Factors affecting the establishment, growth and survival of native woody plant communities on the Canterbury Plain, New Zealand

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The native plant communities of Canterbury, South Island, New Zealand have been severely modified and degraded and the Canterbury Plain (750,000 ha) retains few remnants of its original forest and other ecosystems. The research presented here considers the mutualistic roles of exotic and indigenous species in the process of restoring degraded landscapes. Exotic species may have an important role in the (re) establishment of desired indigenous species, and may influence succession through to a forest dominated by them.

One aspect of this work describes indigenous plant community regeneration facilitated by exotic willow (*Salix* spp.) woodland on the Canterbury Plain. Natural colonisation of the willow woodland by native plants was investigated, with respect to variation in the physical environment in the willow stand. Key factors in the success of willow woodland as a nursery for regeneration of native vegetation include: distance to the nearest seed source, the ability to attract seed dispersers (recruitment only occurred under perch sites), flooding potential (higher recruitment in areas less likely to flood) and possibly light availability. Control of vertebrate (and invertebrate) herbivory is also necessary for successful restoration.

A second aspect was a field experiment in open pasture and in a non-native remnant woodland which was then used to investigate the effects of shelter, plant spacing, mulching and fertiliser on growth and survival of planted native woody species. With minimal management, the selected mid-late successional plants established poorly in the open pasture and had low survival rates (e.g.,*Dacrycarpus dacrydioides, Pseudopanax arboreus, Aristotelia serrata, Melicytus ramiflorus*). Only narrow-leaved species (e.g., *Plagianthus regius, Hoheria angustifolia, Hebe salicifolia, Cordyline australis*) survived this open pasture planting. In contrast, most species (broad and narrow-leaved) established under the sheltered sites.

Exotic nursery vegetation and the establishment of native species, which will, in time, act as a seed source, will be important in successfully restoring a sustainable indigenous element in the cultural landscape of Canterbury. Ecological restoration requires an integrated approach, identifying and understanding the component processes of regeneration, and of the particular aspects/characteristics of the sites involved. This research shows that naturally established plants where existing shelter is available (in this case established willows) tend to have higher growth rates than individually planted plants in open situations, and that the availability of a suitable seed source can also contribute to successful establishment and growth rates.

The meeting of restoration targets on the Canterbury Plain may be accelerated, and costs reduced, through the utilisation of areas where exotic species occur (for instance, extensive willow stands in riparian areas adjacent to waterways) and more particularly, where a local seed source is also available. The findings of this research can contribute to restoration management in helping identify the best practices, based on research, that can lead to the restoration of original plant and animal communities

Key words: restoration, indigenous species, Canterbury, forest succession, *Salix cinerea*

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Chapter One

Introduction

The plant communities of the Province of Canterbury, New Zealand have been severely modified and degraded and the Canterbury Plain (750,000 ha) retains few remnants of its original forest and other ecosystems. Approximately 500 ha of native vegetation now survives on the Plain (C. Meurk pers. comm.) as well as 3610 ha on Banks' Peninsula (Wilson 1992). In pre-European times, the Plain was largely occupied by scrub and short tussock lands on the free-draining, dry sites, podocarp forests on more stable, moist or fine-textured, recent soils (Molloy 1993), and open wetlands in areas with high water tables where forest had not had time to re-establish following disturbance (Meurk & Norton 1988). Since European occupation (c.1860s), severe attrition and fragmentation of these original ecosystems has occurred. The original flora has largely been replaced by pastoral, arable, horticultural and non-native forest habitats which has resulted in local extinctions of native fauna in the region (Keesing & Wratten, submitted). Ecologically, the Plain is degraded.

In the last ten years, a worldwide increase in research interest in ecological restoration/ restoration ecology has occurred. Restoration ecology is evolving rapidly, but the field is still in its early experimental stages. A common view of ecological restoration is that it is an attempt to reinstate/restore biotic communities to their original pre-human state (Atkinson 1988). There is much to be said for this view as an idealistic model, but it can seldom be seen as an achievable goal, unless a very loose definition of what constitutes a pristine state is adopted (Atkinson 1990). To restore in an ecological sense means no more than to put back what has existed before and this depends on the kind of biotic community that is identified as of particular interest. Fundamentally the choice will depend on value judgements of stakeholders (Diamond 1987), although scientific analysis can assist in identifying and characterising communities worthy of being restored. The research presented here considers exotic and indigenous species in terms of facilitating the process of restoring degraded landscapes. Exotic species may have an important role in the re-establishment of desired indigenous species.

The study examines indigenous plant community regeneration facilitated by exotic woodland and active restoration in a farm agroecosystem on the Canterbury Plain, South Island, New Zealand. Some of the processes at the early stages of colonisation and plant succession are examined. Natural colonisation of a non-native, swamp woodland site by native plants is investigated, with respect to variation in the local environment. This early seral study site is referred to here as the Tai Tapu Study site. Artificial colonisation of an agricultural site, by experimental planting of native plant species, is explored and compared with the natural colonisation of the Tai Tapu Study site. The agricultural site is referred to as the Leeston site. Both study sites are located in the Low Plains Ecological District of Canterbury (McEwen 1987).

Goals of the research

1. To investigate native species establishment under *Salix* species.

2. To determine the most successful species and methods for encouraging native plant growth and survival in exotic farmland.

Aims of the research

* to quantify the community structure and population dynamics of regenerating native vegetation in a *Salix* spp. woodland and to measure growth and survival of individual woody plants.
* to investigate the effect of environmental variables on native plant recruitment and mortality in the *Salix* spp. woodland.
* to quantify how different planting regimes affect growth and survival of native vegetation in a farmland restoration scheme.
* to assist with successful management of fragments of regenerating native vegetation in lowland Canterbury.

Specific objectives

1. Tai Tapu Study site

* quantify establishment, growth and survival of native seedlings
* determine species composition of seed rain
* compare seed banks from the *Salix* site and from adjacent paddocks

2. Leeston Study site

* quantify growth and survival of plants under different planting, fertiliser and other experimental regimes

Hypotheses to be investigated

1. Tai Tapu Study site

* *Salix* spp. act as a nursery for native woody vegetation.
* Under favourable conditions, regeneration of native woody vegetation in *Salix* spp. will occur.
* Flooding has a negative influence on germination and seedling survival.

2. Leeston Study site

* Narrow-leaved species establish and survive more successfully than do broad-leaved species.
* Mid-late indigenous successional species can be used to bypass several natural successional stages, to form a self-sustaining community in isolated, exposed farm sites.
* Planting native woody species under existing tree cover enhances survival of broad-leaved species in comparison with similar species in an open paddock.
* Mulch enhances survival, and promotes growth.
* Planting native woody species in groups as opposed to dispersed individuals enhances establishment, promotes better long-term survival of plants, and causes faster canopy closure.

Scope of the study

This study was carried out over a three and a half year period on a part time basis, with fieldwork carried out between November 1994 and November 1997. Relevant measurements of adventive species (for example, numbers and heights of *Sambucus nigra* L.) were taken, but the quantitative data collection was mostly but not entirely, restricted to woody indigenous species. In this thesis, the rationale and methods of an initial approach to biotic community restoration in both a *Salix* spp. woodland and an intensively farmed location are presented. Both studies were carried out with minimal weeding or regular irrigation.

Structure of the thesis

This thesis is divided into five chapters. Following an introduction in Chapter 1, Chapter 2 describes the Low Plains Ecological District of Canterbury, where the study was based, and the two field areas where most of the field work was undertaken. Chapter 3 describes the Tai Tapu Study site methods, results and discussion, and Chapter 4, the Leeston Study site methods, results and discussion. Chapter 5 is a general discussion of restoration in the Canterbury context. Botanical nomenclature follows Poole & Adams (1963), Healy (1984), Webb *et al.* (1988), Beever (1991), Parsons *et al.* (1997); avian nomenclature follows Turbott (1990); and mammalian nomenclature follows King (1995).

Background

Overview of ecological restoration

The United Nations Conference on Environment and Development (UNCED) (The Rio de Janeiro "Earth Summit" 1992) highlighted the need for *in-situ* preservation and/or conservation of (especially) indigenous biodiversity. This is not, however, only of recent concern. A century ago, Walsh (1896) described the destruction of the New Zealand bush in a conservation context, and advocated restoration by natural re-establishment. The Convention on Biological Diversity (Anonymous 1994), as a result of the "Earth Summit" has outlined the challenges facing those concerned with the conservation of New Zealand's biodiversity. The nation's conservation effort can no longer be merely concerned with the legal protection of native species and the preservation of their remaining habitats. The active management of persistent threats, for example introduced mammals, and the restoration of natural communities and processes (ecological restoration) is now required (Resource Management Act 1991). Some of the most exciting prospects for conservation <u>and</u> (my emphasis) ecological restoration in New Zealand currently relate to the possible re-establishment of locally extinct plant or animal species on the mainland (Clout & Saunders 1995). This strategy will make significant contributions to the conservation of biodiversity in New Zealand in the 21st century.

The effects of human activity have been to homogenize and simplify landscape patterns by cultural land use. This has reduced tree species diversity and habitat diversity for wildlife (Urban *et al.* 1987). Wildlife conservation has the potential to be enhanced by linear features of remnant or planted vegetation within fragmented landscapes (Spellerberg & Gaywood

1993). The Canterbury landscape is a good example in this context as the indigenous vegetation is highly fragmented.

The initial focus of New Zealand's indigenous species extensive restoration efforts has concentrated on offshore islands (see Towns *et al.*1990). Increasingly, restoration on the mainland has become more relevant, especially in the urban or peri-urban environment. Much of the restoration research conducted in the past in New Zealand, however, has focused on degraded mine sites and their rehabilitation (for example, Mew & Ross 1989, 1992, Davis *et al.* 1995, 1997, Langer *et al.* 1997, Mew *et al.* 1997, Ross *et al.* 1995, Simcock & Ross 1995, 1996). More recently, restoration ecology that has focussed on conservation of biodiversity in agro-ecosystems has appeared in relevant literature. Presently, many restoration projects in New Zealand have been initiated by private landowners, "landcare" groups and regional, district and city councils. Contemporary work in Canterbury, for example that of Keesing & Wratten (1997, submitted) and Keesing *et al.* (1997) has generally concentrated on "community" restoration in an agro-ecosystem through enhancing abiotic and biotic resources. This is a long-term strategy that will eventually examine the invertebrate as well as botanical components of restoration.

The need to conserve and sustainably use agricultural biodiversity is recognised in the 1994 United Nations Convention on Biological Diversity. In the primary production arena, small cumulative improvements can also deliver progressive gains. If every landowner aimed to maximise diversity of plants and animals, enormous gains could be reaped (The Ministry for the Environment 1997). If remnant forest areas were left and fenced, wetlands protected or even recreated, riparian strips left free from use, steep erosion prone land retired, diverse tree species including natives planted, and more mixed crop and crop/animal combinations used, then production would be more economically and environmentally resilient and, over time, more sustainable (The Ministry for the Environment 1997). The less a farmer has to rely on importing agrichemicals to sustain farming, the better the result for the local and wider environment. Further, over time, such farming should cost less, too.

Meurk and Norton (for example, see 1988) have described the initiation of many restoration projects for conservation purposes, both in Canterbury and further afield. Other local projects are also under way, for example, the Lucas Associates' (1995) Agenda 21 project, Hinewai Reserve, Banks Peninsula (Wilson 1994), the Christchurch City Council's Avoca Valley project (R. Barker pers.comm.), Matawai Park Restoration Project (Franklin & Thompson 1983, Stewart & Woods 1997), the latter one of the oldest restoration projects in Canterbury (G. Stewart pers.comm.) and the Travis Swamp restoration project (Travis Wetland Trust 1992). On the Christchurch Port Hills, Voyce (1998) has looked at *Sambucus nigra* as a facilitator of succession in Hoon Hay Valley and Reay (1996) at the success of three restoration plantings at Kennedy's Bush. Restoration of indigenous plant species on Quail Island is now underway (C. Potter pers. comm).

Internationally, there are studies comparable with those described in Chapters 3 and 4, from Australia (for example, Crome *et al.* 1994) and the USA (for example, McClanahan & Wolfe 1993).

Restoration projects within New Zealand's regulatory environment

The current framework of environmental management in New Zealand is structured by the requirements of the Resource Management Act 1991 (the Act) (Milne, 1992). The legislation is underpinned by the principle (section 5) of sustainable management of the nation's physical and natural resources. Section 5 outlines the purpose of the Act in terms of the requirements of the Act. section 5(2) states that:

"sustainable management" means managing the use, development and protection of natural and physical resources...

The Act's requirement is that this is done in a way and, and at a rate that ensures social wellbeing. However, this must be undertaken whilst attending to intergenerational equity, safeguarding the life-supporting capacity of the environment and

Avoiding, remedying, or mitigating any adverse effects of activities on the environment (s.5(2)(c)).

Section 17 outlines the duty that every person has to avoid, remedy and mitigate any adverse effect on the environment. Effects are defined in section 3 of the Act and include temporary or permanent effects; past, present or future effects and cumulative effects.

There are activities, whether in terms of land use (s.9), the Coastal Marine Area (s.12) or in regard to activities in lakes and rivers (s.13) where restoration projects may be a legitimate means by which effects on the environment can be remedied or mitigated.

The Act does not define what protection of natural and physical resources entails (s.5(2)) (see quote above). The Conservation Act (1987) however, does define protection in terms of its requirements:

"Protection", in relation to a resource, means its maintenance, so far as is practicable, in its current state; but includes-

(a) Its restoration to some former state; and

(b) Its augmentation, enhancement, or expansion.

The emphasis in the Conservation Act is not resource use (as in the Act) but resource conservation. The direction in the Conservation Act is clearer in terms of the place of restoration projects in conservation. However, the restoration of the landscape (at many scales) has a place within the regulatory environment of New Zealand (e.g., restoration of mine sites, trade offs between land for development and reserves, restoration of wetlands because of unauthorised development).

Ecological principles

The ecological principles underlying and informing the restoration process have often not been taken into account (Hobbs & Norton 1996, Keesing & Wratten 1998). Failure of restoration projects as a result of this is costly. Understanding ecological processes which underpin restoration activity will generate site-specific knowledge, enabling effective management. The ultimate aim of ecological restoration is to restore community functions. This is through the restoration of biotic communities in which links between important species or groups of species approximate the former state of the system (Towns & Atkinson 1991). The challenge of ecological restoration, therefore, is to develop strategies and plans which allow management to be appropriately prioritized and scheduled. There is a spectrum of aims in ecological restoration (Hobbs & Norton 1996), but generally, the goal is to re-establish or improve the ecological status of lost and damaged indigenous plant and animal communities (Jordan *et al.* 1987). Whatever the aims of rehabilitation, problems arise in relation to genetic resources. To maintain the local genetic pool for self sustaining communities, there is a need for the genetic resources to make it possible. Therefore, wherever possible, genotypes from the appropriate provenance should be used (for example, see Heslop-Harrison & Lucas 1979, McCombs 1993, Keesing & Wratten 1997).

Plant succession

Early concepts of plant succession described seres (or different seral stages) as unidirectional vegetation changes over time (for example, see Bazzaz 1996 and Burrows 1990). However, seral succession is not necessarily a one-way sequence of vegetation change, but a much more complex process where each vegetation type may be invaded by one of a number of different communities (Chapman & Reiss 1992). The crucial question is whether the vegetation changes or seres progress to an end point or final community type. In the early stages of the concept, it was believed that succession progressed steadily from communities with small simple plants like mosses, through to a final stable community with large, dominant plants such as trees. This final community type was called the climax community, which was generally believed to contain a high degree of biodiversity and be self-maintaining. From a restorationist's perspective, it would be ideal to work on systems that are typified by predictable directional changes in structure during community development (Palmer *et al.*, 1997). In such a system, any attempt to restore an altered community could be viewed as an attempt to manipulate natural successional processes. Such manipulations might attempt to accelerate natural succession, so that the community develops along the same lines as it would in the absence of intervention, but the desired endpoint is reached sooner. Manipulations might also attempt to bypass some of the stages of natural succession, for example by establishing some late successional species in the initial plantings. In such cases, the goal is to accelerate the rate of natural succession so that the desired community is reached sooner than later.

However, the issue of whether biodiversity and ecosystem function are causally related has emerged recently as a highly topical issue in ecology. Recently, some evidence has been presented that suggests that indicators of ecosystem performance may in fact be governed by species composition, not by species diversity (Hooper & Vitousek 1997, 1998). There is also increasing scientific evidence (e.g., Young & Mitchell 1994) that some elements of biodiversity are not sustainable in the long term, because in many places, there are insufficient habitat/community fragments (in both number and size) left for any to be sustainable. New ways to restore viability that are supported by the ecological community need to be found, including establishing linkages between, and buffers around, key fragments both on and outside land reserved for conservation purposes (Department of Conservation, 1995).

Community ecological theory may play an important role in the development of a science of restoration ecology (Palmer *et al.* 1997). It has been suggested that research is needed on the relationship between species richness and community stability of restored sites and on functional redundancy among species in the regional colonist pools (Figure 1), i.e., the source of the seeds.

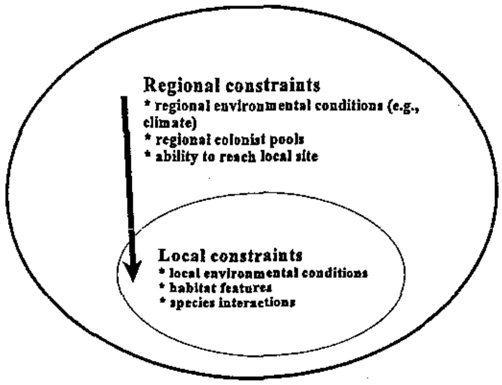


Figure 1: Successional restoration of a community depends on both regional and local factors. There must be an intact supply of colonists at the regional scale that can survive locally significant environmental conditions and reach (disperse to) local sites. For species to become locally established, there must be suitable conditions; favourable environmental and habitat features, including natural disturbance regimes such as flooding or fires. Finally, species interactions may preclude species establishment locally (from Palmer *et al.* 1997).

Efforts targeted at restoring ecosystem function must take into account the role of individual species, particularly if some play a disproportionate role in processing material, or are strong interactors within the system. If community development is highly predictable, it may be feasible to manipulate natural succession processes to accelerate restoration. These ideas were incorporated into this study, although the relationships between species richness and community stability, and species richness and functional redundancy among species were not addressed in a quantitative sense.

Understorey composition of native vegetation in exotic stands

There are few experimental or observational data on the effect of different planted tree species on plant understorey composition in New Zealand, although Norton (1989) reviewed indigenous plants in an exotic plantation (*Pinus radiata* D.Donn, *Pseudotsuga menziesii* (Mirbel) Franco, *Larix decidua* Miller, *Pinus ponderosa* P. Lawson et Lawson and *Eucalyptus* spp.) forest of the Canterbury Plains and Allen *et al.* (1995b) investigated indigenous species richness in three areas of different ages, in a *Pinus radiata* plantation on the central North Island Volcanic Plateau. Norton (1998) also reviewed indigenous biodiversity in New Zealand's plantation forests. However, species diversity under *Salix* spp. has not been documented in New Zealand.

It has been suggested that alongside the promotion and use of exotic plantations as tools of forest restoration, there is a need to obtain information on how different tree species may influence patterns of plant understorey colonization (Guariguata *et al.* 1995). This study documents the understorey colonisation of *Salix cinerea* L. in Canterbury. *S. cinerea* is not a plantation tree species; however, its presence along riparian margins in the agricultural landscape (as a flood control measure) suggests that it could be used as a restoration tool along waterways, for example.

To achieve the desired state of restoration, therefore, a knowledge of ecological processes in plant and animal communities is required. These processes will vary in both space and time in different terrestrial environments. To actually understand these processes is a much larger aim however.

Chapter Two
Study area

Central Canterbury region

The following descriptions are taken largely from McEwen (1987) in a summary of the Ecological Regions and Districts of New Zealand. The identification and description of these Regions and Districts are summarised in the Protected Natural Areas Programme (PNAP) set up in 1983. The aim of the PNAP was to identify and protect examples of the full range of the indigenous and landscape features of New Zealand to maintain the distinctive character and biological diversity which makes this country unique (Park & Kelly 1986). The Canterbury Plains Ecological Region is one of 85 Ecological Regions identified in this summary. It is divided into three Ecological Districts (Figure 2). Most of the Region is intensively farmed (sheep, cattle and crops) and some areas are planted in exotic forest. There are numerous small settlements in the region, plus the major urban centre of Christchurch. The study described in this thesis was conducted in the Low Plains Ecological District of Canterbury Plains Ecological Region.

Location and topography of the Low Plains Ecological District, Canterbury

The Low Plains Ecological District (code 56.02, McEwen 1987) is one of 268 Ecological Districts in New Zealand. It is a large area of a series of coalesced fans, north and south of Banks Peninsula, ranging from sea level to about 300m above sea level. Pleistocene glacial outwash gravels and Holocene alluvial deposits are typical of the area. Soils are shallow, stony and subject to drought on terraces and coalescing low angle fans. Poorly drained, gleyed, silty and clayey soils are typically found on lower parts of the fans. There are alluvial soils on river flats and low terraces, ranging from very well drained stony sands to well drained deep silty soils. On local scales, areas of more poorly drained alluvial and peaty soils occur where water tables are high.

Vegetation

Indigenous vegetation types and Yellow-Grey Earths soils reflect a dry climate (DSIR 1967). Pre-European vegetation was mainly lowland short tussock–land with some floodplain forest. Also, there were extensive areas of *Phormium tenax* J.R.& G.Forst, *Cyperus* spp. and *Cordyline australis* (G.Forst.) Endl. swampland on swamp deposits. Dry riparian *Sophora*-mixed hardwood woodland flanked the major rivers.

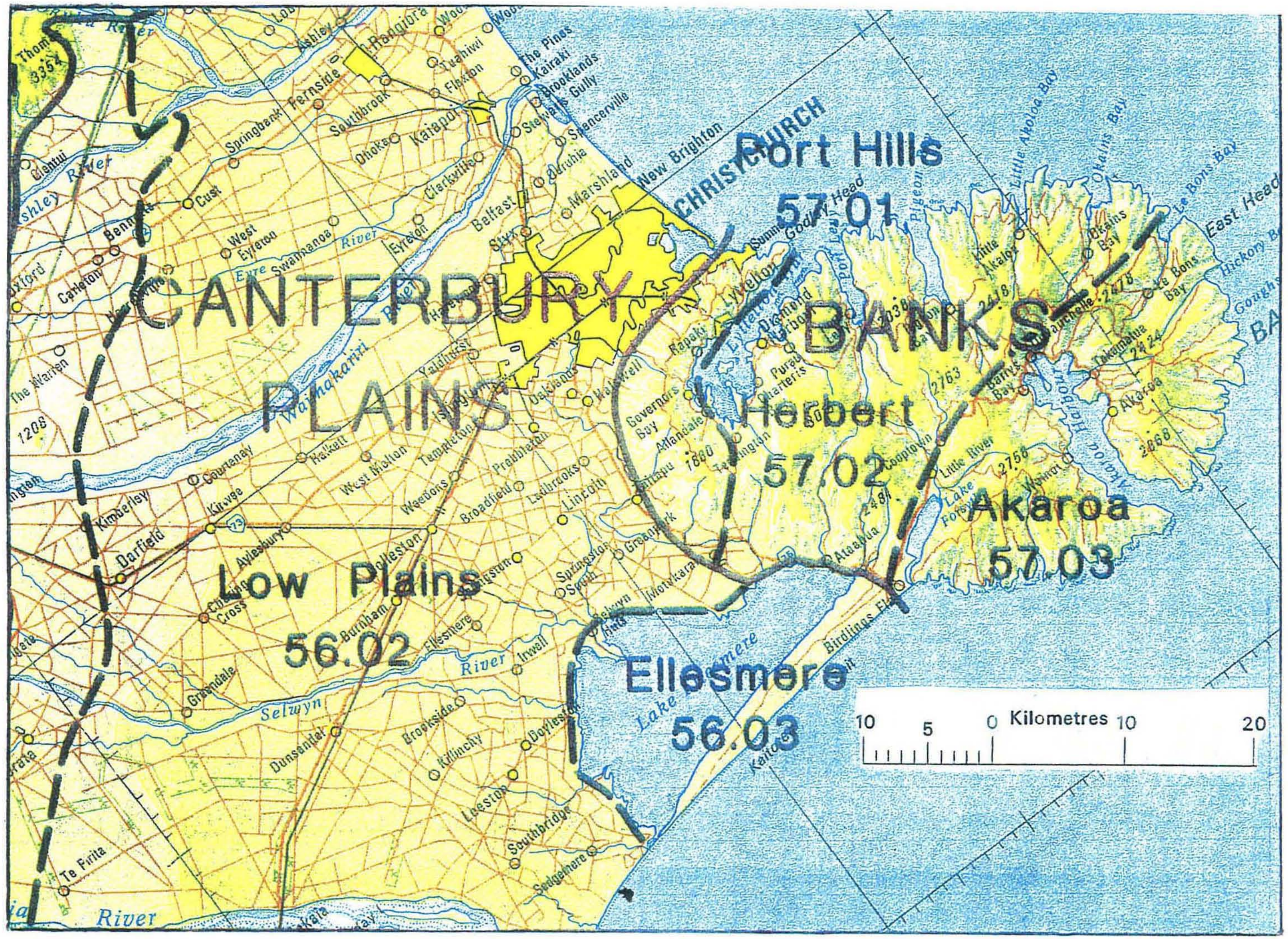


Figure 2: Map of Low Plains Ecological District, Canterbury

Climate of the Low Plains Ecological District

The region has cold winters and warm summers, with average monthly temperatures ranging from 6° C in July to 17° C in January, although temperatures can rise to above 32° in summer (Cherry 1997). Frosts are common in winter in spite of the proximity of the area to the sea (at Lincoln, an average of 90 days of ground frost per year occurs (Cherry 1997)). Wind incidence in the area is high, with south-westerly and north-easterly winds predominating (Norton 1985). In early spring in Canterbury, warm northwest (*föhn*) winds can be followed abruptly by cold, frosty weather. Rainfall for the region is low. Records (Norton 1985) show a mean annual rainfall of approximately 600mm, with little variation throughout the year. Storms causing flooding may occur in any season. In winter, occasional light snowfalls occur.

