

There are at least four points to consider in any vulnerability scoring system for tree species: its distribution; its vulnerability at different life stages; the causes of mortality; and the significance of loss of the species to ecosystem processes. *L. bidwillii* is probably more vulnerable than *N. fusca* to undesirable shifts in demographic processes because:

- Its distribution is more limited

- It is vulnerable at more stages of its life cycle—its foliage and male cones are browsed by possums and disturbance-regenerated stands tend to die off at the same time

- Possums may be implicated in its mortality

However, both species are equally important in terms of their loss to ecosystem processes because they are dominant agents of energy capture, and their litter strongly influences nutrient cycling.

The significance of such concerns can then be addressed in the following sequence of questions (Allen et al. 2002):

- Is recent mortality an issue at local, regional, or national scales?

- Does regeneration match death at these scales?

- Is the affected species a stand dominant?

- Is the process affected by introduced mammals?

- Are there feasible intervention options?

Standard techniques of forest surveillance and monitoring can help answer the first three questions although there may be many methodological challenges (Bellingham et al. 1999a; Allen et al. 2002). The bottom-line for population viability is a balance between average mortality and recruitment rates, applicable at any scale. However, the pervasive influence of disturbance means that there is no reason to expect mortality and recruitment to be in balance at local scales, or even at large spatial scales. Nevertheless, imbalances at large scales raise the strongest doubt about a species' survival.

In terms of the fourth question, it is extremely difficult to separate natural from herbivore-influenced mortality or to ascribe undesirable shifts in tree recruitment to unnatural causes. Some of the factors that contribute to the lack of consensus by researchers on the topic are: high environmental heterogeneity making generalisation from local studies difficult; uncertainty over which historical benchmarks to use in measuring recent change; a failure to recognise the interpretative limitations of most methods of data gathering; and an acknowledged bias toward either natural processes or introduced mammals as explanations for change when complex and variable interactions are likely.

Deriving historical benchmarks is difficult because pollen diagrams show large swings in the apparent abundance of tree species over millennia and even centuries. Forest structures and composition have been strongly influenced by several centuries with only minor vertebrate herbivory which prevailed between the extinction of large birds and the introduction of mammals (Wardle 1991; see also Clout & Hay 1989). The various methods researchers use to document and understand compositional changes in forests have strengths and

weaknesses that influence the interpretative robustness of any role attributed to pests in the recent modification of our forests. Scoring stand density and composition from aerial photos provides comprehensive and potentially representative coverage of large areas, but is qualitative and imprecise for accurately determining composition states and changes. Plot-based data on stand composition and structure are often used to interpret regeneration processes and past disturbances, however, these data represents a snap-shot in time. Interpretations of the mechanisms of change may be compromised by establishment of plots at post-peak animal densities, or during and after the main phase of dieback and canopy collapse (Allen et al. 2003). Evidence of change may remain for different time periods (species grow and decompose at different rates). Lag-effects operate over different time frames and have poorly understood causes. The lack of representativeness of a few plots may be because of high topographically driven variability in forest composition. Repeat measures of permanent plots give unequivocal historical data for the measurement period, but also suffer the interpretative shortcomings of one-off plot-based data described above. Perhaps because of this, no consensus has emerged amongst researchers on how to apportion the various manifestations of defoliation, dieback, and stand regeneration to natural turnover processes or to pest impacts, even in one region (e.g. Veblen & Stewart 1982b; Rose et al. 1992).

Burdened by this imprecision, forest managers finally need to ask whether or not there are feasible intervention options if undesirable shifts in demographic processes are evident. This is also a vexed question. Successions triggered by dieback events may have trajectories favouring plants unpalatable to pest animals that then promote poorer quality litter, and consequent changes in ecosystem nutrient status (Wardle et al. 2002; Coomes et al. 2003). For example, a proliferation of unpalatable ground ferns has apparently obliterated the microsites in which beech seedlings establish in the Ruahine Range (June & Ogden 1975). The effects of cohorts of unpalatable species may be long-lasting—for example, *Pseudowintera axillaris* can live for at least 300 years.

3. Mortality of *Libocedrus bidwillii* at selected localities

3.1 EGMONT NATIONAL PARK

3.1.1 Current status

Libocedrus bidwillii populations within Egmont National Park varied widely in terms of recruitment of seedlings or saplings; similarly, old or standing-dead trees were patchily distributed at a scale of several hundred hectares. Our preliminary site visit during November 2002 revealed that:

- Most stands were short-stature forest, mostly comprising trees < 5 m tall. Taller stands occurred in more sheltered aspects elsewhere in the park, where *L. bidwillii* and *P. hallii* were the emergent canopy dominants.

- Most stands in the Ahukawakawa Swamp catchment were extremely misshapen by wind, resulting in flag-form trees. There has been obvious abrasion or salt damage on the windward side of exposed stands.

- There were many standing dead trees that appeared to persist for several decades, if not longer. These comprised 20–30% of the total adult trees in some stands.

- Large patches (in the order of several hundred hectares) of vigorously regenerating *L. bidwillii* were observed near some ridge tops and protected slopes.

- There was little evidence of possum browse, although a more detailed survey would be required to confirm this observation.

These observations suggest that there is no major cause for concern in this area, and that there are patches of *L. bidwillii* forest both regenerating and in decline. Several factors are likely to contribute to the spatial patchiness of *L. bidwillii* stand structure within the park including disturbance (e.g. landslips, volcanic activity, lahars, windthrow) and desiccation by wind or salt. In addition, synchronous mortality may have occurred for *L. bidwillii*–*P. hallii* forest in this area because of a large-scale disturbance c. 600 years ago, but aerial photos spanning more than 35 years suggest that recent declines are occurring, possibly as a result of pests (Rogers 1997: fig. 3). Pollen evidence shows that over the longer-term, *L. bidwillii* abundance has fluctuated at the scale of centuries, being more abundant c. 3000–2500 years ago and again 1400 –450 years ago (McGlone et al. 1988).

Thirty-eight permanent National Vegetation Survey (NVS) plots are located within Egmont National Park. However, none of these plots contains *L. bidwillii*. This is not entirely surprising considering the restricted elevational zone that the species occupies (c. 950–1100 m; Clarkson 1986), and the spatial patchiness of populations at a landscape scale. We expect that *L. bidwillii* might cover only c. 5% of the landscape in this location (Clarkson 1986; Rogers 1997). This is not a downfall of systematic sampling or location of NVS plots *per se*, but rather points to the need for adequate sampling effort in order to characterise forests or vegetation-types within parks, and that more detailed information or focused sampling is required if we want specific evidence of population imbalance for *L. bidwillii* at this location.

