Restoration of mountain beech (*Nothofagus solandri* var. *cliffortioides*) forest after fire

Nick Ledgard and Murray Davis
Forest Research, P. O. Box 29 237, Fendalton, Christchurch, New Zealand
(E-mail: nick.ledgard@forestresearch.co.nz)

Abstract: Fire occurs relatively frequently in beech (*Nothofagus*) forest in drought prone eastern areas of the South Island, New Zealand. Because beech is poorly adapted to fire, and is slow to regenerate, forest is normally replaced by scrub or grassland. Seeding was investigated as a means of restoring mountain beech (*N. solandri* var. *cliffortioides*) forest after fire destroyed 300 ha of forest at Mt. Thomas, Canterbury, in 1980. A mixture of mountain beech, *Leptospermum scoparium* and other small tree and shrub species was sown within a year of the fire in the presence and absence of pasture species as a cover crop, and fertiliser. Seeding of mountain beech and *L. scoparium* was successful, but other species were of limited success. Competition from pasture species inhibited establishment of all native species. Fertiliser increased *L. scoparium* plant numbers in the first year but had no other beneficial effect on establishment of native species. *Leptospermum scoparium* provided a dense shrub cover in plots where the native species were sown in the absence of pasture species, but mountain beech had begun to overtop the shrub canopy by 20 years after seeding. Browsing by insects or small animals in the first 2 years is suggested as the main cause of mortality in mountain beech. Mountain beech seeded at 1.4 kg/ha resulted in about 1800 saplings/ha at age 20. It is suggested that seeding the wider burn area more than 2 years after the fire would have been unsuccessful because of competition from herbaceous species, especially *Agrostis capillaris*, which rapidly invaded the burnt area. A strategy is outlined for establishing mountain beech over large areas when limited quantities of seed are available.

Keywords: competition; fertiliser; fire; *Leptospermum scoparium*; *Nothofagus solandri* var. *cliffortioides*; rehabilitation; seeding.

Introduction

Forests dominated by beech (*Nothofagus*) species cover about 2.9 million ha and account for almost half of the total area of indigenous forest in New Zealand (Wardle, 1984). The beech forests thus form a major carbon pool (Hall *et al*., 2001) and repository of indigenous species, and are an important economic, soil protection and recreation resource (Wardle, 1984). New Zealand beech species are poorly adapted to fire (McQueen, 1951; Wardle, 1984) and even low temperature burns can lead to forest destruction. Forest areas destroyed by fire, particularly in drier eastern regions, can be very slow to regenerate as seed is infrequently produced and is not adapted for long distance dispersal (Wardle, 1984; Allen and Platt, 1990). Forest regeneration following fire is characteristically by slow marginal spread, although instances of long distance spread from forest have been recorded (Burrows and Lord, 1993). Hence, fires often result in a new vegetative cover of scrub or grassland (Dick, 1956), often dominated by exotic species. If land managers wish to accelerate the recovery of beech forest after fire, active rehabilitation may be required, but there is little information available on possible rehabilitation procedures.

The Canterbury region in the eastern South Island of New Zealand commonly experiences dry summer conditions and is especially prone to forest fire. Since 1970 there have been at least 9 fires in mountain beech (*N. solandri* var. *cliffortioides*) forest in the region (Table 1). The largest of these was at Mt. Thomas when approximately 300 ha of forest was burnt in October 1980. Wiser *et al.* (1997) followed natural recovery for 15 years after the fire. They observed some recovery of woody species, but it was mostly from onward growth of seedlings that survived the fire, rather than establishment of new seedlings out from the forest margin. Following the Mt. Thomas fire, a study was initiated to determine whether forest recovery could be assisted by sowing seed of mountain beech and some associated native shrub and small tree species. Much of the forest destroyed by fire elsewhere in Canterbury has been replaced by grassland.

Table 1. Fires in Canterbury beech forests since 1970.

Location	Year	Area burnt (ha)[1]
Mt. White	1972	16
Oxford	1974	20
Boyle River	1975	29
Flock Hill	1977	15
Flock Hill	1979	5
Mt Thomas	1980	300
Bealey	1981	<1
Cass	1995	100
Mt. Richardson[2]	1995	11

[1]Areas to 1981 are from records of the former New Zealand Forest Service. Areas for Cass and Mt Richardson are from Department of Conservation
[2]This fire occurred within the area originally burnt at Mt. Thomas in 1980.

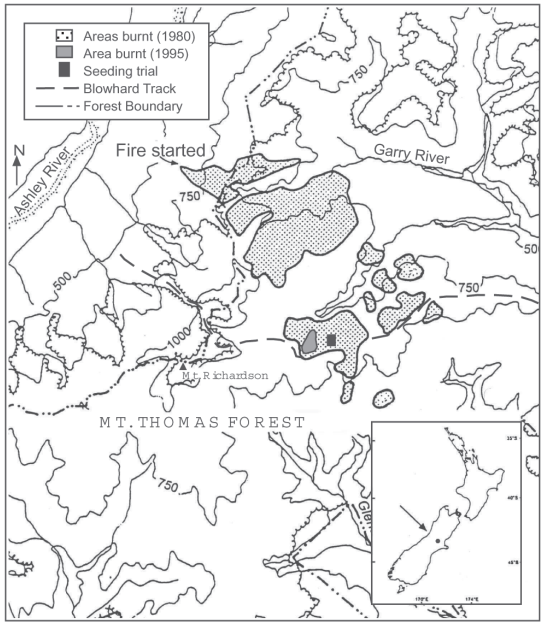

Figure 1. Location of seeding trial in the forest burnt at Mt. Thomas in 1980.