Study area (1): Indigenous plant community regeneration facilitated by exotic willow (*Salix* spp.) woodland in Tai Tapu

Tai Tapu Study site (Study site 1)

Location and topography

The Tai Tapu Study site (the main study site for this part of the investigation) is a small *Salix* woodland which acts as a native seed "sink" and nursery for regenerating native vegetation. It is a Canterbury Regional Council wetland reserve. It is situated immediately adjacent to the Christchurch - Akaroa highway (State Highway 75) between the Ahuriri Road and Motukarara (grid reference: NZMS 260 M36:748216) (Figure 3). It is on the western side of the road, approximately 1km south from the Ahuriri Road turnoff and is low-lying (< 20m above sea level) and flat. During this study the water table in some places in the Tai Tapu Study site was above ground only in winter and spring (personal observation).

Parent soils are Motukarara sandy and silty loams (DSIR 1967), with a high organic content. Results from mechanical analysis of the soil (0- 100mm depth) taken from the area underneath the *Salix*, but from two distinctly different sites, are presented in Table 1 (clay particles = < 2μm, silt particles = 2-20μm, sand particles = >20μm). This analysis was carried out by the author and Leanne Hassalls, Lincoln University, 1998.

Table 1: Mechanical analysis of soils from two different sites

Sample site	% clay	% silt	% sand
Site 1: (nearest the road, taken from an area very prone to inundation in winter/spring)	27	27.5	45.5
Site 2: (furthest from road from an area with high organic matter but not prone to inundation)	6	72	22

Experiments carried out at this site are described in Chapter 3.

Vegetation

A mature (c. 8-9m high), dense *S. cinerea* stand occupied about one third (approximately 2240m^2) of the area of the reserve. A few individual mature (9-10m) *Salix fragilis* L. occurred throughout the *S. cinerea*. To the south the *Salix* spp. were bounded by *Populus nigra* L cv. "Italica". Results from increment coring 10 of the largest (*S. fragilis* and *cinerea*) trees

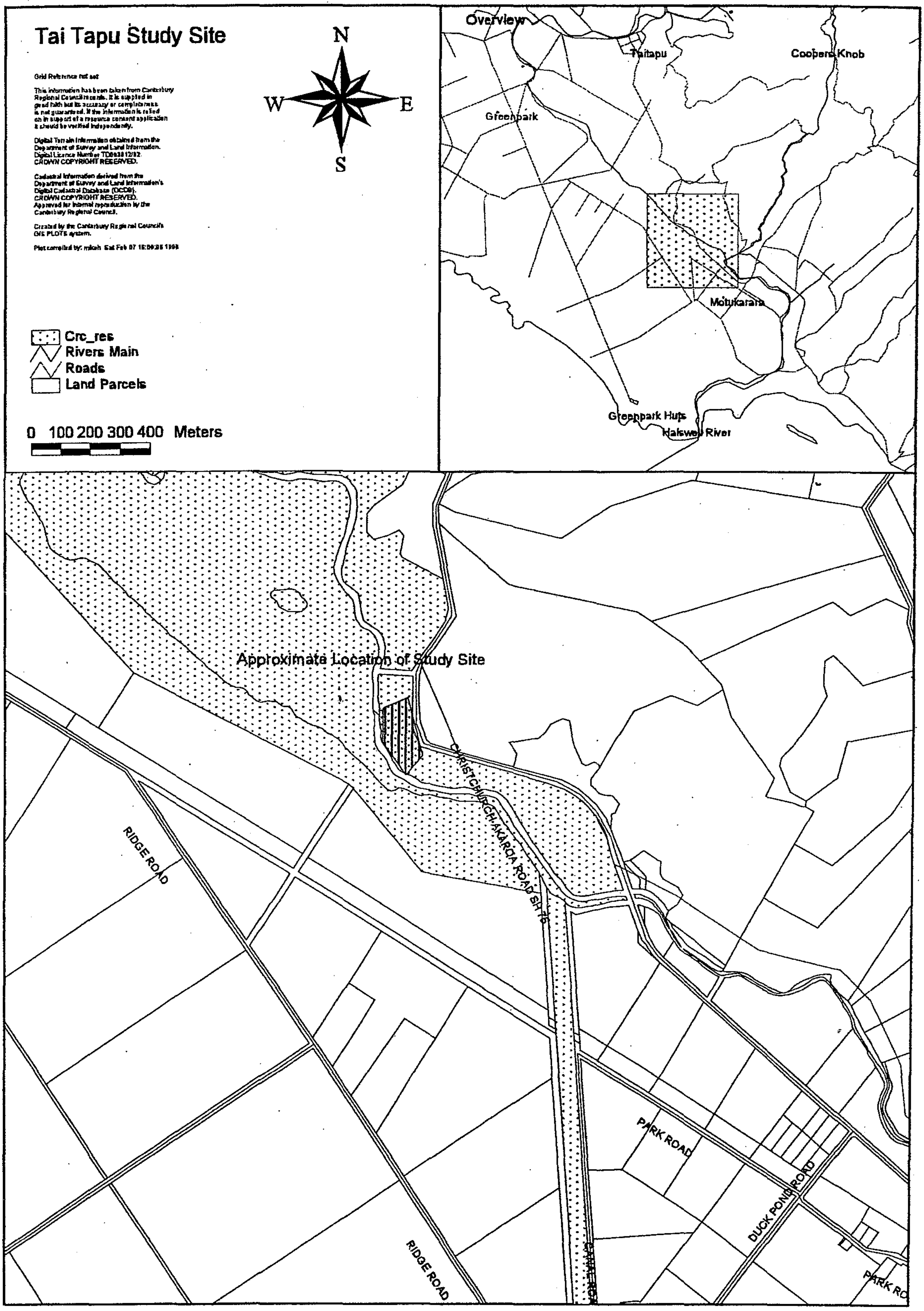


Figure 3. Location of Tai Tapu Study site (Source: Unpublished data, Canterbury Regional Council GIS PLOTS system)

indicated that the oldest were 35-40 years. This coring and analysis of the cores was carried out by the author and Phil Suisted, Landcare Research, 1997. Approximately 20 species of native woody (predominantly juvenile) plants occurred throughout the area, as well as several woody adventives (Appendix 1) and herbaceous species. *Sambucus nigra* (elder) and *Coprosma* spp. seedlings occurred at the highest mean density of all woody plants present throughout the site. This density was approximately four times that of the mature *S. cinerea*, although *Muehlenbeckia australis* (G.Forst.) Meissn. was prominent both as mature vines through the canopy of the *Salix* spp. and as seedlings.

Present status/use

Experimental work was conducted in this reserve which is part of the Canterbury Regional Council (CRC)-managed Ahuriri Reserves complex. It is one of several scattered reserves fringing an area which was once Ahuriri lagoon. It is a swamp in a triangular block of approximately 2ha. It has been leased to Langleydale Station in the past, but is now excluded from the lease, and was fenced by CRC in 1992/3 to exclude stock. The lagoon contains remnant lake edge (Te Waihora/ Lake Ellesmere) wetland vegetation and locally rare plant species (B. Dimbleby pers.comm.). It is also part of Ahuriri Lagoon, formerly a large wetland and very important food-gathering place (mahinga kai) (Evison 1994). The relevance of this will be discussed in Chapter 3.

Plate 1: Tai Tapu Study site in spring

Plate 2: Tai Tapu Study site in winter

Study area (2): Field experiment in open pasture and in a non-native woodland to investigate the effects of shelter, plant spacing, mulching and fertiliser on growth and survival of planted native species

Leeston farm Study site (Study Site 2)

Location and topography

Leeston is situated in the Ellesmere district of the Canterbury Plain. It lies in the rain shadow of the Southern Alps. Geologically speaking, the area is relatively young; the soils have formed as a result of the differential action of soil processes acting on parent materials under the influence of climate, vegetation and slope of the land (Knox 1969). The soils of the Ellesmere district are mostly derived from alluvial material but vary in depth, texture, moisture content and fertility (DSIR 1964). Since European occupation (c.1860s), almost all of the natural vegetation of the area has been cleared and the soils drained. Soils are mapped as Waterton clay loam, strongly gleyed phase (DSIR 1964). Pasture has been established and exotic tree shelter (e.g. *Cupressus macrocarpa* Gordon, *Populus* spp., *Salix* spp. *Pinus radiata* has replaced the indigenous flora (the wetland species *Phormium tenax, Typha orientalis* C.Presl, and *Carex secta* Boott in Hook. f., and *Cordyline australis, Kunzea ericoides* (A. Rich.) J. Thompson, *Discaria toumatou* Raoul, *Carmichaelia* spp., *Sophora microphylla* Ait., *Pteridium esculentum* (Forst.f.) Cockayne, *Cortaderia richardii* (Endl.) Zotov and *Poa* spp. (DSIR 1964). Leeston experiences blustery, strong, dry, warm northerly winds in summer and cold, rain-bearing southerly winds in winter (Norton 1985).

The experiments described below were carried out at a paddock site in soil of heterogenous composition. For example, some areas had been disturbed by the burial of rubbish (house and farm material), and by the presence of carcass pits. In other areas of the property, the soil was very stony. Non-native vertebrates, for example *Oryctolagus cuniculus* L. (European rabbit), *Lepus europaeus occidentalis* de Winton (European hare) and *Trichosurus vulpecula* Kerr (brushtail possum) are present in small numbers. Experiments carried out at this site are described in Chapter 4.

The property (Winfield) is situated on Lochhead's Road and is low-lying (elevation 15m above sea level) and flat; grid reference: NZMS 260 M36:562147 (Figure 4). The land is dominated by Temuka silt loam, gley soils (DSIR, 1967).

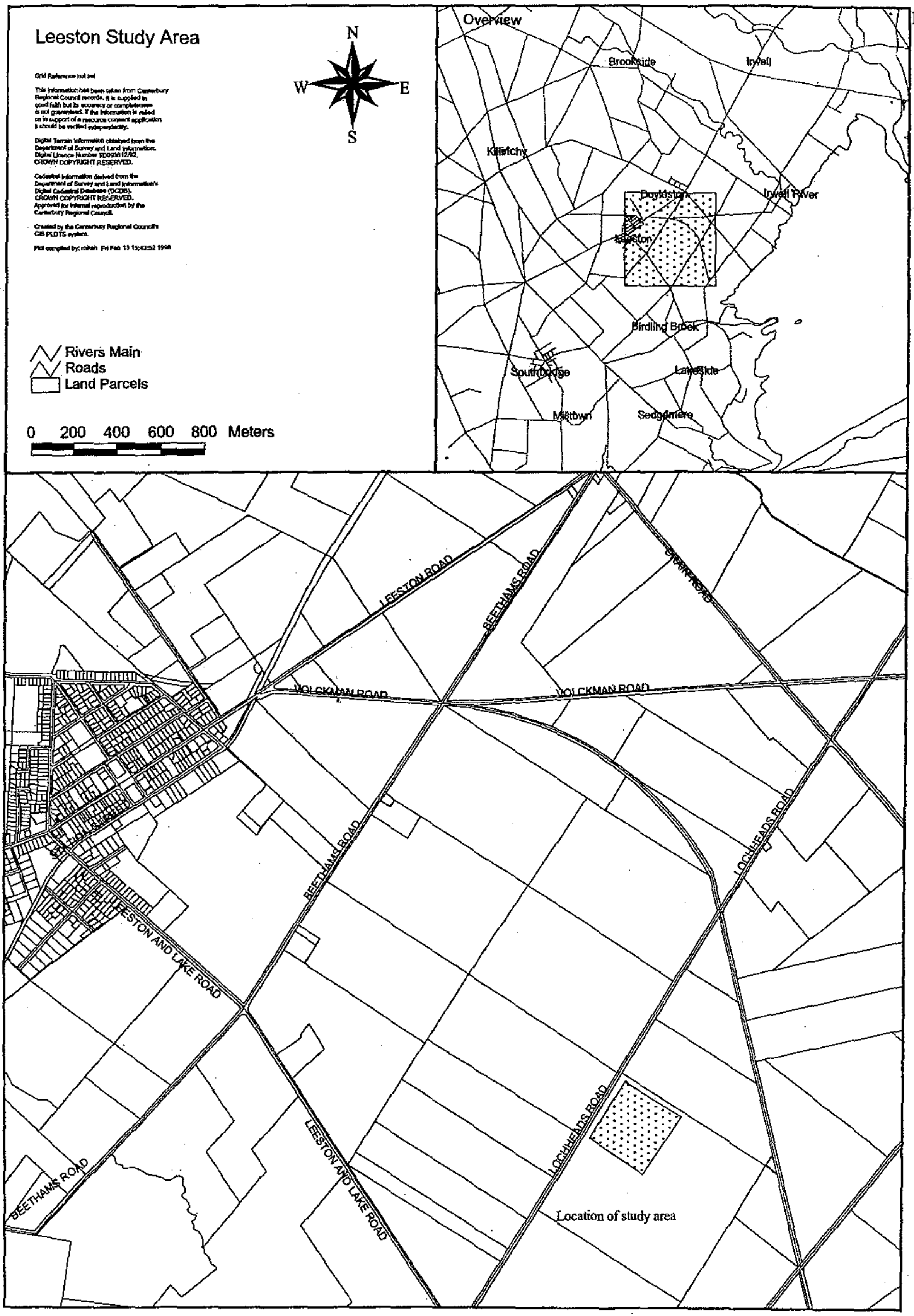


Figure 4. Location of the Leeston Study site (Source: Unpublished data, Canterbury Regional Council GIS PLOTS system)

Vegetation

At this study site there are predominantly shallow, stony soils with some patches of sandy to loamy deeper soils on the Selwyn floodplain (C. Meurk pers.comm.). These soils previously supported a diverse range of vegetation communities. They ranged from *Typha orientalis* and *Phormium tenax* swamp to tall podocarp (*Podocarpus totara,* D. Don in Lamb, *Prumnopitys taxifolia* (D.Don) Laubenf.) forests. In present times, *Lolium perenne* L., *Trifolium* spp., *Dactylis glomerata* L., *Plantago lanceolata* L. and *P. major* L. are the most abundant plant species present. *Cirsium* spp., *Rubus fruticosus* L. and *Ulex europaeus* L. are also present but are not abundant, except along hedgerows in the latter case.

European history

In pre-European times (c.150 years before present), the area was dominated by a shrubby landscape of *Kunzea ericoides, Coprosma propinqua* A. Cunn, *Cordyline australis, P. tenax* and *Poa* spp. (Meurk 1989). There is also evidence of taller podocarps (Meurk 1989), for example, *Prumnopitys taxifolia Dacrycarpus dacrydioides* (A. Rich.) Laubenf. and the hardwoods *Hoheria angustifolia* Raoul, *Griselinia littoralis* Raoul, *Plagianthus regius* (Poit.) Hochr and *Pittosporum* spp.

Present status/use

Experimental work was conducted on the site of an old homestead returned to pasture near Te Waihora (Lake Ellesmere), Leeston. The landowner donated an area of land (a paddock and a *Salix/Prunus* woodland area) and fenced it from stock for these experiments to be carried out.

Plate 3 : Intensive agricultural land at Leeston

Chapter Three

Tai Tapu Study site - natural regeneration of a forest community among *Salix* spp.

Introduction

Salix, Populus, Eucalyptus, pine and macrocarpa (*Pinus* and *Cupressus* spp.) are the most abundant tree species of the Canterbury region. *Salix* spp. tend to be dominant in riparian margins and in other damp areas, but not exclusively so. A few, scattered remnants of native vegetation also occur in the area, mainly on the nearby Port Hills. Some remnants are protected and managed by Regional or District Councils, others are in private ownership. *Salix* woodlands have often been described as a threat to native vegetation and as consisting only of "weeds"; for example, *S. cinerea* severely threatens the Whangamarino wetland (Department of Conservation 1994). The same species in other sites, however, could be unexpectedly useful as nurseries for bird-dispersed native tree seeds (Meurk, unpublished report). The trees act as habitat for birds by providing perching, roosting and nesting sites, and the woodland itself provides food. It is hypothesised that there is no native seed bank from former forest, at the Tai Tapu *Salix* swamp (Study site 1) and it is assumed (by the author) that birds disperse the native seeds into the area from Ahuriri Summit and AhuririValley Reserves. The evidence of which source, which bird species and at what dispersal/colonisation rate is inconclusive, but see Burrows (1994a). This dispersed seed then germinates under favourable conditions. Colonisation of any site is usually both deterministic and stochastic and it has been argued that many sites available for natural colonisation are of considerable ecological value (Bradshaw 1983). This is the case at this site, as native vegetation in Canterbury is severely fragmented and only a few small remnants remain. Objectives of this research included the investigation of the colonisation and successional processes occurring at this site.

Plant succession

In a broad sense, succession refers to any sequence in which the structure and floristic content of vegetation change through time, but the concept can be narrowed to mean sequences that have begun on bare surfaces (Bazzaz 1996, Burrows 1990,Wardle 1991). Plant succession can be defined as the non-seasonal, directional and continuous pattern of colonisation and extinction on a site by species populations (Begon *et al.* 1986). This general definition encompasses a range of successional sequences which occur over widely varying time-scales and often as a result of quite different underlying mechanisms. Just as the relative importance of species varies in space, so their patterns of abundance may change with time. In either case, a species will occur only where and when (i) it is capable of reaching the location, (ii) appropriate conditions and resources exist there, and (iii) competitors and predators do not exclude it.

Many studies have found that effective succession facilitators promote the establishment of seedlings by both attracting animal-dispersed seed, and providing a suitable environment (for example, Lee *et al* 1986, Wilson 1994, Green 1995). Voyce (1998) found that in Hoon Hay Valley, Canterbury, New Zealand, *Sambucus nigra* facilitates regeneration of native plant communities by increasing the dispersal of seed to these early successional communities and by providing a favourable environment for plant establishment at this site. Other examples of qualities that might determine the value of succession facilitators are the perceived weediness and the ability to fix nitrogen. *Ulex europaeus* (e.g. at Hinewai, Banks Peninsula, Wilson 1994), *Sambucus nigra* (in Hoon Hay Valley, Port Hills, Christchurch, Voyce 1998), *Cytisus scoparius* L. (Port Hills, Christchurch) and *Salix* spp. (on the Canterbury Plain) all have the potential to be succession facilitators, but could all be perceived as weeds.

Another way that plants can facilitate succession is by excluding plant species that inhibit the establishment of later-successional species. In this way, the facilitating species affects the environment indirectly through other species. Plants that facilitate succession often increase the concentration of many other mineral nutrients in the soil, by fixing nitrogen e.g. *Chamaecytisus palmensis* (Christ) Bisby & K. Nicholls (Brown 1996). *C. palmensis* also suppresses weed species with high light-intensity requirements (Brown 1996). The success of *Ulex europaeus* and *Cytisus scoparius* as facilitators of succession to forest could also be attributed to their suppression of competitive grasses (Lee *et al* .1986, Wilson 1994). *Salix* canopies as succession facilitators may shade out species with high light intensity requirements, for example, *Pteridium esculentum.* This results in more bare ground. The canopies must however, allow sufficient light through to native forest species, which are less light demanding than scrub vegetation. If environmental conditions strongly limit seedling establishment, then a small change in the environment caused by a different canopy species (e.g. *Sambucus nigra*) could greatly enhance the success of seedling establishment.

It was originally perceived that succession leads to a stable climax, in which diversity, complexity and quantities of carbon, nitrogen and other nutrient elements are at a maximum, with resident species maintaining themselves in constant proportions (e.g.,Clements 1904, 1916). Currently however, the view is that these factors may be at a maximum at earlier stages of succession not at climax (e.g., Prentice 1992, van Hulst 1992, Veblen 1992, Bazzaz 1996). According to Wardle (1991), no community has all the optimal attributes of the ideal climax, which is an asymptote never reached in nature, in other words the absence of climax is because the majority of ecosystems are inherently dynamic and stability is never achieved.

Different species, especially those of different seral stages, have different resource requirements and tolerances to environmental stress. Some sites may never be suitable for later-successional species to establish, but in many situations the environmental conditions are modified by early seral plant species to suit late colonisers. Modification may include changing the structure of the substrate/soil, accumulation of organic matter and soil nutrients (e.g. nitrogen fixation), protection from drought or exposure to wind and snow. A common method of facilitating the succession of vegetation is by increasing the available nitrogen in the soil by nitrogen fixation. *Chamaecytisus palmensis* does this (Brown 1996), although *Salix cinerea* does not.

A conceptual model has been developed (Meurk *et al.* 1997) to test the hypothesis that it is possible to ecologically restore sustainable systems of indigenous species and communities in lowland cultural landscapes in Canterbury. The forest succession model, LINKNZ (G. Hall pers.comm.), has been used as the basis for this work. LINKNZ is a New Zealand version of the LINKAGES model developed by Pastor & Post (1985). It generates autecological responses of plant species to both the environment and competition, depending on certain assumptions. The model of Meurk *et al.* adapts a forest gap model to potential restoration problems by determining what the "natural" or "potential" vegetation of the site was or should be. It uses intra-patch succession and inter-patch dispersal of woody plant propagules to determine that the Leeston area of lowland Canterbury was a totara-matai ecosystem and that it would become dominated by these podocarps once propagules arrive and trees reach maturity. This model does not take into account disturbance events, e.g. those caused by fire or people.

The remainder of this chapter describes quantitative measures of recruitment of individual native plants, and the growth and survival in a fenced area of mature *Salix* spp. To quantify growth, recruitment and mortality, permanent plots were set up and individual trees recorded. Soil seed bank experiments were also carried out to determine the components of the local seed bank. The benefit of investigating the seedbank is that it allows indirect examination of seedfall over a longer period of time. The water table level was monitored throughout the duration of the study to determine the effect of flooding on plant survival. In Chapter 5 of this thesis, plant survivorship, in the Tai Tapu *Salix* site is compared with that of the experimental planting at Leeston (Chapter 4). The Tai Tapu site differs from that of Leeston, however, because considerable native plant survival occurs under the exotic nursery without human intervention (except for the exclusion of cattle).

Methods

1. Vegetation survey and environmental inventory

Pilot study

To determine the appropriate sampling strategy, a pilot survey was carried out. Six points on the southern poplar border of the *Salix* area were selected using random number tables. From each of these points, a 10m transect was laid out in a south-north orientation. Ten pegged, permanent quadrats (0.5m x 0.5m) were spaced at 5m intervals along each transect (systematic sampling) to give a total of 60 plots. A matrix to define topography and surrounding bush density was developed to describe each of these plots. The combinations of environmental variables are presented in Figure 5.

1 Low-lying ground Surroundings - open	**2** Higher lying ground Surroundings - open
3 Low-lying ground Surroundings - dense	**4** Higher lying ground Surroundings - dense

Figure 5: Design of environmental inventory matrix

The ground level was objectively defined as low or high. A low- and high-lying category was assigned on a relative scale, usually in relation to the surrounding topography. Low areas were subject to frequent flooding in winter, the high areas generally were not, although one flood (August 1995) caused inundation of the whole site for 10 days. Plots which were low-lying were assigned an index of one or three, those that were not were assigned an index of two or four. The bush density surrounding each plot was measured to assess the probable accessibility of the plot to cattle. Prior to fencing, cattle roamed freely through the site but had restricted access to areas surrounded by dense vegetation (usually *Salix* spp. or *Sambucus*

Plate 4: Example of a quadrat

Plate 5: Example of a quadrat

nigra). Plots which were surrounded by dense vegetation (and thus were assumed to have had limited access by cattle) were assigned the index of three or four, depending on whether they were low-lying or not. Plots which were not surrounded by dense vegetation and therefore assumed to be open to cattle grazing were assigned the index of one or two. The matrix was designed to investigate whether certain species were present or absent in a given location of either low- or higher-lying ground or with open or dense surroundings, or whether the variable determined growth and/or survivorship. Flooding of quadrats was later measured independently, which probably gives a more accurate assessment of topography or relative elevation, or indication of drainage conditions of a particular site. Canopy cover of each plot was measured on a scale of 1-3, where 1 was allocated to an absent or sparse canopy, 2 described a moderate canopy, and 3 described dense canopy cover. These were measured to describe light availability and to assess bird perch availability. It was not a measure of perch preference (cf. McClanahan & Wolfe 1993). Measurements of topography (matrix), surrounding bush density (matrix), canopy cover (scale of 1-3) and the distance of the centre of the quadrat from the nearest tree (usually *Salix* spp.) were also recorded at each plot. These were measured to quantify the differences in establishment patterns between individuals growing near the base of the tree-trunk and those growing further away. Each quadrat was described by using the same topography/denseness matrix, canopy cover and "distance to nearest *Salix*" parameters (see Appendix 2 for data sheet).

