Natural colonisation and rehabilitation of a copper smelter desert, Mount Lyell, western Tasmania

By
Roger Dawson (BSc.Hons.)

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Declaration

This thesis contains no material that has been accepted for the award of any other degree or diploma, nor any material previously published or written by another person except when due reference is made.

Roger Dawson
25/3/97

Abstract

This thesis investigates patterns of soil contamination and natural colonisation in an area deforested by a long-established, but decommissioned, copper smelter at Mount Lyell in western Tasmania. The rehabilitation of this area is investigated by modifying colonisation and species performance in relation to edaphic factors in accordance with succession management models.

The distribution of soluble metals in soils along transects of increasing, downwind displacement from the smelter stacks provided evidence of Cu/Zn contamination and Al mobilization. A higher than expected soil pH suggested a partial reversal of the process of soil acidification by SO_2 deposition. This may, in part, explain relatively low, near-smelter contamination levels at Mount Lyell in comparison to similar sites in the vicinity of smelters of world-renown. Nevertheless, colonising species grown in Mount Lyell soils exhibited severe growth abnormalities and these were linked to elevated water-extractable metal concentrations.

A survey of the vascular flora, and multivariate analysis, were used to describe the composition and distribution of vegetation colonising the denuded western slopes of Mount Lyell. Compositional trends were explored in relation to edaphic factors by vector fitting, and a numerical classification of survey sites was used to group sites by characterising species coincidences. The classification was used to describe and map spatially distinct colonising communities that differed demonstrably in their tolerance to the phytotoxic metals present in the soils. Species with differential tolerances to the contaminants were identified. The naturalised, exotic grass *Agrostis capillaris,* was a wide-spread, metal-tolerant colonising species. In contrast, many local trees and shrubs were intolerant of the extreme site conditions. A strong spatial correspondence was evident between the colonising communities and contamination patterns in the vicinity of the smelter installations.

Rehabilitation methods for severely eroded sites adjacent to the original smelters were investigated. Emphasis was placed on broadcast sowing as a means of native species re-introduction. A novel, non-destructive, mechanical seedbed preparation method was compared to a conventional method and was found to provide comparable seedling establishment. Low impact, seedbed preparation and application methods for steep, eroded slopes were also compared and dissimilar methods were found to enhance the establishment of colonising *Acacia* species. The ameliorative effects of field applications of lime on the growth of colonising seedlings were evaluated.

The reclamation of Mount Lyell is discussed with reference to patterns of natural colonisation.

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Table of Contents

Chapter 1

The climate, geology and vegetation of Mount Lyell, the legacy of environmental impacts as a result of copper mining and smelting and the aims and structure of this thesis

Chapter 2

Literature review

Chapter 3

Soil characteristics along transects of increasing displacement from the Mount Lyell smelters

Chapter 4

Environmental factors influencing the occurrence and abundance of colonising taxa within the new community at Mount Lyell

Chapter 5

Amelioration of phytotoxic soil characteristics: the addition of lime

Chapter 6

Seedbed preparation and the use of a mechanical roller-aerator to enhance the lodgment, germination and early survival of broadcast sown seed at an eroded, near-smelter location with phytotoxic soil characteristics

Chapter 7

An assessment of minimum impact, seedbed preparation and seed application methods for the establishment of colonising species on steep, eroded slopes at Mount Lyell

Chapter 8

Conclusions

References

Appendices

List of figures

Chapter 1

Chapter 3

Chapter 4

Chapter 5

Chapter 6

Chapter 7

List of Tables

Chapter 5

Chapter 6

Chapter 7

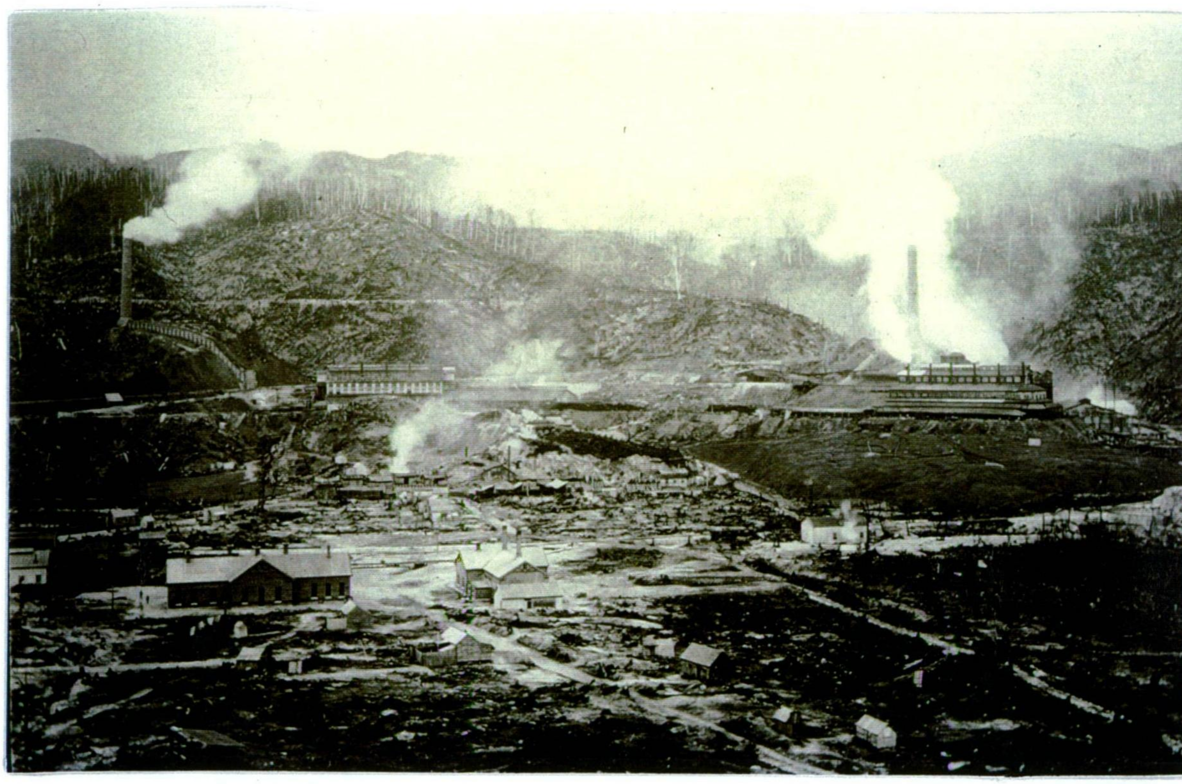
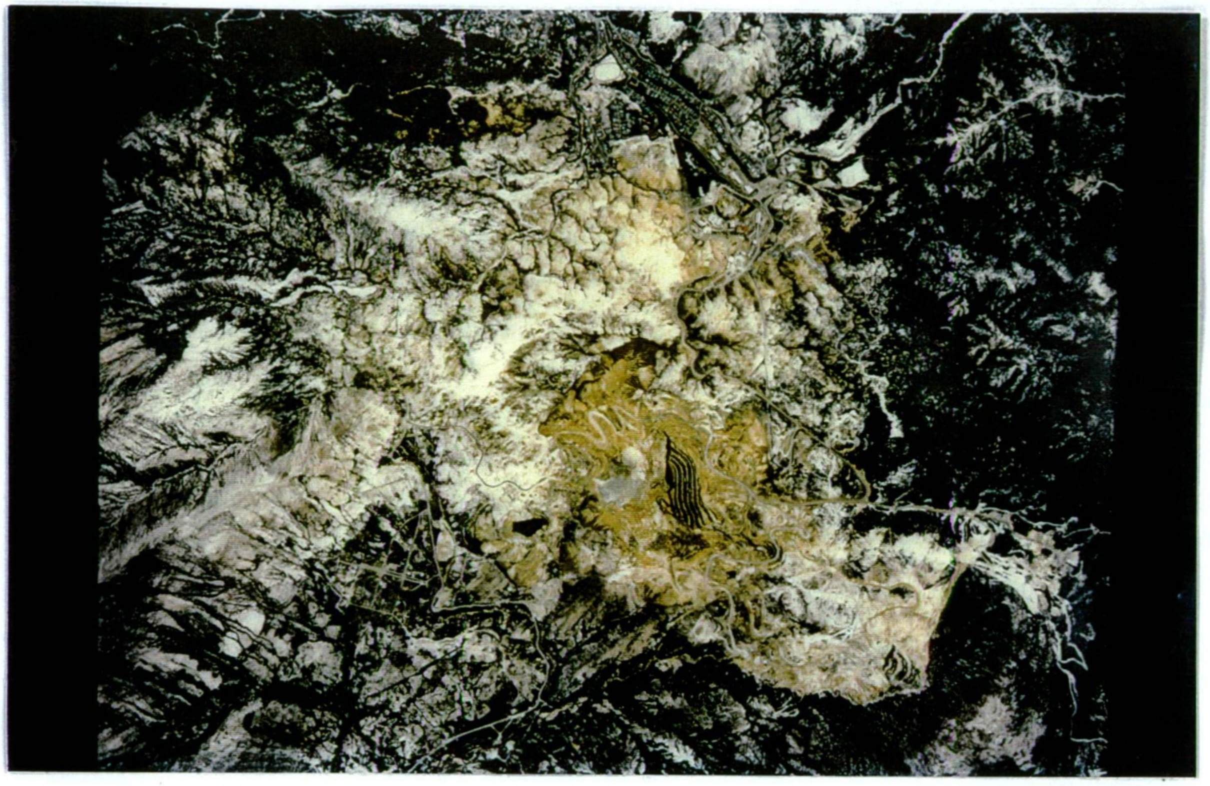


Photo of the Queen River valley depicting the site of the Mount Lyell Mining Company blast furnaces c1899. The deforested lower slopes of Mount Lyell can be seen in the hinterground.