Competition from herbaceous species may inhibit establishment of beech (Wardle, 1984), and germination of shrub species (Wiser *et al.*, 1997). In contrast, Ledgard (1976) and Ledgard and Baker (1988) have shown the importance of the presence of herbaceous cover in assisting establishment of conifer species on eroded mountain soils. Seeding was therefore examined in the presence and absence of herbaceous pasture species. As site fertility may influence establishment and growth of native species, and also the vigour of competition from herbaceous species, seeding treatments were examined in the presence and absence of fertiliser. The trial was sown within a year of the fire, and results up to 19 years afterwards are reported here.

Study area

The study area is fully described by Wiser *et al.* (1997). Mt. Thomas Forest (Fig. 1) is a remnant of formerly extensive Canterbury foothill forests and covers several thousand hectares with an elevation range of 400–1000 m. Black beech (*N. solandri* var. *solandri*) dominates forests below 500 m elevation, while mountain beech is dominant above 500 m (Wardle, 1984). Rainfall is more than 1000 mm and is relatively evenly distributed throughout the year, although dry periods are common in the summer. The soils are mapped as Hurunui steepland soils (New Zealand Soil Bureau, 1968) which are Acid Brown Soils in the classification of Hewitt (1993). They are developed from indurated greywacke and are mostly stony silt loams.

The fire in October 1980 burnt understory shrubs and the forest floor litter, and scorched the canopy trees, though initially much of the foliage was left intact. Some surviving trees developed a flush of growth within 6 months, and by 18 months a small percentage also developed small epicormic shoots (Wiser *et al.*, 1997), but all except for a few trees close to the unburnt forest margin had died within 5 years. The northern slopes of the burnt area adjoin pasture-land in Lees Valley (Fig. 1). Initially these slopes were were grazed periodically by sheep (*Ovis aries*). The southern area is surrounded by intact forest and had minimal grazing by sheep. Low numbers of red deer (*Cervus elaphus*), possums (*Trichosurus vulpecula*) and hares (*Lepus europaeus*) were present on both northern and southern areas.

Methods

A trial site 200 m from the nearest mountain beech margin was selected on a north-facing slope within the southern part of the burn at an altitude of 840 m (Fig. 1). Seed of locally collected mountain beech and associated small tree and shrub species was sown in plots in September 1981, 11 months after the fire. Apart from mountain beech, all seed was collected following the fire. Control plots, not sown with native woody species, were included in the trial. The influence of herbaceous species on establishment of woody species was examined by sowing the native woody species and control plots with or without a mixture of pasture grass and legume species. Molybdenised superphosphate (1 t/ha) and di-ammonium phosphate (80 kg/ha) fertiliser was applied to the lower half of each plot. The fertiliser treatments were thus sub-plots of a split plot design. Plots were not cultivated or otherwise disturbed, and measured 5 x 10 m, the short side being parallel to the slope. Within the trial area, which was completely bare of vegetation at the time of sowing, individual plots were located between dead beech stems on areas with a relatively even surface. There were 4 replicates of each treatment arranged in a randomised block layout. The trial area measured 50 x 50 m and was fenced to exclude deer, but tree fall resulted in the fence becoming ineffective after five years.

The small tree and shrub species used in the trial (Table 2) all occur in beech forest at Mt Thomas and were selected mainly on the basis of seed availability following the fire. *Leptospermum scoparium* seed was stored and sown dry, but all other species were stratified moist for 7 weeks prior to sowing. The seed from berry fruits of *Griselinia littoralis*, *Rubus cissoides*, *Coprosma* "sp (t)", *Carpodetus serratus* and *Aristotelia serrata* were stratified in moist sand. The seed was not tested for viability. No information was available on appropriate field seeding rates, and rates used were based only on quantities of seed available. The pasture species included *Agrostis capillaris*, *Dactylis glomerata* and *Trifolium repens*, each sown at 5 kg/ha. *Trifolium repens* seed was inoculated with *Rhizobium* immediately prior to sowing. Rain fell within 48 hours of sowing.

In each plot all visible mountain beech seedlings were located and marked (using an aluminium peg with numbered tag attached) during the winter following trial establishment. The survival and height of seedlings (< 1.35 m tall, Allen and McLennan, 1983) and saplings (> 1.35 m) were recorded periodically thereafter. The percentage cover of native woody species, pasture species, other exotic herbs, litter, woody debris (including bark) and bare ground were visually estimated (to the nearest 5%) and recorded periodically in each of 20 contiguous 50 x 50 cm quadrats located along a transect bisecting the plot. Ten quadrats were located in each of the unfertilised and fertilised sub-plots. Within each quadrat the presence of all species was recorded, along with the number of native woody species.