3.1.2 Research recommended

The age of standing dead trees is needed to determine their proportion in the population. Many can be increment-cored and their tree rings cross-referenced with live trees to determine their date of death and hence length of time they have been standing dead (see Xiong & Palmer 2000). If trees live for c. 600 years, and standing dead persist for 100 or more years, then dead adult trees would be expected to constitute 20–30% of the total adult population. The long-term persistence of *L. bidwillii* spars and logs is probably a feature of this ecosystem. For example, the preferred habitat of an endemic fern in New Zealand, *Hymenophyllum malingii*, is *L. bidwillii* spars, suggesting that this is

an evolutionarily reliable substrate (P. Bellingham, pers. comm.). Tree ages can also be determined by tree cores, to confirm this and to back-calculate the annual tree mortality rate.

Our preliminary observations suggest that some stands are actively regenerating while others may be undergoing dieback at a spatial scale of several hundred hectares. Ageing dead trees and calculating tree mortality rates could be done for a range of stand conditions in Egmont National Park. The most obvious factor that could control tree recruitment there is disturbance. This could be double-checked by assessing whether recruitment was approximately synchronous within stands, and whether this regeneration matches dates for the last major disturbance. Work by McGlone (1988) in Egmont National Park, and elsewhere by Norton (1983), Veblen & Stewart (1982a), and Wells et al. (2001), suggests that cedar regeneration may be driven by large-scale disturbances.

To determine whether patches of regenerating *L. bidwillii* are sufficient to maintain the population at the landscape scale, both tree mortality and recruitment rates should be measured for the stands of *L. bidwillii* within Egmont National Park. These can be assessed by tagging adult trees and following survivorship (mortality rates) and measuring establishment within plots (recruitment). This information can then be used to determine whether there is evidence for a population decline, or whether local diebacks are being offset by adequate regeneration at other sites.

3.2   HIHITAHI FOREST SANCTUARY

3.2.1   Current status

The Hihitahi Forest Sanctuary is the site in this study with the greatest evidence for *L. bidwillii* population imbalance. Several critical observations were made during our site visit, including:

- Extremely low seedling numbers. Only about a dozen *L. bidwillii* seedlings were encountered, and these were restricted mostly to dead logs at a single location. Similarly, very few saplings or pole-sized individuals were observed.

- No *L. bidwillii* seedlings were recorded in 50 one-metre-square plots (Rogers 1989: table 3) or in a larger sample of 8 ha (Rogers 1997).

- Most *L. bidwillii* trees had a large diameter (i.e. > 50 cm dbh), rounded crowns, and some branch death. About half of all trees were 'standing dead' individuals (see also Rogers 1997: table 3).

- Similarly, Rogers (1989: fig. 5) found that most *L. bidwillii* were > 35 cm dbh and > 300 years old in a 0.2 ha *N. fusca* stand.

- Little evidence of possum browse was observed on any trees (see also Hawcroft 2003).

- Fresh fallen branches were observed to have been damaged by an insect borer. Branch samples were examined by John Dugdale, Landcare Research, Nelson (see Appendix 1). Briefly, wood-boring larvae of a native moth were encountered that created damage just below the bark. The tunnels left by

this insect were subsequently colonised by a range of fungi and other organisms, which in turn caused damage to, or blocked, the water-conducting tissues of the wood (i.e. analogous to a stroke in plant tissues).

- There seems to be a 'missing' canopy tier. *L. bidwillii* is the only emergent dominant at higher elevations (> 800 m), but there is a gap of 5–8 m between this canopy and the understory, which is currently dominated by thickets of *Pseudowintera colorata* (see front cover) and *Aristotelia serrata* at lower abundance, possibly as a consequence of deer browse.

- An analysis of 1978 aerial photographs by Rogers (1997: fig. 4) shows that up to 75% of the *L. bidwillii–P. hallii* montane forest within Hihitahi undergoing moderate to severe dieback.

Data previously collected from 12 permanent NVS plots (20 × 20 m) located in the Hihitahi reserve confirm our field observations that there is a lack of recruitment for *L. bidwillii*. Although it was recorded in four of the twelve NVS plots, no seedlings or saplings were recorded on any plot. Only seven adult trees were recorded, all of which were > 40 cm dbh (Fig. 1). No pole-sized trees were recorded (i.e. all trees were > 40 cm diameter).

Figure 1. Size class distribution of *L. bidwillii* at Hihitahi.
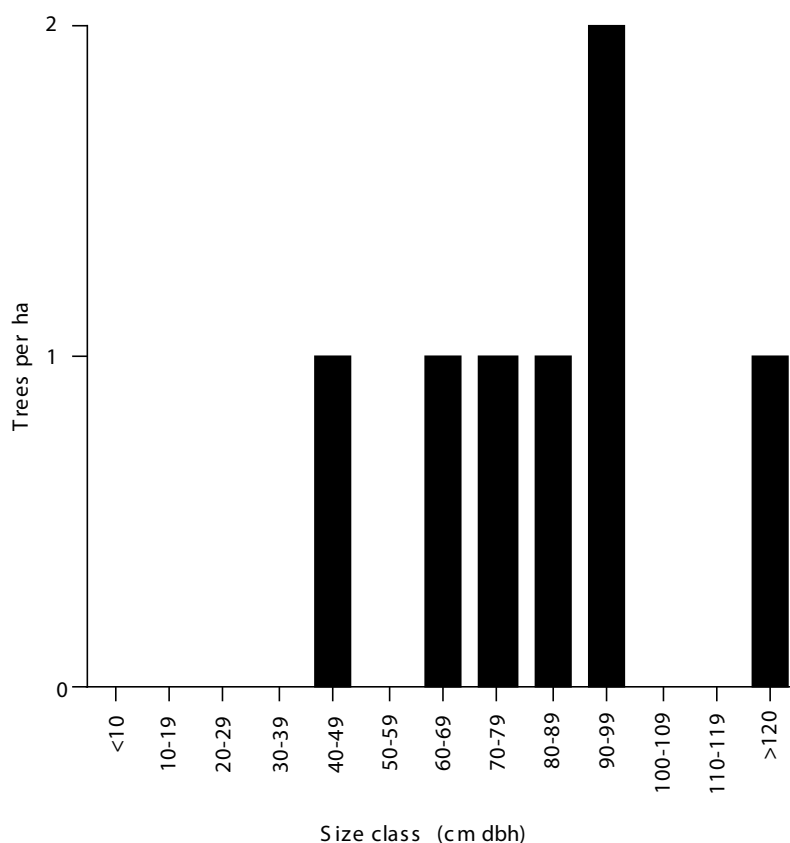
Together, our field observations, previous data (e.g. Rogers & McGlone 1989; Rogers 1997) and the NVS data show that there is an extremely low rate of recruitment into the *L. bidwillii* population at Hihitahi. Furthermore, most of the population consists of older or dead standing trees, which may be susceptible to insect borers, pathogens or disease, i.e. most of the population appears moribund.