Plant identification

To describe the species composition and size-structure of the vegetation and examine the floristic composition of the study site, each individual (native and non-native) woody species rooted in each quadrat was identified and its height measured. Stem diameter of individual plants was not measured as most of the individuals were seedlings. A representative sample of individuals of all species present throughout the site was tagged with pieces of coloured plastic-coated wire. Individual plant height was measured as the distance between the highest point of a live leaf to the ground. All individuals except the *Salix* spp. and the vines, *Muehlenbeckia australis* and *Hedera helix* L. were measured in this way. Height of *Salix* individuals was estimated and *M. australis* and *H. helix* were not measured, because it was not possible to measure vines that grew into the canopy. For a species list of woody plants occurring in the Tai Tapu *Salix* site, see Appendix 1.

Limitations of the pilot study

Data from the pilot survey indicated that either more sampling plots were required or the data set needed to be more comprehensive, because sample sizes were not large enough. It was decided to increase the plot size to 1m x 1m and also set up two further transects to make a total of 80 plots. Data presented in the results section are derived from only 64 of these. The other 16 were excluded from the final analysis because they were outside the *Salix* boundary (Figure 6).

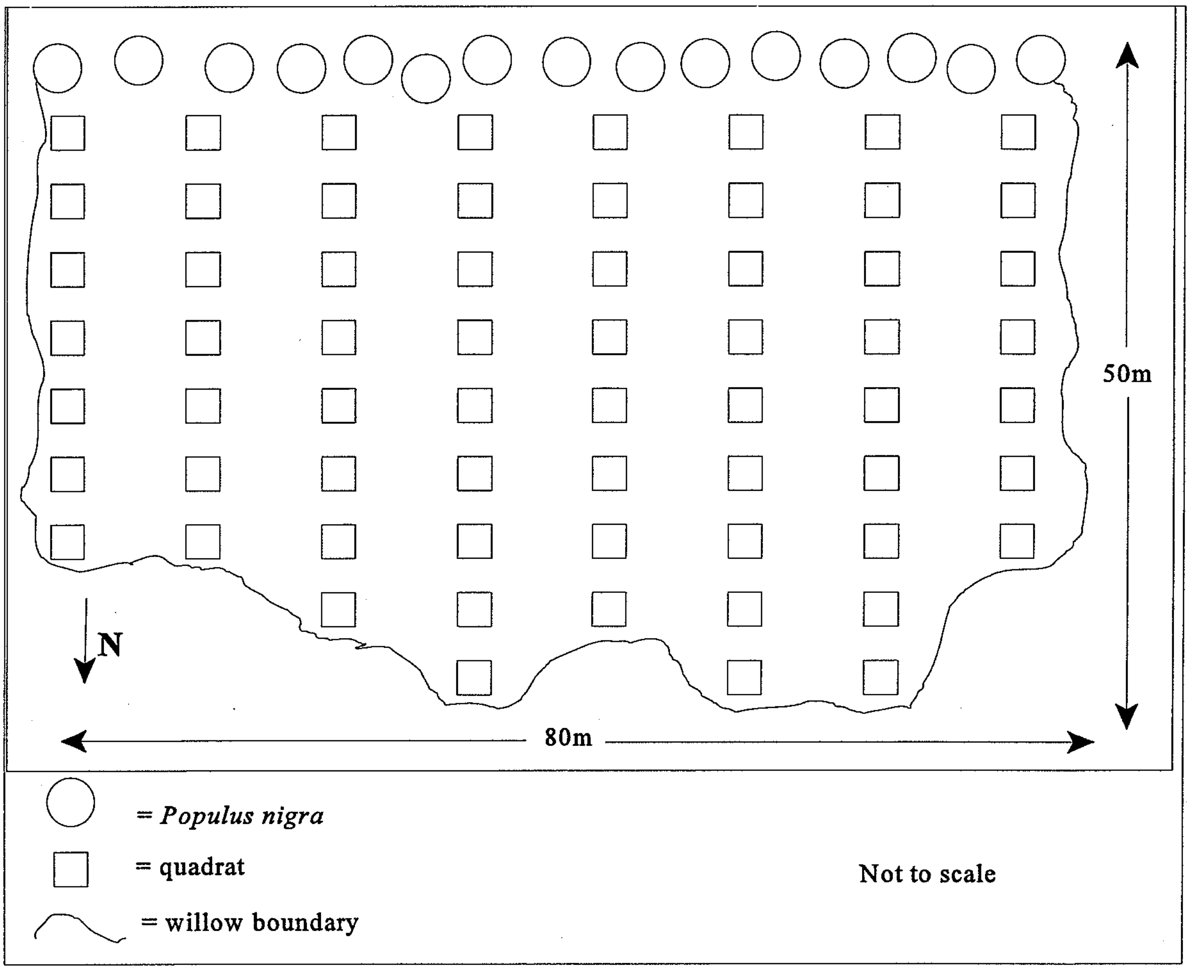


Figure 6. The experimental design of the Tai Tapu *Salix* site

Flooding effects

To determine the influence of periodic flooding on recruitment and mortality, each quadrat was assigned a value (once per month) using a scale of 1-4 to describe the proportion of the area inundated over the autumn - spring period (May - December) of 1995 and 1996. The water table was generally near ground level (+/- 20cm) and a score of one was allocated to a plot that was one quarter flooded and four allocated to a plot which was completely inundated. These values were then multiplied by the length of time (days) for which the sites were inundated, to give a flooding index for each quadrat. These time-weighted measures of figures of 1995/96 flooding data are presented in Appendix 5. These indices were then divided into four flood classes; 0-0.25, 0.26-0.5, 0.51-0.75 and 0.76-1.0, where the flood class 0-0.25 equates with least inundation and the class 0.76-1.0, the most. Recruitment and mortality of individuals for each of these classes was then calculated.

2. Soil seed banks

The seed bank of a site is made up of seeds of a variety of plant species dispersed from adjacent sites or dropped locally by the mature individuals present at the site. Seeds of many plant species, including those of late-successional trees, can be dispersed at any time into successional habitats depending on time of seed production and proximity of seed source. The initial stages of dispersal during recovery are aided by wind; later, other agents, for example birds become important dispersal agents (Bazzaz 1996).

To investigate the best method of sampling the soil seed bank, a pilot study was carried out in August 1996. Coring (as per McClanahan & Wolfe 1993) with a PVC pipe proved to be too difficult under the *Salix*, because of dense matted roots. A sharp spade and knife to obtain cubic samples proved effective instead. The chosen method for investigating differences between the seed bank of the *Salix* area and the adjacent paddocks is described below. In September 1996, 16 soil samples 0.25m x 0.25m x 0.1m ($6.25 \times 10^{-3} m^3$) were removed from selected sites and transferred to a glasshouse on the Landcare Research campus (Lincoln) to record germination of native seeds. In the *Salix* area itself, four samples were taken from areas prone to flooding and four taken from areas not prone to flooding. Each sample was dug from an area immediately adjacent to the northernmost edge of a permanent plot. Care was taken to ensure that the sites were similar in each case. Four samples were also taken from the adjacent paddock and four were taken 5m to the south of the poplar boundary. These sites were randomly chosen and were assumed to be representative of the area. The first eight

Plate 6: Example of a quadrat

Plate 7: Seedtrap beside a quadrat

samples were therefore taken from an area with a potential bird perch above them; the second eight were taken from areas without. The heated glasshouse at Lincoln, where the samples were assessed after their removal from the study site, had external glass and internal plastic (Agphane) walls. There was shading on the external walls and on the roof. Temperature ranged from 15° to 30°C. Each sample was placed in a tray, and the 16 trays were then housed in a plastic cell (approximately 4m x 4m x 4m) within the glasshouse. This cell kept the specimens isolated from potential pathogens or seeds from other plants in the glasshouse. The trays were watered every two days. As some samples came from paddocks, it was possible that slugs were present. To remove them, so that seedlings were not consumed before their identification, a molluscicide was applied. Each tray received 15 pellets of Watkins Slug Slam™, active ingredient 30g/kg metaldehyde in the form of a pellet, as soon as the soil was transferred to the glasshouse. Weekly observations of the trays were made for one year (until September 1997) and each individual which germinated or was present in the tray was documented. Grasses and other weeds were removed every two weeks to allow light into the tray. Following germination, any woody native species were recorded and removed.

3. Seed trapping

A seed trap experiment was set up to monitor the bird-dispersed seed arrival, presumably from the nearby seed sources of Ahuriri Summit and Valley Reserves. Burrows (1994b) designed and constructed light, durable seed traps with easily exchangeable catch pots. As the present experiment was carried out for one year only, it was decided to keep the equipment simple and cheap. Rectangular plastic containers were used as seed traps. Sixteen of these seed traps each with a trapping area of 36cm x 36cm ($0.13 \times 10^{-3}m^2$) were secured to the ground, placed immediately next to quadrats in a random fashion but secured in places that positively indicated a potential bird perch above. This was to maximise the chances of trapping seeds in bird faeces. From June 1995 - December 1996, traps were emptied monthly and trap contents analysed to record the timing of seed arrival. In the laboratory, leaves and other debris were washed away from the sample. The remaining litter was dried in a Westinghouse Motor Sentinel drying oven for 24-36 hours at approximately 35° C. Seeds of native woody plants were sorted and identified under a microscope.

4. Mist netting

By year-round observation, presence and absence of bird species was recorded for three years (Appendix 6). Incidental observations were made during each visit to the site. Absolute abundance data were not collected. In the second year (1996) for one day per week between January and June, when ripe native fruit was locally available, mistnetting at Tai Tapu was undertaken. Mist nets were erected and the droppings of any birds caught in the nets were collected. This was done to identify which bird species were dispersing which seeds. *Turdus merula* Linnaeus (blackbird), *Passer domesticus* Linnaeus (sparrow), *Turdus philomelos clarkei* Hartert (song thrush) but mainly finches, were caught and faeces analysed.

5. Methodology for analysis

Data from these experiments were largely descriptive and analysed by parametric (ANOVA and Fishers Least Significant Difference test) methods. Species diversity of the native vegetation (using the Shannon-Wiener (1949) index) present in the *Salix* swamp was determined, using

$$H = \sum_{i=1}^{S} (p_i)(\log_2 p_i)$$

where H = Index of spp. diversity

S = No. of species

p_i = proportion of total sample belonging to the ith species.

The Shannon-Wiener measure of diversity should be used only on random samples drawn from a large community in which the total number of species is known (Krebs 1989). The Shannon-Wiener measure H increases with the number of species in the community.

Results

1. Vegetation survey and environmental inventory

80 (1m x 1m) plots were present in the study site, although data presented here are derived from only 64 of these. The other 16 were excluded from the final analysis because they were outside the *Salix* boundary. Therefore, each of the 64 plots was potentially below a bird perching site. Collectively, they represented approximately 3% of the area under observation. For the *Coprosma* species, results and analysis presented here have been derived from pooled data. This is of practical relevance, as many *Coprosma* individuals in the site were hybrids and difficult positively to identify to species level. Data presented in this section comprise tagged and non-tagged individuals occurring in plots, but not the tagged individuals outside a plot.

In each of 1995, 1996 and 1997, wholly-native vegetation occurred in just two of the 64 plots. Wholly-exotic vegetation occurred in 12 plots in 1995, 10 plots in 1996 and 12 plots in 1997. A mixture (native and non-native) occurred in 35 plots in 1995, in 32 in 1996 and 38 in 1997.

The floristic composition of the site is presented in Table 2. Out of five exotic and ten native plant species, a total of 315 woody plants occurred within all plots in 1995, 313 in 1996 and 335 in 1997. This is approximately five plants/m^2 in all years (mean density for the three years is 0.08 (+/- 0.01) exotic species/m^2) and 0.15 +/- 0.03 native species/m^2).

In 1995 and 1996, *Coprosma* spp. occurred at the highest mean density throughout the site of between 1.4 and 2.0 individuals/m^2. In 1997, *Sambucus nigra* was most common, at 1.89 individuals/m^2.

Table 2: Number (n) and mean density of woody species (all growth stages) present in the Tai Tapu *Salix* site.

Species	n (1995)	n (1996)	n (1997)	Individuals/m² 1995 n/64	Individuals/m² 1996 n/64	Individuals/m² 1997 n/64
Exotic species						
Rosa rubiginosa	14	16	8	0.22	0.25	0.13
Lycium ferocissimum	1	1	1	0.02	0.02	0.02
Sambucus nigra	85	61	121	1.33	0.95	1.89
Salix cinerea	17	14	13	0.27	0.22	0.20
Hedera helix	1	2	12	0.02	0.04	0.19
Native species						
Coprosma spp.	92	129	95	1.44	2.02	1.48
Cordyline australis	37	27	36	0.58	0.42	0.56
Pseudopanax arboreus	3	5	4	0.05	0.08	0.06
Griselinia littoralis	10	10	9	0.16	0.16	0.14
Pittosporum eugenioides	1	1	1	0.02	0.02	0.02
Dacrycarpus dacrydioides	10	10	6	0.16	0.16	0.09
Prumnopitys taxifolia	1	1	1	0.02	0.02	0.02
Myoporum laetum	0	1	0	0	0.02	0
Podocarpus totara	4	3	3	0.06	0.05	0.05
Melicytus ramiflorus	39	32	25	0.61	0.50	0.39
TOTAL	**315**	**313**	**335**			

Species diversity

Species diversity was calculated using the Shannon-Wiener (1949) function. Species diversity (H) was 1.91 in 1995, 1.86 in 1996 and 1.81 in 1997.

Mean height in quadrats

Mean heights of most species for 1995, 1996 and 1997 are presented in Table 3. *Salix cinerea* and *Hedera helix* are omitted from this summary because absolute measurements of these species were not taken. Mean height is calculated from measurements of all individuals of each species present in a quadrat. Therefore height change from year to year would be influenced by recruitment of new smaller plants, and by mortality.

Table 3: Mean height of exotic and native seedlings in 1995, 1996 & 1997.

Species	Mean height (cm) 1995 ± S.D.	Mean height (cm) 1996 ± S.D.	Mean height (cm) 1997 ± S.D.
Exotic species			
Rosa rubiginosa	33.4 ± 27.60	36.1 ± 23.71	49.3 ± 30.37
Lycium ferocissimum (n=1)	96.0	103.0	81.0
Sambucus nigra	46.7 ± 47.50	67.3 ± 55.42	49.5 ± 56.40
Native species			
Coprosma spp.	21.6 ± 17.27	20.7 ± 19.0	31.1 ± 26.94
Cordyline australis	63.5 ± 27.43	74.1 ± 19.32	62.8 ± 30.40
Pseudopanax arboreus	11.7 ± 2.87	16.4 ± 10.00	25.5 ± 6.10
Griselinia littoralis	18.4 ± 12.21	23.2 ± 17.95	27.4 ± 21.55
Pittosporum eugenioides (n=1)	29.0	40.0	51.0
Dacrycarpus dacrydioides	9.5 ± 2.91	11.3 ± 4.05	15.5 ± 3.91
Prumnopitys taxifolia (n=1)	27.0	29.0	32.0
Myoporum laetum	-	8.0 ± 0	-
Podocarpus totara	7.8 ± 1.92	13.7 ± 4.03	15.3 ± 5.25
Melicytus ramiflorus	27.2 ± 11.66	32.6 ± 12.41	32.6 ± 12.60

Mean height for all species from 1995 to 1996 increased, except for *Coprosma* spp., where a decline in mean height (-0.99cm) occurred. The greatest change in mean height for an exotic species occurred in *Sambucus nigra* (20.68cm) and, for a native species, in *Cordyline australis* (10.51cm). Mean height from 1996 to 1997 increased for all species except *Rosa rubiginosa* L. (-8.6cm), *Lycium ferocissimum* Miers (-22cm) and *C. australis* (-11.21cm). The greatest change in mean height for an exotic species occurred in *R. rubiginosa* (13.12cm) and for a native species in *Pittosporum eugenioides* A. Cunn. (11.0cm), although this species had a sample size of only one.

Height class distribution

Analyses of height class distribution of *Coprosma* spp., *C. australis* and *Melicytus ramiflorus* J.R. & G. Forst are presented below (see Figures 7,8 and 9). These indigenous species were chosen because they occurred at the highest density in the site in all three years. Most *Coprosma* spp. (1995, 1996 and 1997) were <20cm tall. Most *C. australis* (1995, 1996 and 1997) were 60-100cm tall. Most *M. ramiflorus* (1995, 1996 and 1997) were 21-40cm tall.

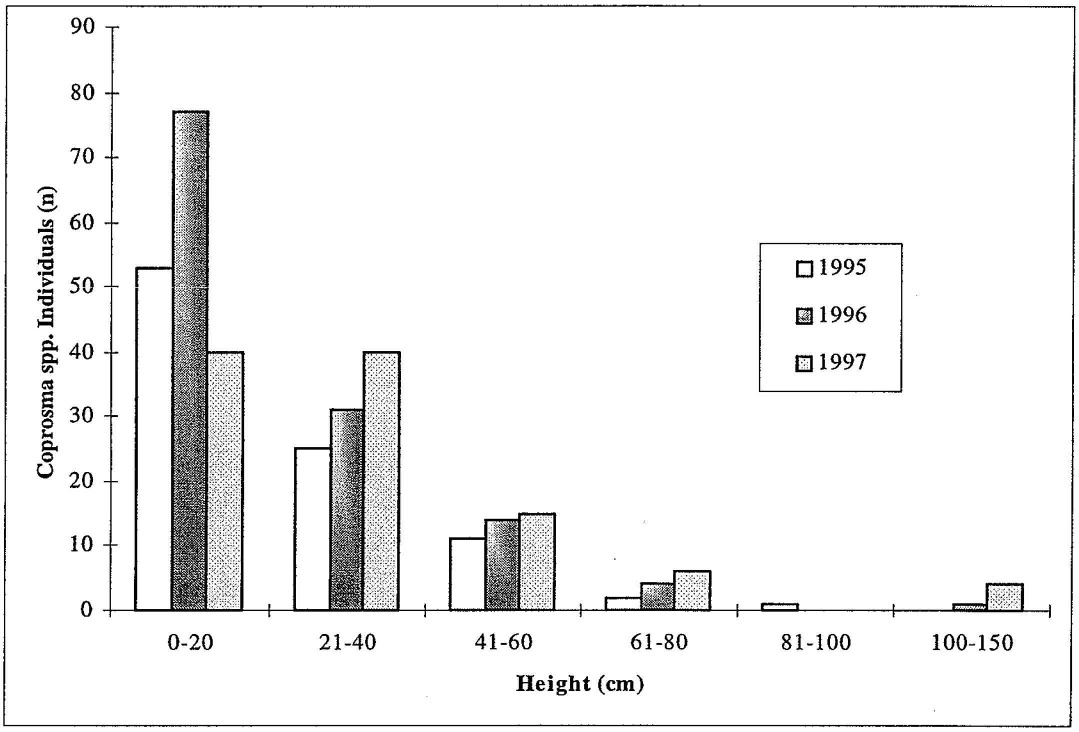


Figure 7: Height class distribution of *Coprosma* spp.

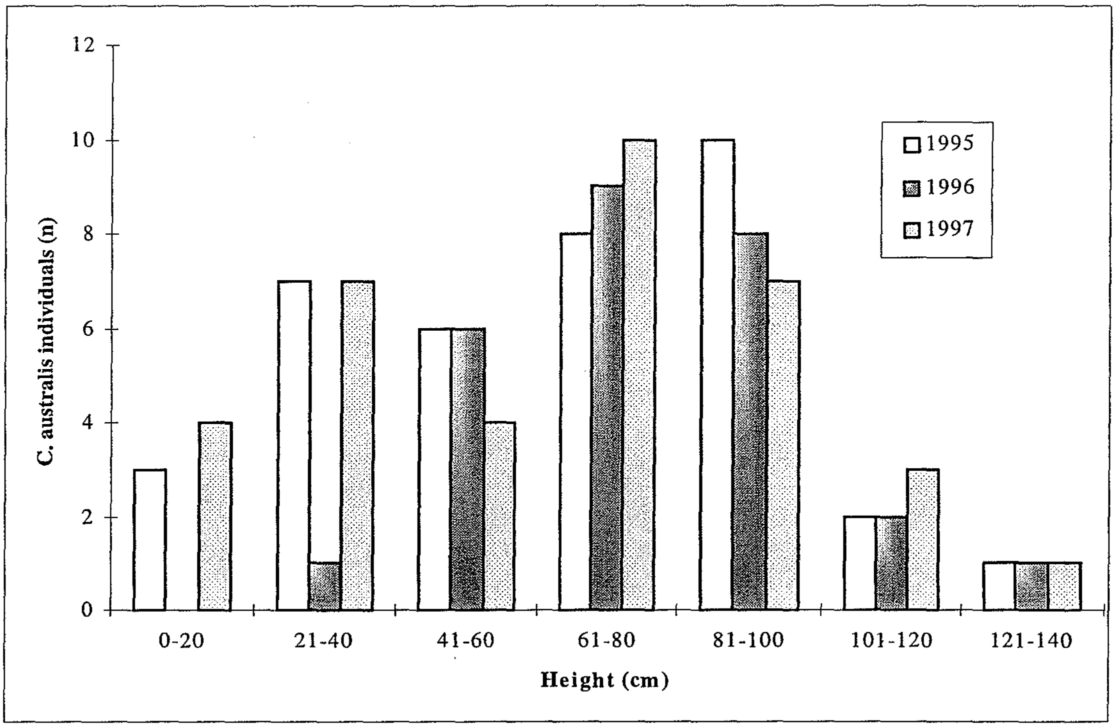


Figure 8: Height class distribution of *Cordyline australis*

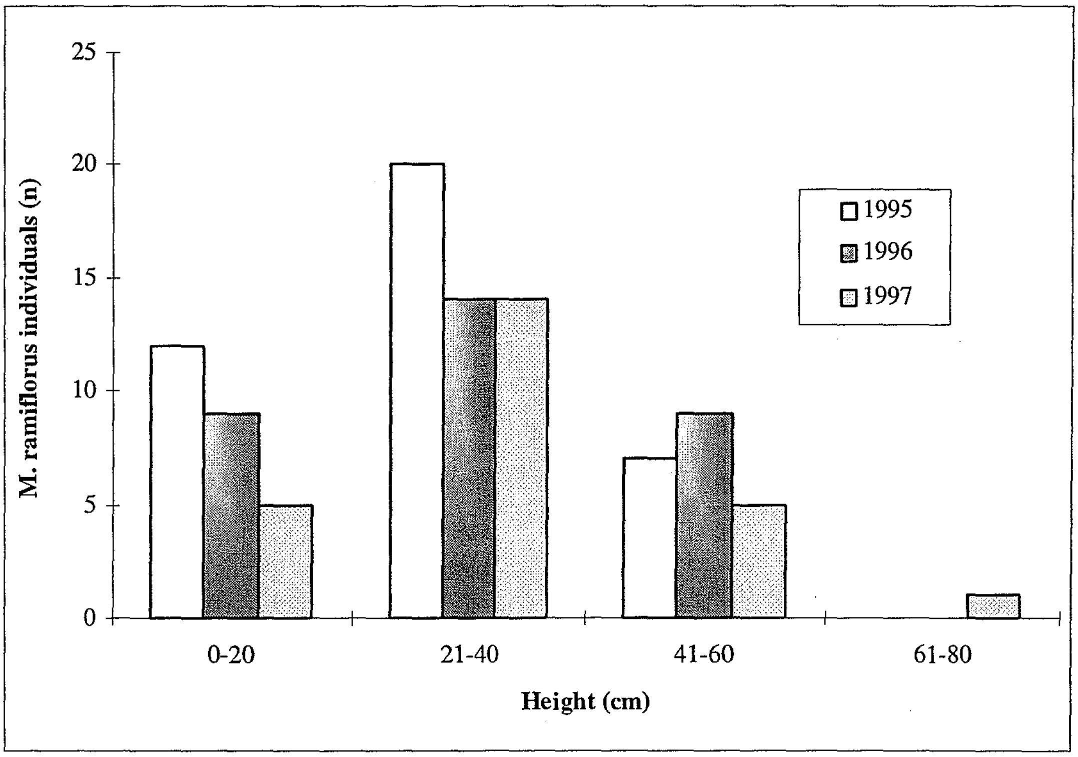


Figure 9: Height class distribution of *Melicytus ramiflorus*

Height growth data are taken from established individuals (tagged and positively identifiable) that changed in height from one year to the next. They do not include newly recruited individuals or those that were unidentifiable from one year to the next.