Aerial photograph of Mount Lyell in 1991. Queenstown is situated in the lower-middle of the photograph. The West Lyell open cut and the western slopes of Mount Lyell are located in the centre and centre-top of the photograph, respectively.

Chapter 1

The climate, geology and vegetation of Mount Lyell, the legacy of environmental impacts as a result of copper mining and smelting and the aims and structure of this thesis

1.1 Introduction

Mining began in prehistory as a means of supply of unrefined minerals. Some minerals later developed trade value and were greatly sought. In special cases, such as that of salt, minerals assumed a value equal to that of currency. Subsequently, world population growth, trade and the recognition of new materials, ensured that the demand for minerals, and therefore mining, would expand to most areas of the world.

Mining activities, whether surface or deep, have often resulted in significant, and sometimes spectacular, alterations to a landscape (Bradshaw, 1992). Generally, the significance of these alterations grew in line with demand and the technology-limited scale of the extraction procedure. However, until very recent times concern over permanent, or semi-permanent, alterations to the appearance of landscapes, or the function of their ecosystems, rarely arose. Extraction took place until an ore body was exhausted (Bradshaw, 1992), access became difficult or the market collapsed. Thereafter, a mine was simply abandoned. There were no contingency plans, for example, for mine closure and site rehabilitation.

In terms of the history of mining, the development of a copper mine at Mount Lyell late last century is recent. Never the less, the initial mining methods employed were very much pre-20th century (Blainey, 1993); driven by man, horse or crude steam power; and the mining ethos that of the pioneer and the then, none too distant, Australian gold rush. In this age, when forests were set alight in order to aid prospecting and overburden left where it fell, the maintenance of environmental quality was not an issue (Blainey, 1993). Indeed, control over the environment, even where it produced tangible degradation, such as air or water pollution, was accepted and considered a necessary sign of progress.

Mount Lyell was the subject of unrestrained exploitation for copper. One company, the Mount Lyell Mining and Railway Company (MLMRC), formed in 1893, was to dominate the development of the site. The environmental impact that this exploitation was to have, much of it unrecognised in the early years, was visually complete around 1900 (Mount Lyell Company Museum). In the space of a decade, eucalypt forest and rainforest vegetation on the mountain had been reduced to an eroded graveyard of blackened stumps, bathed in the acrid fumes of a then new, pyritic-smelter works (Blainey, 1993). These developments occurred before environmentalism, and later formalisations such as environmental impact assessments, pollution control regulations and mine closure plans became publicly accepted, and legitimate and unavoidable concerns of the mining industry.

In 1990, the Mount Lyell Mining and Railway Company (MLMRC), in recognition of major environmental concerns, decided to address some of the historic, environmental legacies of one hundred years of copper prospecting, extraction and smelting at Mount Lyell (Falkner, pers. comm.). The Company formed an Environmental Department which - among a raft of serious environmental problems, some of which appeared technically, let alone financially, insoluble, such as acid mine drainage emanating from innumerable mapped and unmapped drains, fissures and addits throughout the mining lease - undertook to redress the most visually apparent environmental legacy of mining at Mount Lyell; the denuded hillsides surrounding the mine and smelter site. These hills were known locally as the 'Queenstown Desert'.

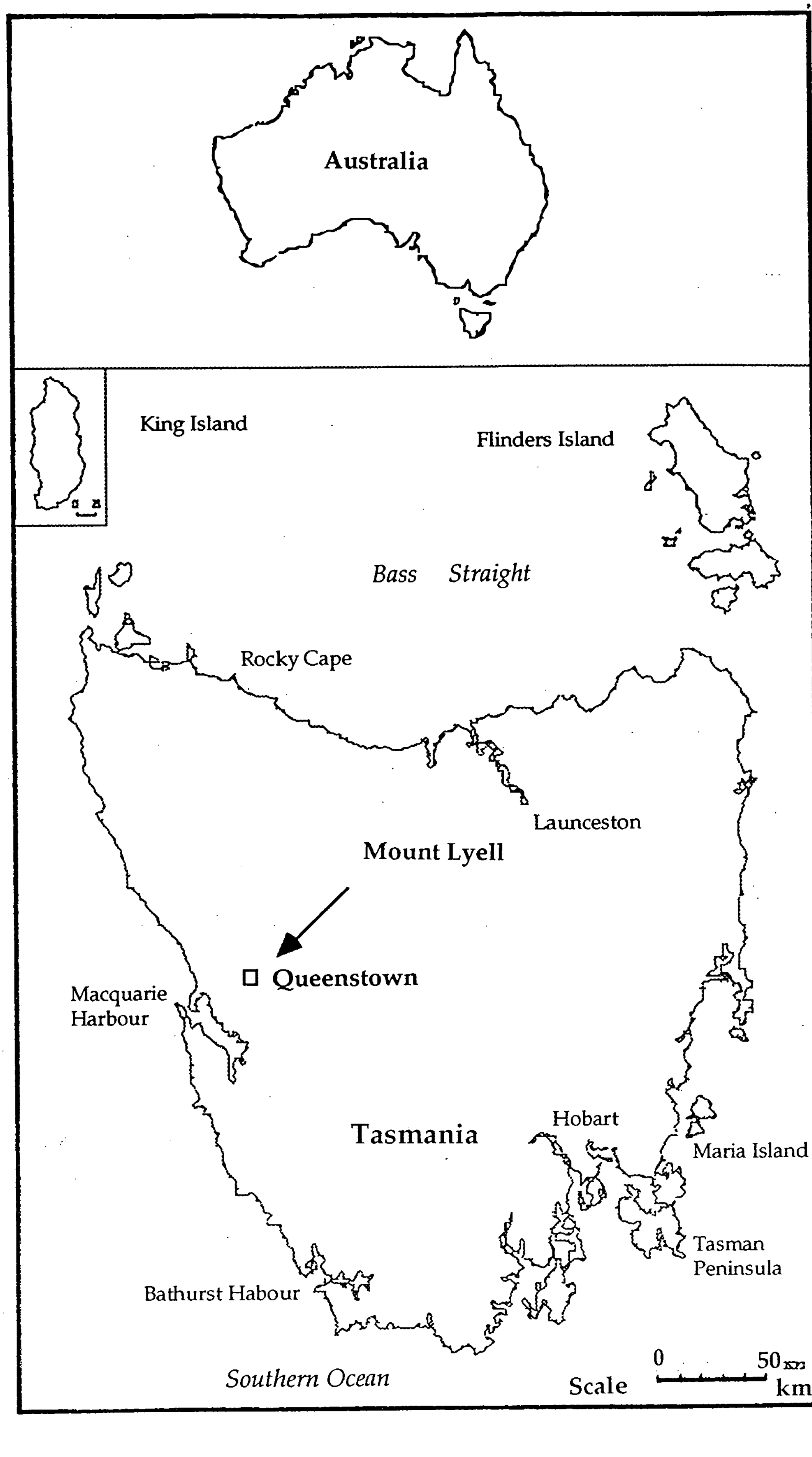
The present study formed a part of the MLMRC's environmental program. The intention of this part of the program was to provide a research-based platform on which the revegetation of the denuded hills could be based. Consequently, this thesis was directed towards an understanding of the vegetation re-colonising Mount Lyell and developing means to accelerate revegetation.

1.2 Location, climate and geology

1.21 Location and climate

Mount Lyell is located 39 kilometres inland from the west coast of Tasmania and a few kilometres to the north east of the present-day, mining township of Queenstown (Fig.1). The mountain separates the catchments of the King River, to the east, from the Queen River to the west. The summit is reached at 917 m. The majority of mine workings are found in the Queen River catchment, on the western face of Mount Lyell, between the 160 m (Queen River) and 550 m contours (Fig. 2).