3.2.2 Research recommended

Several approaches can be used to better understand the drivers of *L. bidwillii* decline at Hihitahi, and whether it can be reversed.

Recruitment limitation of *L. bidwillii* needs to be understood. First, seed production and viability need to be determined for adult trees. Seeds could be collected using seedfall traps over several seasons to avoid problems with variability in seed production (e.g. seed masting). Seed viability can either be scored as the proportion of 'filled' seed, or by testing germination under ideal glasshouse conditions. Second, collected seed can be sown on a variety of substrates to determine whether there are limited 'safe sites' for seedling establishment. Substrates could include dead logs, scarified soil, and soil with trenching to reduce below-ground competition from roots. Because rodent pests can destroy or cache the majority of tree seeds, rodents should also be excluded in part of the plots or in additional plots.

NVS plots need to be re-measured and additional plots established to measure both seedling recruitment and tree mortality rates. Currently, only four established NVS plots contain *L. bidwillii*, representing a sample size of $n = 0$ for seedlings and saplings, and $n = 7$ for adult trees. An absolute minimum of 120 'tree years' of monitoring is required to detect a high mortality rate of 3% per annum (Fig. 2). A realistic approach would be to establish at least 20 additional permanent plots and monitor these over at least 5 years. To detect lower mortality rates or to have greater confidence in mortality estimates in the face of spatial and temporal variation in tree deaths, additional plots or length of monitoring period would be needed.

Figure 2. Minimum sample size required (number of trees × years sampled) versus annual tree mortality rate (% yr⁻¹). This relationship shows the minimum sampling effort in 'tree years' (number of trees × time) in order to detect a given tree mortality rate. More samples are required to account for temporal or spatial differences in mortality rates, and cumulative tree deaths (reduced sample size through time). Sample size is calculated using the minimum detectable effects based on statistical *t*-distributions (almost identical results are obtained with chi-square distributions).
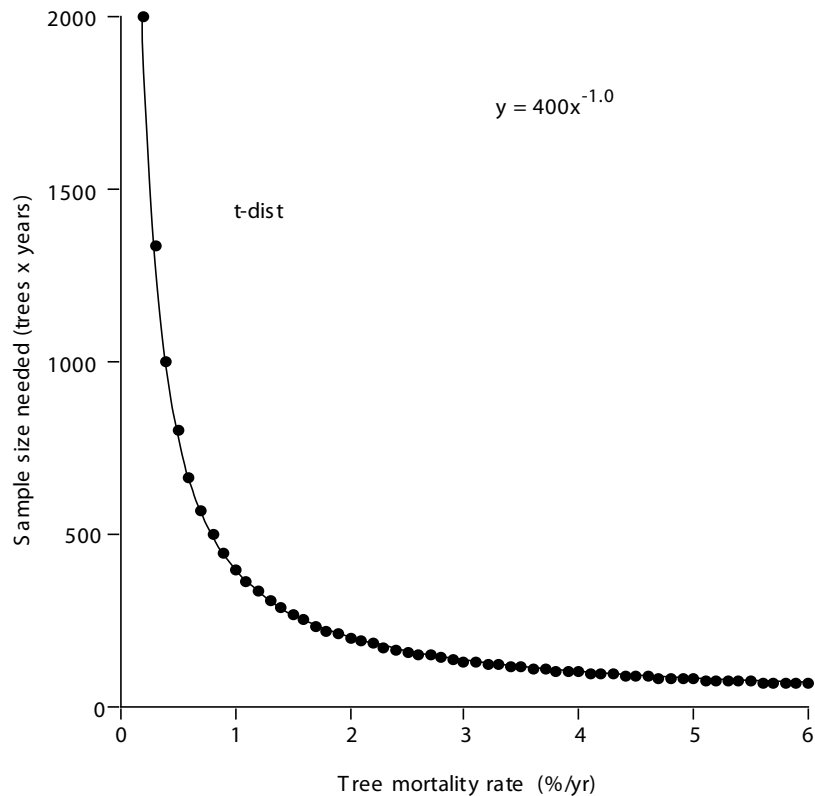
The role of shrub thickets, particularly *Pseudowintera colorata*, in regulating tree recruitment can be assessed using shrub and litter removals in combination with seed additions. *P. colorata* seedlings and saplings occur on 76% and 96% of the NVS plots respectively, at a density of 19 700 (seedlings) and 4200 (saplings) per hectare. All other woody species seedlings are from 3-fold to 20-fold less abundant, and saplings are at least 69-fold less abundant! It is unknown whether there is a 'missing' subcanopy or tall understorey layer. For example, deer have locally removed palatable understorey species (e.g. *Schefflera digitata*, *Pseudopanax arboreus*) in indigenous forests (Forsyth et al. 2002; Coomes et al. 2003). The previous presence and composition of putative missing elements of the subcanopy could be assessed by collecting macrofossils (e.g. partially preserved wood) or phytoliths (i.e. silica bodies from plant leaves). Soil fertility (N, P) or pollen data (e.g. Rogers & McGlone 1989) could also be measured to help understand which species may have dominated this tier. An intriguing possibility is that there could be a 'serial dieback' at Hihitahi. That is, if subcanopy species were lost first, and these species facilitated *L. bidwillii* regeneration better than *P. colorata* does, this could explain the lack of observed *L. bidwillii* seedlings and saplings. On the other hand, the absence of small trees suggests there has been low recruitment for more than 100 years, given the size class distributions in Rogers (1997). The first step in understanding this process is to determine whether *P. colorata* thickets are suppressing *L. bidwillii* regeneration.

Fresh twigs and foliage material should be collected to determine the extent of borer damage and related agents of decline, i.e. pathogens. The initial examination of borer-damaged twigs by John Dugdale suggests that a native moth is causing the initial damage to twigs by burrowing just below the bark, but that subsequent colonisation by other species including fungi may contribute to tree death (more details are in Appendix A). Whether this is a cause or consequence of tree decline is unknown, but many pathogens are viewed as attacking weakened or senescent individuals (Manion 1991). Further work on the natural history of this borer-pathogen system is sorely needed.