Table 4: Mean plant height growth from 1995 to 1996 and from 1996 to 1997

Species	Mean growth (cm/year) 1995/96 ± S.D.	Mean growth (cm/year) 1996/97 ± S.D.
Exotic species (n)		
Rosa rubiginosa (5)	10.9 ± 17.56	8.6 ± 7.71
Lycium ferocissimum (1)	7.0	-22.0
Sambucus nigra (40)	15.2 ± 14.69	13.4 ± 24.08
Native species (n)		
Coprosma spp. (54)	5.2 ± 3.08	9.6 ± 15.76
Cordyline australis (16)	0.8 ± 3.08	-1.2 ± 15.74
Pseudopanax arboreus (3)	2.0 ±	9.5 ± 3.5
Griselinia littoralis (9)	4.1 ± 7.06	2.9 ± 5.13
Pittosporum eugenioides (1)	11.0 ±	11.0 ±
Dacrycarpus dacrydioides (6)	2.7 ± 2.62	3.0 ± 0.71
Prumnopitys taxifolia (1)	2.0 ± 0.0	3.0 ± 0.0
Podocarpus totara (3)	5.0 ± 2.83	2.5 ± 1.5
Melicytus ramiflorus (17)	11.0 ± 4.55	6.6 ± 7.77

From 1995 to 1996, mean height growth for all species was positive. Maximum mean growth in an exotic species occurred in *S. nigra* (15.2cm/annum). Maximum mean growth in a native species occurred in both *P. eugenioides* and *M. ramiflorus* (11.0cm/annum). From 1996 to 1997, mean growth for all species was positive, except for *L. ferocissimum* (-22.0cm/annum) and *C. australis* (-1.15cm/annum). Maximum mean growth in an exotic species occurred in *S. nigra* (13.36cm/annum). Maximum growth in a native species occurred in *P. eugenioides* (11.0cm/annum), but there was only one individual of this species.

Figure 10 presents the trend in growth predicted from 1995 to 1996 data and the actual growth (from 1996 to 1997) of the above 12 species.

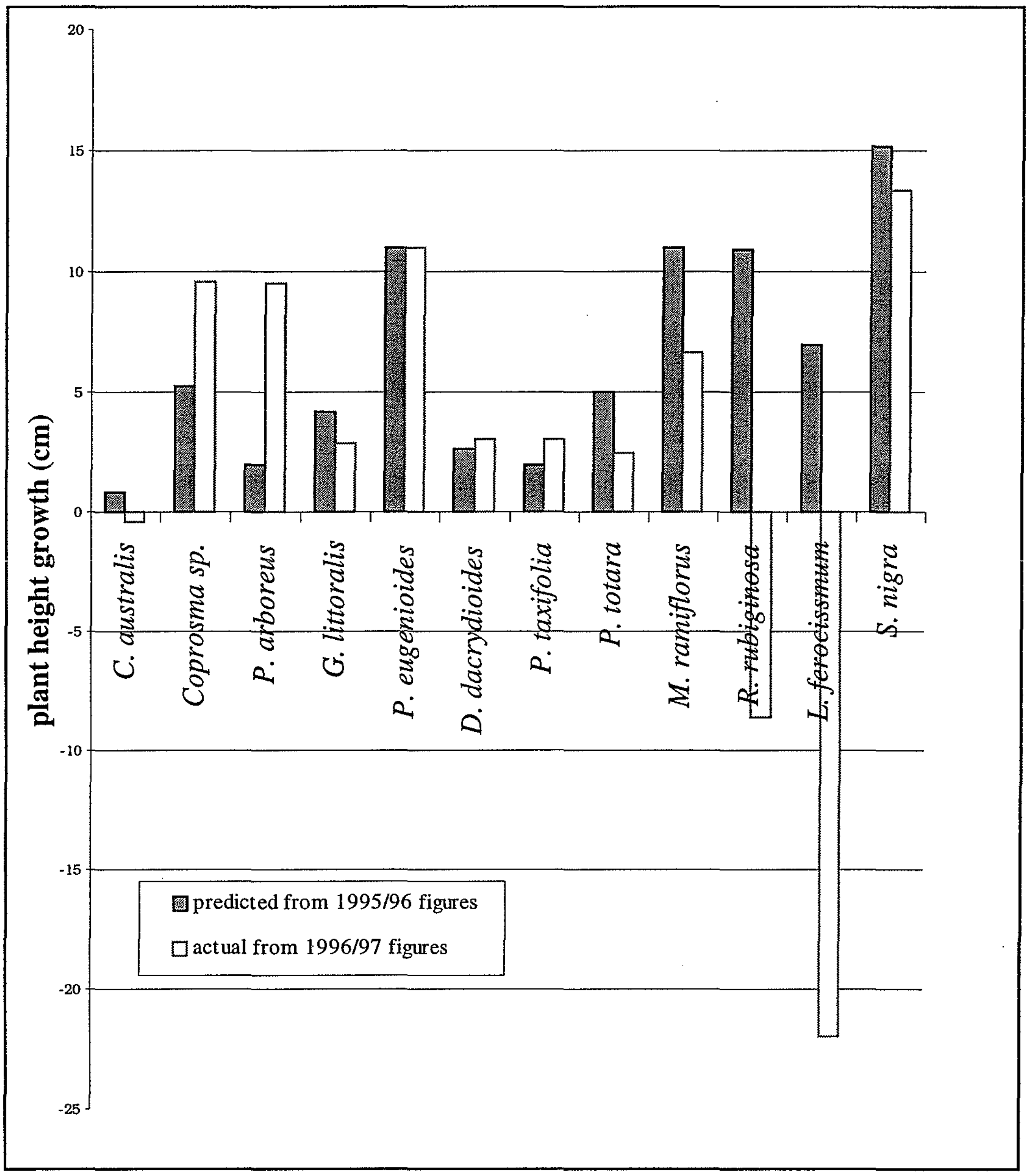


Figure 10: Predicted and actual height growth of all species (1996 to 1997).

Environmental variation

Summarised below is the description of the *Salix* understorey species in relation to the environmental variables assigned to each quadrat at the outset of the study (see page 25 for scoring system). Table 5 presents the number and proportion of each pair of environmental conditions in the total number of quadrats (64). The density of surroundings as a variable has been omitted from the final analysis, because it was decided that it was unnecessary, i.e., did not add anything to the analysis.

Table 5: Number and proportion of plots in different environmental categories

Value	Matrix	Number of plots with this matrix value	Proportion of total number of quadrats (64)
1	Low ground, open surroundings	21	32.80%
2	High ground, open surroundings	19	29.70%
3	Low ground, dense surroundings	4	6.25%
4	High ground, dense surroundings	20	31.25%

Most ground (61%) was categorised as being unlikely to flood (matrix values 2 and 4), the remainder (matrix values 1 and 3) was low-lying and subject to frequent or extended flooding especially during winter (Table 11, see flooding section).

In 1995, 1996 and 1997, most established individual plants (83%,84% and 87% respectively of total established plants) occurred on high ground, and fewest on low-lying ground, although the small sample size of four (for low-lying ground with dense surrounding vegetation) would have been a bias.

Plant densities have been calculated separately for each environmental type (Table 6). In 1995, *S. nigra* occurred at the highest density of 1.24 individuals/m^2 in low-lying ground with open surroundings. *Coprosma* spp. occurred at the highest density of 1.26 individuals/m^2 in higher ground with open surroundings. *Coprosma* spp. also occurred at the highest density of 2.25 individuals/m^2 in low-lying ground with dense surroundings, again at the highest density of 2.70 individuals/m^2 in higher ground with dense surroundings. In 1996, this trend was repeated.

Table 6: Density of individuals (per m²) in each environmental category (matrix values 1-4), 1995, 1996 and 1997

	1995 (matrix value below)				1996 (matrix value below)				1997 (matrix value below)			
	1	2	3	4	1	2	3	4	1	2	3	4
Species												
Exotic species												
R. rubiginosa		0.21	0.25	0.45		0.32	0.75	0.35		0.16	0.25	0.20
L. ferocissimum			0.25				0.25				0.25	
S. nigra	1.24	0.74	0.50	2.15	0.62	0.74	0.50	1.60	1.00	1.95		3.15
S. cinerea	0.14	0.26	0.50	0.35	0.14	0.21		0.35	0.10	0.21		0.35
H. helix		0.05				0.05		0.05		0.05	0.25	0.50
Native species												
Coprosma spp.	0.24	1.26	2.25	2.70	0.48	1.32	3.25	4.05	0.29	0.89	1.75	3.25
C. australis	0.05	0.32		1.50	0.05	0.11		1.20	0.05	0.32		1.45
P. arboreus		0.05	0.25	0.05			0.50	0.15		0.05	0.25	0.10
G. littoralis	0.05	0.11		0.35	0.05	0.11		0.35	0.05	0.11		0.30
P. eugenioides				0.05				0.05				0.05
D. dacrydioides		0.16		0.35		0.16		0.35		0.11		0.20
P. taxifolia		0.05				0.05				0.05		
M. laetum						0.05						
P. totara				0.20				0.15				0.15
M. ramiflorus		0.79		1.20		0.53		1.10		0.47		0.80
Total individuals	**36**	**76**	**16**	**187**	**28**	**69**	**21**	**195**	**31**	**83**	**11**	**210**

In 1997, however, *S. nigra* still occurred at the highest density of 1.00 individuals/m² in low-lying ground with open surroundings but also occurred at the highest density of 1.95 individuals/m² in higher-lying ground with open surroundings. *Coprosma* spp. occurred at the highest density of 1.75 individuals/m² in low-lying ground with dense surroundings. Again, *Coprosma* spp. occurred at the highest density of 3.25 individuals/m² in higher ground with dense surroundings, but *S. nigra* occurred at 3.15 individuals/m².

Recruitment

Recruitment is defined here as the germination and survival of individuals from one year to the next. It does not relate to recruitment of an individual from one height class to the next. Net, rather than absolute recruitment is presented, because of the difficulty in correctly identifying one individual plant from year to year. Therefore, absolute recruitment can be masked by high mortality. In 1996, five species had been recruited into plots ($64m^2$) in the *Salix* swamp. Three of these were native. Some recruited species (*Coprosma* spp., *R. rubiginosa, Pseudopanax arboreus* (Murray) Philipson) had already been present in plots in 1995, others (*H. helix, Myoporum laetum* G. Forst) were absent. Among the native species recruited, *Coprosma* spp. were recruited in the highest numbers (37 individuals, i.e. 40% of the baseline figure, see Table 7), mostly into high-lying ground with open surroundings. This represents net recruitment, as some *Coprosma* spp. may have died but at least 37 individuals were recruited to give the net recruitment of 37 individuals in 1996. *P. arboreus* was recruited at 66% of the baseline figure (three were present in 1995 and two were recruited in 1996 to give a total of five). A single *M. laetum* was recruited onto high ground with open surroundings, which was the first *M. laetum* individual recorded at the site. A single *H. helix* was recruited onto high ground with dense surroundings, although other individuals were present in the site, but did not occur in a plot. Most recruitment occurred in high ground with dense surroundings (all species pooled) and least in low-lying ground with open surroundings.

Table 7: Per cent recruitment of individuals in 1996

Species	baseline n (1995)	n recruited	% recruitment
Rosa rubiginosa	14	4	28.6
Coprosma. spp	92	37	40.2
Pseudopanax arboreus	3	2	66.7
Hedera helix	1	1	100.0
Myoporum laetum	0	1	100.0

In 1997, three woody species had been recruited into plots. Only one of these was native. Each of them had been present in plots in 1995 and 1996. Nine individuals of *C. australis* were recruited, mostly into high-lying ground with dense surrounding vegetation (Table 8). Ten individuals of *H. helix* were recruited, again mostly into high-lying ground with dense surroundings. Sixty *S. nigra* individuals were recruited into both high- and low-lying ground.

Table 8: Per cent recruitment of individuals in 1997

Species	baseline n (1996)	n recruited	% recruitment
Cordyline australis	27	9	33.3
Hedera helix	2	10	500.0
Sambucus nigra	61	60	98.4

Most recruitment occurred in high ground with dense surroundings (all species pooled) and least (zero) in low-lying ground with open surroundings.

Mortality

Mortality is defined here as the non-survival of individuals from one year to the next. All data presented here are as net, rather than absolute mortality. This is because of the difficulty in correctly identifying one individual plant from one year to the next. Thus, absolute mortality may be masked. Of the native species present in 1995, seven *M. ramiflorus* (18% of the 1995 total), 10 *C. australis* (27% of total) and one *P. totara* (25% of total) had died by 1996 (Table 9). Many exotic species also died including two *R. rubiginosa* (14% of total), three *S. cinerea* (18% of total) and 24 *S. nigra* (28% of total). *S. nigra* survival was the lowest overall, expressed as a proportion of individuals dying per m^2 per annum and as a proportion of the number originally present. Most mortality occurred on high ground with open surroundings (all species pooled), with the lowest on high ground with dense surroundings.

Table 9: Per cent mortality of individuals in 1996

Species	baseline n (1995)	n mortality	% mortality
Rosa rubuginosa	14	2	14.3
Sambucus nigra	85	24	28.3
Salix cinerea	17	3	17.6
Cordyline australis	37	10	27.0
Podocarpus totara	4	1	25.0
Melicytus ramiflorus	39	7	17.9

Of the native species present in 1996, 34 *Coprosma* spp. (26% of the 1996 total), one *P. arboreus* (20% of the total), one *G. littoralis* (10% of the total), one *C. australis* (4% of the total), four *D. dacrydioides* (40% of the total), one *M. laetum* (100% of the total) and seven

M. ramiflorus (22% of the total) had died by 1997 (Table 10). Of the exotic species, eight *R. rubiginosa* (50% of the total) and one *S. cinerea* (7% of the total) had died. The single *M. laetum* seedling died in 1996. *D. dacrydioides* survival was also low (40% mortality occurred). Most overall species mortality occurred on high-lying ground with dense surroundings and least on low-lying ground with open surroundings.

Table 10: Per cent mortality of individuals in 1997

Species	baseline n (1996)	n mortality	% mortality
Rosa rubuginosa	16	8	50.0
Salix cinerea	14	1	7.1
Coprosma spp.	129	34	26.4
Cordyline australis	27	1	3.70
Pseudopanax arboreus	5	1	20.0
Griselinia littoralis	10	1	10.0
Dacrycarpus dacrydioides	10	4	40.0
Myoporum laetum	1	1	100.0
Melicytus ramiflorus	32	7	21.9

Flooding data

Plots were allocated the time-weighted flooding index if they had been inundated for any length of time over the two-year period. Out of the 64 plots, 29 were allocated the flooding index and 35 plots were not. Some of these plots were without any woody vegetation. The following descriptions have been used in the text to describe the community structure of plant species; native-only: only native vegetation is concerned; exotic-only: only exotic vegetation is concerned; mixed vegetation; both native and exotic species are concerned.

Flooding: 1996 results

In 1996, woody vegetation occurred in 13 of the 29 flooded plots (Table 11). Of these, native-only vegetation occurred in one, exotic-only occurred in four and a mixture of native and exotic vegetation occurred in eight. In 1996, plant mortality occurred in a total of eight plots. Native-only mortality occurred in two, exotic-only in four and a mixture of the two occurred in two plots. Recruitment of individuals occurred in just four plots, with native-only in two, exotic-only in one and a mixture also in one.

No flooding: 1996 results

In 1996, woody vegetation occurred in 23 of the 35 unflooded plots. Of these, native-only vegetation occurred in one, exotic-only occurred in seven and a mixture of native and exotic vegetation occurred in 15. In 1996, plant mortality occurred in a total of 15 plots. Native-only mortality occurred in five, exotic-only in six and a mixture of native and exotic occurred in four. Recruitment of individuals occurred in a total of 15 plots. Native-only recruitment occurred in four, exotic-only in seven and a mixture of native and exotic also in four.

Table 11: Pooled 1995/96 flooding data with 1996 and 1997 vegetation data

	Number of plots 1996	Number of plots 1997
Flooded plots with native-only vegetation	1	2
Unflooded plots with native-only vegetation	1	0
Flooded plots with native-only recruitment	2	2
Unflooded plots with native-only recruitment	4	3
Flooded plots with native-only mortality	2	1
Unflooded plots with native-only mortality	5	7

	Number of plots 1996	Number of plots 1997
Flooded plots with exotic-only vegetation	4	6
Unflooded plots with exotic-only vegetation	7	7
Flooded plots with exotic-only recruitment	1	6
Unflooded plots with exotic-only recruitment	7	10
Flooded plots with exotic-only mortality	4	1
Unflooded plots with exotic-only mortality	6	4

	Number of plots 1996	Number of plots 1997
Flooded plots with mixed vegetation	8	6
Unflooded plots with exotic-only vegetation	7	7
Flooded plots with mixed recruitment	1	0
Unflooded plots with mixed recruitment	4	3
Flooded plots with mixed mortality	2	3
Unflooded plots with mixed mortality	4	5

Flooding: 1997 results

In 1997, woody vegetation occurred in 14 of the 29 flooded plots (Table 11). Of these, native-only vegetation occurred in two, exotic-only occurred in six and a mixture of native and exotic vegetation occurred in six also. In 1997, plant mortality occurred in five plots, one native-only, one exotic-only and three a mixture of native and exotic. Recruitment of individuals occurred in eight plots with native-only recruitment in two, exotic-only in six and there was zero recruitment in "mixed".

No flooding: 1997 results

In 1997, woody vegetation occurred in 24 of the 35 unflooded plots. Of these, plots with native-only vegetation did not occur, exotic-only occurred in seven and a mixture of native and exotic vegetation occurred in 17. In 1997, plant mortality occurred in 16 plots, with seven native-only, four exotic-only and five a mixture of native and exotic. Recruitment of individuals occurred in 16 plots with native-only in three, exotic-only in 10 and a mixture of native and exotic also in three.

Flooded plots were originally allocated a flood class and data are presented in Table 12 with the 1996 mortality and recruitment of individuals associated with plots in each flood class.

Table 12: 1996 mortality and recruitment for each flood class.

Flood class	mortality	recruitment
0-0.25	*Sambucus nigra* x 1	
	Sambucus nigra x 4	*C.*spp. x 1
		Cordyline australis x 1
	Sambucus nigra x 1	
	*C.*spp. x 1	
		*C.*spp. x 1
	*C.*spp. x 1	
	Sambucus nigra x 1	
	*C.*spp. x 1	
	*C.*spp. x 4	
	Sambucus nigra x 1	
	Sambucus nigra x 16	
0.26-0.5		*C.*spp. x 2
		Sambucus nigra x 1
0.51-0.75		*C.*spp. x 3

In 1996, 22 individuals died in plots allocated a floodclass. Nine were recruited into plots with a floodclass. However more individuals overall, died in plots without a floodclass than those with. More individuals were recruited into plots without a floodclass than into those with. Most mortality (77%) occurred in *S. nigra*, and most recruitment (78%) occurred in *Coprosma.* spp.

The four flood classes of 1997 and the mortality and recruitment of individuals associated with plots in each flood class are presented in Table 13.

Table 13: 1997 mortality and recruitment for each flood class.

	Mortality	Recruitment
0-0.25		*Sambucus nigra* x 6
		Sambucus nigra x 1
		Sambucus nigra x 2
	Pseudopanax arboreus x 1	*C.* spp. x 1
	Melicytus ramiflorus x 1	
		C. spp. x 1
		Sambucus nigra x 2
	C. spp. x 6	*Hedera helix* x 1
	Sambucus nigra x 2	
	Rosa rubiginosa x 2	
	Pseudopanax arboreus x 1	
0.26-0.5	*Sambucus nigra* x 1	
	C. spp. x 3	
0.51-0.75	*C.* spp. x 1	
	Salix cinerea x 1	
		C. spp x 1

In 1997, 17 individuals died and 15 were recruited, in plots allocated a flood class. However more individuals overall, died in plots without a floodclass than those with. More individuals recruited into plots without a floodclass than into those with. Most mortality (53%) in flooded plots occurred in *Coprosma.* spp., and most recruitment (73%) occurred in *S. nigra*.

A basic ANOVA was carried out on the flooding index data which showed that between the four matrices (environmental variation), there was a significant difference ($p < 0.001$) between matrices in response to flooding. Subsequent analysis (Fisher's Least Significant Difference test) showed that matrix 1 was significantly different from the other three matrices. This is probably due to its relatively low elevation.

Distance to *Salix*

An ANOVA was carried out on this data which showed that between the four matrices, there was no significant difference ($p = 0.17$) between the distances to the nearest *Salix* tree. In this case, therefore, it appears that the mature *Salix* spp. has no value as a nursery for existing plants, so it can be assumed that moisture, topography or flooding are the major controlling factors on plant growth and survival. This accords with Yin *et al.* (1980) although the *Salix* spp. provides the shelter and the initial perching site (see next section) from which birds defecate seed.

2. Soil seed banks

All plants which germinated (native and non-native) from the soil samples taken are listed in Appendix 3. A total of 35 species, mainly adventative weeds and grasses, germinated. For the purpose of this research, however, these species were of limited interest. From all of the soil samples taken, only *M. laetum* (one individual or the equivalent of 10/m^3), *M. australis* (33 individuals or 330/m^3), *Coprosma* spp. (eight individuals or 80/m^3) and *C. australis* (nine individuals or 90/m^3) germinated and survived among the native species to be identified over an eight-month period (August 1996 - April 1997). Each of these germinated from samples taken from below the *Salix* trees. The origin of these seeds is presented in Table 14. No germination of any native woody species occurred from soil samples taken from either the poplar boundary fence line or the open paddock.

Table 14: Glasshouse germination of native woody species in soils collected from the study site

Germinating native species	Number of plants germinating from wet sites under *Salix*	Number of plants germinating from dry sites under *Salix*
Muehlenbeckia. australis	21	11
Cordyline australis	6	3
Myoporum laetum	-	1
Coprosma. spp	-	8

3. Seed trapping

Seedtraps were left uncovered during the course of the experiment, therefore seed predation by insects, birds and rats was not excluded. Thus the seed trap data described here refer to the positive identification of seed after predation losses. Results of this experiment were pooled because the seed sample sizes were small and no difference was found between traps at different locations. (Seeds were identified with the help of Dr Colin Webb.)

Table 15: Seedfall in 1995/96 (no. seeds collected)

	Jan-Mar	Apr-Jun	Jul-Sep	Oct-Dec
1995	-	-	*M. australis* (2)	*M. australis* (19)
1996	*M. australis* (24)	*M. australis* (2)	*M. australis* (8)	*M. australis* (23)
	C. spp. (5)		*C.* spp. (2)	*S. nigra* (3)

4. Mist netting

No whole seeds of native woody plants were identifiable from any samples collected.

Discussion

In New Zealand, exotic species such as *U. europaeus*, *S. nigra* and *P. radiata* have been studied for their contribution to native regeneration, but *Salix* spp. have received no quantitative analysis. The presence of *U. europaeus* has been considered desirable to re-establish native forest on Banks Peninsula (Wilson 1994) and *S. nigra* enhances the re-establishment native forest in the Port Hills region of Canterbury (Williams 1983, Voyce 1998). Allen *et al.* (1995a), Clout & Gaze (1984) and Ogden *et al* (1997) have described native biodiversity in *P. radiata* plantations.

Management of exotic woody species e.g., *Salix* spp. and *S. nigra* depends on an understanding of their ecology, especially their role in the enhancement of succession of native vegetation, whenever the aim of vegetation management is to encourage the return to native forest (Meurk *et al.* 1997). Under the *Salix* spp. at Tai Tapu, 22 bird-dispersed native species occur. The assumption is that the birds use the *Salix* trees to perch in and disperse the seed into the area from nearby sources (Ahuriri Summit and Ahuriri Valley Reserve). It affords shelter from the north west and southerly winds and winter frosts, and as the area is now fenced, protection from grazing stock encourages survival of the native seedlings. In this instance *Salix* spp. appear to provide a good nursery area for native woody vegetation, although data collected in the experiment suggested otherwise.

1. Vegetation survey and environmental inventory

Environmental variability in both space and time has been shown to promote species diversity in plant communities (Grubb 1977, Huston 1994). Similarly, at the population level, theoretical analyses have shown that environmental heterogeneity contributes to the maintenance of genetic variability in plant populations (Bazzaz 1996). In this study, a basic analysis of environmental variability was undertaken alongside a vegetation survey over a three year period. The pH across the site ranged from 6.5 - 7.2; calculated from 5 soil samples from each the four environmental categories (no significant difference between the sites).

Preliminary observations in the Tai Tapu Study site suggested that the seedlings of most species generally had a clumped distribution. These are probably related to preferred perching sites of birds, although the hypothesis was not tested. In any event such places often corresponded to high (unflooded) microsites and were near a tree trunk. The result of early succession being facilitated this way, is a heterogenous landscape with *S. cinerea* acting as a successional facilitator. This pattern has been termed nucleation after the phenomenon in the physical sciences (Yarranton & Morrison 1974). Different patches may have different levels of light, moisture, nutrients, herbivore and pathogen loads. All these may promote variability in populations, but in this study they were not measured. The amount of light reaching the forest floor was not quantitatively measured, but was estimated to be fairly constant throughout the site including taking account of the deciduous nature of *S. cinerea*. Except for some possible differences in reflectivity, there is no *a priori* reason to suspect that most plots are inherently different from one another, in the amount of radiation received at ground level. Light levels could have been measured to quantify those differences. Any differences among habitats, therefore, are largely the product of the nature of the vegetation itself, the topography of the microenvironment, and how light quantity and quality change as radiation penetrates the canopy. As the *Salix* canopy senesces due to old age, opportunities for the native understorey to form the new canopy will be created.

The importance of both the number of individuals and their spatial distribution has long been recognised in community ecology. However, research in successional communities has emphasised biomass as the measure of change, as there are many cases where biomass and density of a species can have very different patterns (Bazzaz 1996). In this study, the biomass option was not considered an appropriate measure and data collection was focussed on diversity and abundance.