Fig.1: Map of Tasmania indicating the location of Mount Lyell



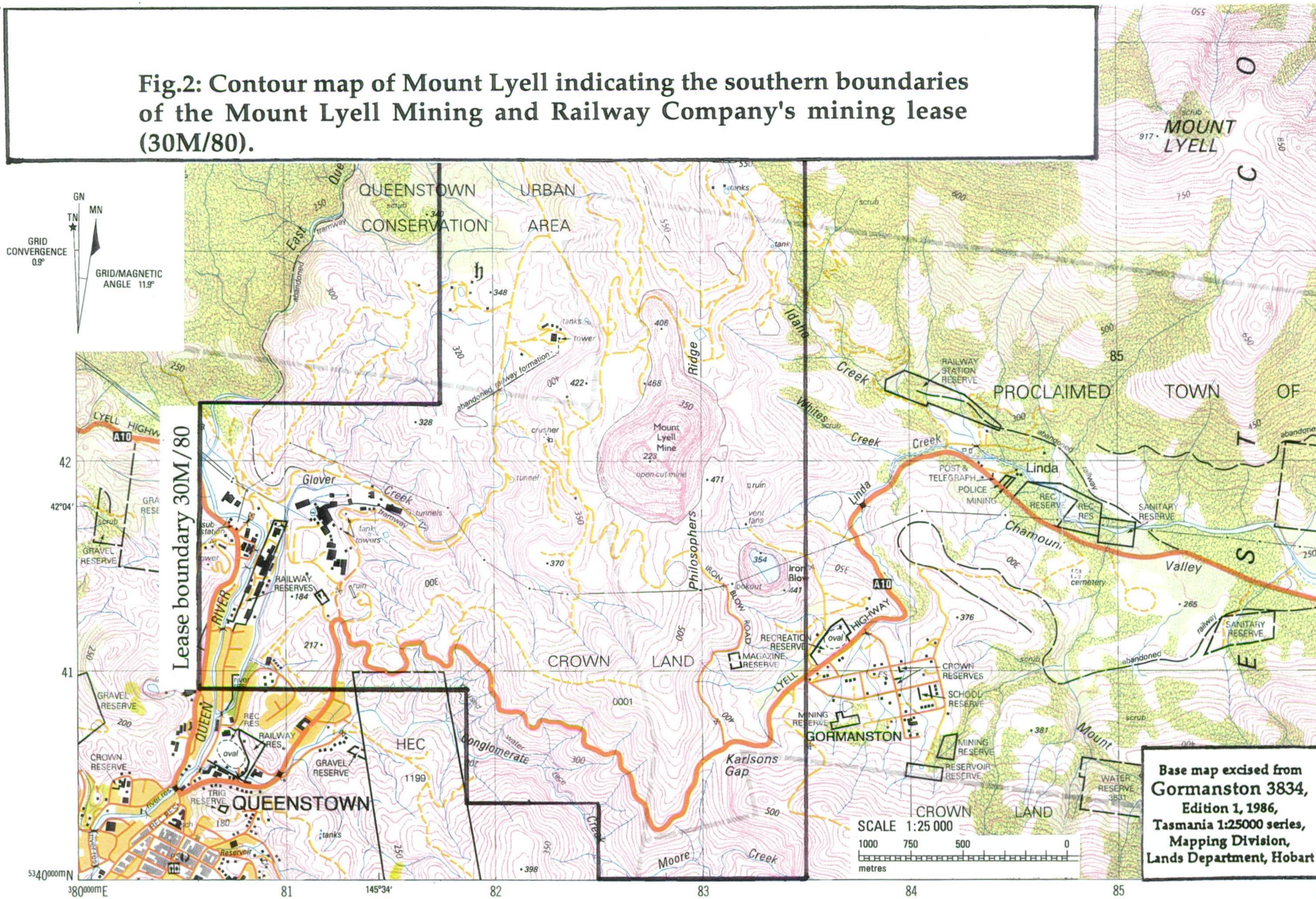


Fig.2: Contour map of Mount Lyell indicating the southern boundaries of the Mount Lyell Mining and Railway Company's mining lease (30M/80).

The temperate maritime climate of the west coast of Tasmania results in a climate at Mount Lyell that is humid with cold winters and mild summers. The warmest month is February and the coldest July. Snow can fall on the mountain in any month above approximately 600 m, but there is no permanent snowline. The nearby town of Queenstown (Lat. 42° 06' S, Long. 145° 33' E), at an altitude of 129 m above sea level, has mean daily maximum temperatures of 22.0 and 11.6 °C in Feburary and July, respectively (Bureau of Meteorology, Hobart). The equivalent mean daily minimum temperatures are 12 and 2 °C.

The town has a long-term, mean annual rainfall of 2517 mm per annum. This increases to approximately 3500 mm with altitude. Rainfall is relatively evenly spread throughout the year (240 days/year) with a Feburary minimum (121 mm) and an August maximum (263 mm). Evaporation may exceed rainfall for only one or two months per year.

1.22 Regional geology

The Dundas Trough is a sedimentary and volcanic region which extends, over a 20-30 km wide strike for 250 km from Elliot Bay, south of Macquarie Harbour, to Deloraine in the north of the State (Green, 1990). Situated near the southern end of this unit is the Cambrian-aged, Mount Read Volcanic belt. This is a sequence composed of silicic lavas, volcaniclastics, massive flow breccias and minor intrusives and dykes (Mount Lyell Technical Review, 1993; Solomon, 1989). Sedimentary lenses occur within the formation.

A major north-south fault, the Henty, divides the Volcanic sequence into two distinct stratigraphic zones. To the north and west of the fault zone is the

Central Volcanic Complex, itself overlain by the mainly andesitic Dundas Group. To the east and south, composed of feldspar-phryic rhyolitic volcanics, is the South Central Volcanic Complex, and the volcano-sedimentry Western Sequence. These latter units are overlain by the Tyndall Group, a sequence of quartz and feldspar-phryic lavas and pyroclastics (Mount Lyell Technical Review, 1993). Mount Lyell, and the area contained within the MLMRC lease, are situated within the South Central Volcanic Complex.

1.23 The geology of Mount Lyell

The geology of Mount Lyell lease is characterised by the contact of two dissimilar surface geologies. The central and eastern areas of the mountain are part of the Owen Conglomerates of Ordivician origin. These conglomerates consist of fine to very coarse siliclastics of mixed fluvial and marine facies (Mount Lyell Technical Review, 1993). The presence of haemitite gives a distinctive pink colouration to much of the unit. A fault, the Great Lyell, running approximately north-south along the western flank of Mount Lyell, divides the conglomerates from the Mount Read Volcanic sequence. The majority of economic deposits have been found adjacent to this fault zone.

Metamorphism, and associated faulting and folding, occurred in the Great Lyell fault zone during the Tabberabberan Orogeny of the Middle Devonian (Mount Lyell Technical Review, 1993; Wade and Solomon, 1958). Ore deposition took place toward the end of this period of alteration (Wade and Solomon, 1958). Since the late 1960's, most geologists have accepted the deposits as subaqueous-exhalative in origin (Mount Lyell Technical Review, 1993).

1.24 Mine mineralogy

The mineral deposits occur in the zone of alteration entirely within the Mount Read Volcanic sequence (Solomon, 1989, Corbett, 1981; Solomon and Elms, 1965; Wade and Solomon, 1958). Schistose pyroclastics, composed mainly of quartz, sericite, chlorite and siderite are host to the deposits. These host rocks are known locally as 'felsic volcanics'. Units of 'intermediate-mafic' and 'mafic' volcanics occur throughout the felsic sequence.

The felsic volcanics consist of fine grained quartz chlorite-sericite rocks. They are variably haemitite-pink to grey in colour. In thin section, the ground mass consists predominately of fine-grained, mosaic textured quartz crossed by a network of sericite±chlorite (Mount Lyell Technical Review, 1993). Pyrite, chalcopyrite, magnetite, monazite and apatite occur as accessory minerals.

The intermediate-mafic and mafic volcanics consist of fine-grained, dark green to grey, chloritic-sericitic-quartz schists. Trace element geochemistry suggests that these are intensely altered andesitic to basaltic volcanics (Mount Lyell Technical Review, 1993). In thin section these rocks are composed of intergrown chlorite and sericite with disseminated quartz, accessory pyrite, magnitite, haematite, monazite and apatite.

The mineral deposits occur as iron and copper sulphides in three main combinations; massive pyrite ($Fe\ S_2$)-chalcopyrite ($Cu\ Fe\ S_2$), disseminated pyrite - chalcopyrite and chalcopyrite-bornite ($Cu_5\ Fe\ S_4$; Solomon, 1989; Corbett, 1981; Solomon and Elms, 1965). The smaller, massive pyrite-chalcopyrite deposits, as found at the Blow, occurred near the top of the mine sequence at relatively high grades. Disseminated chalocpyrite was located stratigraphically beneath the massive sulphide. This larger, lower grade deposit (0.72-2.38% Cu), accounted for approximately 86% of the total ore deposits (Mount Lyell Technical Review, 1993). By 1994, 111 million tonnes of this ore deposit, mostly from the West Lyell open cut and the Prince Lyell underground mines, had been extracted.

1.3 Vegetation

1.31 Tasmanian rainforest classification

The modern form of rainforest in Tasmania is distinct from the other three rainforest types represented in Australia and is described as cool temperate. The accepted definition is that of forest dominated by members of a small suite of 'Antarctic flora' tree species. Jarman and Brown (1983), add that the members of this small group do not require disturbance for regeneration. It exists principally in the western and central areas of the State where it reaches its greatest extent and diversity.

Within Tasmanian rainforest, two main floristic groups are recognised; the Myrtle-Beech alliance, dominated by hardwood and conifer species from the genera *Nothofagus, Eucryphia, Atherosperma, Athrotaxis, Lagarostrobus, Phyllocladus* and/or *Diselma;* and the less extensive, Pencil Pine alliance, dominated by *Athrotaxis cupressoides* (Jarman *et al.*, 1984). The Myrtle alliance, which occurs from the lowlands to about 1000 m (Jarman *et al.*, 1991), is dominated by the species *Nothofagus cunninghamii* on optimal sites (Read, 1991). It is comprised of three sub-alliances which form a floristic and structural continuum; namely, callidendrous, thamnic and implicate (Jarman *et al.*, 1991). These names are without ecological conotation and were chosen to reflect the physiognomy of the groups. These authors recognised further division in each sub-alliance by understory type.

1.32 Aspects of rainforest succession

Succession in Tasmanian rainforest is believed to be controlled by climate, soil fertility and fire (Jackson, 1968; Macphail, 1991). The relative importance of each factor is dependent on time and geographic scale. Accordingly, climate, and to a lesser extent soil fertility, is understood to affect rainforest composition and, ultimately, its regional range, over the long term. Conversely, fire is capable of altering the composition and range of rainforest in the short term.

Tasmanian rainforest is believed to have developed from early post-glacial 'alpine' scrub some 10 000 - 11 000 years ago. Pollen records indicate that the composition of this scrub was dominated by a *Nothofagus, Phyllocladus* and *Eucalyptus* association (Macphail, 1991). Records for the intervening, warm interglacial period to the present day, collated from a variety of sites around Tasmania, indicate that the relative proportions of major rainforest species shifted in both time and space, resulting in a succession of rainforest associations. Based on his understanding of the pollen record for Tasmania, Macphail (1991) concluded that there is not, and is unlikely to be, a stable, floristic entity in Tasmania called temperate rainforest.