The age of standing dead trees is needed to determine whether there have been patches of synchronous tree mortality, and what proportion of the population is standing dead. Many standing dead trees can be increment cored and their tree rings cross-referenced with live trees to determine date of death. Although current tree mortality rates can be measured using additional permanent plots or tagged trees (suggested above), reconstructing tree deaths will give insights into whether dieback has been gradual or synchronous. Synchronous diebacks could then be matched with disturbance, drought, the introduction of pests, or other events (but see Bellingham et al. 1999b). Similarly, other complimentary techniques can be used to determine whether tree population declines are recent or longer-term. These include historical aerial photographs (Rogers 1997), decay rates of fallen logs (and their approximate age) or, over the very long-term, pollen evidence for long-term changes in abundance.

3.3 RUAHINE CORNER

3.3.1 Current status

Ruahine Corner contained the most vigorous stands of *L. bidwillii* encountered over an area of several thousand hectares. There was active regeneration at all locations visited. Some additional observations include:

- *L. bidwillii* crowns appeared healthier and crown volumes much larger than any stands encountered at either Egmont National Park or Hihitahi.

- There appeared to be relatively few trees that were standing dead or with partially dead canopies (i.e. < 10% of the canopy emergents), although data presented in Rogers (1997: tables 3, 4) suggest that 25% of *L. bidwillii* stems are standing dead, although these represent only 9% of standing basal area.

- It appears that *Halocarpus biformis* (pink pine) facilitates *L. bidwillii* regeneration on ridges, as many young seedlings were observed directly underneath *H. biformis* canopies. Perhaps *H. biformis* act as nurse plants to mitigate the harsh environmental conditions on ridges.

- Discrete patches of *N. fusca* and *N. solandri* are intermixed throughout the *L. bidwillii* forest (see Rogers 1989: fig. 6, and table 2). *Nothofagus* patches are relatively small (i.e. most patches are < 1.0 ha), and contain a mixture of tree sizes. Relatively few *Nothofagus* seedlings or saplings were observed.

- Thickets of *P. colorata* were not present in the understorey of *L. bidwillii* forest at Ruahine Corner, but are widespread along the western fringe of the Ruahine Range south of Ruahine Corner.

If there is a population imbalance at Ruahine Corner, it is not tree dieback, but rather tree expansion (i.e. a positive tree population imbalance). If the causes of successful regeneration can be assessed at Ruahine Corner, this may provide clues for management options at other sites, particularly Hihitahi.

Data for tree size distributions and abundance for *L. bidwillii* in 46 permanent NVS plots established in the northern Ruahine range in 1983 reveal that it occurs in only three (6.5%) of the plots, and that only seven trees were recorded, but that these individuals were generally in much smaller size classes than the trees found at Hihitahi (Fig. 3). These data suggest that *L. bidwillii* is extremely patchy on the landscape scale, as found elsewhere. *N. fusca* was the overwhelmingly dominant species across the northern Ruahine NVS plots, occurring in 71.7% of the plots at a density of 202 stems/ha. The wide range of *N. fusca* tree size classes present suggests these populations have a well-mixed age distribution in this area. *P. colorata* dominated the understory in the majority of plots and comprised about one-third of all plant stems (789 stems/ha).

More detailed survey data from three 0.4 ha plots within this area by Rogers (1989: table 2) showed that most *L. bidwillii* stems were < 75 cm diameter at breast height, but some individual trees > 75 cm dbh persisted. This more equitable size-class distribution of trees compared to Hihitahi suggests that recruitment has been more continuous over the past few centuries at Ruahine Corner. Rogers (1989: table 3) recorded 81 *L. bidwillii* seedlings in 74 small (1 m²) plots. Nearly all of these seedlings were found on tree bases, stumps or decaying logs (73 seedlings), but a few were found on moss-covered ground

Figure 3. Size class distributions for *Libocedrus bidwillii*, *Nothofagus fusca* and *Pseudowintera colorata* at Ruahine Corner.
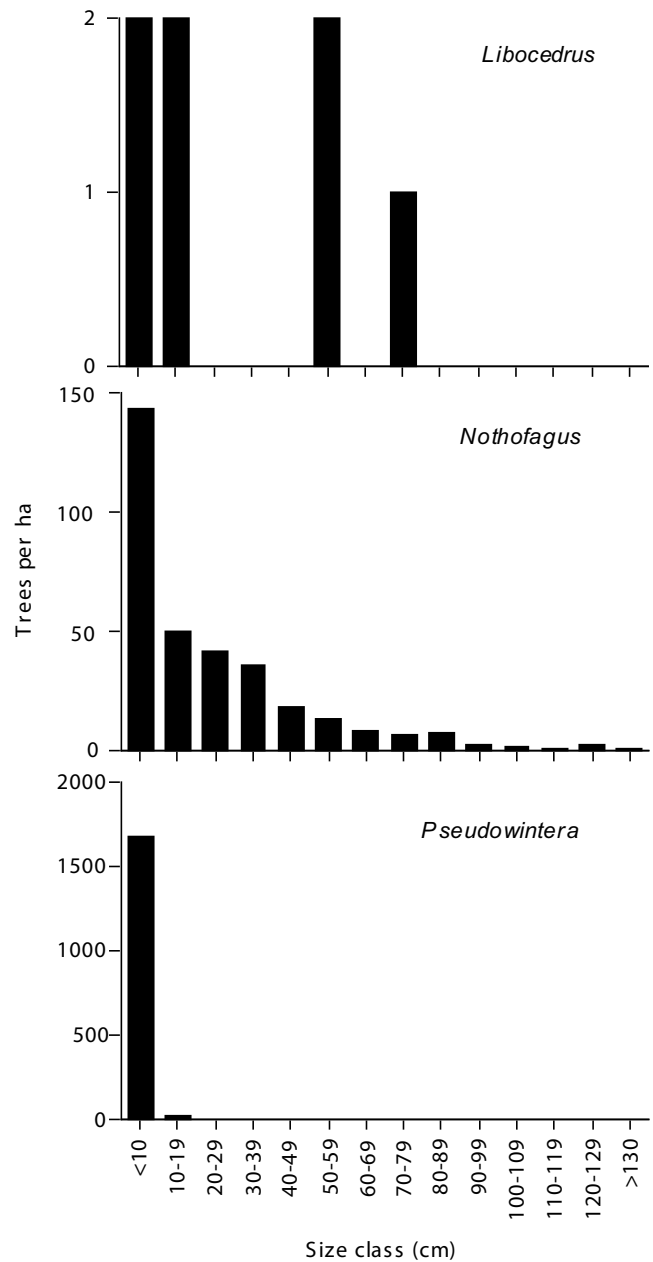
(8 seedlings). In addition, Rogers (1997: table 4) showed that seedlings and saplings occur at densities of c. 353 and 174/ha respectively. Together, these data and our observations suggest that the *L. bidwillii* populations at Ruahine Corner are not undergoing dieback, and have much higher and consistent rates of recruitment through recent history.