Mean height of a species is determined by the recruitment and mortality within the species, in as much that taller older plants could die and new recruitments are in early stages of growth. Individually tagged plants however can demonstrate absolute growth which is not skewed by measurements of mortality and recruitment of smaller plants. Insufficient individuals were tagged at the outset of the study to generate survivorship curves of recognisable cohorts. The most widely used way of depicting patterns of mortality are survivorship curves (Begon *et al.* 1986).

Plate 8: Regeneration under the *Salix* spp.

Plate 9: Regeneration under the *Salix* spp.

It is notable that all native species present in the study site have seeds that are bird dispersed. Wind dispersed species such as *P. regius* and *Leptospermum scoparium* J.R. et G. Forst. are absent. An absence of mature plants (of bird-dispersed species) producing seeds in the site, offers strong support to the hypothesis that birds are responsible for the dispersal.

In practice, for biological communities, the Shannon Wiener index H does not seem to exceed 5 (Krebs 1989). The results from this study conform to this and even though species diversity decreases slightly over the course of the study, it gives a useful basic index for comparison with other reserves in the area.

Ecology of native species

Data on plant ecology in Canterbury which have been used in this thesis largely come from the National Indigenous Vegetation Survey (NIVS) database held at Landcare Research, Lincoln (Forest Research Institute 1989). Temporal data such as those acquired from the NVS database are useful for comparisons with short-term datasets as there is little relevant literature available. Data from NIVS is compared with data from Tai Tapu for *G. littoralis* and *Dacrycarpus dacrydioides*. Analysis of these (NIVS) data by Hall (1994) shows that mean growth of *G. littoralis* at Mt. Oxford (eight years' data of same size classes as Tai Tapu) is 0.28cm/annum. This is considerably less than at Tai Tapu (mean growth of *G. littoralis* is 4.14 and 2.88cm/annum for 1995/6 and 1996/7 respectively). Differences in environmental conditions between the two sites are probably responsible, although higher levels of mammalian herbivory at Mt. Oxford cannot be excluded. The seed coat of *G. littoralis* could also have an effect on potential dispersal. Thin-coated *G. littoralis* may be more vulnerable to bird digestive processes which may explain the moderate-only density in Tai Tapu compared with, for example, that of *M. ramiflorus.*

Mean growth of *D. dacrydioides* at Mt. Thomas (eight years' data of same size classes as Tai Tapu) is 0.11cm/annum (NIVS database). This is considerably less than at Tai Tapu (2.66 and 3.0cm/annum for 1995/6 and 1996/7 respectively). Differences in environmental conditions (e.g., precipitation, temperature regime) are probably responsible, although mammalian herbivory at Mt. Thomas cannot be excluded.

M. ramiflorus is considered shade-tolerant (Williams & Buxton 1989), and it has a similar germination rate in the dark and light (Burrows 1995). This could explain its relative abundance in the Tai Tapu site compared with other species, for example *Fuchsia* *excorticata* (Forster & Forster f.) L. f., which have high light requirements for germination. Willow, being deciduous, is perhaps the optimum kind of shade for *M. ramiflorus*.

Ecology of exotic species

There is little or no literature on the ecology of *Salix* spp. in New Zealand (although some Eurpopean data exist), but the ecology of *Sambucus nigra* will be briefly discussed here, because it occurs at high densities in the Tai Tapu Study site. It is the predominant exotic species present in the site that has the potential to be ecologically important in the area. Using the facilitation/ successional model (Connell & Slatyer 1977), *S. nigra* is able to take over the role of native species that are adapted to early successional environments and therefore act as a facilitator of succession. *S. nigra* is a light-demanding species and therefore will probably continue to dominate early succession in this site, because when the ability of seedlings to establish is marginal, other characteristics of the species may increase its likelihood of success, for example, being a prolific seed producer and having the ability to reproduce vegetatively.

Flooding

In 1996, more individuals died in flooded areas than were recruited. It is likely that seeds that could have been present and been potential new recruits would probably have rotted over winter or new recruits with different root depths in areas flooded for lengthy periods would have died due to persistent inundation. However, more individuals overall died in non-flooded areas compared with flooded areas. In 1997, more individuals died in flooded areas than were recruited. As in 1996, more individuals overall died in non-flooded areas compared with flooded areas.

In 1996, most recruitment into plots allocated a floodclass occurred in *Coprosma* spp. The following year, most mortality in these plots occurred in that species. Seeds that manage to germinate and survive from one year to the next, may not be able to tolerate flooded conditions at certain stages in their life cycle. This could be due to individual characteristics of the plant, or to the degree of flooding they were subjected to, for example, only few species can develop aerenchymas tissue in response to waterlogged soils (e.g. *Leptospermum scoparium)* (Cook *et al.*, 1980).

Within forested areas, where most seeds of these woody species are deposited there should be, if conditions are moist, a correspondingly high potential for the occurrence of abundant seedlings at times when moisture stress is least likely to affect both germination and establishment of seedlings (Burrows 1993). This characteristic is likely to occur in an exotic woodland, where seeds are dispersed into the area. Frequent observation shows that each autumn, winter and spring, bare ground in mixed forest on Banks Peninsula is carpeted with small seedlings. Most of these disappear in summer probably mainly because of dry conditions typically experienced in the region (Burrows 1993). Similar patterns occur at the Tai Tapu site (personal observation).

The flooding study was probably done at the wrong scale as it is difficult to fully explain and interpret recruitment and mortality as a function of flooding, with the data collected. It probably should have been continued throughout the duration of the fieldwork.

2. Soil seed banks

In all samples taken, native woody species were poorly represented in the persistent seed bank relative to exotic herbs and adventives (Appendix 3). Differences in germination behaviour of seeds, may relate as much to phylogenetic history as to ecology, although the seed dispersal mode of each species has some bearing on the germination behaviour (Burrows 1996). The complete absence of woody native species germinating in any of the paddock or poplar border samples supports the assumption that birds are dispersing seeds to the *Salix* spp. area and using these trees as sites for perching and defecating. Germination delays and patterns have evolved as a series of strategies to cope with constraints imposed by the seed's environment, for example dispersal, biotic hazards, climatic hazards and competition from surrounding plants. Evolved germination delay strategies (e.g., impermeable seed coat, fleshy pericarp tissues or primary dormancy) can affect germination rates in inter- and intra-specific germination. Factors causing germination delay permit temporal spread of germination, but for certain species (e.g., *F. excorticata, G. littoralis, Macropiper excelsum* (G. Forst.) Miq. and *M. ramiflorus*) are insufficiently strong to permit the development of long-term seed banks (Burrows 1995). Seeds of *Griselinia littoralis* allowed to dry in laboratory conditions showed reduced rates of germination typical of recalcitrance in seeds (Bannister *et al.* 1996). They also have a thin seed coat which could make them more vulnerable to normal bird digestive processes. In the experiment described in this thesis, germination of four species occurred in soil taken from dry sites in the area, and of two species in soil taken from wet sites. As each soil sample received the same glasshouse treatment, it is difficult to make comparisons of the effects between the originating conditions (dry/wet site) of these samples on germination. To address this, viable seeds could be placed in distinctly wet and distinctly dry sites and monitored to test for germination rates between the two.

The four native woody species which germinated in this glasshouse experiment were *C. australis*, *M. australis*, *M. laetum* and *Coprosma* (unknown species). Results from previous experimental work (Burrows 1993) showed that most *C. australis* seeds germinate within six months (although some overwinter). *M. australis* seeds germinate within 6-9 months and those of *M. laetum* germinate sporadically in the first 10 months, with the rest germinating over a short period. The *Coprosma* spp. tested germinated within six months. Relevant (to this study) species tested in which germination did not occur until after the first year, were *Melicope simplex* A. Cunn., *Parsonsia heterophylla* (G.Forst), *Myrsine divaricata* A. Cunn, *Pseudowintera colorata* (Raoul) Dandy and *M. laetum*. Most of these species also had low germination success rates in the dark. The glasshouse experiment conducted for this thesis could have been extended, so that the duration of monitoring was more than one year, but it was assumed that most seeds present would have germinated in this time. Seeds of different species germinate at different rates which are characteristic for the species. Some germinate quickly in summer or autumn, while others are slower, germinating in autumn, winter or early spring (Burrows 1993). The germination of each native seed in this experiment occurred in the first six months. This period was followed by six months of no new germination. To detect species with very small seeds which are difficult positively to identify in a seedtrap investigation (for example, *F. excorticata*), the glasshouse method is valuable. Species that have special germination requirements that were not provided, however, could be under-represented in these results.

Preliminary findings of studies quantifying the properties of soil seedbanks in the temperate forests of New Zealand are similar to patterns found for forests elsewhere (Sem & Enright 1996). There is little similarity in species composition between the soil seedbank, which is typically dominated by species characteristic of disturbed areas (ruderal species) and the extant forest vegetation. In the work described in this thesis, the soil seedbank of all samples (i.e., those taken from both the willow and non-willow areas) was dominated by ruderal species, mainly adventive weeds. This is not surprising given the local conditions/landuse.

Originally, this study site was a very important food-gathering place or mahinga kai (Evison 1994). It consisted of wetland vegetation only, with little or no woody species except perhaps *Coprosma propinqua* A. Cunn (C. Meurk pers.comm). Generally speaking, the survival of seeds produced on-site by species present at an earlier stage of forest stand development would suggest a seedbank inheritance (Sem & Enright 1996). The complete absence of a woody native seedbank in the paddock or poplar border samples, the high germination success rates and lack of long germination delays for many native seed species (Burrows 1993) and positive germination from the seedbank samples taken from the area, further support the hypothesis that birds are dispersing seeds to the *Salix* stand. Therefore, I have concluded that there was no native woody vegetation present at the site immediately prior to European arrival.

3. Seed trapping

The seed traps were set up to monitor seed arrival. This technique is likely to detect fewer species than would the seed bank analysis because of possible predation from the seedtraps and the difficulty of sorting and analysing the seedtrap contents. Few seeds of native woody species were identified and most of these were *M. australis*. These would probably have fallen from the canopy anyway. The only other two species present were *S. nigra* and *Coprosma* spp. Both were likely to have been dispersed into the area as there are no plants mature enough to be producing seed in the study site.

Rates of seed input were probably too low to account for the actual recruitment of individuals over the duration of the study or the higher seed densities in the soil. This could be indicative of extreme year to year variability in seed inputs, not enough seed traps, not in the right places or predation, as survival of seeds produced on-site by species present at an earlier stage of forest stand development (i.e., seedbank inheritance) is highly unlikely.

4. Mist netting

Mist netting was carried out to collect faecal samples of birds visiting the area. No whole seeds of native woody plants were identifiable from any samples collected. As finches are seed predators (and they were the most common species caught), this is not surprising. However, it had been hoped that other frugivores may have provided identifiable seeds. Nineteen species of bird were observed in the area from November 1994 to February 1997. Of these, 15 were known to eat fruit. In the 1994/5 breeding season, three *Turdus philomelos* (song thrush) nests, one *Zosterops lateralis lateralis* Latham (silvereye) and one *Carduelis chloris chloris* Linnaeus (green finch) nest with chicks were observed. In 1995/6, seven song thrush nests, one greenfinch and two blackbird nests with chicks were observed. Originally, this research aimed to provide a preliminary understanding of the dynamics of ornithocorous (bird-dispersed) native seed in the area. There is no direct evidence however, from these results that birds were dispersing seeds into the area. Birds could have been shot and had their stomach contents analysed for seeds, but this was not considered a priority and was not undertaken. Attracting frugivores to restoration areas has several advantages. Frugivores bring in seeds of their favoured fruits, thus attracting further conspecifics as new plant food matures. In this study, from the total of 42 bird-dispersed woody plant species in Ahuriri Summit Reserve (Burrows 1994b), 22 were represented in the willow swamp (i.e., 52%) and three in the seed trap catch. Some plants (including endangered species) depend on particular dispersal species, and the local assemblage of frugivores can exert a profound effect on the ultimate structure and composition of forest regrowth (for example, see Crome *et al.* 1994, Crossland 1996, Lee 1996 and Williams & Karl 1996). Providing or protecting bird perches is one simple tool therefore, for accelerating ecological succession and reforesting disturbed land (McClanahan & Wolfe 1993).

Insectivorous species present at the site such as *Gerygone igata* Quoy & Gaimard (grey warbler) and *Rhipidura fuliginosa fuliginosa* Sparrman (fantail) are known to occasionally feed on fruit (O'Donnell & Dilks 1989), but this observation was made in extensive indigenous forest in South Westland and fruit comprised <1% of their diets. Neither of these species were caught in this study, so conclusions as to their seed contribution cannot be determined, but it is likely (personal observation) to be negligible if any.

Many New Zealand plants appear to depend on frugivores for seed dispersal (Lee *et al.* 1991). Non-endemic and adventive birds disperse indigenous fruits into early successional vegetation (Williams & Karl 1996), and the importance of their seed rain for conservation of biodiversity therefore depends on the site. At the Tai Tapu willow site, it is probably adventive birds that are carrying out most of the dispersal because in three years of observations, native species which eat fruit were only seen/heard on a few occasions. *Hemiphaga novaeseelandiae* Gmelin (New Zealand Pigeon) are present year round in both Ahuriri Summit Reserve and Kennedy's Bush Scenic Reserve (personal observation) but were never sighted at the Tai Tapu study site. *Sturnus vulgaris* Linnaeus (starlings) eat indigenous fruit (Williams & Karl 1996), but because none were caught in this study it is not possible to speculate on their contribution to the native regeneration occurring in the swamp. On certain occasions (mainly during summer), large flocks of starlings used the willows as a roosting site. An absence of mature plants producing seeds in the site though, offers strong support to the hypothesis that birds are responsible for the dispersal of native seeds from

Ahuriri Summit and Ahuriri Valley Reserves.

Other considerations

Phenology in Ahuriri Summit Reserve

Year-round (1994-97) observations were carried out in Ahuriri Summit Reserve and surrounding areas to determine flowering and fruiting phenology of the native vegetation occurring there. Species were recorded as fruiting if any ripe fruit was observed on any individual. At the same time, incidental bird observations indicated which birds ate which fruit and when. Faeces from *H. novaeseelandiae* in Ahuriri were collected by following the birds and scraping new faeces into pottles on 16.2.95 and 24.2.95. The seeds present were cleaned, and identified as *Fuchsia excorticata*, *Myrsine australis* (A. Rich.) Allan and *Coprosma rotundifolia* A. Cunn (C. Burrows pers. comm). The reason why this was performed was because the original research plan included the quantification of the seeds that were dispersed by birds into the Study site 1, and these casual observations in Ahuriri Summit Reserve would be useful to the overall research.

The seeds of most of the woody plant species of Ahuriri Summit Reserve (42 of 56 species, or 75%) have the potential to be dispersed by birds (Burrows 1994b). In Ahuriri Valley Reserve 49 native woody species occur (Molloy 1980). Extrapolating Burrows' estimate, 38 of these have the potential to be bird-dispersed. In the Tai Tapu study site, 22 bird-dispersed native woody species occur (52% and 58% of the total available from Ahuriri Summit Reserve and Ahuriri Valley Reserve respectively). Each of these species occurs in one or both of the reserves. This characterisation of the flora strongly suggests that in the absence of a long-term seedbank, the seeds have been recently dispersed by birds into the site. Interesting is the absence from the Tai Tapu site of *Fuchsia excorticata*, *Schlefflera digitata* J.R. & G. Forst., *Macropiper excelsum* and *Pennantia corymbosa* J.R. & G. Forst. The small seed size of *F. excorticata* and limited energy reserve may result in the species being intolerant of shade, leading to its absence from the site because of low light levels. Its germination rate is much lower in the dark than in the light (Burrows 1995). *F. excorticata*, *S. digitata*, and *M. excelsum* do not thrive in overly wet conditions, but *P. corymbosa* persists in the seed bank for a few years (C. Meurk pers. comm.). However, its ecology is not well enough understood to account for its absence in the Tai Tapu site.

Plate 10: Ahuriri Valley reserve from the Tai Tapu Study site

Two *Pseudopanax ferox* Kirk individuals were present at the Tai Tapu Study site, although they were not recorded in any quadrat. This species is included in a register of rare and endangered species (Given 1976).

Pollen analysis

A single sample of peat was taken from the study site from 40cm depth for a pollen analysis. Pollen analysis is an important and widely used paleolimnological technique (Moore *et al.*, 1991). Results from this sample suggested that the peat is of European age, i.e., formed after European settlement, as pollen of Cupressaceae, *Pinus*, and *Salix* was present. Seeds of *Juncus* spp. and *Carex* spp. were also present in the sample (J. Wilmshurst pers.comm.). The results of this coring also support the hypothesis that there was no woody native vegetation present at the site prior to European human arrival.

"Nurse" plants for the regeneration of native forest

The Canterbury Regional Council (CRC) (1996) advocated the use of undisturbed stands of *Ulex europaeus* L. (gorse) or *Cyticus scoparius* (L.) Link (broom) which can be retained to act as "nurse" plants for the regeneration of native forest. It is a promising and realistic land management option for gorse and broom infested areas of Banks Peninsula and much of the Canterbury foothills. The best results arise from sites with gorse or broom stands that have received no mechanical or biological control (i.e., just left undisturbed) for 10 years or more. Undisturbed stands progressively exclude grasses and herbage; often under a nitrogen enriched litter layer, providing suitable site conditions for the germination and growth of native flora. CRC has suggested that it is possible to classify land according to its potential to develop secondary successions through to native forest from dense gorse to broom. A scoring system is used to give an indication of potential for success, using rainfall, aspect and distance from seed source as the key factors. A further factor that affects the rate of succession is the density of gorse or broom. Dense stands that have crowded out grass and herbaceous competition will move more quickly to succession. Succession in open stands of gorse or broom with denser grass or other groundcover will be slower (CRC 1996).

Providing gorse and broom are sufficiently dense to eliminate grass and herbaceous ground cover and rainfall is adequate, emergence of native shrub-hardwood species takes as little as 3-5 years in the absence of all grazing animals. Complete smothering and replacement of gorse and broom by native shrub-hardwood forest could take place within 10-30 years (CRC 1996, Wilson 1994). This study shows that perhaps *Salix* stands could also be used as nurse plants for the regeneration of native forest. The *Salix* trees in this area are 35-40 years old, but as cattle were excluded from the area in only 1992/93, it is too early to make predictions regarding the potential for complete smothering and replacement of the *Salix* by native shrub-hardwood forest.

Key factors

For restoration success in riparian margins of Christchurch, Morland (1996) has identified several key factors. These are soil moisture, riparian class, canopy and slope. Key factors for *Salix* communities to act as a nursery for regeneration of native vegetation could include, distance to nearest seed source, the ability to attract seed-dispersers, the control of vertebrate (and invertebrate) herbivory, flooding potential and possibly light availability. An understanding of the ecology of *Salix* is also important. *Salix* spp. can be used if regeneration is the objective, but the lower the potential due to the key factors being unfavourable, the more important it is that both farm and feral grazing animals are excluded. Since many vegetation associations have been severely diminished on the Canterbury Plain, attracting seed-dispersers into regeneration areas assists with conservation of species and habitats, and is cost-effective. It should not be overlooked however, that the manner in which a restored site (as its plant cover develops) becomes more attractive to alien rather than to native mammals and birds (and weeds), as its plant cover develops is another topic that deserves study (Atkinson 1997).

Chapter Four

Leeston farm site - experimental regeneration of forest communities in an agroecosystem

Introduction

An interest in enhancing ecological diversity on the Canterbury Plain provided the impetus for this part of the experimental work. The Plain in general is depauperate in native species of both plants and animals. Some farmers are enthusiastic about restoring small areas or field margins of their land to a semi- or largely native system in order to recreate a more diverse landscape. Presently, field margins on the Plain are characterised by post and wire fences, with heavy herbicide use at their bases. Woody field margins, where they exist, usually comprise *Populus* spp., *P. radiata, C. macrocarpa, S. nigra, Ulex* spp. and occasionally *C. australis*, *Muehlenbeckia complexa* (A. Cunn.) Meissn. and *M. australis*; the vertebrate and invertebrate communities are also sometimes dominated by non-native species, although high numbers of native and endemic invertebrate species can also occur (Topping & Lövei, 1997). To change this situation and set about restoring small areas in Canterbury, land managers need to know the best way to go about it. A growing recognition of the landscape and other benefits from trees and shrubs on farms and lifestyle blocks has led to a recent upsurge in planting in rural areas, for example, in Australia, planting of trees has reduced the saline content of soils by lowering the water table (the three highest priority environmental issues in Western Australia have been identified as land salinisation, salinisation of inland waters and maintaining biodiversity (Environment Western Australia 1998)). Some surveys in New Zealand have suggested however, that a high proportion of trees (possibly up to 50%) planted in rural areas die within two years of planting (Giller 1994).

Ideal species for restoration purposes need to be site tolerant and well-adapted to the local environment. In general, in the open, the most valuable species are those that are natural pioneers or at least those of the second shrubland-woodland phase. Usually, species should be chosen in relation to the particular qualities of the site such as soil pH and climate. It is important to choose plants which are tolerant of the moisture regime of a given site so that supplementary watering is not required. With these criteria in mind, species were selected for the experiment.

Often, there is little potential for designing restoration projects as controlled experiments. Replanting tends to be in an *ad hoc* fashion without scientific input into the scheme, and site selection is often poor (Atkinson 1990). He has concluded that opportunities for proper experiments with replicated treatments and controls are likely to be the exception and not the rule. The work described here represents an initial controlled experimental approach to ecological restoration in a farmland situation in Canterbury. It is however, part of a longer-term experiment to examine the botanical and invertebrate components of the restoration scheme (S Wratten, pers. comm.). The ultimate aim of the work is to devise a planting protocol for future use by interested farmers in Canterbury. There are many publications (e.g., Simberloff 1990, Porteous 1993, Atkinson 1994, Lamb 1994) which document procedures for ecological restoration, but few present data on the success of the methods.

Nutrients

Little is known in New Zealand about nutrient requirements of indigenous seedlings in the field, especially in a restoration context. Nutrients are the principal cause of failure of land rehabilitation (more so in mine sites) due to deficiency (Bradshaw & Chadwick 1980). The critical nutrients are nearly always nitrogen and phosphorus. A wide range of fertilisers (slow and fast release) are available for promoting plant growth and after determining the particular nutrient deficiencies, appropriate fertilisers should be chosen to suit individual problems. Chicken manure was chosen for this experiment because of its availability, low cost and ease of use. Since properly grown planting stock will usually be well-fertilised and carry its own supply of nutrients for the first year, the main need for fertiliser will, in general, be in the second and subsequent years (Bradshaw & Chadwick 1980), although in this experiment it was applied in year one to help promote initial growth.

Mulches

Mulches of a fibrous material such as chopped straw, shredded bark or wood pulp are widely used temporarily to protect the surface (reduce weed invasion) of land under restoration and improve soil condition by adding organic matter (Bradshaw & Chadwick 1980). There is little literature in New Zealand, relating mulch or fertiliser type to native plant performance in a restoration context, although recent research in Westland suggested that neither the addition of compost, nor blood and bone had any significant effect on native plant growth or survival (Ogle 1996). The research described in this thesis was therefore carried out with an empirical approach to plant success in relation to mulch and fertiliser.

Leaf size

There is a strong correlation between leaf size in plants and distance from the equator (Parkhurst & Loucks 1969). The largest-leaved plants tend to be in moist equatorial regions, and leaf size steadily decreases in cooler and more variable climates towards the poles. Water loss experiments have shown no clear proof that the small-leaved species are more drought resistant than their large-leaved relatives (Wilson & Galloway 1993), and some small-leaved species for example, *Coprosma rotundifolia* are in fact quite prone to frost damage. Also, many hardwoods in other parts of the world are deciduous which is another strategy against water loss. At Leeston, the hypothesis that narrow-leaved plants survive better in lowland Canterbury than broad-leaved plants, by planting both types with broad leaves, was investigated.

Three sites were used at an old homestead ("Winfield") returned to pasture near Te Waihora (Lake Ellesmere), Leeston. It is an exposed farmland site, provided by the Osborne family who have farmed there since the mid nineteenth century. Unlike the Tai Tapu *Salix* site (Chapter 3), there is no significant native seed source (except *P. regius* and *M. australis*) nearby. Consequently, native seed dispersal into the area by birds is presumably negligible.

It is necessary to choose species known to be well-adapted to the local environment. In general, in the open, the most valuable species are those that are natural pioneers or at least those of the second shrubland-woodland phase. Usually, species are chosen in relation to the particular qualities of the site such as soil pH and climate. A range of mid-late successional plant species (Appendix 4) originally part of the Canterbury Plain flora (C. Meurk pers.comm.) were planted. They were obtained from the Wai-ora Trust nursery and the Department of Conservation nursery at Motukarara, both of which propagate plants of local genetic source. It is important to maintain genetic integrity in restoration work (for example; McCombs 1993) and to choose species known to be adapted to the local environment to prevent resource wastage. Planting in Leeston occurred in October 1995 and monitoring (see below) commenced after planting.

Planting took place according to the recommended method of Porteous (1993), with reference to the guidelines for establishing trees and shrubs in rural Canterbury (Giller 1994), in three experimental areas. Two of these (Area 1 and 3) were planted with a range of species and one area (Area 2) was planted with a single species (*Hebe salicifolia* (Forst. f) Pennell).