Support for the role of soil fertility in rainforest succession comes from Jarman *et al.* (1991). These authors, in an examination of floristic data from western

Tasmanian rainforest sites, concluded that within a climatic region, geology, through its effect on soil fertility, is the most important determinant of variability in rainforests at the level of the three major sub-alliances. They found that fertile sites overwhelmingly supported structurally and floristically simple, Myrtle-dominated, callidendrous forest while, conversely, less fertile sites supported thamnic or implicate communities. These findings were in accordance with those of Jackson (1968) and Kirkpatrick (1977), who reported that the infertile soils of west and south-west Tasmania supported a comparatively complex rainforest community dominated by Myrtle (*Nothofagus cunninghamii*), and Sassafras (*Atherosperma moschatum*) with Leatherwood (*Eucryphia lucida*) and Celery Top pine (*Phyllocladus aspleniifolius*).

1.33 Fire frequency and its influence on the communities of western Tasmania

In comparison to the effects of climate and soil fertility, fire is capable of altering the composition and range of Tasmanian rainforest in the short term. This is due to the relative lack of adaptations to fire displayed by each member of the small group that characterises the rainforest (Jackson, 1968). Whereas a mean fire frequency of about 400 years maintains Tasmanian rainforest (Duncan, 1991), and allows for the successional development of the climax sub-alliances described by Jarman *et al.* (1991), increased fire frequencies result in rainforest replacement or their confinement to fire shadows. Jackson (1968) estimated that rainforest occupies only 23% of its potential range in Tasmania. Elsewhere, fire is believed to maintain the presence of fire-sensitive or fire-dependent vegetation (Jarman *et al.*, 1982; Brown and Podger, 1982; Bowman and Jackson 1981; Kirkpatrick, 1977; Jackson, 1968). Fire-dependent vegetation of this type has been described as being composed of fire disclimax communities (Kirkpatrick, pers comm.).

In general, fire disclimax communities occur when the mean fire interval is reduced below that required by the dominant rainforest species to attain sexual maturity. Where rainforest is excluded, typical fire-dependent communities develop and their composition is strongly related to the interval between fires. A single fire-event in low altitude forest can result in a temporary increase in species richness with the invasion of wind dispersed shrubs and species from adjacent vegetation types (Kirkpatrick, 1977). Subsequent events result in typical, fire-dependent communities. In western Tasmania, rainforest has been replaced over much of its range by eucalypt forest, scrubland and sedgeland; the most frequently burnt, lowland areas being occupied by heathy sedgeland (Kirkpatrick, 1977). These communities, some containing residual or regrowth rainforest species, are themselves understood to be successionally related (Duncan, 1991; Brown and Podger, 1982; Kirkpatrick, 1977). For example, sedgeland maintained by fire on sites suitable for rainforest, may form scrubland when woody species re-invade forming a closed canopy (Duncan, 1991). In the absence of fire scrubland may in turn be invaded by rainforest species. Where subsequent fires occur, *Eucalyptus* species may invade.

Prehistorical evidence of the effect of fire upon rainforest communities can be also found in the pollen record. Based on species disappearances from the record, Macphail (1991), presents an Holocene example of the extinction of Tasmanian endemic conifers and *Nothofagus gunnii* under an increasing fire frequency. The author concludes that local extinction of rainforest species and their replacement by a button-grass sedgeland (*Gymnoschoenus* *sphaerocephalus*), at a high altitude cirque in the Denison Ranges, is symptomatic of the species susceptibility to closely-spaced wildfires.

1.34 King Billy pine and the vegetation of Mount Lyell prior to the arrival of the Europeans

King Billy Pine can be found in all three Tasmanian rainforest sub-alliances - callidendrous, thamnic and implicate (Jarman *et al.*, 1984; Jarman *et al.*, 1991) - as a co-dominant of tall forest with *Nothofagus cunninghamii*, and *Atherosperma moschatum* over a fern understory (callidendrous), a dominant or co-dominant with *Phyllocladus aspleniifolius*, and *Eucryphia* spp. over typical understory species *Agastachys odorata* and *Richea pandanifolia* (implicate) and a dominant in low forest with *Nothofagus cunninghamii*, *Atherosperma moschatum* and *Phyllocladus asplenifolius* over *Agastachys odorata* (implicate). The species is also found in montane forest where, at altitude, it forms low forest with *A. cupresssoides* over *Nothofagus gunnii* and *Richea pandanifolia*.

Evidence of the composition of the pre-European forest at Mount Lyell comes from a variety of sources; relict, photographic and written. In 1895, a two-furnace smelter and ore processing facilities were constructed on a terrace of the Queen river just north of the present day town of Queenstown. Photographs of the construction and early operation of the installations take in the pre-European vegetation of the lower (between 200 to 300 m in altitude), western slopes of Mount Lyell prior to clearance (Mount Lyell Company Museum). Although the photographs are not distinct to species level, they depict rainforest and mixed-forest; the latter over-topped by eucalypt emergents. The emergent trees, confined mainly to the spurs, were probably Smithton Peppermint (*E. nitida*) or Messmate (*E. obliqua*). The former is a widespread Tasmanian endemic referred to in Mount Lyell's historical records (Blainey, 1993). Mixed forest of this type is maintained at fire frequencies between 100-400 years (Duncan, 1991). Blackwood (*A. melanoxylon*) was probably present as an understory species on fire prone sites. In the wetter drainage lines, the photographs record the presence of rainforest. *Nothofagus cunninghamii, Atherosperma moschatum* and *Eucryphia lucida* were undoubtedly present as canopy dominants.

On the middle slopes of the Mountain (250 to 600 m), most notably on its western face, the presence of numerous King Billy pine stumps (*Athrotaxis selaginoides*), representing a range of size classes, attest to the significance of this species in the pre-European forest. The presence of this fire-sensitive species indicates that fire had been absent from the middle slopes of Mount Lyell for some centuries prior to the arrival of European man. King Billy pine probably occurred as a dominant or co-dominant component of implicate rainforest. Likely co-dominant species at this altitude are Sassafras (*Atherosperma moschatum*), Myrtle (*Nothofagus cunninghamii*) and Celery Top pine (*Phyllocladus aspleniifolius*). The early prospectors are recorded as having fashioned sluice boxes from the former in the vicinity of the Iron Blow (Blainey, 1993). Blackwood (*Acacia melanoxylon*) was probably locally abundant in exposed, fire-prone sites.

On the upper slopes of Mount Lyell (above 650 m), King Billy pine is likely to have dominated low, montane forest. Remnants of this forest type, including King Billy pine stags, persist on Mount Lyell today.

1.4 A review of the mining history of Mount Lyell in relation to environmental impacts

1.41 Early exploration and timbergetting

The earliest man-made disturbances to the face of Mount Lyell were those of the geologist Charles Gould, who at the request of the Tasmanian Government, undertook to survey the area between Macquarie Harbour and the Eldon Ranges (Blainey, 1993). In 1862, on his second attempt to locate gold in payable quantities, Gould used the contemporary prospecting method of 'scrub' firing, followed by pit and alluvial sampling, in the Linda Valley. This valley takes its headwaters from a saddle running between the southern slopes of Mount Lyell and adjacent Mount Owen.

The consequences of this early prospecting were doubtless severe on those species not able to withstand fire. In his diary Gould remarked that tea-tree (*Leptospermum* spp.), a readily combustible genus, covered much of the Linda Valley bottom while the upper slopes (presumably of both Mount Lyell and Mount Owen) were barren (Hobart Mercury, 1862). The prospecting fires, which burnt for days under favourable conditions (Blainey, 1993), would have been lit among the scrub and allowed to progress uphill, through rainforest on the middle slopes, to the alpine zone. Fire-sensitive rainforest species, notably the long lived, but slow growing King Billy Pine (*Athrotaxis selaginoides*), would have been killed. Although Gould spent many days sampling in the Linda Valley area, he did not identify the deposit that was to become known as the Iron Blow; a formation with a minor gold and silver deposit. Later excavation of the Blow led to the discovery of the Mount Lyell copper deposits.

Further disturbances to the slopes of Mount Lyell probably did not occur until 1883 when the McDonough brothers and partner Steve Karlson, pegged the Iron Blow and washed sufficient gold from the head of the Linda Valley to cause a minor rush numbering sixty prospectors (Blainey, 1993). In order to wash the fine gold the prospectors built dams, water races and sluice boxes. Blainey (1993) records at least one of the latter was formed from a hollowed out sassafras (*Atherosperma moschatum*) trunk. Reference is also made to small peppermint gums (*Eucalyptus nitida*) growing on the rocky outcrop that formed the Iron Blow.

In 1885, with the financial aid of new shareholders, the Iron Blow was blasted and excavated. At the same time the surface soils of the nearby slopes at the head of narrow Linda Valley were sluiced for fine reef gold. In the favoured areas soils were sluiced to a depth of more than a metre. Three years later, a dray road was constructed and a stamp mill brought to the Linda Valley by the Mount Lyell Gold Mining Company. Its boiler was fired by local timber. In 1892, William Orr and Bowes Kelly bought a controlling interest in the company. Their interest was copper ore, assayed at 4.5 to 7%, rather than gold. One year later, in order to raise more capital, the Mount Lyell Mining Company was formed (March 1893). This resulted in the development of more tunnels and shafts.

The initial clearing of vegetation at Queenstown had allowed for the construction of camps and provided firewood. As mining activity increased so did the demand for timber and the extent of vegetation clearance. Timber, taken from the slopes of Mount Lyell, and subsequently, from as far away as the King River valley, was used in the construction of tunnels and shafts, tramways, water races, and dwellings.