Table 2. Native woody species and application rates used in the seeding trial.

	Common name	Application rate kg/ha	seeds/m^2
Nothofagus solandri var. *cliffortioides*	Mountain beech	1.4	31
Leptospermum scoparium	Manuka	6.0	6000
Coprosma "sp (t)"		0.1	2.3
Carpodetus serratus	Putaputaweta	5.4	171
Rubus cissoides	Bush lawyer	nd[1]	-
Aristotelia serrata	Wineberry	nd[1]	-
Griselinia littoralis	Broadleaf	2.0	8.6

[1]not determined

Analysis of variance was used to determine significance of treatment effects on vegetative cover and numbers of shrubs and small trees. Cover estimates and counts were averaged across the 10 quadrats in each sub-plot and the sub-plot means derived were used for the analysis. Log-transformed data were used for analysis of variance. Untransformed means with standard errors are shown in tables.

Results

Vegetative cover development

The sowing of pasture species significantly hastened the establishment of a vegetation cover (Table 3, Fig. 2). Six months or one growing season after sowing (1.5 years after the fire) plots not sown with pasture species remained largely bare of vegetation, whereas cover in plots sown with pasture species amounted to about 20% in the absence of fertiliser and 60% where fertiliser was added (Fig. 2a). At 2.5 years (3 growing seasons) after sowing, cover in pasture plots averaged 90% irrespective of whether fertiliser had been added, while the cover in plots not sown with pasture species ranged between 30 and 50% (Fig. 2b). Pasture species invaded plots where they were not sown and cover of these species reached 30–60% after 4 growing seasons (Fig. 2c). At this stage total vegetative cover on all plots was similar, ranging between 70 and 90% (Fig. 2c). Pasture cover declined between years 3.7 and 18.7 in all plots and was replaced by native woody species (Figs. 2c, d), and woody debris arising from fall of dead stems (data not shown). *Agrostis capillaris* was the dominant pasture species throughout the trial.

Cover development of native woody species was slow, with no plots having more than 1% native cover 6 months after sowing (Fig 2a). After 2.5 years, however, the plots sown with native species alone (without grasses and legumes) had a native woody cover of around 20% (Fig. 2b). Only about half of this could be attributed to sowing, however, as native woody species from natural regeneration were also present in the control plots. After 18.7 years, the plots sown with native species alone had a cover of approximately 80% of native woody species present (Fig. 2d), compared with a cover of 20–40% in the

Table 3. *F* and *P* values from analysis of variance of the effects of seeding pasture (P) and native woody species (N) and fertiliser (F) application on the cover of pasture and native woody species at four assessment times.

Years after seeding	Source	Pasture species		Native woody species	
		F	*P*	*F*	*P*
0.5	Pasture	381.75	< 0.01	12.67	< 0.01
	Native	0.10	0.77	44.27	< 0.01
	Fertiliser	203.61	< 0.01	0.60	0.45
	P x N	0.19	0.78	10.56	< 0.01
	P x F	41.72	< 0.01	0.02	87.85
	N x F	0.19	0.67	0.22	0.65
	P x N x F	2.43	0.16	0.02	0.88
2.5	Pasture	139.97	< 0.01	142.07	< 0.01
	Native	0.04	0.84	2.72	0.11
	Fertiliser	0.08	0.79	0.20	0.65
	P x N	0.03	0.87	4.28	0.05
	P x F	0.04	0.83	0.00	0.97
	N x F	1.60	0.24	0.05	0.83
	P x N x F	1.58	0.24	0.04	0.84
3.7	Pasture	17.03	0.03	120.45	< 0.01
	Native	4.22	0.13	7.85	0.01
	Fertiliser	3.56	0.16	0.03	0.87
	P x N	2.41	0.22	7.85	0.01
	P x F	2.23	0.17	0.03	0.87
	N x F	6.00	0.04	0.92	0.34
	P x N x F	1.01	0.34	0.35	0.56
18.7	Pasture	14.94	0.03	30.33	< 0.01
	Native	81.86	< 0.01	19.40	< 0.01
	Fertiliser	6.69	0.08	0.16	0.69
	P x N	11.57	0.04	0.01	0.90
	P x F	1.10	0.32	0.89	0.35
	N x F	0.40	0.54	0.68	0.42
	P x N x F	2.11	0.18	0.00	0.95

control plots. The suppressive effect of the pasture species on development of both sown native species and natural regeneration was clearly evident at all assessments (Table 3, Fig. 2). Native woody cover was not significantly affected by fertiliser application at any stage (Table 3).

Establishment of mountain beech

Mountain beech seeding was successful when sown in the absence of pasture species (Table 4). No beech seedlings were observed in the treatments where it was not sown, and only one ephemeral seedling was observed (2.5 years after seeding) where mountain beech and other native species were sown in the presence of pasture species. Fertilisers did not significantly affect seedling numbers (Table 4, Fig. 3a), or height growth (Fig. 3b).