Databases such as the NZBN (New Zealand Bibliographic Network), Netscape CRI libraries catalogue and SIRIS were searched for ecological data (e.g., growth rates, survival, susceptibility to herbivory etc) on the species planted in this experiment. Searches and analyses on data held in the National Indigenous Vegetation Survey (NIVS) database at Landcare Research, Lincoln (Forest Research Institute 1989) were also carried out. These data were originally collected by the New Zealand Forest Service. This was carried out in order to make some comparisons between observations from this research and those already documented.

Methods

A range of mid-late successional plant species (Appendix 4) originally part of the Canterbury Plain flora were planted in 3 areas in the paddocks. Each was established using researchers and students as the labour force and the areas then received no further management, apart from irrigation (Area 1 only, see below) on two occasions immediately after planting, to alleviate drought during establishment. Three experimental areas (see Figure 11 for layout) were chosen.

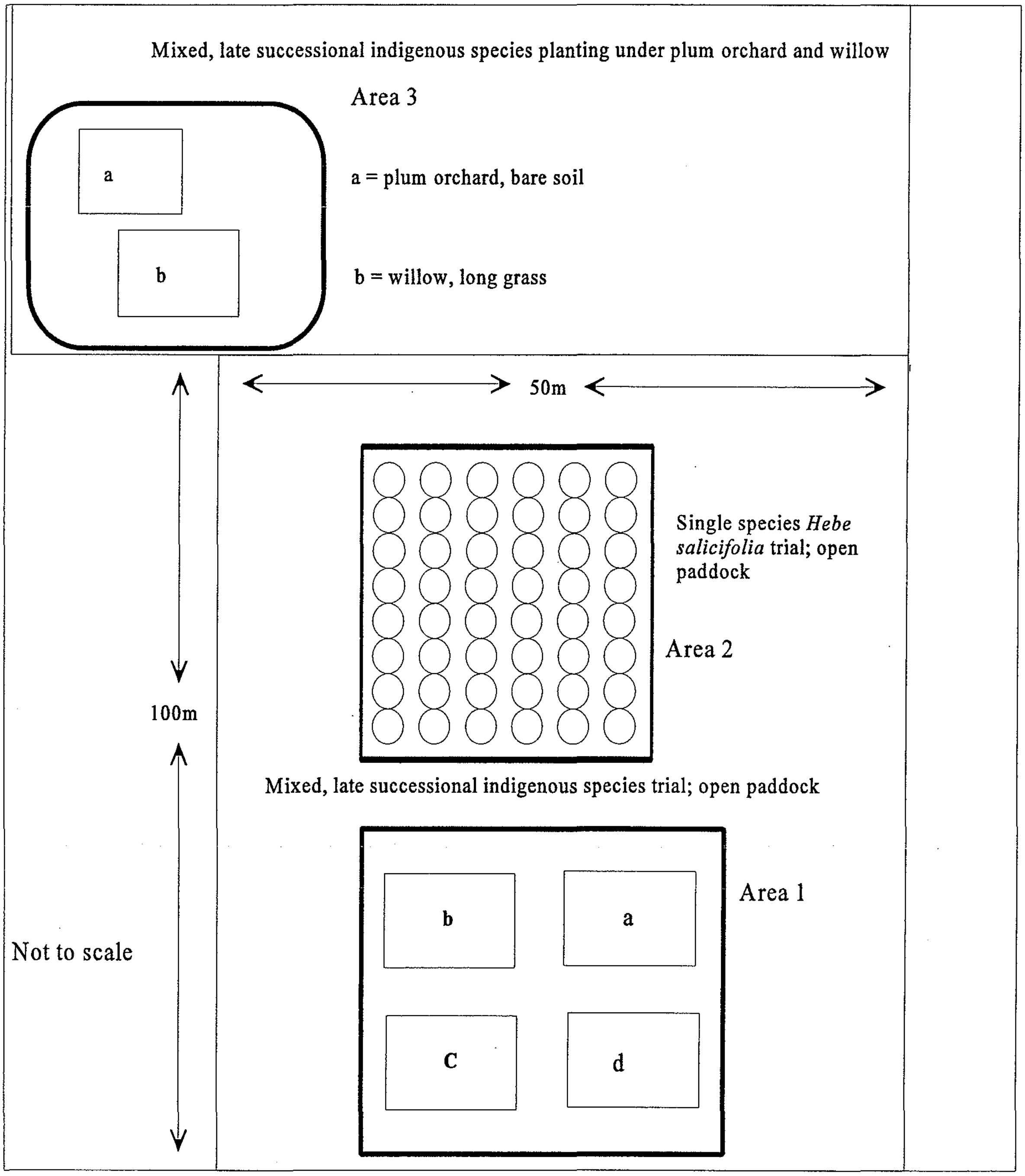


Figure 11: Layout of the Leeston field site

An area of 100m x 100m x 50m formerly grazed paddock was fenced by the farmer and Areas 1 and 2 were set up within this boundary. An existing *Salix* spp./ *Prunus domestica* (L.) orchard was used for the Area 3 trial.

Area 1:	mixed, mid-late successional indigenous species planting trial; open paddock.
Area 2:	single indigenous species (*Hebe salicifolia*) planting trial; open paddock.
Area 3:	mixed, mid-late successional indigenous species planting trial under existing exotic vegetation (abandoned plum orchard surrounded by *Salix* and poplar).

Area 1 - Open paddock; late successional indigenous species planting.

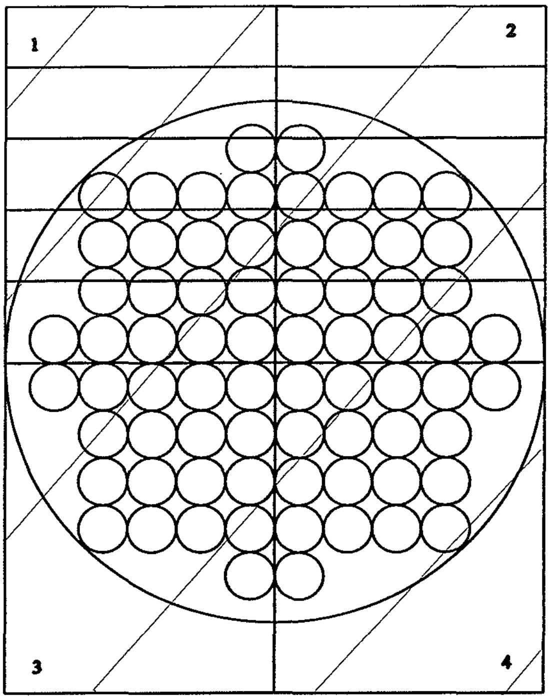
A factorial experiment of clumped and regularly spaced plantings was set up. Treatments comprised either 1) bark or straw mulches and 2) fertiliser. Area 1 was sprayed with Roundup[c] (10ml/l of water; active ingredient = 360g/l glyphosate as the isopropylamine salt) before commencement of the trial. It consisted of four plots (nominated a-d), each 12m x 28m (Figure 12), each divided into four quarters (numbered 1-4). In each of these 4 plots, 76 individuals of 11 different mid-late successional native species were planted in approximately equal numbers (some quarters received one or two more individuals of one species than others, due to plant supply constraints). Each plot consisted of a mixture of broad-leaved sub-canopy plants e.g., *Coprosma robusta* Raoul, *Aristotelia serrata* (J.R. & G.Forst.) W. Oliver, *M. ramiflorus*, *P. arboreus*, *P. eugenioides*, *G. littoralis,* and taller canopy trees e.g. *Hoheria angustifolia* Raoul, *D. dacrydioides*, *P. totara* and *P. regius*.

Plot a (clumped)

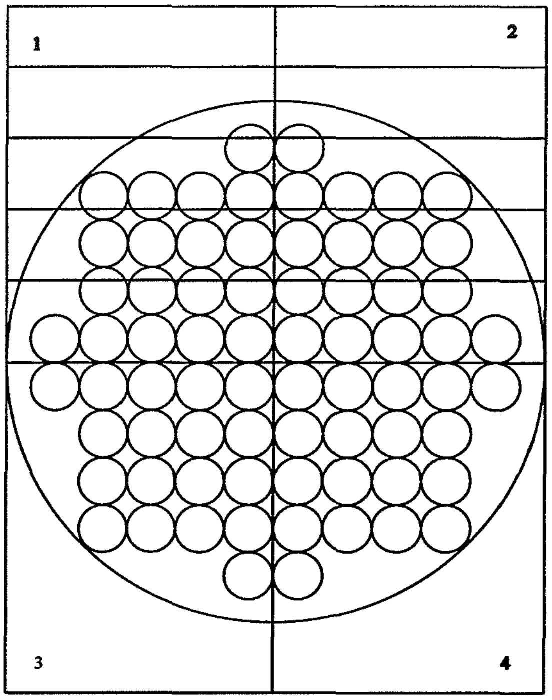
All species were planted within a 5m radius of a central peg. Each plant was placed 0.5-0.7m away from its neighbour. The southern half (quadrats 1 and 2) was covered with fertiliser (chicken manure) to a depth of approximately 0.002m. Then, pine forest bark mulch was added to cover the 10m diameter circle to a depth of approximately 0.15m.

Plot b (clumped)

All species were planted within a 5m radius of a central peg. Each plant was placed at 0.5-0.7m away from its neighbour. The southern half (quadrats 1 and 2) was covered with fertiliser (chicken manure) to a depth of approximately 0.002m. Then, a barley straw mulch sourced from the study farm (six bales at 500kg/bale) was spread over the entire 12m x 28m area.

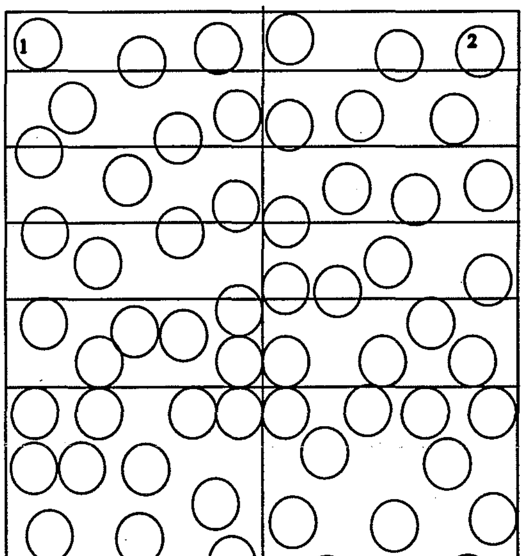


Plot b: Clumped planting with fertiliser, covered with barley straw mulch

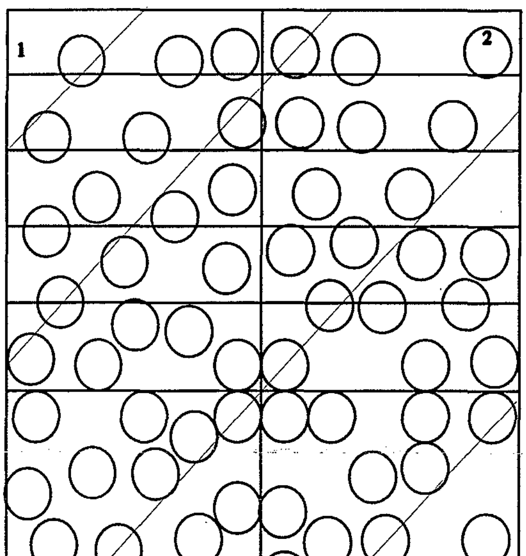


Plot a: Clumped planting with fertiliser, covered with forest bark mulch

Not to scale



Plot c: Distributed planting with fertiliser, covered with forest bark mulch



Plot d: Distributed planting with fertiliser, covered with barley straw mulch

Figure 12: Leeston restoration trial (Area 1, native mix area, mid-late successional species plots)

Plot c (dispersed)

Plants were planted in a dispersed manner with regular spacing (approximately 1-1.5m) between each plant. This was carried out in each quadrat to attain maximum possible spacing between plants over the entire 12m x 28m. The southern half (quadrats 1 & 2) received a covering of fertiliser to a depth of 0.002m. Then, bark mulch was placed to a depth of 0.15m and in a 0.5m radius around each plant.

Plot d (dispersed)

Plants were planted in the dispersed manner of plot c (above). The southern half (quadrats 1 & 2) received a covering of fertiliser to a depth of 0.002m. Then, a straw mulch was spread over the entire area.

Plant survival was initially frequently monitored; three times in the first month after planting (December 1995), then twice in January 1996 and once only in February 1996, June 1996, September 1996 and December 1997. The survival categories used were: healthy, stressed or dead, although the plants were usually alive or dead. Data from the first two months (December 1995, January 1996) were pooled, as were data from the next two recording dates. Four periods are presented in the results. Data were analysed with multivariate statistics (ANOVA - Systat). Analysis of the data provides information on the differences between narrow-leaved and broad-leaved plant survival under the different planting regimes. It also demonstrates the treatment effects of bark mulch, straw mulch, and the application of fertiliser.

Area 2 - Open paddock; single species (*H. salicifolia*) planting

The trial was set up as in Figure 13. The experiment consisted of six replicates of eight plots per replicate. This factorial design was used to investigate the influence of 2 factors (single plantings or cluster plantings) for four different treatments. Using one species (*Hebe salicifolia*) only, allowed for better comparisons of the effect of mulch or fertiliser than using a range of species as in Area 1. Prior to planting, an area of 24m x 36m was pegged out. A total of 48 1m x 1m plots were then established, with 3m spacing between each plot. Each plot received one of eight treatments:

1. single *Hebe* with no mulch
2. single *Hebe* with bark mulch
3. single *Hebe* with straw mulch
4. single *Hebe* with straw mulch and chicken manure (approximately 1kg/plot)

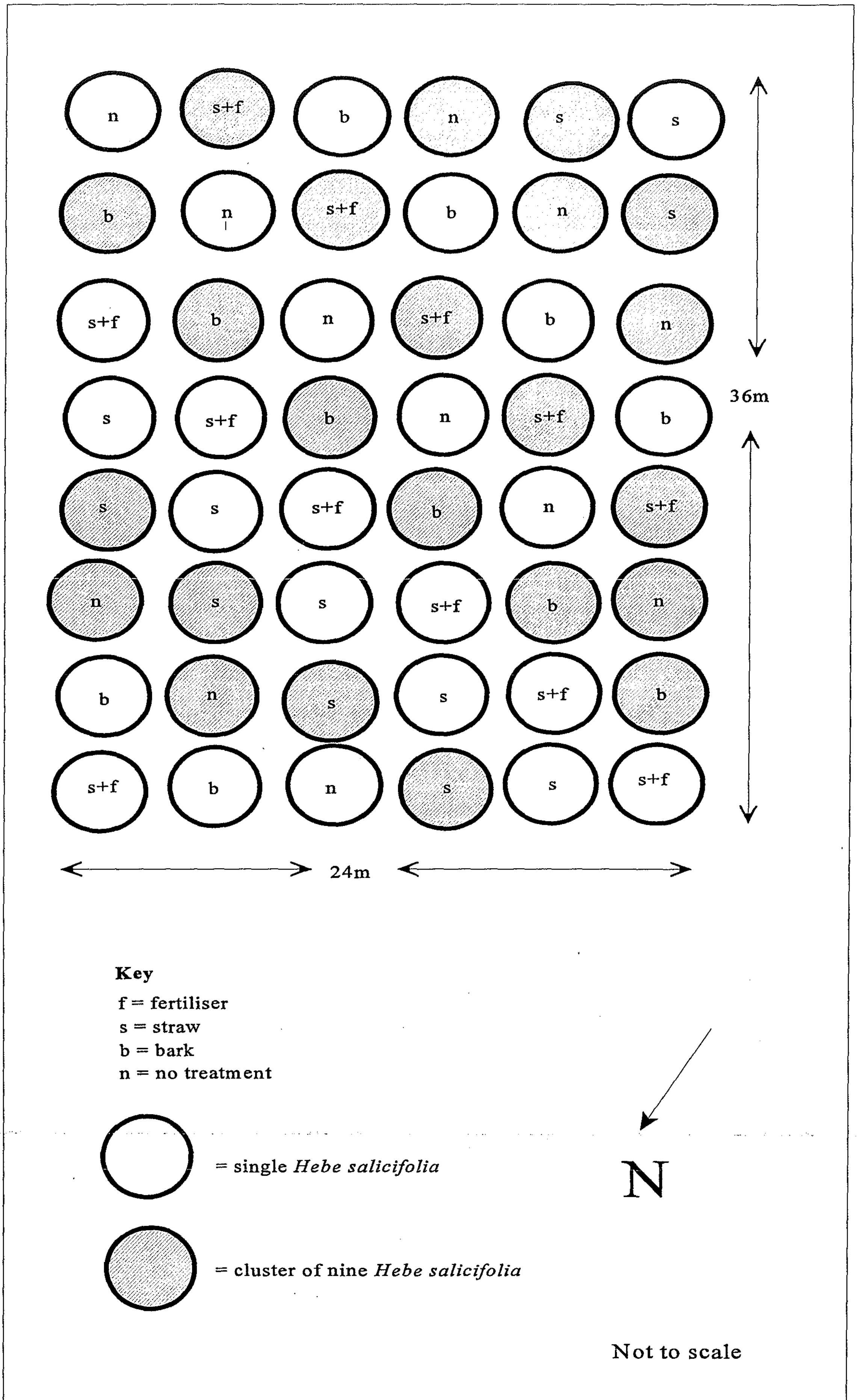


Figure 13: Leeston Study site: Area 2; *Hebe salicifolia* trial - to investigate the effects of straw, bark and chicken manure and the control (no treatment)

Plate 11: Leeston open paddock trial, Area 1 plot d

Plate 12: Leeston open paddock trial, Area 1 plot d

Treatments 5-8 were identical to treatments 1-4 except that instead of a single plant, a cluster of *Hebe* was used, consisting of nine plants arranged in a one-m^2 plot about a central marker peg, i.e., a square pattern of three plants by three. The treatment order of the first replicate (easternmost) was randomised. The following replicates (from east to west) were not random. Each treatment was positioned one quadrat lower (more northerly) than it had been in the previous replicate, i.e., in a Latin square design. This was to allow for soil variation and edge effects. Thus, the treatment at "position 1" in replicate one went to "position 2" in replicate two. The treatment at "position 8" in replicate one was moved to "position 1" in replicate two. After planting, mulches were applied to a depth of approximately 0.1m for the bark, or 0.3m for straw, to cover an area extending to approximately 1m out from the plots.

After one year (October 1996), each plot was visited and the number of surviving plants recorded. The height, diameter, and a "health" rating were recorded for each individual plant, whether planted as a single plant or as part of a cluster. Height was measured as the distance between the highest shoot pulled erect and the ground; diameter was taken at the widest point of the plant crown and the health rating was scored on a scale of 1-5, where 5 was allocated to a vigourous plant with no herbivory (i.e., growing tips, vigorous foliage, no discolouration and good habit) and 1 was allocated to a plant which was failing to thrive, i.e., unhealthy growing tips or foliage and leaf discolouration. Data were analysed with univariate statistics (ANOVA and a Fisher's Least Significant Difference test). The site was revisited in November 1997 and the same methods of measuring applied.

Area 3 - *Salix* and *Prunus* shelter; mixed mid-late successional species planting

Area 3 was planted under an abandoned *Prunus domestica* orchard surrounded by *Salix* spp. and *Populus* spp.. This area had been fenced from grazing stock by the farmer. A range of mid-late successional, indigenous species (60 individuals per plot) were planted. Two plots were set up to incorporate the same species as Area 1. They were also set up in the spaced (distributed) regime (i.e., similar to plots c and d in Area 1) but neither plot received any fertiliser, or straw or bark mulch. One plot was planted underneath *Prunus* (Plot a). This site consisted of bare soil and leaf litter. The other was planted underneath *Salix* (Plot b). This site consisted of tall non-native grasses. This *Prunus/Salix* area is similar to the Tai Tapu *Salix* site (Study site 1) and comparisons of plant success between the two areas can be made. Both the sites are sheltered from prevailing winds and the mature exotic trees afford shelter to the naturally colonised (Study site 1, Tai Tapu) and planted (Area 3, Leeston) native seedlings Plant survival was measured in October 1996 and December 1997. In each of the trials at Leeston, plants were not secured with stakes; they would probably be less at risk from the wind and competition from weeds, had they been staked. The temperature of the whole site (Areas 1,2 and 3) was monitored using a "Tiny-talk" thermo recorder at the soil surface, under the mulches, in the open paddock, under the *Prunus* shelter and in the adjacent long grass. Temperature was recorded for one week in summer (December 1995) and one week in autumn (April 1996). In mid summer (February 1996), soil moisture was measured by taking samples from under mulches, open paddock, under the *Prunus* shelter and in the long grass. Three 15mm x 50mm cores (volume $8.83cm^3$) of soil per site (Areas 1-3, plus from a nearby grazed paddock) were weighed wet, then dried at 100°C for 24 hours and re-weighed. These data, supported by local meteorological data (Cherry 1997) indicate some of the conditions that may influence plant survival and growth.

Methodology for analysis

Univariate repeated measures analysis of variance was used to analyse the data. Single-degree-of-freedom polynomial contrasts (polynomial test of order) were used for the plant survival data to test for linear and/or quadratic relationships between the treatments. In addition, Fisher's Least Significant Difference test was used in the *Hebe* trial. The data were analysed using SYSTAT software.

Plate 13: Planting in long grass underneath *Salix* canopy

Plate 14: Planting in long grass underneath *Prunus* canopy

Results

Area 1 - Open paddock; mid-late successional indigenous species planting.

Survival of plant species varied greatly. Most or all of the individuals of approximately half of the species planted had died after one year. Of the remaining species, approximately 50% of all individuals planted survived after one year. Generally, species that died were broad-leaved (for classification, see Table 11). Of all the individual plants planted in October 1995, all *C. robusta* (14 individuals) and *A. serrata* (40 individuals), were dead within two months of planting. *M. ramiflorus* and *P. arboreus* also failed to establish. Only one *M. ramiflorus* individual out of 14 planted was surviving in December 1995. By June 1996, this individual had died and each *D. dacrydioides* (14 individuals) had also died (Table 16). *P. regius* survival rate was best (nearly 80% of those planted were still alive after one year, the same figure after two years, in 1997), although all *H. salicifolia* (20 individuals) and *C. australis* (eight individuals), planted in December 1995 to replace the dead *A. serrata* and *M. ramiflorus* respectively, were also alive in September 1996. Survival of *C. australis* remained at 100% after two years. The *H. salicifolia* survival rate was 100% after one year, but had decreased to 75% in 1997. The survival rate of *H. angustifolia* was 70% after both one and two years and of *P. totara*, 71% after the first year and 43% after the second. Contributing factors to the death of plants were assumed to be frost damage, wind damage (mechanical damage and desiccation) and damage caused to the bark by straw rubbing, but none of these were quantified. No noticeable vertebrate or invertebrate herbivory occurred in the open paddock.

Table 16. Area 1; plant survival summary

Species	Common name	Leaf classification	n planted 10.10.95	% survival 1.6.96	% survival 26.9.96	% survival 12.12.97
Plagianthus regius	ribbonwood	narrow	33	87.9	78.8	78.8
Podocarpus totara	totara	narrow	28	92.8	71.1	43.0
Hoheria angustifolia	lacebark	narrow	37	78.4	70.0	70.0
Coprosma robusta	karamu	broad	14	0	0	0
Griselinia littoralis	broadleaf	broad	28	75.0	46.4	14
Pittosporum eugenioides	lemonwood	broad	40	55.0	17.5	15.0
Kunzea ericoides	kānuka	narrow	28	25.0	21.1	3.0
Dacrycarpus dacrydioides	kahikatea	narrow	14	0	0	0
Pseudopanax arboreus	fivefinger	broad	28	7.1	0	0
Aristotelia serrata	wineberry	broad	40	0	0	0
Melicytus ramiflorus	whiteywood	broad	14	0	0	0
Hebe salicifolia (planted 12/1995)	koromiko	broad	20	100.0	100.0	75.0
Cordyline australis (planted 12/1995)	cabbage tree	narrow	8	100.0	100.0	100.0

The relationships between leaf size, mulch type and planting regime on survival are presented in Table 17 for 1995/1996 only. The figures are derived from a univariate ANOVA. The proportion of plants surviving from those originally planted at each time interval are presented here.

Table 17: The effect of planting regime, mulch type, fertiliser and leaf size on % plant survival (T1= time 1 (1.12.95), T2=time 2 (1.6.96), T3= time 3 (26.9.96) T4= time 4 (12.12.97) F=fertiliser, NF=no fertiliser, t=thin leaved plant, b=broad leaved plant)

Planting regime	mulch	T1Ft	T1Fb	T1NFt	T1NFb	T2Ft	T2Fb	T2NFt	T2NFb
clumped	bark	0.89	0.40	0.83	0.32	0.61	0.35	0.50	0.27
clumped	straw	0.79	0.37	0.75	0.38	0.68	0.11	0.69	0.29
dispersed	**bark**	**0.95**	**0.58**	**0.88**	**0.52**	**0.58**	**0.21**	**0.82**	**0.43**
dispersed	**straw**	**0.68**	**0.47**	**0.94**	**0.33**	**0.58**	**0.16**	**0.65**	**0.24**

Planting regime	mulch	T3Ft	T3Fb	T3NFt	T3NFb	T4Ft	T4Fb	T4NFt	T4NFb
clumped	bark	0.5	0.1	0.61	0.18	0.69	0.1	0.61	0.05
clumped	straw	0.42	0.1	0.44	0.05	0.38	0.1	0.33	0
dispersed	**bark**	**0.56**	**0.11**	**0.76**	**0.19**	**0.65**	**0.14**	**0.67**	**0.11**
dispersed	**straw**	**0.47**	**0.1**	**0.47**	**0.14**	**0.41**	**0.1**	**0.62**	**0.11**

Univariate (distribution or mulch) and multivariate repeated measures analysis of the data from Table 17 is presented in Table 18 as an ANOVA summary.