By chance, the western face of Mount Lyell offered an irresistible resource; an extensive stand of King Billy Pine (*A. selaginoides*), a strong, durable species with a sizable, tapered-pole form that was remarkably easy to mill and of renowned durability. The use of this resource in construction, and subsequently as a furnace fuel during the refining processes, resulted in the complete exploitation of the stand. The existence of this stand is today recorded by the presence of numerous stumps. In 1912, the development of the Lake Margret power station, as a supply of hydro-electricity to the mine, reduced fuel demands and timbergetting all but ceased (Blainey, 1993).

1.42 Smelting

Blainey's work, 'The peaks of Lyell', provides the only comprehensive history of mining and smelting at Mount Lyell during the early years. This section summarises the developmental history of Mount Lyell and draws exclusively on Blainey's excellent synthesis.

The Mount Lyell Mining Company initially sent ore to Adelaide and Swansea (Wales) for smelting. In order to minimise costs, particularly those of transportation, two 150 ton blast furnaces designed by Robert Sticht were constructed by the Company during 1895. These were located in the Queen River Valley at the foot of the western slope of Mount Lyell. To support its construction a saw mill, brick kilns and a crushing plant were built. Wood for the boilers was hauled from all the nearby valleys. The ore was transported from the nearby mountain on a steam tramway connected to an aerial haulage line.

The smelters, fired in June 1896, relied on coke and high-sulphur content iron pyrite ore as a fuel source. They were hailed as a success. Shares soared, a boom ensued, and by 1898 42 companies (only four of which were ever to find payable ore) had taken up adjacent leases. Soon thereafter, the Company quarried many of the hills around Queenstown for hundreds of thousands of tons of siliceous smelter flux, a mineral that was lacking in the Iron Blow ore. In late 1902 Sticht perfected pyritic smelting had almost eliminated the need for coke as a blast-fuel.

By 1899 the number of blast furnaces operated by the Mount Lyell Mining Company on the Queen River site had reached 11. At the time, Sticht claimed that his smelters would treat 1000 tons of ore a day. The black heap of nearby slag grew. Blainey (1993) records that seven or eight furnaces in continuous blast would have produced 120 000 tons of sulphur dioxide per annum (unconfirmed sources suggest figures of 200 000 tonnes per annum). On still days the sulphur fumes filled the Queen River Valley, thickening the morning fogs of winter. When the prevailing westerlies blew, the sulphur pall extended toward Gormanston, and beyond, its influence said to be seen in the corroded telegraph poles ten miles to the east of the Linda Valley. The fumes polluted drinking water, caused respiratory problems and yellowed or 'burnt-off' regrowth (Sarson, pers comm.).

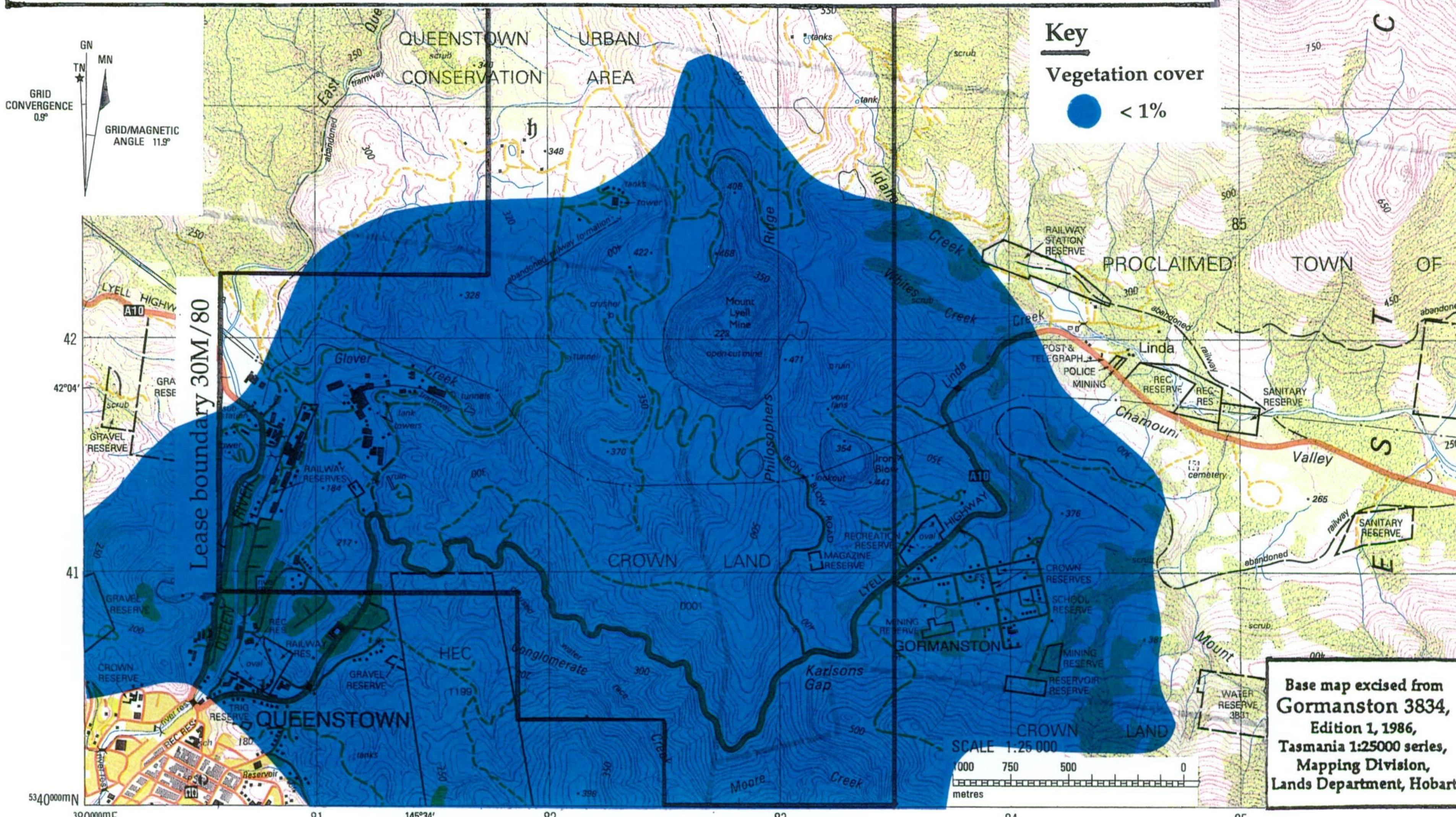
Bushfires became a frequent summertime occurrence. From 1896 to the turn of the century many of the leases were burnt numerous times. The upper slopes of Mount Lyell burnt. Hundreds of houses, including the entire town of Penghana, which supplied and sheltered the Company's smelting men, were burnt out of existence. Trees that had not been selected for mine construction, infastructure, housing or fire wood were doubtless consumed. These probably included Myrtle (*Nothofagus cunninghamii*), a rainforest species not preferred for construction. Exposed, the organic rich soil dried in the prevailing winds, burnt and washed away in the heavy rain. Blainey (1993) describes the landscape at the turn of the century as "black and desolate, a cemetery of black stumps" in which "two beautiful valleys had become as ugly as battlefields". Although no records remain of the exact extent of the deforested areas during this period, it is likely to have exceeded 4 000 hectares. Wood (1991) estimated the extent of the highly disturbed area from aerial photographs at approximately 1000 ha in 1953 (Fig. 3).

In 1899 the Mount Lyell Mining Company's payroll averaged 2 600 men and two years later, became the largest copper producer in the Southern Hemisphere. However, the concentration of the Iron Blow deposits rapidly dropped from 6.04% in 1896 to 2.59% in 1900. Between 1899 and 1901, with concentrations in the Iron Blow still falling, the Company reluctantly accepted the predictions that Mount Lyell was a large, low grade copper field and began to purchase copper-rich siliceous ore from a rival company, the North Lyell Company, and a number of rich but small satellite mines.

In 1903, after considerable animosity and manoeuvring, the Mount Lyell Mining and Railway Company was formed from the two largest, competing companies at Mount Lyell; the prime assets of which were North Lyell's high grade ore and Mount Lyell's efficient smelters. North Lyell's ore required iron and

Fig. 3: Area of denudation at Mount Lyell in 1953.

(Map redrawn from Wood, 1991)



sulphur to separate the copper. The low grade deposit in the Iron Blow became a flux quarry. It averaged 48% sulphur and, with the exception of that recovered from the mine for superphosphate production, none was recovered from the smelter gases. As the copper grades declined the percentage of both lead and zinc increased. Neither were recovered, and were lost as smelter emissions.

Declining ore grades demanded alternate metallurgical processing. In 1916, the first flotation plant at Queenstown concentrated low grade schist ore. This process produced waste tailings which were piped directly into the nearby Queen River. From 1922 until 1934 the concentrate was sintered of sulphur, then smelted without the aid of pyrites in one small furnace. Electrolytic refining was also used from 1928. Semi-pyritic smelting later eliminated sintering and a dust-collecting apparatus was installed. This was claimed to eliminated 95% of the 52 tons of particulate emissions produced each 24 hours. Semi-pyritic smelting continued until the smelters were finally shut at the end of 1969.

1.43 The modern era

After 1969, a selective flotation method was used to form a copper concentrate which was shipped to Australian and overseas smelters. This method of separating the copper pyrite minerals from the host rock was used until the closure of the mine in December 1994 (Falkner, pers. comm.). During this period approximately 4 000 tonnes of tailings, including residual metals, were produced per day and discharged directly into the Queen River. The tailings, alkaline due to the addition of lime to the flotation slurry, partially neutralized the acidified waters of the Queen (Locher, 1995; Wood, pers. comm.). Eventually, the tailings deposited in deeper water of the Macquarie Harbour, via the lower King River, forming a sub-aerial delta measuring approximately 2.5 km^2 (Taylor *et al.*, 1996) in area at the river mouth.