At nine months after sowing a total of 41 seedlings (0.7% of seed sown) were individually marked in the 4 plots where native species were sown alone, and a further 2 seedlings were marked the following year. At the final assessment (18.5 years after sowing) only 21 of these (49%) were still alive (Fig. 3a). Initial mortality was high with 15 seedlings (68% of total mortality) being lost during the first 2 years. Much of this early mortality may have been due to browsing. Nearly half (49%) of the marked seedlings were browsed in the first 2 years, and 60% of seedlings lost during the first 2 years were severely browsed, most to the point of having all leaves removed. Browsing could be attributed to either insects or animals, most likely by hares or possums, as larger animals were excluded.

A further 16 unmarked seedlings were present at the final assessment. All of these seedlings were located in plots where woody species were sown in the absence of pasture species. Twelve of these fell within the height range of the surviving marked seedlings (1.4–3.2 m), indicating they were probably present, but not sighted, at the initial assessment. The remaining four seedlings were smaller and may not have been present initially. Three of these ranged in height between 1.0 and 1.3 m while the fourth was only 0.24 m, suggesting it might have been a relatively recent recruitment. At the final assessment 36 mountain beech saplings were

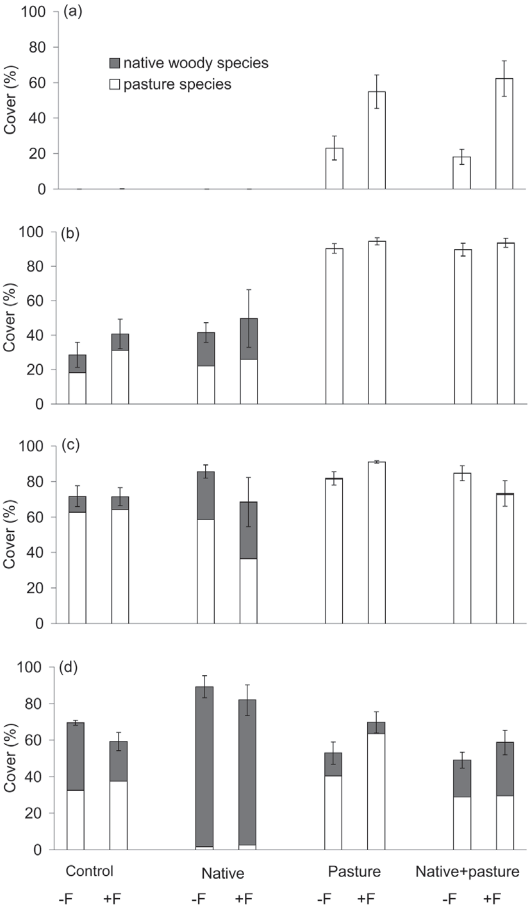

Figure 2. Effect of seeding treatment and fertiliser on ground cover of pasture and native woody species at (a) 0.5, (b) 2.5, (c) 3.7 and (d) 18.7 years after seeding. Bars show standard errors, $n = 4$. (–F = no fertiliser applied, +F = fertiliser applied at sowing).

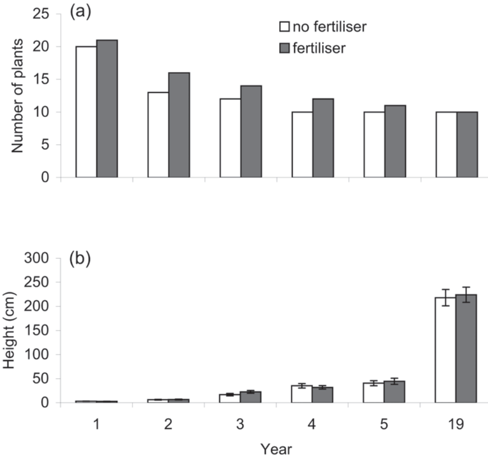

Figure 3. Change over time in total number (a) and mean height (b) of mountain beech plants located and marked 1 year after seeding in all plots where native species were seeded with and without fertiliser in the absence of pasture species. Bars show standard errors.

present in the 4 plots where it was sown in the absence of pasture species giving an establishment rate of 0.6% of total seeds sown, and a stocking of 1800/ha. The saplings had a mean height in excess of 2 m (Fig 3b). No seedlings succeeded in any of the other treatments.

Establishment of small trees and shrubs

Initial establishment of *Leptospermum scoparium* was much more successful than that of all other species, with nearly 40 seedlings/m^2 being recorded at the end of the first growing season in fertilised plots where only native species were seeded (Fig. 4). Although *L. scoparium* seedlings were present at the end of the first growing season in plots seeded with native and pasture species, the presence of pasture species greatly decreased *L. scoparium* plant numbers (Table 4, Fig. 4). The effect was less pronounced but still evident at the final assessment. Fertiliser increased numbers of *L. scoparium* seedlings initially, but this effect was soon lost (Table 4, Fig. 4). Up to 3.7 years after seeding no *L.scoparium* plants were recorded in plots where it was not sown, however small numbers of plants arising from spread from the initial seeding were present in unsown plots after 18 years (Table 5). *Leptospermum scoparium* grew rapidly (plant heights averaged 25 cm and 60 cm after 2.5 and 3.7 years respectively) and contributed most of the native cover on plots where native species were sown alone. *Leptospermum scoparium* saplings were up to 2.5m tall at the final assessment. The greater success of *L. scoparium* is probably attributable to the very high seeding rate used, as the number of seedlings 6 months after sowing in the treatment with highest establishment (natives sown alone, with fertiliser) amounted to only 0.6% of seed sown.