3.3.2 Research recommended

Permanent NVS plots need to be remeasured or established with a particular focus on understanding what is driving successful recruitment at Ruahine Corner. Data on both biological and environmental factors associated with seedling establishment should also be collected: availability of dead logs (e.g. Rogers 1997), soil fertility (e.g. Rogers 1989), etc. Another option would be to carry out a more purpose-built study on controls over tree recruitment rather

than setting up additional NVS plots. In either case, this information will enable us to determine whether recruitment rates are higher at Ruahine Corner, and what controls successful recruitment (at least to the seedling stage).

L. bidwillii seed could be collected to determine viability. Measures of both seed production and viability are needed to understand whether seed availability is driving population expansion at Ruahine Corner, or whether factors later in the life history stage are responsible. This would be particularly useful to compare against seed production and viability at Hihitahi.

A random sample of *L. bidwillii* could be aged using increment cores to determine if populations have established synchronously or continuously. Synchronous establishment would suggest that it is recruited after a particular event (e.g. major disturbance, seed masting year). Continuous recruitment would suggest that regeneration is possible at any time, although recruitment rates could still vary with resource availability or competition. This information can be used to confirm whether the tree population is increasing, and together with existing data (e.g. Rogers 1997), whether any increase is because of lower adult mortality and/or higher recruitment than in other areas.

Existing permanent NVS plot data could be made electronically available for investigations such as this. This would provide information on seedling distribution and abundance of *L. bidwillii*, and also generate data that could be used to determine whether *P. colorata* is invading or becoming more abundant at the site.

4. *Nothofagus fusca* collapse, Pohangina Valley

4.1. CURRENT STATUS

N. fusca was, and in part remains, the dominant tree in montane and subalpine forests of the upper reaches of the Pohangina Valley. This is somewhat unusual in this landscape, particularly at high elevations, as the sites would appear more suitable to *N. menziesii*. Near treeline, the *N. fusca* canopy is formed by trees < 4 m tall with heavy branching near ground level. Dead patches of *N. fusca* trees occur in the mid and upper reaches of the Pohangina River. These dead patches vary in size and the standing dead spars vary in stage of decay. The most recent examples still have fine branchlets indicating death over the last few years. Thus the evidence is that this *N. fusca* dieback has been an ongoing process for several decades. Observations suggest *N. fusca* trees died earliest on land faces, particularly near large landslides, with recently dead patches appearing to be more commonly located on ridges. Recent ecological and experimental work shows that when *N. fusca* trees die from drought, insects, earthquakes and timber harvesting, residual trees often continue to die from secondary damage, for example, from *Platypus* attack of damaged trees (Wiser et al. 2003; but see Grant 1984; Hosking & Kershaw 1985; Haase 1999).

Virtually all live *N. fusca* trees in the head of the Pohangina have relatively low crown area and the foliage is intensively damaged by what appears to be a microfungus (i.e. many small perforations in the majority of leaves for some trees). Canopy foliage was denser in *N. fusca* forest on the western slopes of the Ngamoko Range in the western Ruahines than the head of the Pohangina Valley, where there was enormous patchiness in density and size, and most stands were dominated by few, large-stemmed trees. Much of the forest appeared to have a relatively open canopy (50–60% cover). Similarly, Rogers & Leathwick (1997: table 3) show that, based on analyses of aerial photographs, most *N. fusca-Libocedrus* forest at higher elevation has experienced some loss of canopy cover over the past 50 years, and that some of this forest type has become subalpine scrub.

The *N. fusca* forest understorey has extensive areas dominated by *Chionochloa conspicua*, *Blechnum discolor*, and *Pseudowintera colorata*, including gradations with mixtures (Fig. 4; see also Batcheler 1983). Based on a limited ground survey, *N. fusca* seedlings could be found at varying density in most types of ground cover. Unstable past landslide surfaces were dominated by *C. conspicua* and any *N. fusca* seedlings were concentrated on the upper part of the slope where there was less cover. Gully heads that were once probably dominated by *Fuchsia* are often now dominated by *Pseudowintera colorata*, containing very little *N. fusca* regeneration. Although in forests elsewhere *N. fusca* often regenerates in thickets, only a few patches of dense regeneration occur—often on ridges. Virtually all seedlings and saplings appear stressed,
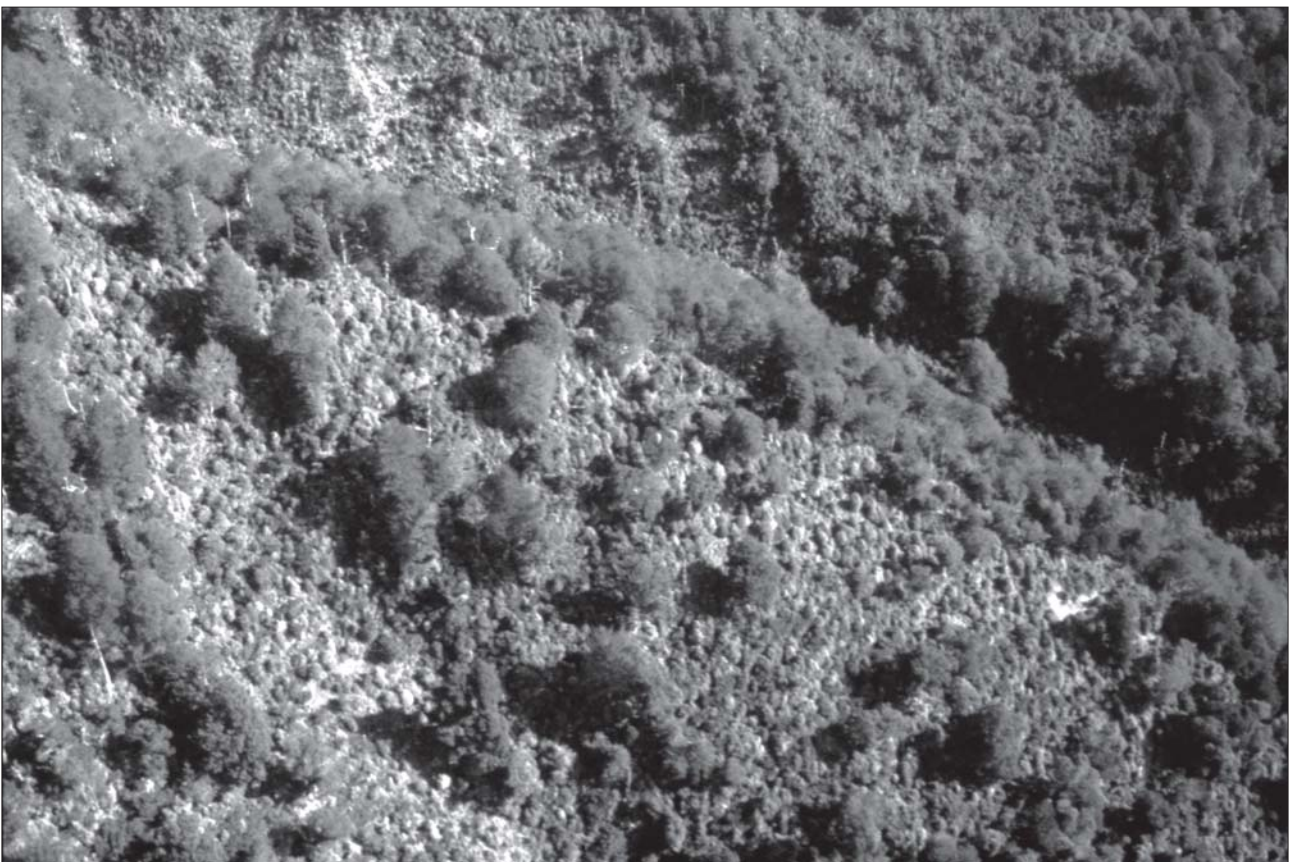