Today, unabated acid mine drainage (Miedecke, *et al.*, 1996), the tailings delta, and dissolved metals such as Cu (Teasdale *et al.*, 1996) from the mine workings pose ongoing risks to the local environment and population (de Blas, pers. comm). As I write (1995), these risks are the subject of The Mount Lyell Remediation Research and Demonstration Program funded jointly by the Federal (Office of the Supervising Scientist) and the Tasmanian Governments. This multi-disciplinary project is to focus on the impacts of acid-mine drainage and tailings deposition in the Queen River and delta in Macquarie Harbour. Concurrently, Copper Mines ot Tasmania (CMT) has taken over the old Mount Lyell lease with plans for further copper mining (3.5 Mt per annum at a head grade of 1.45% Cu). The proposal includes refining using alternate technology and the construction of a tailings dam to the west of Queenstown.

1.5 The contemporary environment of Mount Lyell

1.51 Surface geology

During the life of the mine, severe erosion exposed subsoils over extensive areas of the lease (Wood, pers. comm.) and resulted in the deposition of an estimated 10 Mt of top soil in the Queen and King River systems (Locher, 1995). The loss of top soil throughout the Mount Lyell mining lease has resulted in a surface layer typically composed of either exposed, partially-weathered sub-soils or consolidated parent materials.

The surface geology of the lease, and much of the western face of Mount Lyell, is characterised by partially weathered, metamorphic assemblages of quartzes, sericites, chlorites and sulphides in various proportions (Mount Lyell Technical Review, 1993). A felsic sequence (quartz sericite-chlorite) dominates the lease. At intervals, irregular units of intermediate-mafic volcanics occur throughout the felsic sequence. The units exhibit proportionally less quartz and range from 20-30 cm up to tens of metres thick (Mount Lyell Technical Review, 1993).

In the felsic volcanics, weathering of the more succeptible minerals, such as haemitite or chlorite, has produced a pale pink to white, quartz-dominated skeletal material with a sandy texture. In places, fractured veins of quartz have formed deep drifts composed solely of coarse fragments. In the mafic volcanics, weathering has produced emerald green to orange-iron red, sericitic clays. On this geology, partially-decomposed volcanic fragments and pebbles litter the surface (author's observations). Folded, iron rich, volcanics occur as jagged, poorly weathered outcrops throughout the lease.

1.52 The present-day vegetation of Mount Lyell

The fire frequency of the Mount Lyell region increased with the arrival mining prospectors of mid-last century (Blainey, 1993). Today the vegetation of the Mount Lyell region is typified by fire-disclimax communities (Kirkpatrick, pers. comm.) of eucalypt forest (dominated by *Eucalyptus nitida*), blackwood forest (dominated by *Acacia melanoxylon*), scrubland (dominated by *Leptospermum* spp.) and buttongrass sedgeland (dominated by

Gymnoschoenus sphaerocephalus). Rainforest has been eliminated or restricted to fire shadows.

None of the above community descriptions, however, adequately describe the present-day vegetation of Mount Lyell, particularly along its western, southern and south-eastern flanks. Here, the combined effects of prospecting fires, intensive timber-getting and prolonged smelter contamination have had their greatest impact. Assessment of period aerial photographs (Wood, 1991) revealed that approximately 15 km^2 were denuded of vegetation in the early 1950's with and aditional 25 km^2 affected. In the intervening years, colonisation by a restricted number of plants has occurred and this has resulted in the formation of a depauperate 'community'of tolerant species. In the areas of greatest impact, typically close to the smelter stacks, vascular vegetation is non-existent or severely restricted, and sparse colonisation by bryophytes, algae and a few, residual higher plants represent the only visible signs of life (Author's pers. obs.).

The upper slopes of Mount Lyell have been less altered by direct and indirect mining activities. However, wildfires have degraded the vegetation and exposed the thin soils to erosion. Today, an open montane shrubland exists. In fire shadows, a few King Billy Pines remain.

1.6 The scope of this work

The investigations reported on in this study were confined to the deforested and eroded areas falling between the Mount Lyell smelter site and the eastern boundary of the Mount Lyell lease (30M/80). The study area totalled approximately 400 hectares, or about 1/10 of the total area of the lease. About 75% of this area has slopes between 15 and 25°. Excluded were any areas directly disturbed by recent mining operations, such as surface excavations and waste rock dumps. Prominent exclusions from the study area were the workings of the West Lyell open cut and its associated waste dumps.

The general aims of this work were two-fold: 1) to explore compositional trends in the colonising vegetation in relation to the edaphic environment, and 2) evaluate rehabilitation methods using local, colonising species.

Chapter 2 of this thesis begins with a review of the literature. The topics discussed include the deposition emissions from base-metal smelters, the effects of sulphur dioxide on vegetation and soils, metal phytoxicity and metal tolerance, the assessment of metal contamination and the rehabilitation of acid, metal-contaminated terrestrial ecosystems.

Chapter 3 describes soil characteristics along transects of increasing displacement from the Mount Lyell smelters. Particular attention is paid to soil-metal concentrations. Comparisons of metal distribution patterns and contamination levels are drawn with other world-renowned, base metal smelters. Phytotoxic contaminants are discussed.

Chapter 4 explores the relationships between the composition of the colonising community and edaphic variables. Soil and floristic data were collected in a survey of the environment of Mount Lyell. Occurrence was used as a measure of the differential performance of species and related to edaphic variables with the aid of numerical analyses. The colonising community was mapped with the aid of a classification. The chapter contributes towards greater understanding of the process of re-colonisation in metal-contaminated areas, and in doing so, provides a guide to the site conditions that local species are able to tolerate.

Chapter 5 examines the phytotoxicity of the Mount Lyell soils along a metal concentration gradient. Seedling growth in pot trials was used as a measure of tolerance. Pot and field trials are used to assess the usefulness of a conventional, neutralising agent as a means of phytotoxicity amelioration.

Chapter 6 compares two mechanical seedbed preparation methods designed to enhance the lodgement, germination and early survival of local, colonising species. Seed was broadcast sown at an eroded, near-smelter location with phytotoxic soil characteristics.

Chapter 7 explores low-impact methods of seedbed preparation for the establishment of local, colonising species on Mount Lyell's steep, eroded slopes. The chapter compares several non-mechanical methods of seedbed preparation and seed application at a near-smelter location with phytotoxic soil characteristics.

Chapter 8 provides a summary of the thesis. It considers appropriateness of the strategies of reclamation and neglect to Mount Lyell and gives recommendations.

Chapter 2

Literature review

2.1 Overview

Throughout the world, waste gases from certain industrial smelting operations have been linked with the contamination or alteration of terrestrial and freshwater ecosystems. Many of these smelter gases have contained a high percentage of sulphur dioxide. In terrestrial ecosystems the exposure of vegetation to sulphurous smelter gases has caused injury and death in sensitive plants. Persistent exposure has resulted in reduced site productivity and altered species diversity. In soils, the deposition of SO_2 emissions has been associated with significant shifts in elemental solubility, equilibrium and speciation chemistry, often culminating in accelerated cation leaching. Soil acidification has also been reported.

In the case of base metal smelters, the deposition of metal particulates has lead to elevated concentrations of soil metals. Elevated soil-metal concentrations have been linked to phytotoxicity. This may be of especial concern in low pH environments as soil acidity influences metal bio-availability. In the vicinity of base metal smelters, vegetation subjected to prolonged metal contamination has been completely eliminated. In some locations, vegetation elimination has contributed to severe physical degradation, such as extensive soil erosion. Disturbance at these sites is catastrophic and results in drastically simplified ecosystems. Locally, these simplified ecosystems have been referred to as deserts.

Natural recolonisation occurs in areas contaminated by base metal smelters. However, altered selection pressures may result in communities distinct from those following a natural disturbance. Recolonisation usually begins when emissions are significantly reduced or eliminated. Commonly, such reductions result from mine or smelter closure. Species richness in recolonised areas is commonly low, and a restricted number of survivors of the original flora, together with pre-adapted colonising species, are likely to be present. Individuals are often stunted due to the presence of stressful environmental conditions. Even when smelter emissions are eliminated, recolonisation may be retarded for many decades, especially in soils exposed to high levels of phytotoxic contamination.

The reclamation of lands degraded by smelter emissions and related mining activities has been attempted in a number of areas of the world. An underlying principle is to reduce soil toxicity. To this end, neutralizing agents and fertilizers have been applied. Species re-introduction has been attempted by replanting or resowing. Success, however, has been limited. This may be due to the loss of many of the qualities of a pre-existing environment. Reclamation success may have to meet both objective measures, such as survival and growth rate, and local community expectations. Meeting reclamation requirements may require major intervention and involve great cost.

2.2 The composition, dispersion and deposition patterns of gaseous and particulate emissions from smelter chimneys

2.21 Smelting products

Smelting typically results in both gaseous and particulate atmospheric products. The smelting of sulphide ores produces gaseous sulphur dioxide. Gaseous SO_2 may react with atmospheric water to ultimately form sulphuric acid, or, following chemical or photo-oxidisation in the atmosphere, produce secondary particulate sulfates (Brown, 1982). Within a plume oxidation can occur on the surfaces of soot, liquid-coated particles or droplets. Particulate products may also be represented by various metallic elements as either process losses or by-products. Common examples of metallic particulates found in smelter emissions are copper, zinc and lead.