Table 18: Effects of planting regime, mulch type, fertiliser and leaf size on survival (*= significant to p<0.05, **= significant to p< 0.01)

Single factor analysis	P value	significance
distribution	0.346	
mulch	0.123	
Multi factor analysis		
time	0.001	**
time × distribution	0.401	
time × mulch	0.214	
fertiliser	0.159	
fertiliser × distribution	0.639	
fertiliser × mulch	0.728	
thin (t) vs broad (b) leaf	0.004	*
t/b × distribution	0.237	
t/b × mulch	0.035	

Mulch type or distribution were not significant factors influencing plant survival. The interactions of time and distribution and time and mulch were also not significant. Mean plant survival (all species) was, not surprisingly, significant ($p < 0.05$) with time, but from these results, it cannot be said why. At T=1, mean survival was 84% and at T=2, mean survival was 60%. At T=3, mean survival was 43% and at T=4 mean survival 40%. In 1996 it was thought that fertiliser may be having a significant effect on survival ($p = 0.077$), but by 1997, $p = 0.159$. Narrow-leaved species survived significantly better than did broad-leaved ones ($p<0.05$). The interaction between thin/broad leaf and plant distribution or between thin/broad leaf and mulch type was not a significant factor influencing plant survival.

Area 2 - Open paddock; single species (*H. salicifolia*) planting

The measurements of *H. salicifolia* height, diameter (crown spread) and health resulted in some statistically significant differences between treatments over the two years (Table 19). In 1996, plant distribution had a significant effect on both plant height and plant health ($p < 0.05$).

Table 19: Effects on *Hebe salicifolia* height, diameter and health, 1996 & 1997 (* = significant to p<0.05, ** = significant to p< 0.01).

Factor analysis 1996	**height p value**	**significant**	**diameter p value**	**significant**	**health p value**	**significant**
plant distribution	0.042	*	0.323		0.008	**
treatment	0.032	*	0.001	**	0.164	
distribution × treatment	0.824		0.583		0.238	
Factor analysis 1997						
plant distribution	0.004	**	0.005	**	0.081	
treatment	0.054		0.068		0.062	
distribution × treatment	0.043	*	0.276		0.631	

There were statistically significant treatment differences after one year in the height, diameter and health of *H. salicifolia*, planted singly and in clumps. Single plants were taller and healthier than plants planted in clumps ($p < 0.05$). Heights and diameters, independent of grouping are presented in Figure 14.

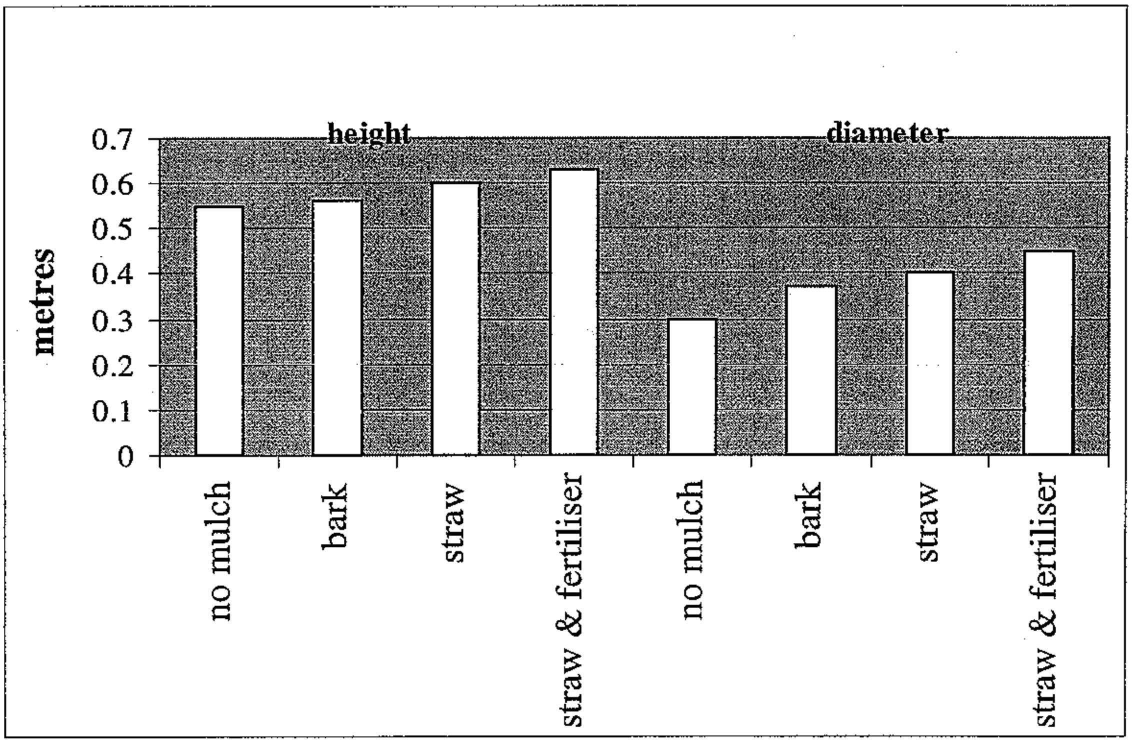


Figure 14: Mean *Hebe salicifolia* height and diameter after one year in relation to treatment

In year two, the increase in height in some plants appeared to be related to the application of fertiliser and mulch type (Figure 15).

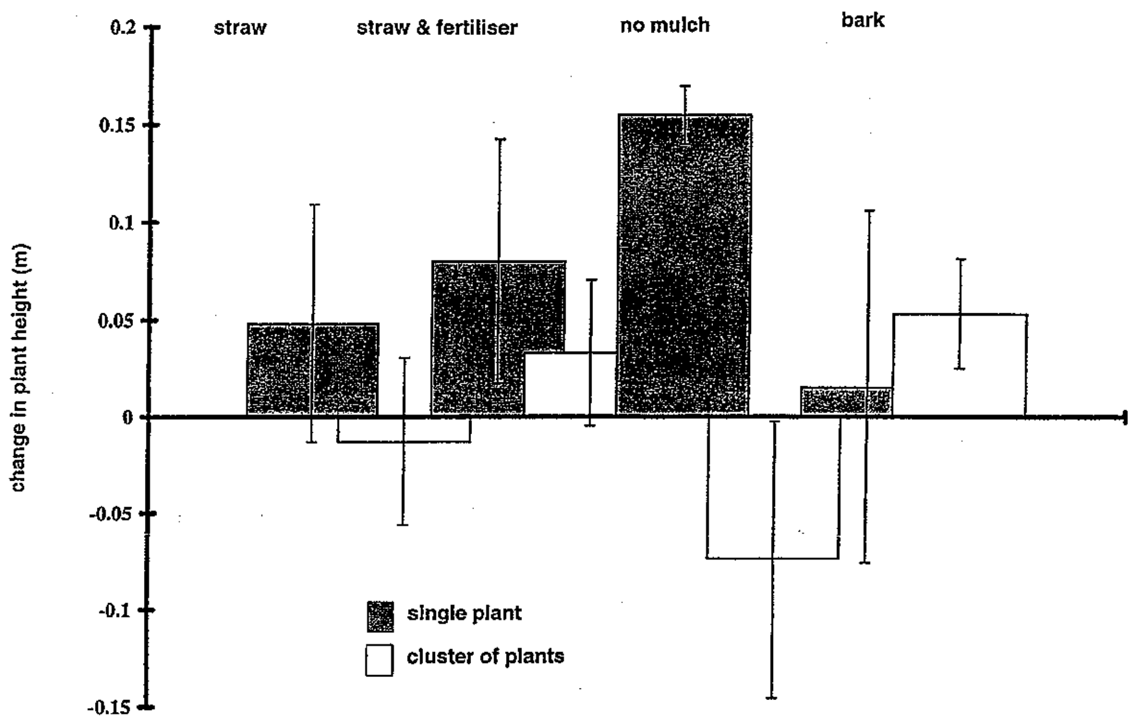


Figure 15: Change in mean *Hebe salicifolia* height (m) between year one and year two in relation to treatment (error bars are standard error of the means).

Growth was promoted by the presence of straw mulch compared with bark or no mulch. Greatest growth occurred with the combination of straw mulch and fertiliser ($p<0.05$) and when plants were planted as single individuals as opposed to clusters.

Plant diameter (crown spread) was also significantly affected by mulch type and the application of fertiliser ($p<0.05$). Plants growing with a straw mulch and fertiliser attained the greatest spread (Figure 16).

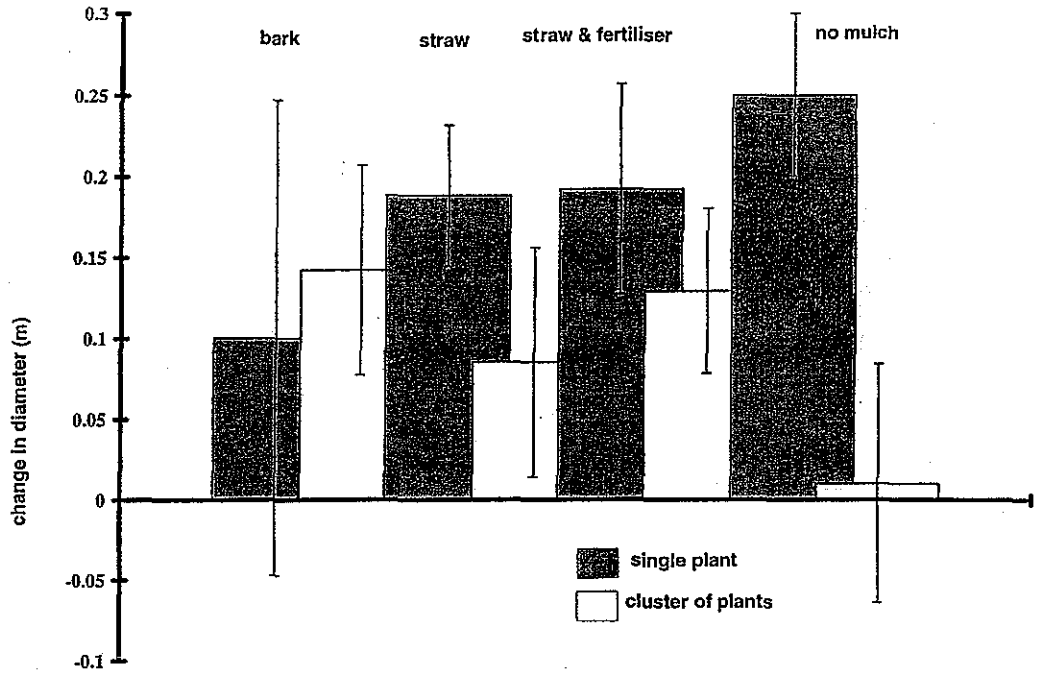


Figure 16: Change in mean *Hebe salicifolia* diameter (m) between year one and year two in relation to treatment (error bars are standard error of the means).

Health ratings did not appear to be related to either mulch treatment. They did, however, appear to be related to the planting regime. Single plants were more healthy than individuals planted in clusters ($p<0.05$) (Figures17 and 18).

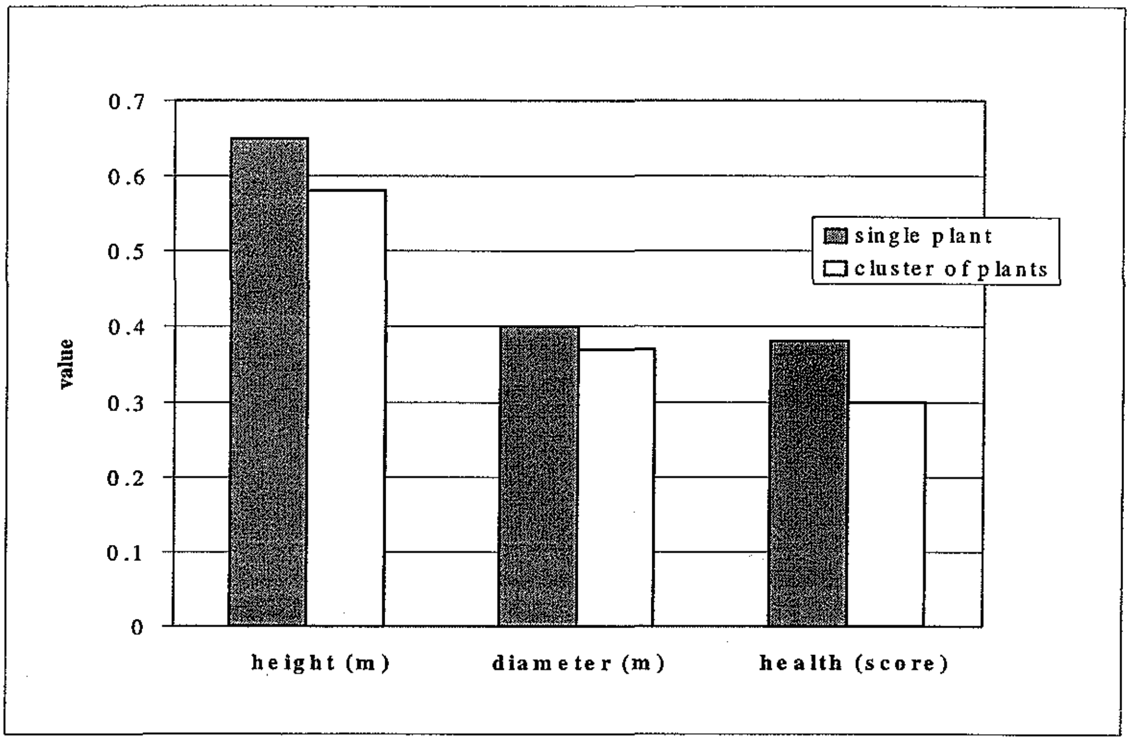


Figure 17: Effect of plant grouping on mean height (m), mean bush diameter (m), and mean health (index score) after one year

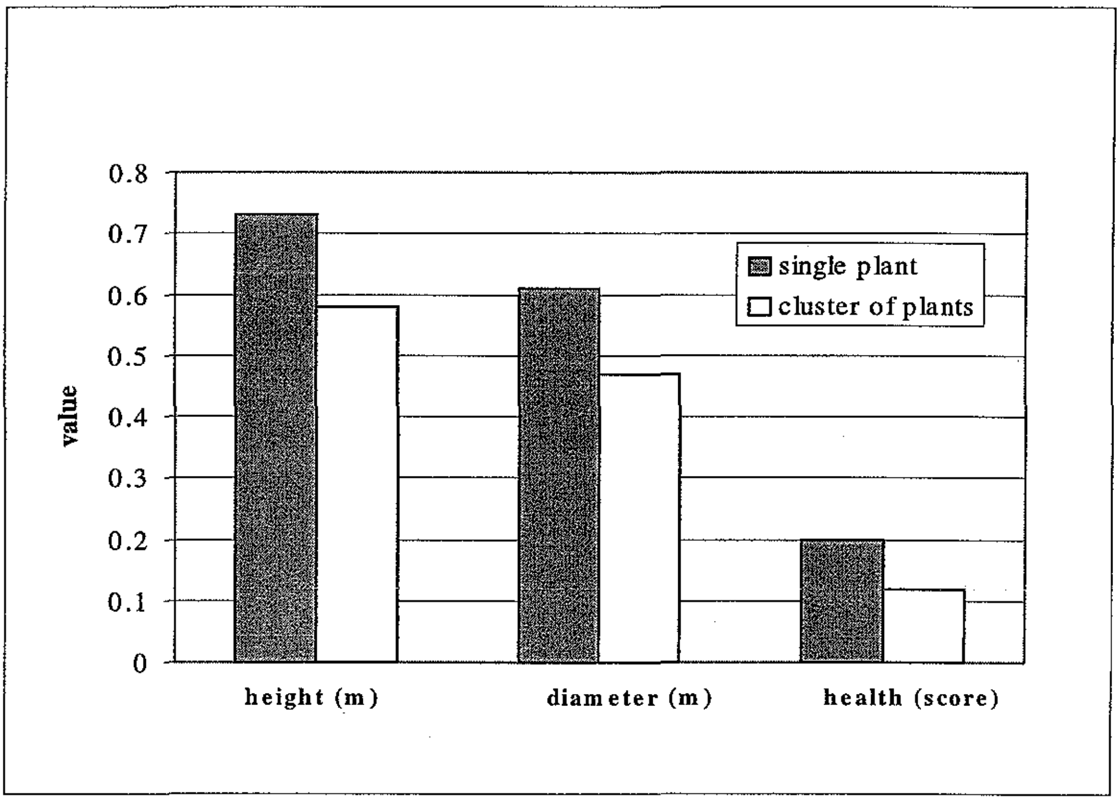


Figure 18: Effect of plant grouping on mean height (m), mean bush diameter (m), and mean health (index score) after two years

Area 3 - *Salix* and *Prunus* shelter; mixed mid-late successional species planting

Survivorship in Area 3 varied between species (Table 20).

Table 20: Total survivorship of plants in Area 3.

Species	n planted 20/10/95	% survival 22/10/96	% survival 12/12/97
Plagianthus regius	3	66.7	33.0
Podocarpus totara	12	58.3	42.0
Hoheria angustifolia	16	93.8	63.0
Griselinia littoralis	10	60.0	10.0
Pittosporum eugenioides	17	100.0	100.0
Kunzea ericoides	16	50.0	13.0
Dacrycarpus dacrydioides	6	50.0	33.0
Pseudopanax arboreus	12	8.3	0
Aristotelia serrata	22	54.5	36.0
Melicytus ramiflorus	6	50.0	50.0

For plots a (*Prunus*) and b (*Salix*) combined, *P. eugenioides* survived best (100% after year one and two). The survival rate of *H. angustifolia* was 94% after year one, but had decreased to 63% by year two. *P. arboreus* survived the least well. After year one, the survival rate was 8% and after year two, all individuals planted were dead. However, at least half the individuals of all species except *P. arboreus* were surviving one year after planting. By year two, only *H. angustifolia, P. eugenioides* and *M. ramiflorus* had survival rates of 50% or above.

Table 21: Survivorship of plants in Area 3 plot a (*Prunus* canopy, bare soil).

Species	Narrow/ broad leaved?	n planted 20/10/95 plot a	% survival 22/10/96	% survival 12/12/97
Plagianthus regius	n	2	50	50
Podocarpus totara	n	6	100	83
Hoheria angustifolia	n	8	88	38
Kunzea ericoides	n	8	50	13
Dacrycarpus dacrydioides	n	3	33	33
Griselinia littoralis	b	4	75	25
Pittosporum eugenioides	b	10	90	90
Pseudopanax arboreus	b	6	0	0
Aristotelia serrata	b	10	40	10
Melicytus ramiflorus	b	3	67	67

Table 22: Survivorship of plants in Area 3 plot b (*Salix* canopy, long grass).

Species	Narrow/ broad leaved?	n planted 20/10/95 plot b	% survival 22/10/96	% survival 12/12/97
Plagianthus regius	n	1	100	0
Podocarpus totara	n	6	17	0
Hoheria angustifolia	n	8	100	88
Kunzea ericoides	n	8	50	13
Dacrycarpus dacrydioides	n	3	67	33
Griselinia littoralis	b	6	50	0
Pittosporum eugenioides	b	7	100	100
Pseudopanax arboreus	b	6	17	0
Aristotelia serrata	b	12	75	58
Melicytus ramiflorus	b	3	33	33

Discussion

The experiments described here in Chapter four were carried out to quantify how different planting regimes affect growth and survival of native vegetation in a farmland restoration scheme and to assist with successful management of fragments of regenerating native vegetation in lowland Canterbury.

Different planting regimes and their effect on growth and survival of native vegetation in a farmland restoration scheme have been quantified. Mid-late successional under-canopy species establish poorly on the open Canterbury plains, probably because of the dominant dry north-westerly winds and lack of water. Existing shelter is necessary for the establishment of broad-leaved forest species under a low management regime. Narrow leaved species, for example, *Plagianthus* spp., *Hoheria* spp. and some podocarps e.g., *P. totara* generally survive and are the best group to use.

Of the treatments used, clumping, without fertiliser had no significant effect on plant survival. Single applications of fertiliser helped establishment, but may have retarded later development. The use of long-term mulch types such as bark were more useful than faster-degrading types (straw). No mulch and no fertiliser resulted in poorest growth initially, but in the best growth after establishment (two years). Low management at such sites leads to high mortality in some species. This work has shown experimentally how best to utilise native New Zealand woody species under a low-management regime likely to be typical of low budget, but large scale restoration attempts.

Area 1 - Open paddock; mid-late successional indigenous species planting.

Results for the mid-late successional plants clearly illustrate the difficulty of establishing native trees and shrubs on the Canterbury Plain. The most successful species in the open paddock were *Hebe salicifolia*, *Plagianthus regius*, *Cordyline australis* and *Hoheria angustifolia*. The planting of (native woody) vegetation in clusters as opposed to separated individuals has not enhanced initial survival, as was originally thought. Planting density may have been too high, although this is unlikely. This result is similar to Ogle's (1996) work where little difference in growth and survival was detected in a planting density trial. In any restoration attempt, planting density must be a compromise between achieving an early cover and expense, for example, a dense stand may require thinning later. Monitoring over the next

5-10 years would determine whether planting in clusters promotes better long-term survival of the remaining plants.

Tree establishment procedures and timing of planting

Much establishment failure is not so much due to the province's environment, as to failure by planters to follow a number of basic tree establishment procedures. Plants were planted according to the recommended method of Porteous (1993), with reference to the guidelines for establishing trees and shrubs in rural Canterbury (Giller 1994), and therefore were afforded the best possible establishment process. Some local surveys have suggested that possibly up to 50% of trees (excluding forestry) die within 2 years of planting and some parts of Canterbury (especially those with light stoney soils) are simply too harsh for most broadleaved species (Giller 1994). The results of the Leeston trial correspond well here, in that plant survival for most or all of the individuals of approximately half the species planted was zero after one year. Of the remaining species, approximately 50% of all individuals planted were surviving after one year. Generally, greater species diversity is associated with better soils and/or higher rainfalls than Canterbury provides but once shelter is developed (e.g., Riccarton Bush) diversity in species number and community structure can occur.

The saplings planted in this experiment may not have been of sufficient size to determine whether planting density is affecting growth and survival. Further measurements need to be taken in the future to assess this factor. Weed mat could also have been used to suppress weed growth. Ogle (1996) found significantly greater growth and survival in plots with weed mat compared with plots that had hand weeding or no weeding.

The timing of planting can also have a bearing on establishment success. On heavy soils in cold and exposed sites, planting should be carried out in early spring (Giller 1994). Most planting at Leeston was carried out at this time, therefore plant survival is considered to be at its optimum as far as timing of planting is concerned.

Broad- versus narrow-leaved species

Broad-leaved species (*Coprosma robusta, Griselinia littoralis, Aristotelia serrata, Melicytus ramiflorus, Pseudopanax arboreus*) survival in the Leeston experiment was poor. This could have been due to the light stoney soils, but extreme winds could also have been responsible. Frost was also most likely to have been a factor causing mortality. *A. serrata* and *M.*

ramiflorus were damaged by frost. The Leeston site is very exposed to high light intensity (including UV) and hot north-west wind and north-east wind in summer and to the cold southerly wind in winter; leaves of *C. robusta, A. serrata, M. ramiflorus, P. arboreus* and *G. littoralis* exhibited damage from wind. Neither vertebrate or invertebrate herbivory was observed throughout the duration of the study, except for some invertebrate (of unknown species) herbivory on *G. littoralis*. The most successful species planted in the open paddock were *P. regius, H. angustifolia, C. australis* and *H. salicifolia*, which are all narrow-leaved.

Fertiliser

Fertiliser may have had an effect on plant growth and survival. If there were more replicates, or a more sensitive measurement than just survival was used, a more definitive statement perhaps could be made. Further measurements need to be taken in the future to assess this factor. Different fertilisers e.g., organic compost or blood and bone could have been trialed to detect which give the best results in Canterbury conditions.

Mulch

In year one, straw mulch is high in carbon, but low in nutrients (P. Hart pers. comm.). As soil microbes and macrobes begin to denature the straw in year two and use the soil nutrients in the process, this could in turn cause a nutrient deficiency for the plants which results in low growth in year two. Straw rub damage was also observed on the stems of *P. regius, P. totara, H. angustifolia* and *K. ericoides*; the latter may have been minimised by staking.

Environmental conditions

The prime cause of plant death in this area was almost certainly a result of long periods of hot, dry, north-westerly winds (67% of the wind directions are NW in Canterbury in summer; V. Keesing pers.comm), although it was not tested however. There is also little or no rain during summer (on average 15-16mm / month which results in approximately 24 days of every summer month being a "dry soil day" (Cherry 1997)).