Seeding of *Coprosma* sp. "t" did not significantly increase plant numbers in the first 3.7 years after sowing, but some limited success was achieved from seeding of *Carpodetus serratus*, *Rubus cissoides* and *Aristotelia serrata* (Tables 4, 5). All 3 of the latter species failed to establish where pasture species were

Table 4. *F* and *P* values from analysis of variance of the effects of seeding pasture (P) and native woody species (N) and fertiliser (F) application on numbers of plants of five sown native woody species at four assessment times.

Years after seeding	Source	*Nothofagus solandri* v. *cliffortioides*		*Leptospermum scoparium*		*Coprosma* spp.[1]		*Carpodetus serratus*		*Rubus cissoides*[2]	
		F	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*
0.5	Pasture	16.73	<0.01	22.25	<0.01	0.86	0.36	9.70	<0.01	7.23	0.01
	Native	16.73	<0.01	228.44	<0.01	0.01	0.93	6.04	0.02	1.62	0.22
	Fertiliser	0.01	0.91	5.01	0.04	0.40	0.53	4.06	0.06	0.02	0.88
	P x N	16.73	<0.01	22.25	<0.01	2.41	0.14	1.49	0.24	1.62	0.22
	P x F	0.01	0.91	0.95	0.34	0.51	0.48	1.22	0.28	0.02	0.88
	N x F	0.01	0.91	5.01	0.04	1.10	0.31	0.12	0.74	1.62	0.22
	P x N x F	0.01	0.91	0.95	0.31	0.04	0.84	0.00	0.99	1.62	0.22
2.5	Pasture	13.86	<0.01	127.69	<0.01	9.82	<0.01	30.85	<0.01	52.57	<0.01
	Native	13.86	<0.01	154.90	<0.01	0.10	0.75	0.64	0.43	52.57	<0.01
	Fertiliser	0.02	0.89	0.00	0.96	0.17	0.68	1.65	0.21	0.18	0.68
	P x N	13.86	<0.01	127.69	<0.01	1.31	0.27	0.64	0.43	52.57	<0.01
	P x F	0.02	0.89	0.00	0.96	0.28	0.64	1.65	0.21	0.18	0.68
	N x F	0.02	0.89	0.00	0.96	1.19	0.29	0.03	0.87	0.18	0.68
	P x N x F	0.02	0.89	0.00	0.96	0.07	0.79	0.03	0.87	0.18	0.68
3.7	Pasture	14.10	<0.01	38.75	<0.01	5.31	0.03	26.59	<0.01	56.53	<0.01
	Native	14.10	<0.01	76.34	<0.01	0.11	0.74	0.84	0.37	31.21	<0.01
	Fertiliser	0.06	0.80	0.49	0.49	0.02	0.90	1.54	0.23	0.06	0.81
	P x N	14.10	<0.01	38.75	<0.01	3.81	0.07	0.84	0.37	31.21	<0.01
	P x F	0.06	0.80	0.43	0.52	0.61	0.44	1.54	0.23	0.06	0.81
	N x F	0.06	0.80	0.49	0.49	0.86	0.37	0.00	0.99	0.16	0.69
	P x N x F	0.06	0.80	0.43	0.52	0.00	0.99	0.00	0.99	0.16	0.69
18.7	Pasture	110.86	<0.01	14.79	<0.01	14.29	<0.01	19.65	<0.01	-	-
	Native	110.86	<0.01	121.85	<0.01	2.17	0.16	3.53	0.08	-	-
	Fertiliser	0.64	0.43	2.83	0.11	2.97	0.10	0.01	0.93	-	-
	P x N	110.86	<0.01	11.82	<0.01	6.28	0.02	3.53	0.08	-	-
	P x F	0.64	0.43	1.26	0.28	0.32	0.58	0.01	0.93	-	-
	N x F	0.64	0.43	0.21	0.65	0.94	0.35	0.24	0.63	-	-
	P x N x F	0.64	0.43	2.34	0.14	0.80	0.38	0.24	0.63	-	-

[1]*Coprosma* spp. were *C.* "*sp (t)", C. linariifolia*, *C. pseudocuneata* and *C. microcarpa*; *C.* "sp (t)" was the only species sown, and the only species present in the first three assessments. In the last assessment it was still dominant, but other species were also present.
[2]*Rubus cissoides* plant numbers were not counted at the final assessment in plots where it was sown in the absence of pasture species. The habit of rooting from trailing stems precluded identification of individual plants.