2.22 Plume patterns and plume dispersion

Many factors influence the pattern and dispersion of a plume originating from an atmospheric point-source of emission. The most influential are the prevailing atmospheric conditions, as these control the diffusive effects of wind speed and turbulence (Gifford, 1976). Dry atmospheric conditions produce a number of characteristic plume patterns (Oke, 1978) with certain patterns associated with either localised or long-distance dispersion. For example, neutral atmospheric conditions produce streaming, which may result in dispersion over long-distances, while stable atmospheric conditions cause fumigation and this tends to produce localised deposition.

The dispersion of smelter emissions has been modelled for short time periods (e.g. 1 hour). The most widely applied plume dispersion model for a continuous, point-source emission is the Gaussian model (Hanna, 1982). This model includes the factors: point-source concentration, height of emission, effective plume height, lateral distance from plume center-line, wind speed at effective plume height and lateral and vertical dispersion. All of the models contain a number of assumptions and do not attempt to account for chemical transformations such as sulfate conversion rates.

A factor which may influence the dispersion from a point source of emission is the local terrain (Oke, 1978). Elevated terrain down-wind of a point source of emission may act to reduce the effective plume height. Physical influences on plume dispersion interact with the prevailing meteorological conditions. For example, in stable meteorological conditions with a positive, vertical temperature gradient (Mitchell, 1982; Pasquill, 1962) a plume may impact a hill in its path, whereas in neutral or unstable conditions a plume will ride up over a hill (Hanna, 1982).

2.23 Wet and dry deposition

The deposition of both gaseous and particulate sulfate compounds requires some means of vertical transport to ground level. Two forms of sulphur deposition are generally recognised; wet deposition and dry deposition. Under dry conditions deposition usually occurs by gravitational settling, although it may also occur under certain unstable atmospheric conditions. Wet deposition occurs due to atmospheric condensation and precipitation scavenging.

The dry deposition of both gaseous and particulate sulfate compounds occurs under the influence of gravitation. Garland (1977) considered dry SO_2 deposition the most significant means of sulphur transport from air to land in the UK, with an average deposition velocity of 0.85 cm s^{-1}. However, the deposition rates of most particulate sulfates are an order of magnitude smaller than those of gaseous sulphur as most particulates are <1μ (Maul, 1982, Garland, 1978). Slow deposition velocities result in long atmospheric residence times and pollutants are known to be transported up to several thousand kilometres from primary sources of emission (Brown, 1982). For these particles precipitation scavenging is considered the only significant removal mechanism (Maul, 1982). Small deposition velocities have led to the omission of sulfate particulates from deposition models (Maul, 1982; Sehmel, 1980).

Dry deposition may also occur under unstable atmospheric conditions. Convective turbulence may produce a looping plume pattern due to the development of large eddy structures in daytime during summer (Oke, 1987). This may result in relatively undiluted emissions reaching the ground in the vicinity of a smelter. Another atmospheric condition in which undiluted emissions may reach ground level is known as fumigation. Fumigation usually occurs under stable conditions on winter mornings when surface warming causes mixing under an inversion layer. Under these conditions fumigation results in SO_2 fogs. Sulphur dioxide fumigation of this type reportedly occurred in a mountain valley at Trail, British Columbia near a lead and zinc smelter (Dean *et al.*, 1944) following the overnight, down-valley drainage of poorly diluted pollutants.

In comparison to dry deposition, wet deposition tends to be localised and may be the dominant mode of sulphur deposition in areas of high precipitation. Wet deposition occurs as atmospheric water condenses and scavenges gaseous sulphur dioxide and particulate sulfate compounds above (rainout) and below (washout) the cloud base (Brown, 1982; Maul, 1982). In the case of gaseous emissions the reaction with hydroxide ion radicals results in sulphurous acid (Brown, 1982). Sulphurous acid rapidly oxidises to form sulphuric acid or acid rain, usually by the time it reaches the ground (Maul, 1982). Some gaseous emissions may be removed below the cloud base (washout) and occur as sulfite ions in rainwater (Maul, 1982). Wet deposition of sulfate particulates is dominated by rainout, with washout only important for the largest particles i.e. >2μ (Garland, 1978).

2.24 Deposition envelopes

Although there appear to be no definitive studies, it is plausible that different deposition envelopes exist for each constituent of a plume as the chemical form of emissions affects particle size which in turn interacts with the mechanisms of deposition and influences atmospheric residence times. For example, some authors have suggested that the deposition envelopes of sulphur dioxide and metal particulates resulting from point sources of emission are not necessarily the same (Hutchinson and Whitby, 1974). The supposition appears to be that under dry conditions, sulphur dioxide is more likely to be deposited closer to a source than metal particulates due to its greater depositional velocity. It has also been suggested that chemical separation within a plume resulting in distinct deposition envelopes may be reflected by the receiving soil environment. Overrein (1972), in an attempt to explain the occurrence of the lowest soil pH at some distance downwind from a smelter source, suggested that various basic particulates, themselves capable of acid neutralisation, tended to deposit closer to a point source than acid particulates.

2.3 The effect of sulphur dioxide fumigation and particulate deposition on leaf tissue

At elevated concentrations, gaseous sulphur dioxide is an unspecific toxicant affecting many cell functions in higher plants (Knabe, 1976), although sensitivity is demonstrably species-specific (Murray, 1984; O'Connor, 1974). In sensitive species, exposure to sub-lethal SO_2 concentrations in laboratory fumigation trials has resulted in reductions in net CO_2 assimilation rates (Addison *et al.*, 1984) and chlorosis (Knabe, 1976). Sulphur dioxide enters mainly through leaf stomata and injury occurs when dissolved SO_2 diffuses through leaf tissue via cell walls (Knabe, 1976). Dissociation products include the highly phytotoxic sulfite ion (Knabe 1976). Within the cell the presence of the sulfite ion results in numerous biochemical interactions, notably adversely affecting the permeability of the cell membrane and the function of cell chlorophyll. Other authors have reported on the inhibitory effects of SO_2 and sulfite on cellular function and photochemical processes (Veeranjaneyulu *et al.*, 1990).

Acute SO_2 injury results in cell collapse, and has been known to cause defoliation in the vicinity of a source of SO_2 emission (Preston, 1988). Initially, in broad-leaved plants necrosis begins in the more susceptible marginal and interveinal areas of the leaf and extends inward (Knabe, 1976; O'Conner, 1974). The affected areas appear dark green, with a water-soaked discolouration as cell contents leak into intercellular spaces, become flaccid, and, upon drying, can range in colour from white to red-brown or black (Taylor, 1971).

Laboratory fumigation chamber experiments using seedlings have been used to rank species sensitivities in response to elevated levels of SO_2. O'Conner *et al.* (1974) have published the only comprehensive SO_2 fumigation study of Australian native plants. None of the West Tasmania rainforest endemics were included in the study. However, six widespread species present in the Queenstown area were examined. At concentrations of up to 3 ppm (7860 $\mu g m^{-3}$) over four hours, the seedlings of four species were rated resistant to extremely resistant to SO_2 fumigation(*Acacia dealbata, Acacia melanoxylon, Eucalyptus obliqua* and *Melaleuca squamea*), while two other species were rated two moderately sensitive (*Melaleuca squarrosa* and *Banksia marginata*).

2.4 The effects of sulphate deposition on soils

2.41 Sulphate

Sulphur enters the soil system from the atmosphere either directly, as dry deposition on the surface of foliage, soil, or litter or dissolved in precipitation (Brown, 1982). It may also enter the soil indirectly following assimilation into plant tissue and subsequent release by litter decomposition (Brown, 1982).

Sulfate accumulation is not known except in dry regions (Brown, 1982). The major pathway for the removal of sulphur from a soil system is by sulfate leaching. However, additional losses can occur where crops are harvested or microbial volatisation is significant. Leaching losses depend mainly on soil sulphur status, rainfall and soil texture, and losses from bare soil are generally more severe than from cropped land (Brown, 1982). In some soils, sulphate leached from upper horizons may be retained by adsorption in subsoils.

Acidified streams and lakes draining catchments exposed to elevated sulphur deposition have been widely reported (e.g.Wright and Gjessing, 1976).

In contrast to the phytotoxic metals, direct toxic effects due to sulfate contamination are uncommon in well-drained soils (Brown, 1982). More usually, toxicity is caused by H_2S formation during waterlogging, or by the accumulation of soluble sulfates in poorly drained acid soils (Halsted and Rennie, 1977). Sulphate deposition may indirectly result in increases in metal-ion solubility in response to soil acidification (Malmer, 1976) and cation leaching (Reuss *et al.*, 1987).

2.42 Cation leaching, acidification and metals

The initial effect of sulphur deposition on soil is to increase the concentration of H^+ and SO_4^{2-} in the soil solution (Reuss *et al.*, 1987). Raised hydrogen and sulfate-ion concentrations result in two recognised, direct geochemical consequences: the displacement and subsequent leaching of soil nutrient cations (Wookey and Ineson, 1991; Heute and McColl, 1984; Malmer, 1976) and increased metal-ion solubility (Nelson and Campbell, 1991; Reuss *et al.*, 1987; Malmer, 1976). These responses are the result of acid-neutralising reactions at cation exchange sites or on mineral structures.

Cation leaching in response to sulphur deposition has been widely reported, and has been confirmed by laboratory acid-loading experiments (Dahlgren *et al.*, 1990; Heute and McColl, 1984). Dahlgren *et al.* (1990) reported that strong acid inputs were initially neutralised by SO_4^{2-} adsorption and the release of basic cations from the exchange complex. The contribution of base cation release to acid-neutralisation was related to the base saturation of the soil. Prolonged loading resulted in the depletion of the base cation reserve while at the same time the adsorption/release of SO_4^{2-} found a new equilibrium. Thereafter, aluminium dissolution became the primary acid-neutralisation mechanism. Matzner and Prenzel (1992) reported that the principal H^+ buffering process in a German forest subject to heavy acid deposition was the release of Al ions.