Area 2 - Open paddock; single species (*H. salicifolia*) planting

The use of mulch and fertiliser for *H. salicifolia* planted in the open habitats was initially beneficial for establishment. It enhanced soil water content by as much as 5-6 mg of water per gram of soil (doubling the surface soil's water content), reduced variation in soil temperature, and suppressed weed growth, although these benefits did not lead to a statistically significant increase in survival in the mixed-indigenous plots of Area 1. However, this *H. salicifolia* trial showed that after the first year, the effects of mulch type and fertiliser became complex to interpret. Growth and survival appeared to benefit from the treatments over the first year, but appeared to be disadvantaged by those same treatments through the second year. One reason may be the different rate of breakdown of mulch types, and the consequent timing of nutrient release. Straw which breaks down quickly ceases to have any beneficial influence after one year, while bark mulch, for a period , may initially absorb nitrogen from the soil, possibly suppressing plant development (Pliszaka and Scibisz, 1989) and only slowly returns this and more nutrients (especially calcium) to the soil as it decays (see below); however, it does maintain its moisture-retaining functions and soil temperature regulation much longer than does straw mulch.

In Area 2, plant growth and health were better with straw mulch and fertiliser than with bark mulch or no mulch treatments in the first year. This result initially supported the idea that particular mulch types and fertiliser can enhance growth and health of indigenous species (cf. Bradshaw & Chadwick 1980), and, at least partly, moderate competition from other species. However, after two years the straw mulch had largely decomposed.

After one year, clumped *H. salicifolia* plants with fertiliser were larger and healthier than those in clumps without fertiliser. Single *H. salicifolia* plants with fertiliser also grew, in one year, more than single plants without fertiliser. Single plants, with or without fertiliser, all grew better than did any clumped plants. This suggests that clumping increased nutrient demand beyond that which the unimproved soil could meet, and supports the use of fertilisers.

By the end of the second year the fertiliser's effects had diminished and the clumped plants which had received fertiliser were not performing better than those in clumps which had not received fertiliser. Those with bark mulch had grown most in height and were in better health than were other clumped plants. It may be that those plants that received fertiliser may actually have been disadvantaged (see Pliszaka and Scibisz, 1989). Those with added, readily available nutrients may have had less-established root systems and greater foliage development (Ayuso *et al.*, 1996) than those under low-nutrient conditions, which may have developed their root system at the expense of foliage development.

The planted seedlings may not have been of sufficient size to determine whether planting density is affecting growth and survival. Further measurements need to be taken in the future to assess this factor. Weed mat could also have been used to suppress weed growth. Ogle (1996) found significantly greater growth and survival in plots with weedmat compared with plots which received hand weeding or no weeding. Some farmers in the Leeston area use split-open plastic fertiliser sacks as mulch around newly-planted native trees on farmland (T.Chamberlain pers. comm.)

Area 3 - *Salix*, plum shelter; mixed mid-late successional species planting

Planting native woody vegetation under existing tree cover enhances the survival of the broad-leaved species. This is likely to be through moderation of wind desiccation and frost. The broadleaved species did better in Area 3 than in Area 1. This is probably due to the shelter afforded by the willow/plum trees in this area. *G. littoralis* (Clarkson & Clarkson 1995), *P. eugenioides* and *P. tenuifolium* (Bannister *et al.* 1996) have each been described as frost tolerant. *M. ramiflorus* is not frost tolerant and possibly does not compete well (like *D. dacrydioides)* with weeds. Some *M. ramiflorus* mortality is likely to be due to the weed competition. Individual plants could have been secured by stakes to detect whether this ensures better success. Many species also suffered more herbivory in this area than in the open paddock, both by vertebrates and invertebrates.

Kunzea ericoides plays an important seral role and dominates early phases of succession in forested regions and usually occurs on well-drained soil (e.g., see Burrell 1965). It requires good drainage (Burrell 1965, Cook *et al.* 1980) and does not tolerate waterlogging. *K. ericoides* seedlings are probably more tolerant than *Leptospermum scoparium* (manuka) of shade and root competition.

Some *K. ericoides* mortality in this area is likely to be due to herbivory by rabbits (V. Keesing pers. comm.). Also, it is not shade-tolerant and in some places the willow/plum canopy may provide too much shade for survival. Some *P. arboreus* mortality is likely to be due to herbivory by possums (personal observation). Possums have also damaged some trunks of *A. serrata*. Some *D. dacrydioides*, and *P. totara* mortality is likely to be due to herbivory by hares (V. Keesing pers. comm). Some *A. serrata, M. ramiflorus, G. littoralis* and *P. arboreus* mortality is likely to be due to lepidopteran (unknown species) herbivory (V. Keesing pers. comm.).

Podocarpus totara did well under the *Prunus* canopy in bare soil, but not under the *Salix* canopy in long grass. This could have been due to nutrient, moisture or light competition from weeds.

Temperatures and soil moisture

Temperatures and soil moisture levels were measured throughout the field trial, but no significant effects were found. The ground frosts of winter 1997 (Cherry 1997) and the el niño summer of 1997/98 with frequent north-west winds, will have contributed to plant mortality, especially in the open pasture.

There would have been a moderation of these wind and frost effects under the willow canopy, but these effects were not tested.

Other factors

Mycorrhiza may have some importance in restoration, particularly on disturbed ground (see Hall 1977), but this was not investigated in this experiment. Competition from weeds and vertebrates have a major influence on growth and survival (e.g., see Ogle 1996), although was not tested here, and the effect of invertebrates also requires further evaluation. Exclosure plots to detect the effects of vertebrate herbivory could have been used in Area 3.

The results of this study will contribute to the large amount of information available to assist with successful management of fragments of regenerating native vegetation in lowland Canterbury.

Chapter Five

General discussion

Introduction

Restoration means recreating both the structural and functional attributes of a modified, disturbed or lost ecosystem. This requires a reconstruction of ecological complexity based on a synthesis of knowledge. Restoration aimed at a predisturbance condition is not frequently a viable option, because ecosystems are the product of climatological and biogeochemical sequences that are probably unique. The predisturbance condition is therefore seldom known or if it is, is unattainable. All restoration efforts are exercises in approximation and in the reconstruction of assemblages of organisms and their predisturbance chemical/physical environments. The aim of these projects is to enhance self sustaining ecological processes. Restoration ecology, although a practice with a long history, is now emerging as a recognised discipline (Pickett & Parker 1994). The status of theoretical science for restoration ecology - that is, the development and testing of a body of theory for repairing damaged ecosystems - at the landscape level is far from robust (Atkinson 1990). However, ecological restoration efforts may be ideal for testing important hypotheses in unique ways. For example, large-scale experimentation, including manipulations, may be more acceptable at restoration sites than in pristine settings. Further, interpretation of experimental outcomes may be easier because restoration sites often harbour simpler communities (Palmer *et al.* 1997). The practicality of theoretical restoration science will remain uncertain, until predictive models have been developed and validated. For full benefit to be derived from restoration, experimental results need to be analysed over the long-term and the models adjusted (Pickett & Parker 1994). This uncertain situation will not change quickly, because of both the temporal and spatial dimensions of restoration efforts (Cairns 1991). Therefore, every restoration project should be used to improve the status of both theoretical and applied aspects of the science. This could be achieved by the documentation of the methods, results, conclusions and recommendations from these projects.

The potential value of ecological restoration as a test of basic ecological theory has been previously highlighted. A feature of the recent upsurge in tree planting and revegetation is that they are increasingly prescribed within both catchment and farm plans to achieve multiple objectives and to complement agricultural land use. It is now not unusual for local groups to undertake projects for protecting remnants of particular ecosystems or vegetated riparian margins or to create bush corridors and to tackle causes of vegetation degradation such as destructive land uses, fire or disease. Given the increasing commitment to catchment planning and management, there should be an increase in ecological reconstruction. If local commitment to landcare is strong, a strategy of facilitating group empowerment to ensure that development will continue towards the ultimate goal of sustainable land use. It is predicted that the total landscape approach being adopted by some landcare groups in their catchment planning will provide "grass roots" support to reconstruction of ecological values (Goss & Chatfield 1993).

Relevance of this study

Habitat fragmentation has been identified as one of the greatest of all threats to biodiversity (Saunders *et al.*, 1991). In Canterbury, this is of great concern as the indigenous vegetation is highly fragmented. Northern New Zealand forest fragments of smaller than one hectare do not support forest interior conditions or vegetation associations (Young & Mitchell 1994); it is necessary to have approximately 9ha to produce forest interior conditions. However, restoration improves the overall viability of habitat generally, such as the interlinking of existing fragments, so small areas still have value.

It can be argued that successful restoration of a disturbed ecosystem is an "*acid test*" of our understanding of that system (Bradshaw 1987). In this reconstruction, attention can be paid to whatever theories are thought to be important, and individual components or functions considered to be critical can be incorporated. The value of restoration is a test of ecological understanding i.e., what determines the development of ecosystems from very skeletal beginnings and what may restrict it? The essential quality of restoration is that it is an attempt artificially to overcome the factors that are considered to restrict ecosystem development. This provides a powerful opportunity to test in practice the understanding of ecosystem development and functioning. Species life history attributes, such as dispersal abilities and length of time plant seed remains viable, will influence both plant and animal composition at the landscape scale and must be taken into account in any attempt to model overall biodiversity patterns (Allen *et al.* 1995b).

Since Europeans settled in New Zealand, many vegetation associations have been severely diminished and there is a need to regenerate buffer zone areas around small reserves, connect wildlife corridors, and to reconstruct endangered habitats. Attracting native seed-dispersers into regeneration areas assists with conservation of species and habitats, and is cost-effective (Green 1995). Of all New Zealand's major cities, Christchurch is by far the poorest in terms of the numbers and species richness of native "bush" birds living within the urban area (Crossland 1996). Had it not been for the naturalisation of silvereyes and blackbirds last century, the link between birds and seed dispersal in Banks Peninsula forests would now be very tenuous (Burrows 1994b). Colonisation is both deterministic and stochastic. Many sites left to colonise naturally are of considerable ecological value (Bradshaw 1983). Primary succession tends to be slower than the more rapid processes of secondary succession. These are two distinctive aspects to succession which must be considered separately.

Future research

Tai Tapu

Monitoring should be continued to see if the *Salix* spp. - native succession fits the facilitation model of Connell & Slatyer (1977). This describes how later (native) plants can become established and grow only after earlier ones (*Salix* spp.) have suitably modified the conditions i.e., had provided perch sites. Connell & Slatyer (1977) proposed three alternative models of succession: tolerance, inhibition and facilitation. The tolerance model is characterised by floristic changes that are a result of different life history traits and the different abilities of later successional species to tolerate the initial environmental conditions. The inhibition model is characterised by the initial invaders acting as regulators of succession, so that later successional species cannot invade and grow in their presence. The facilitation model is characterised by early successional species modifying their environment and thus facilitating the establishment of later successional species.

The management of remnant patches of native vegetation in Canterbury requires an integrated effort to understand the dynamics of the system. A larger scale study, similar in intent to the present work would highlight the temporal and spatial processes operating both within and between sites. To understand fully the implications of these island communities for restoration, baseline data on processes of colonisation (by plants and animals) is a useful starting point. This research has shown that small land areas are important. For land considered for restoration potential in the broader context, small areas should therefore not be disregarded because of their size. They can be a potential source of viable habitat and later a viable seed source for future dispersal into new habitats. Over time, restoration of small isolated patches into connecting islands can produce a more biologically valuable asset to the region as well as being functional and an aesthetic component of the landscape. Long-term spatial and temporal research is required because of the nature of processes operating here. It would be interesting to thin the *Salix* stand to examine the early effects of thinning on understorey plant dynamics compared to adjacent, unmanipulated sections. Some enhancement work on the Tai Tapu wetlands is being supported by Canterbury Regional Council. The female *Salix* spp. will be identified and removed. This will curb the spread of seedlings. Some planting of native species around the edges of the wetland will take place as an enhancement measure. This is designed to be a Turning Point 2000 project. The results of this study could be compared with other sites of native regeneration under willow, for example along the Waimakiriri river banks.

Documentation of seedlings in Tai Tapu in 1999 would probably reveal high mortality due to the el niño summer of 1997/98 and la niña summer of 1998/99, which caused extremely dry conditions in the region.

Leeston

Results from this initial work have shown the importance of selecting the right species to plant for local restoration projects. If the goal of restoration is to re-establish or improve the ecological status of lost and damaged indigenous plant and animal communities, then the animal components in this system need to be encouraged. It is universally acknowledged that high (invertebrate/ vertebrate) community diversity is generally associated with mid-late stages of vegetation succession. Often in restoration projects, because the late successional species are omitted in restoration planting, early successional (pioneer) species grow to create a shrubland only. In order to accelerate the natural successional process in this experiment, and to increase the plant components of the community (which hopefully will in turn provide habitat for invertebrates and vertebrates), there will be later plantings of *Coprosma* spp., *Cordyline*, *Pittosporum*, *Plagianthus*, *Kunzea* etc. The objectives of a longer term study at the site are to investigate invertebrate dynamics and possibly introduce some local species. For instance, two Banks Peninsula species of tree weta (*Hemideina* spp.) inhabit *Hoheria angustifolia, Kunzea ericoides* and seven broadleaf species (*Hemideina ricta* Hutton) (Townsend *et al.* 1997). The other weta *Hemideina femorata* Hutton, occupies holes in *Kunzea ericoides* trunks. Invertebrates such as these could potentially occupy the Leeston site if local conditions were appropriate.

Management recommendations

Planting plans for habitat restoration projects at several Christchurch City parks or reserves have already been prepared (Meurk 1993) and this piece of work can be integrated into these programmes. These results with other information from Landcare Research, Canterbury Regional Council, Department of Conservation, Canterbury University, Lincoln University, local landscape architects and others, provides information for potential restoration projects in Canterbury. Naturally established plants are nearly always faster growing than are planted individuals, therefore regeneration can be costly and time-consuming, especially on difficult terrain or in remote areas. This has been shown in the Leeston experiments, especially those conducted in the open paddocks. Encouraging native or non-native animals to assist in the dissemination of seeds will enhance the cost-effectiveness of restoration schemes, with the main cost being to exclude browsing mammals and possibly control certain weeds. Ecological restoration requires an integrated approach, focussing on identifying and understanding the component processes of regeneration. This research will contribute to guidelines for the recognition of “useful" (willow, for example) communities where colonisation into these vulnerable areas is of benefit to the region. "Useful" in this sense means having the ability to be colonised. It will therefore contribute towards the targets being met by Regional Councils, Department of Conservation etc. in their Resource Management Act (1991) requirements to assess the impacts of regeneration on biodiversity and ecosystem processes.

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Appendix 1 - Native woody plant species in Study site 1 - Tai Tapu *Salix* spp. site

1. *Dacrycarpus dacrydioides* (A. Rich.) Laubenf. - kahikatea

2. *Podocarpus totara* D.Don in Lamb. - totara

3. *Pseudopanax ferox* Kirk - lancewood, horoeka

4. *Pseudopanax arboreus* (Murray) Philipson - fivefinger, whauwhaupaku

5. *Melicytus ramiflorus* J.R. & G.Forst. - māhoe, whiteywood

6. *Griselinia littoralis* Raoul - broadleaf, pāpāuma

7. *Pittosporum tenuifolium* Sol.ex Gaertn. - kohuhu

8. *Pittosporum eugenioides* A.Cunn. - tarata, lemonwood

9. *Myrsine divaricata* A.Cunn - weeping matipo

10. *Myrsine australis* (A.Rich.) Allan - red matipo, māpou, māpau

11. *Cordyline australis* (G.Forst.) Endl. tī, cabbage tree

12. *Myoporum laetum* G.Forst - ngaio

13. *Coprosma robusta* Raoul - karamū

14. *Coprosma crassifolia* Col.

15. *Coprosma rhamnoides* A.Cunn.

16. *Coprosma rotundifolia* A. Cunn. - round leaved coprosma

17. *Coprosma virescens* Petrie

18. *Coprosma propinqua* A.Cunn. - mikimiki

19. *Pennantia corymbosa* J.R.& G.Forst. - kaikomako

20. *Pseudowintera colorata* (Raoul) Dandy - horopito, pepper tree

21. *Muehlenbeckia australis* (G.Forst.) Meissn.

22. *Phormium tenax* J.R.& G.Forst. - harakeke, New Zealand flax

23. *Prumnopitys taxifolia* (D.Don) Laubenf. - matai

24. *Urtica linearifolia* (Hook.f) Cockayne

25. *Parsonsia heterophylla* (G.Forst.) R.Br.- akakaikiore, NZ jasmine

Non-native woody plant species occurring in Study site 1 - Tai Tapu *Salix* site

1. *Salix cinerea* L. - grey willow

2. *Salix fragilis* L. - crack willow

3. *Sambucus nigra* L. - elder

4. *Rosa rubiginosa* L.- briar

5. *Trachycarpus fortunei* - fan palm

6. *Crataegus monogyna* Jacq. - hawthorn

7. *Hedera helix* L.- ivy

8. *Lycium ferocissimum* Miers - boxthorn

9. *Populus nigra cv Italica* - Lombardy poplar

Appendix 2 - Data collection sheet for study site 1

Observer / date

Weather

Species	quadrat	colour	height	matrix	m from willow	shade

Appendix 3 - Seed germination from study site 1: glasshouse experiment

Plot	Species	Authority	Number
4.10.96	*Callitriche stagnalis*		
5.1	*Holcus lanatus*		
5.3	*Muehlenbeckia australis*		17
5.5	*Muehlenbeckia australis*		10
	Cordyline australis		1
5.7	*Muehlenbeckia australis*		1
	Sambucus nigra		12
6.2			
6.4	*Muehlenbeckia australis*		2
	Cordyline australis		1
6.6	*Myoporum laetum*		1
6.8	*Coprosma* sp.		7
	Sambucus nigra		4
Plot 10.1.97			
5.1	*Callitriche stagnalis*		
	Holcus lanatus		
	Ranunculus sceleratus		
	Ranunculus repens		
	Poa sp.		
	Juncus articulatus		
	Epilobium billardiereanum		
	Cotula coronopifolia		

Plot	Species	Authority	Number
5.3	*Muehlenbeckia australis*		19
	Cordyline australis		1
	Cerastium glomeratum	Thuill.	
	Mimulus guttatus	DC.	
	Solanum dulcamara		
	Ranunculus sceleratus		
	Callitriche stagnalis		
	Rumex obtusifolius		dock
	Juncus articulatus		
	Epilobium billardiereanum		
	Lolium perenne	L.	
5.5	*Muehlenbeckia australis*		11
	Cordyline australis		3
	Mimulus guttatus		
	Ranunculus sceleratus		
	Solanum dulcamara		
	Cirsium vulgare		
	Juncus articulatus		
	Rumex obtusifolius		

Plot	Species	Authority	Number
5.7	*Sambucus nigra*	L.	
	Mimulus guttatus		
	Muehlenbeckia australis		
	Stellaria media		Chickwee
	Epilobium billardiereanum		d
	Cirsium vulgare		
	Solanum dulcamara		
	Portulacca (?) c/f		2
	Holcus lanatus		
	Ranunculus sceleratus		
	Callitriche stagnalis		
	Juncus articulatus		
	Eleocharis		
	Triglochin		
	Lycium ferocissimum		
6.2	*Callitriche stagnalis*		
	Holcus lanatus		
	Juncus articulatus		
6.4	*Cordyline australis*		5
	Sambucus nigra		
	Juncus articulatus		
	Callitriche stagnalis		
	Solanum dulcamara		
	Mimulus guttatus		
	Ranunculus sceleratus		
	Holcus lanatus		
	Cirsium vulgare		
	Portulacca (?) c/f		
	Muehlenbeckia australis		2
6.6	*Sambucus nigra*		3
	Rumex obtusifolius		1
	Myoporum laetum	G.Forst.	1

Plot	Species	Authority	Number
6.8	*Cordyline australis*		
	Coprosma virescens		8
	Epilobium billardiereanum		
	Cirsium vulgare		
	Solanum dulcamara		
	Ranunculus sceleratus		
	Mimulus guttatus		
	Portulacca (?) c/f		
	Lolium perenne		
Paddock	*Holcus lanatus*		
1	*Rumex obtusifolius*		
	Trifolium repens	L.	
	Cirsium arvense		
	Trifolium fragiferum	L.	
	Ranunculus repens		
	Chenopodium album	L.	
	Anagallis arvensis	L.	
	Lolium perenne	L.	
2	*Chenopodium album*		
	Trifolium repens		
	Rumex obtusifolius		
	Polygomun sp.		
	Hypochoeris radicata		
	Galium aparine		
	Cirsium vulgare	spear/scotch thistle	
	Ranunculus repens		
	Elytrigia repens		

Plot	Species	Authority	Number
3.	*Agrostis capillaris*	brown top	
	Holcus lanatus		
	Trifolium repens		
	Cirsium arvense		
	Cerastium glomeratum		
	Rumex obtusifolius		
	Lolium perenne	L.	
	Trifolium dubium		
	Hypochoeris radicata		
4	*Agrostis capillaris*		
	Cirsium arvense		
	Trifolium repens		
	Ranunculus repens		
	Trifolium fragiferum		
	Cirsium vulgare		
	Lolium perenne	L.	
	Chenopodium album		
	Stellaria media		

Appendix 4 - Plant species planted at Leeston (Study site 2, Areas 1, 2 and 3)

1. *Dacrycarpus dacrydioides* (A. Rich.) Laubenf. - kahikatea

2. *Podocarpus totara* D.Don in Lamb. - totara

3. *Pseudopanax arboreus* (Murray) Philipson - fivefinger, whauwhaupaku

4. *Melicytus ramiflorus* J.R. & G.Forst. - māhoe, whiteywood

5. *Griselinia littoralis* Raoul - broadleaf, pāpāuma

6. *Pittosporum eugenioides* A.Cunn. - tarata, lemonwood

7. *Cordyline australis* (G.Forst.) Endl. tī, cabbage tree

8. *Coprosma robusta* Raoul - karamū

9. *Hoheria angustifolia* Raoul - houhere, narrow-leaved lacebark

10. *Kunzea ericoides* (A. Rich.) J. Thompson - kānuka

11. *Aristotelia serrata* - (J.R. & G.Forst.) - mako mako, wineberry

12. *Plagianthus regius* - (Poit.) Hochr. - manatu, lowland ribbonwood

13. *Hebe salicifolia* - (Forst.f.) Pennell - koromiko

Appendix 5 - Time weighted measures of flooding for quadrats

Quadrat	Flooding index (time-weighted measure of the degree of inundation)	Quadrat	Flooding index (time-weighted measure of the degree of inundation)
1.3	0.092	4.5	0.046
1.5	0.322	4.1	0.046
1.6	0.442	5.3	0.046
1.7	0.462	5.2	0.046
1.8	0.226	5.6	0.046
2.3	0.092	8.4	0.046
2.4	0.172	8.8	0.046
2.5	0.822	7.9	0.091
2.6	0.886	6.4	0.091
2.7	0.576	6.5	0.091
4.1	0.046	2.3	0.092
4.3	0.567	1.3	0.092
4.4	0.349	7.8	0.104
4.5	0.046	6.2	0.137
5.1	0.183	2.4	0.172
5.2	0.046	7.7	0.183
5.3	0.046	5.1	0.183
5.6	0.046	1.8	0.226
5.8	0.809	1.5	0.322
6.2	0.137	4.4	0.349
6.4	0.091	1.6	0.442
6.5	0.091	6.9	0.453
6.9	0.453	1.7	0.462
6.10	0.715	4.3	0.567
7.7	0.183	2.7	0.576
7.8	0.104	6.10	0.715
7.9	0.091	5.8	0.809
8.4	0.046	2.5	0.822
8.8	0.046	2.6	0.886

Appendix 6: Bird species documented from Study site 1: Tai Tapu

Species	Common name	Authority
Sturnus vulgaris	Starling	Linnaeus
Alauda arvensis	Skylark	Linnaeus
Turdus philomelos clarkei	Song thrush	Brehm
Larus dominicanus	Black-backed gull	Lichtenstein
Porphyrio porphyrio	Pukeko	Linnaeus
Carduelis carduelis	Goldfinch	Linnaeus
Turdus merula	Blackbird	Linnaeus
Zosterops lateralis lateralis	Silvereye	Latham
Circus approximans	Harrier	Peale
Gerygone igata	Grey warbler	Quoy & Gaimard
Anas platyrynchos	Mallard duck	Linnaeus
Vanellus miles novaehollandiae	Spur-winged plover	Stephens
Fringilla coelebs	Chaffinch	Linnaeus
Gymnorhina tibicen hypoleuca	Australian magpie	Latham
Emberiza citrinella	Yellowhammer	Linnaeus
Larus novaehollandiae	Red-billed gull	Stephens
Passer domesticus	Sparrow	Linnaeus
Carduelis flammea	Red poll	Linnaeus
Hirundo tahitica neoxena	Swallow	Gould
Carduelis chloris	Greenfinch	Linnaeus
Rhipidura fuliginosa fuliginosa	Fantail	Sparrman
Anthornis melanura	Bellbird	Sparrman
Branta canadensis maxima	Canada goose	Delacour