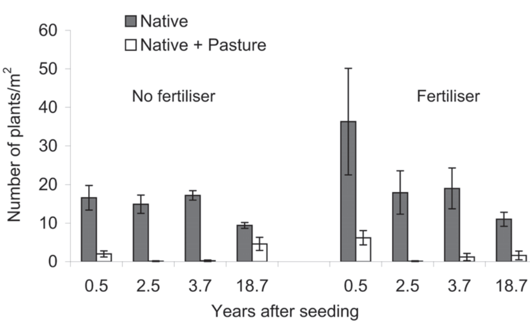

Figure 4. Effect of pasture species and fertiliser on density of *Leptospermum scoparium* plants in plots seeded with native species at 0.5, 2.5, 3.7 and 18.7 years after seeding. Bars show standard errors.

sown (Table 5). Fertiliser did not significantly affect *C. serratus*, *R. cissoides or A. serrata* plant numbers at any assessment time (Table 4, data not shown for *A. serrata*). A number of seedlings of *C. serratus* established from natural regeneration in control plots, but plant numbers were higher where native species were sown after 6 months. Ultimately, however, plant numbers were no higher in seeded plots (Tables 4, 5). Initial establishment of *C. serratus* seedlings, as a proportion of seed sown (1.5% in plots sown with native species with fertiliser), was the highest recorded for the sown species for which the seeding rate (as seed number/m^2) was known. Small numbers of *A. serrata* plants were present, only in plots sown with native species alone, from 0.5 to 3.7 years, but no plants were recorded at the final assessment (Table 5). *Rubus* *cissoides* seedlings were present in control plots, but plant numbers were higher where natives were seeded in the absence of pasture species (Tables 4, 5).

Although seeding of *Coprosma* "sp. (t)" did not significantly increase plant numbers initially, plants of other *Coprosma* species (*C. pseudocuneata*, *C. microcarpa* and *C. linariifolia*) established on the plots through natural regeneration. Similarly, plants of the unsown species *Ozothamnus leptophyllus* established as natural regeneration (Table 5). Plant numbers of *Coprosma* spp. and *O. leptophyllus* increased substantially at the final assessment, and it is noteworthy that plants of the latter species established as well in plots where pasture species were sown as where they were not sown. In contrast, more plants of *Coprosma* spp. were recorded in plots sown with native species without pasture than in the presence of pasture species (Table 5). More plants of *Coprosma* spp. established in unfertilised (4.5 ± 1.19) than fertilised (2.3 ± 0.77) plots ($P = 0.03$). Similarly more plants of *O. leptophyllus* established in unfertilised (2.2 ± 0.69) than fertilised (1.2 ± 0.32) plots ($P = 0.04$).

Other unsown species found as rare individuals up to 3.7 years after trial establishment, but not at year 18, included *Fuchsia excorticata* and *Pseudowintera colorata.* At year 18 individual plants were found of *Myrsine divaricata*, *Cyathodes* sp. and *Olearia* sp.

Table 5. Numbers of small tree and shrub plants in four seeding treatments at four assessment times after seeding. Values are means (± standard error), of 2 fertiliser treatments and 4 replicates.

Species	Years after seeding	Number of plants/m^2			
		Control	Native	Pasture	Native + Pasture
Leptospermum scoparium	0.5	0	26.5 (± 7.55)	0	4.1 (± 1.25)
	2.5	0	16.4 (± 2.87)	0	0.1 (± 0.06)
	3.7	0	11.2 (± 8.00)	0	0.8 (± 0.58)
	18.7	0.1 (± 0.06)	10.2 (± 0.96)	0.5 (± 0.05)	3.1 (± 1.10)
Coprosma spp.[1]	0.5	0.7 (± 0.26)	2.4 (± 0.89)	1.7 (± 1.10)	0.4 (± 0.24)
	2.5	0.6 (± 0.13)	1.5 (±0.66)	0.3 (± 0.20)	0.1 (± 0.05)
	3.7	0.3 (± 0.10)	1.3 (± 0.54)	0.4 (± 0.24)	0
	18.7	2.7 (± 0.98)	7.4 (± 1.17)	2.9 (± 1.70)	0.7 (± 0.26)
Carpodetus serratus	0.5	0.8 (± 0.34)	2.9 (± 0.95)	0.30 (± 0.15)	0.6 (± 0.21)
	2.5	1.1 (± 0.38)	2.2 (± 0.71)	0	0
	3.7	0.8 (± 0.31)	1.8 (± 0.66)	0	0
	18.7	0.8 (± 0.60)	1.0 (± 0.24)	0	0
Rubus cissoides[2]	0.5	0.1 (± 0.07)	0.4 (± 0.16)	0	0
	2.5	0	3.2 (± 0.75)	0	0
	3.7	0.2 (± 0.07)	2.3 (± 0.44)	0	0
	18.7	0.5 (± 0.22)	-[2]	0	0
Aristotelia serrata	0.5	0	0.1 (±0 .05)	0	0
	2.5	0	0.7 (± 0.21)	0	0
	3.7	0	0.4 (± 0.25)	0	0
	18.7	0	0	0	0
Griselinia littoralis	0.5	0	0	0	0
	2.5	0	0	0	0
	3.7	0	0	0	0
	18.7	0.1(± 0.06)	0.5 (± 0.20)	0	0
Ozothamnus leptophyllus (not sown)	0.5	0	0	0.1 (± 0.05)	0
	2.5	0.2 (± 0.07)	0.2 (± 0.10)	0	0
	3.7	0.2 (± 0.07)	0.1 (± 0.05)	0	0
	18.7	1.9 (± 0.51)	0.6 (± 0.24)	2.5 (± 1.22)	1.8 (± 0.66)

[1, 2]See footnotes in Table 4.