Figure 4. *Nothofagus fusca*-dominated stands in the head of the Pohangina Valley, Ruahine Range. *Pseudowintera colorata* and *Blechnum discolor* dominate the understorey.

with small leaves and an absence of vigorous growth, given the light levels. *N. fusca* seedlings did not exhibit browse by possums or deer.

N. fusca occurred in 12 of the 23 NVS permanent plots located in the upper Pohangina Valley. *N. fusca* averaged 81 stems/ha and 23.1 m²/ha basal area across these plots. Its size class distribution shows a large proportion of the population is in small/young stems with no evidence for dieback (i.e. there is a good number of individual stems across all tree size categories) observed across these plots (Fig. 5). Seedlings and saplings were present at densities of 700 and 20 per ha respectively, but these numbers are lower than some *N. fusca* populations elsewhere (e.g. Station Creek, Maruia: 1533 seedlings ha⁻¹). Both seedlings and saplings were found on fewer plots than mature red beech. These data suggest that recruitment of *N. fusca* occurs, but is extremely patchy or varies widely among sites, which corroborates our observations on the ground. The issue remains whether patches in which stand thinning occurs are also those with low seedling and sapling densities.

As at Hihitahi, *Pseudowintera colorata* dominated the seedling and sapling tiers in the NVS plots. It averaged 1420 stems ha⁻¹ across the plots, and had fourfold more seedlings and fivefold more saplings than any other woody plant species.

Figure 5. Size class distribution of *Nothofagus fusca* at the Pohangina Valley.
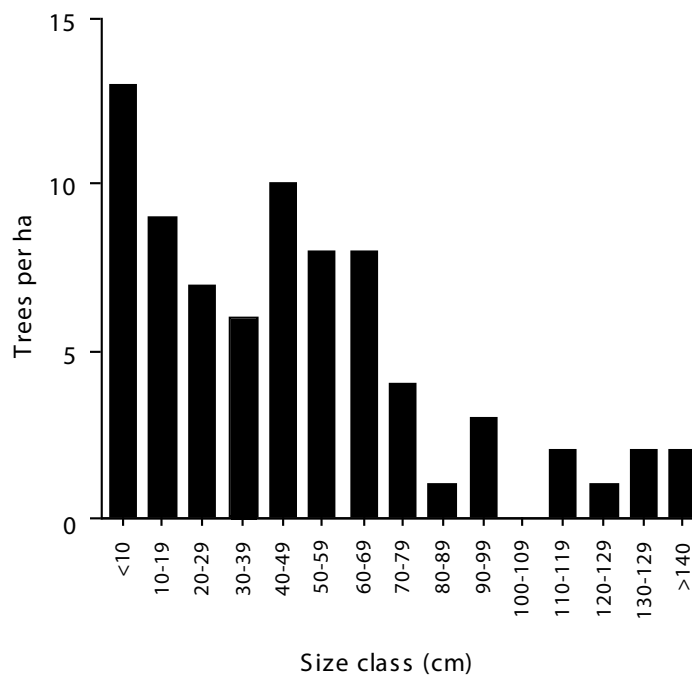
4.2 RESEARCH RECOMMENDED

The 23 permanent NVS plots in the Pohangina Valley need to be remeasured to assess changes in stand structure and to estimate tree mortality rates. These plots were last measured in 1983, and there are sufficient plots (12 containing *N. fusca*) and individual trees (75 trees > 10 cm dbh) to estimate mortality rates over this period.

The adequacy of natural regeneration to sustain current *N. fusca* populations in the Pohangina Valley needs to be determined. Results from the NVS plots show that both seedlings and saplings are present, but that their distribution is extremely patchy. Another interpretation is that regeneration is successful only on a small subset of available sites. Additional environmental and biological information is needed about what controls regeneration of *N. fusca* (e.g. soil fertility, presence of dead logs). This information could be collected at the same time as the NVS plots are remeasured.

Removal of shrub thicket, fern, or grass, or thinning of plots could be done to determine whether *N. fusca* regeneration is suppressed by the dense cover of ferns, native grasses, and *P. colorata*. As in our recommendation for Hihitahi, a combination of root trenching or shoot removal is needed together with seed addition. If *N. fusca* seedling establishment is higher after either seed addition or understorey removal, this would be strong evidence that either seed limitation or competition are regulating tree recruitment.

Fresh foliage should be collected to identify the cause of leaf perforations or holes, a general fungicide could be applied to several infected trees to see if this controls the 'microfungus' suspected to cause it, although reinfection from neighbouring trees would have to be prevented.

The spatial extent of areas with sparse stand density needs to be determined from aerial photos. The extent of the phenomenon should then be compared with *N. fusca* stand density in other parts of the southern Ruahines where *N. fusca* dominates (e.g. Rogers & Leathwick 1997). This will help determine whether stand thinning is occurring only locally, or whether a more widespread problem exists. Spatial patterns of canopy coverage or condition can also be generated using this approach.