Discussion

There have been few reports of successful seeding with native woody species in ecological restoration in New Zealand. Evans (1983) and Porteus (1993) describe procedures for establishing *Leptospermum scoparium* and *Kunzea ericoides* from seed on newly burnt or cultivated ground to provide a nurse cover for other native woody species, but we are aware of no reports of seeding with beech or other tall forest tree species. The results from the present trials show that mountain beech and *L. scoparium* may be re-established by seeding after beech forest destruction by fire.

Suppression of woody plants

The almost complete failure of mountain beech seedlings to establish in plots sown with pasture species, even in the absence of fertiliser, demonstrates their vulnerability to competition from herbaceous species. Wardle (1984) has previously noted the poor ability of beech seedlings to compete against herbaceous species. Establishment of other native woody species from seeding was also inhibited by the presence of pasture species. At sowing there was essentially no competing vegetation present, and control plots were still largely bare 6 months after sowing (17 months or 2 growing seasons after the fire). At 2.5 years (or 3 growing seasons) after the fire, however, unfertilised control plots had a cover of 30%. Although only two thirds of this was made up of pasture species, this amount of cover may have been sufficient to limit establishment of woody seedlings, as evidenced by their lack of initial establishment in unfertilised plots sown with pasture species. Cover of herbaceous species in the wider burn area 2.5 years after the fire appeared similar to that in unfertilised control plots (20–30%). As in the plots, the wider burn area was dominated by *Agrostis capillaris* that probably originated from adjacent grassland (Wiser *et al*., 1997). Thus, in the wider burn area competition from herbaceous species would likely have inhibited establishment of mountain beech and other woody species if sown 3 years (or growing seasons) after the fire. Clearly, seeding within one or two growing seasons of fire is imperative for successful establishment of woody species where competition from herbaceous species is likely to develop.

Leptospermum scoparium establishment

Most cover on plots sown with native species alone was provided by *L. scoparium* throughout the assessment period, although mountain beech saplings were beginning to overtop the *L. scoparium* canopy at the final assessment. The very high seeding rate used for *L. scoparium* clearly contributed to this species success, as its establishment rate as a proportion of seed sown was similar to that of other species for which numbers of seed sown were known. Sessions and Kelly (2000) reported germination rates of 0.2 to 0.5% for *L. scoparium* for a seeding made into *Agrostis capillaris* swards within the Mt Thomas burn in 1999. These rates are similar to the establishment rates recorded at 6 months in the present study. The sowing by Sessions and Kelly (2000) was made prior to winter, however, and no seedlings survived.

Leptospermum scoparium was the only native woody species to respond to fertiliser. Initial plant numbers were increased by fertiliser, but the effect was transient, with plant numbers declining to similar numbers (*c*. 15/m^2) in fertilised and unfertilised plots by 2.5 years after seeding. The initial response is likely to have developed before seedlings became mycorrhizal. Baylis (1972) demonstrated two-fold growth responses by non-mycorrhizal *L. scoparium* seedlings to phosphate in a steamed forest soil in pot culture.

Leptospermum scoparium as a 'nurse' for beech

There is some evidence that, away from the forest margin environment, mountain beech will establish preferentially in communities where *L. scoparium* is present, both from natural seed dispersal (Burrows and Lord, 1993; Easterbrook, 1998) and from planting (Langer and Baker, 1993; Easterbrook, 1998). Easterbrook (1998) showed that transplanted mountain beech seedlings survived better amongst *L. scoparium* scrub than in beech forest edge, short tussock grassland, and open low vegetation (dominated by moss, *Coprosma petriei*, *Gaultheria depressa* and *Leucopogon fraseri*). Mortality of recently transplanted beech seedlings was high in canopy gaps created in the *L. scoparium* (by tying back plants), indicating the beneficial effect of *L. scoparium* was likely to have been through shading. Baylis (1980) suggested that beech may share the same mycorrhizal fungi with *Leptospermum*, which would confer nutritional and water uptake advantages to beech seedlings establishing amongst *L. scoparium*. The occurrence of fruiting bodies of several of the same species of mycorrhizal fungi under both *Leptospermum* and *Nothofagus* stands suggests that at least some species of mycorrhizal fungi may colonize both species (P. R. Johnston, Auckland, Landcare Research, N.Z., *pers comm*).

The present trial could not determine whether the presence of *L. scoparium* as a 'nurse' crop assisted mountain beech establishment and growth, as mountain beech was not sown alone. Since the seedlings established concurrently, it seems unlikely that *L. scoparium* seedlings conferred shade or mycorrhizal benefits to the establishing beech seedlings. The fact that both *L. scoparium* and mountain beech established successfully indicates that sufficient mycorrhizal propagules were present for their infection. Once established, *L. scoparium* grew faster and shaded mountain beech seedlings, which may have been detrimental to the seedlings; Easterbrook (1998) demonstrated more rapid growth of established (8–10 months after transplanting) mountain beech in canopy gap than in shaded environments. Competition from *L. scoparium* for nutrients and water may have also reduced growth of mountain beech.

In plots seeded with native species, mean stem densities of *L. scoparium* at 18.7 years were 31 000/ha and 102 000/ha with and without sown pasture species respectively. In East Coast North Island hill country, Bergin *et al*. (1995) found *L. scoparium*-dominated *L. scoparium*/*Kunzea ericoides* stands on old pasture sites typically contain more than 20 000 stems/ha within 10 years of establishment, the density declining to about 10 000 stems/ha by age 20. Maximum densities ranged up to about 65 000 stems/ha. Bergin *et al*. (1995) found *L. scoparium* declined significantly after age 20, with stands becoming dominated by *K. ericoides*. This result was consistent with other studies that have shown *L. scoparium* to decline from about age 20 years and be replaced by *K. ericoides* (Allen *et al*., 1992; Esler and Astridge, 1974). The present study indicates a similar scenario when *L. scoparium* and mountain beech establish together, with mountain beech beginning to overtop *L. scoparium* after about 20 years.

Early mortality of mountain beech

Most mortality of labeled beech seedlings in the fenced trial occurred within the first 2 years and is attributed to browsing by insects, possums or hares which may have been attracted to the trial area by the sown pasture species. The importance of browsing as an agent for beech mortality is uncertain, however, as the effect of insect or small animal exclosures on survival was not studied. Although plots seeded with native species alone were initially free of herbaceous species, a cover of pasture and other herbaceous species ultimately developed and competition from these species may have contributed to mortality of mountain beech seedlings that occurred beyond the end of the first growing season. A sward of pasture species may inhibit seedling mortality from frost lift, however, as has been shown for conifer seedlings seeded in exposed mountainland subsoils (Ledgard, 1976; Ledgard and Baker, 1988). No frost lift of beech or other woody seedlings was observed in the present study. Fall of dead stems may have accounted for mortality of some beech seedlings, especially after the fourth year when the rate of stem fall increased.

Other woody species

In addition to mountain beech and *L. scoparium*, seeding of *Rubus cissoides* was successful. Although plant numbers were not determined at the final assessment *Rubus cissoides* was common in the understory of plots where it had been sown in the absence of pasture species. It was also present in control plots. The failure of seeding of *Coprosma* "sp (t)" and *Griselinia littoralis* may be at least partly attributed to the low seeding rate used. Viability of seed was not determined so the importance of this factor in establishment failure of these species is not known. Sessions and Kelly (2000) reported much higher germination and survival of *Coprosma* species in *Agrostis capillaris* swards than of *L. scoparium* in their study at Mt Thomas. Seeding of *Carpodetus serratus* and *Aristotelia serrata* increased plant numbers initially in the absence of pasture species, but not in the long term, possibly because of competition from the developing dense *L. scoparium* canopy. Frosting may have also contributed to mortality of *A. serrata* plants as the species is not particularly frost hardy when young (Pollock, 1986).

Establishment of *Coprosma* species from natural regeneration was greatest where native species were sown in the absence of pasture species. The *L. scoparium* cover may have assisted *Coprosma* regeneration. *Coprosma* seed is spread by birds. The *L. scoparium* cover may have provided a roosting place for birds, or alternatively, a more suitable microclimate or soil environment for plant growth. The latter explanation seems more likely as sprouting from burnt stumps contributes substantially to *Coprosma* species regeneration after fire (Wiser *et al*. 1997). Such regeneration was also noted in this study. Regeneration of the wind dispersed *Ozothamnus leptophyllus*, on the other hand, was not significantly affected by sowing treatment, although plant numbers were lowest in plots sown with native species only and highest in plots of pasture only. This indicates that it may be more tolerant than other woody species of herbaceous competition. As such, its role as a primary woody plant coloniser in native woody plant revegetation of grassland or other herbaceous communities may be greater than generally recognised. *Ozothamnus leptophyllus* was first observed 3 years after the burn, the seed probably originating from wind spread from plants 2–3 km from the trial area. As this species is a fast-growing shrub which matures early, many of the plants observed at the final assessment may have been second generation, originating from locally-produced seed.

Conclusions

The study has shown that mountain beech and *L. scoparium* can be established by seeding shortly after fire. The window available for achieving good establishment may be no more than 2 years, after which competition from herbaceous species is likely to develop. Although the trial did not determine whether the presence of *L. scoparium* as a 'nurse' crop assisted mountain beech establishment and growth, in terms of producing a dense cover of native woody species as quickly as possible, the seeding of a combination of *L. scoparium* and beech appears particularly effective. Where mountain beech seed is not immediately available, seeding of the more readily available *L. scoparium* by itself may be useful to enable subsequent introduction of beech, since mountain beech appears to establish preferentially in *L. scoparium* stands. The mountain beech seeding rate of 1.4 kg/ha produced a stocking of 1800 saplings/ha after nearly 20 years. While this would be effective for reforestation, obtaining the quantities of seed required for an area the size of the Mt. Thomas burn (300 ha) would be difficult. An alternative for large areas could be to 'spot-seed' with beech, possibly with *L. scoparium*, with the aim of establishing at least a low density of seed trees.

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