5. Strategic research needs

5.1 QUANTITATIVELY ESTABLISH THE LEVEL OF DECLINE

The maintenance of key tree species is of concern to DOC for a range of species, at a range of sites and at a range of spatial scales (see review in Allen et al. 2002). The issue is not new. The public and resource managers need to be shown that the widespread death of canopy trees is often a natural phenomenon, and part of the ecology of New Zealand forests, but in view of concern over tree dieback, fundamental information is needed to more effectively manage the species of concern. It is important to substantiate the level of decline in species of concern. Several approaches can be used to understand whether tree population imbalances occur for specific species at specific locations. Permanent plots (e.g. from the National Vegetation Survey databank) can be used to assess distributions of different size-classes for tree species, as well as rates of mortality and recruitment. However, the patchiness

of many tree populations means that sample size is often too small to estimate tree mortality or regeneration at local scales. This problem can be overcome by adding more permanent plots, or by tagging several hundred trees and following their survival for at least five years. In this report, for the first time, we show the sampling intensity required to characterise levels of tree mortality with a given level of confidence. Similar analyses need to be undertaken to characterise recruitment rate, although this is a more challenging issue because we are not assessing individuals that exist at the beginning of the study. Already comparative figures are available for tree mortality rates that indicate when managers should be alerted (e.g. Bellingham et al. 1999a). In contrast to permanent plots, specific measurements to characterise rates of mortality and recruitment of a species at specific locations is not field-intensive work. It is essential to bear in mind that there will be strong interpretive limitations from such focused measurements. Further work could be done on sampling design, e.g. how to select individuals to be monitored. We suggest that such an approach be used to efficiently measure tree mortality and recruitment rates, and quantify population imbalances at a particular site where there is immediate concern about the maintenance of key tree species.

5.2   UNDERSTAND THE REASON FOR DECLINE

Effective management responses to tree population imbalances can only be identified through an understanding of the underlying causes. The current emphasis on measuring foliar condition (e.g. Payton 1983; Payton et al. 1999) does not adequately measure the health of tree populations, or capture the complexity of population increases or declines. Research on what drives apparent synchronous mortality in some tree species has been inconclusive, but often shows that declines have several interacting causes. These underlying causes can only be understood using robust long–term data (i.e. > 20 years), exploiting experimental approaches and by using adaptive management within appropriate sites. At least three critical life stages should be assessed in this way, each of which can contribute to demographic rates.

First, seed production and viability for a given tree species at a particular location need to be determined. Determining levels of seed production for trees at specific localities is not a high-cost activity and could be readily implemented where tree regeneration is of concern. Many New Zealand tree species produce seed periodically and such seed crops can suffer high levels of predation by native and introduced seed eaters. It is increasingly recognised that seed predation can markedly reduce the regeneration of some tree species, and these effects can be tested through animal-exclusion plots.

Second, given sufficient seed sources, regeneration depends on safe sites for seedling establishment and recruitment. There are many management options for dealing with recruitment, e.g. by adding viable seeds and removing competitors such as shrub or fern thickets that may suppress seedlings. There is a need for research on the direct and indirect effects of some pervasive understorey changes (such as the increase of ferns and shrubs) on the regeneration of dominant tree species on some sites. An ecosystem-process

context should be used which includes the role of competition for light and nutrients and how this is influenced by herbivory and plant species effects (see Wardle 2002). In addition, plants can have long-term effects on soil processes or communities that may persist long after plant thinning or removal (Wardle 2002). It has been shown that large-scale disturbances may be required for successful establishment of *L. bidwillii* (McGlone et al. 1988; Rogers 1997), but whether relatively small-scale removal treatments or disturbances can be used to create opportunities for establishment is currently unknown. Many plant species in New Zealand appear to require catastrophic disturbance in order to stimulate recruitment or for successful regeneration (e.g. McGlone et al. 1988; Wells et al. 1998, 2001), but the mechanism—increasing light availability to the soil surface, increasing soil nutrient levels, or reducing belowground competition—needs to be investigated.

Third, what drives variability in the mortality of adult trees needs to be understood. Most attention to date has been on tree mortality rather than on regeneration. This bias is expected because adult tree death is visually obvious, often assumed to be caused by pests, and is therefore of immediate concern. The problem with this approach is that there are many reasons for tree death (i.e. pests, pathogens, natural ageing, environmental stress, disturbance, etc.). The most commonly applied management has been pest control, but the gains from reduced tree mortality have not been convincingly demonstrated. We suggest a significant shift in research and management emphasis from mortality to the regenerative phase. However, disentangling the relative importance of pests and natural processes such as disturbance are difficult. This difficulty can be largely overcome by examining population turnover rather than foliage condition on a single life stage of the population (even when the specific agent of decline or increase is not established).

5.3   ESTABLISH WHICH POPULATIONS AND SPECIES ARE MOST AT RISK

Given that there is concern about numerous tree species throughout New Zealand, a system is required that objectively establishes priorities for research and management among these species. A landscape–scale assessment of tree population imbalance is needed. For example, at Hihitahi Forest Sanctuary there was almost no recruitment observed during our field visit or recorded in NVS plots, and the adult population appeared to be in decline across this site. In contrast, Egmont National Park may have had small patches in which *L. bidwillii* was dying back, but there were also regenerating patches within the park. Past evidence shows that this species has been on 'boom and bust' cycles throughout its history (Rogers & McGlone 1989). Similarly, *N. fusca* in the Pohangina Valley consists of a mixture of declining and healthy stands. Priority setting depends on three factors: rate of adult tree mortality, rate of recruitment, and the net population changes summed across all stands in a particular area. Higher-priority areas are those with negative or lowest net population growth across all stands in a given area. An added complication is that population changes may shift along environmental or stress gradients

throughout the distribution of tree species. Allen et al. (2002) list several other considerations in prioritising species management including:

- If a species has a restricted distribution, then potentially it would be more vulnerable than species that are widely distributed and/or only affected over part of their range. In addition, species that have a restricted distribution and are also locally rare would be particularly vulnerable.

- Species are vulnerable at critical stages in their life history. Species that are vulnerable to browsing and predation at several stages in their life history may be more vulnerable than those that are susceptible at only a single stage. Any assessment should be modified by an appreciation of variable longevity.

- If several factors are implicated in the mortality of a species, that species is likely to be more vulnerable. Where natural factors are identified as the cause of mortality, a species may be ranked less vulnerable.

- The loss of the dominant species in an ecosystem can have the greatest impact since those that contribute the largest biomass are often drivers of ecosystem processes. If species that are suffering mortality clearly play a role in key ecosystem processes, they would be scored as more vulnerable.

There are considerable knowledge gaps for implementing such a priority scheme, but this can be seen as a challenge for ongoing research and management.

5.4 DEVELOP AND IMPLEMENT A RESTORATION STRATEGY

A major question is whether sites or forests identified as a priority can be restored to some desirable state by appropriate action. Historically, most conservation expenditure with a goal of maintaining canopy tree species has focused on pest control. An alternative possibility is that in some instances, sites may have undergone irreversible changes beyond historical boundaries (e.g. Coomes et al. 2003; Husheer et al. 2003). For example, shrub thickets, particularly of *P. colorata*, have always been a part of the New Zealand landscape, but may currently be more widespread or abundant because pests have eliminated more-palatable shrub species; this hypothesis needs to be rigorously tested. If *P. colorata* thickets suppress tree regeneration more than the palatable species that have been removed, then we may be observing pest effects that only manifest themselves as imbalances in canopy tree populations decades later. Certainly, the ability of systems to recover is somewhat more complex than merely removing the herbivores because plant understorey changes may take many years to reverse. Increasingly we are aware that herbivores can have major effects on the decomposer subsystem, which in turn is likely to exert long-term effects on the performance of the ecosystem. An open question is whether we should be managing for herbivore–absent ecosystems, or whether additional treatments over-and-above pest removal are needed to redress tree population imbalances (e.g. shrub thicket removal; Simard et al. 2003). While it remains difficult to distinguish what processes are natural or not, DOC needs to consider a wider range of management options if it is to retain current values in sites such as Hihitahi Forest Sanctuary.

6. Acknowledgements

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Appendix 1

Libocedrus CALLUS INDUCER/BROWSER

By John Dugdale, Landcare Research, Nelson

Dieback in stands of mountain cedar (*Libocedrus bidwillii*) near Hihitahi had not diminished after a programme of possum control. Landcare Research has a contract with DOC to find if there are other causative agents beside possum browse. Damage to live branches was noted and samples taken by Duane Peltzer, Landcare Research, Lincoln.

I examined these branches, which were forwarded several months after they had been cut live from affected trees. By this time, any live organisms had dehydrated and died, and the wood tissues had dried out. Despite the desiccation, each area of damage except one showed the same conditions:

An eroded area of irregular shape, and some 4 mm below the level of the surrounding bark-covered tissues, edged with a wall of callus tissue, and floored by stained xylem, superficially weathered grey.

Within this stained region were several tunnels, flattened-oval in cross-section, each bored into the central wood and ending as a straight elongate blind chamber roughly 4 mm wide and 13–15 mm long. The stained areas (particularly dark around the blind chambers) extended only a little beyond the eroded area and did not occupy the whole volume of the branch.

In two instances, a frass-covered gallery of strong white silk was appressed to the callus edge; the silk was found to be very strong and seemingly almost endlessly extensile.

In one instance, removing the bark revealed a flat irregular-sided gallery with frass pellets and some silk strands, and the gallery had been chewed down through the phloem/cambium to the outer sapwood. The caterpillar had long since gone.

The frass pellets and silk indicated that the subcortical (under bark) gallery maker was a lepidopteran.

Three blind chambers each contained a pupa from which the adult moth had long since emerged. In one there was a thin cocoon of wood fibres and widely spaced silk strands. Another chamber contained a hymenopterous parasite cocoon and a shrivelled larva. Two pupal skins still had the caterpillar skin and head-capsule bits with them, and two shrivelled larvae were also found.

The pupal skins were fragile and dull rather than polished; the antennal sheaths curved in and met in midline, the abdomen tergites had no row(s) of backwardly directed thorns, and the hind-leg sheaths projected beyond the antennal sheaths.

In another instance a gallery entirely full of tightly packed frass, typical of cerambycid larval working extended from a decayed branch stub sub-cortically

into what had been live tissue. Lack of reaction tissue there indicates that this gallery was extended subsequent to severing the branch. Such work is typical of conifer-cerambycids, e.g. *Stenopotes pallidus*, but has no pathological implications.

At first I thought that the damage was like that caused by species of *Heterocrossa* (Carposindae) on houhere trunks and branches, and on wound tissue on trunks of many forest trees and old puriri moth burrow mouths (all callus tissue sites). However, carposinid pupae have the antennal sheaths widely separated (Common 1990: 103, and fig. 4), and the integument is, if anything, even flimsier and paler than the ones found in *L. bidwillii*. Also, structures of the shrivelled larvae agreed with structures of the larval skins associated with the pupal skins and did not agree with larval structures of Carposinidae (e.g. maxillary structure, prothoracic pre-spiracular setal number, dorsal modification of the posterior abdominal segments). Lastly, carposinids in New Zealand are not known to bore into the wood of their hosts to pupate, but pupate in a frass- and debris-encrusted cocoon, often some distance away.

I then thought that the larvae and the pupae might not be the primary callus-eaters and chamber borers; galleries in woody stems can be invaded by other arthropods (succesori), and there is one genus of moths whose larvae browse lichens on branches and use old wood-borer tunnels as ready-made refugia. But there was no indication of any other organism. None of the wood-eating biologies known for New Zealand weevils (May 1993) was applicable, as neither belids nor psephalocine weevils use the entry tunnel as an exit tunnel, nor are any callus-inducers / eaters.

The larval chaetotaxy and the disposition of the pupal antennae and other appendage sheaths point to the agent being a gelechioid moth (Hodges 1998: 131; Common 1990: 221, and fig. 79). Gelechioidea is one of the larger Lepidoptera superfamilies. Common (1990: 227–229) describes callus-feeding and pupation in wood for many taxa in the Xyloryctidae, a gelechioid family greatly developed in Australia. I have found that silk galleries and cocoons of Oecophoridae (in the broad sense) are difficult to tease apart, as the silk strands seem almost infinitely extensible.

Conclusion

The damage is like that described for Australian moth groups in Xyloryctidae (Gelechioidea). Characters shown by remains of larvae and pupae also suggest this group. The New Zealand genus *Phaeosaces* Meyrick has a species (*P. coarctatella* Walker) close in size, but note that this species is known as a lichen browser on branches, and as a user of old wood-borer tunnels for a larval refuge.

Recommendations

• The stained wood should be examined to see if there is a pathogen that has stained the wood, and if there are tyloses in the tracheids. This assumes dieback was caused by an interruption to the water column.

• Examination of fresh, live damage, preferably with organisms inside, would be more helpful than looking at more old damage. The fact that all samples

seemed to be of equal condition (with the original agents long gone (two, three, or more seasons ago) suggests that current damage is disguised by bark that has not weathered away, and therefore will not be obvious.

- Caterpillar biology of *P. coarctatella* needs checking.

- There is a strong possibility that a canopy-dwelling species of the genus *Hierodoris* (another probable xyloryctid) is the agent. Dr Robert Hoare (Landcare Research, Auckland) will need to be involved.

References to Appendix 1

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