The other recognised consequence of sulphur deposition on soils is its influence on pH-dependent, metal-ion availability. Non-essential or micro-nutrient, metallic elements are normally non-toxic to plant life at the solubility-restricted concentrations at which they are found in the natural environment. Restricted metal-ion solubility, rather than rarity, usually limits plant availability and thus the potential for phytotoxicity. However, acidic inputs, in combination with base cation leaching, may result in an accumulation of acid cations in the soil solution and result in a lowering of soil pH (Malmer, 1976). Increased acidity may shift the solubility, abundance and speciation of metal-ions and can result in alterations to bio-availability with phytotoxic consequences (Nelson and Cambell, 1991; Malmer, 1976). Examples of metal-ions that display increased solubility with pH decrease are copper (Flemming and Trevors, 1989), zinc (Balsberg Påhlsson, 1989) and aluminium (Andersson, 1988). Species of each of these elements have the potential to be phytotoxic at elevated concentrations or under conditions of increased availability.

2.43 Interaction with the environment

Not all soils are equally disposed to cation leaching and acidification. The sulfate adsorption capacity of a soil may vary between soil types and within a profile in association with clay content, organic matter, free sesquioxides and pH (Heute and McColl, 1984; Malmer, 1976). For example, the sesquioxide rich, weathered soils of the southeastern United States appeared better buffered and more able to adsorb SO_4^{2-} than the immature soils of northern North America and Europe (Reuss *et al.*, 1987). Similarly, Malmer (1976) considered soils high in clay or organic colloidal content more resistant to nutrient cation leaching, and thus less sensitive to acidification than typically acidic, carbonate-free, siliceous soils with low base saturation (Reuss *et al.*, 1987). Krug and Fink (1983) caution that soil acidity does not necessarily confer susceptibility to further acidification. These authors maintained that because H^+ ions in acid rain are inefficient at exchanging bases at pH 4 or lower, the less acid, coarse siliceous soils with low cation exchange capacities are more susceptible to acidification (Krug and Fink ,1983; Malmer,1976). Similarly, acid humic soils may be buffered against further acidification as H^+ hinders the dissociation and solubility of naturally occurring acid-forming humic materials (Richie and Posner, 1982).

Even in the absence of an anthropogenic source of metal-ions increased soil acidity may lead to soil toxicity, as in the case of aluminium. As soil pH falls below 4.5 the solubility of aluminium rapidly increases, accelerating its release from soil minerals (Andersson, 1988; Reuss *et.al.*, 1987; McLean, 1976). The ubiquity of Al as a constituent of soil minerals, and the toxicity of some of its inorganic species, contribute to a potential to influence biota under conditions of acidic deposition.

2.44 Reversal

There is some experimental evidence to suggest that decreases in acid deposition will lead to equilibrium reversal between the soil solution and exchange sites, which over time, may result in reductions in the concentrations of soluble metal ions. In a column leaching experiment using the B horizon from a spodosol, Dahlgren *et al.* (1990) found that, following a decrease in acid loadings, basic cations were retained, SO_4^{2-} was released and concentrations of soluble Al were reduced. Desorption of SO_4^{2-} lead to a lowering of the pH of the leachate. Acidification due to desorption ceased when the SO_4^{2-} leachate concentrations approached the input concentrations. Despite the reductions in Al concentration, the authors concluded that Al dissociation remained the main H^+neutralising process, accounting for acidity as a result ofSO_4^{2-} desorption and cation retention.

2.5 Soil pH and the phytotoxic metals aluminium and copper

2.51 Soil pH

The direct effects of the H ion on plant growth are difficult to assess as other potentially harmful elements are likely to be present in toxic concentations, and other elements may be sub-optimal (Foy, 1992). Foy (1992) indicated that in acid soils below pH 4.0, the toxicity of certain metal ions such as Al and Mn are probably more important than H ion toxicity. Nevertheless, excess H ions are understood to compete with other cations for root adsorption sites, interfere with ion transport and cause root membrane failure. Competition for root adsorption sites may cause nutrient deficiencies such as Ca (Lund, 1970). H ion toxicity has reportedly effected the activity and survival of rhizobia and other soil microorganisms (Richardson *et al.*, 1988a,b; Kamprath and Foy, 1985; Moore, 1974).

Foy (1992) considered clearly identifiable H ion toxicity highly unlikely in higher plants grown on agricultural soils, but conceeded that H ion toxicity could play a role when plants are grown on mine spoils below about pH 3.0.

2.52 Aluminium

Soil pH, aluminium dissolution and speciation

Aluminium is an abundant constituent of soil but is a non-essential element for plant nutrition. At neutral pH levels, Al is almost insoluble (Singer and Munns, 1987) and is therefore unavailable and does not interfere with plant health. However, the dissolution of Al from soil minerals increases as soil pH falls below 4.5 (Foy, 1992), or rises above 8.5 (Fuller and Richardson, 1986), and results in excess soluble aluminium (Andersson, 1988; Reuss *et al.*, 1987).

Excess soluble Al in low pH soils is one of a number of metals that may cause phytotoxicity. Foy (1974, 1984, 1988) and Kamprath and Foy (1985) considered Al toxicity to be probably the most important growth limiting factor for plants in strongly acidic soils and mine spoils. Aluminium toxicity has also been associated with cultivated soils (Foy, 1992) and acidic subsoils. In contrast, high Al concentrations occur naturally in the acid soils of the tropics and the spodosols of the humid temperate regions, primarily under forest (McKeague *et al.*, 1983). Plants in these environments have adapted to high Al concentrations and native tropical plants will grow under conditions that would be fatal to cultivated grains (Foy, 1992).

The soil pH at which Al becomes soluble depends upon many soil factors including the predominant clay minerals, organic matter levels, the concentrations of other cations, anions and total salts, and particularly, the plant species or cultivar (Foy, 1984; Kamprath and Foy, 1985). Shifts in Al speciation in soils as a result of acidification favour the soluble, inorganic monomeric forms, in contrast to polymeric or organically complexed forms. In soil below pH 4.5, Al exists primarily as Al^{3+}, and in this form it occupies a large part of the exchange complex (Foy, 1992). It is generally the monomeric forms that are recognised as phytotoxic (Andersson, 1988; Bell and Edwards, 1896; Blamey *et al.*, 1983; McLean, 1976). In solution, Al may be involved in cation displacement and obstruct exchange in the soil complex due to its high valence and ability to polymerise (McLean, 1976). Bloom *et al.* (1979) concluded that exchange of Al ions from organic matter exchange sites controls the relationship between pH and Al^{3+} activity in soils with a low amount of permanent charge CEC relative to the quantity of organic matter.

McLean (1976) explains that, when the hydrogen ion concentration in the soil solution increases to a pH of 4 or below, hydronium ions (OH_3^+) are formed, resulting in the dissolution of aluminium ions from the edges of partially weathered mineral structures. The aluminium ions become six-fold coordinated with oxygen in OH_2 groups forming aluminohydronium ions, $Al(OH_2)_6$; a substitution in which aluminium ions displace hydrogen from hydronium ions. Sequential dissociation of the aluminohydronium ions results in partially neutralised, hydroxy-aluminium ions (OH-Al). Some hydroxy-aluminium ions remain in the soil solution. Others are adsorbed and polymerise on the surfaces of clay minerals, or become complexed with organic matter. Both adsorbed and complexed hydroxy-aluminium ions obstruct the exchange of other cations, such as calcium, and thereby inhibit neutralisation at exchange sites.

The chemistry of Al transformations in soils has been recently reviewed by Huang (1988). A review of the effects of freshwater acidification on the geochemistry of aluminium and other metals is provided by Nelson and Campbell (1991).

Tests for aluminium toxicity in soils

Numerous measures have been used to relate soil Al to plant growth. These measures include exchangeable Al (using a variety of extractants such as $CaCl_2$, $LaCl_3$, NH_4OAc and KCL), exchangeable Al index (Reeve and Sumner, 1970; NH_4Cl), Al saturation percentage of effective cation exchange capacity (KCL Al/KCL Al plus NH_4OAc exchangeable Ca, Mg, K and Na; Evans and Kamprath, 1970), soil solution Al and Al activity.

Adams and Lund (1966) and Wright (1989) considered conventional Al soil tests (pH, acid exchangeable Al, percentage Al saturation of CEC and salt extractable Al) unuseful for predicting Al toxicity across a wide range of soils. Nevertheless, exchangeable Al has been related to root growth and yield (McKenzie and Nyborg, 1984; Siagusa *et al.*, 1980; Moschler *et al.*, 1960), but the measure does not always relate to plant response because of differences between soils in mineralogy, surface charge, organic matter and other factors (McCray and Sumner, 1990). Some authors have used Al saturation of CEC to predict Al toxicity (Evans and Kamprath, 1970, Blamey and Nathanson, 1977; Farina and Channon, 1980; Kamprath and Foy, 1985). This method, however, must be used within a narrowly defined set of conditions regarding soil type, plant species and genotype (Foy, 1987; Adams *et al.*, 1967). Other authors suggest, that where similar parent materials exist, pH and extracted Al (KCL or other salts) may be useful in predicting Al toxicity for a given plant (Blamey and Nathanson, 1977; McCormick and Amendale, 1983). Pravan *et al.* (1982) showed that plant injury was a function of the activity of Al^{3+} in the soil solution rather than total Al concentration. Brenes and Pearson (1973) and Adams and Lund (1966) also reported close relationships between plant growth and Al activity.

The concentration of the soil solution provides a more direct measure of the conditions experienced by plant roots (McCray and Sumner, 1990). Baker *et al.* (1988) and Bruce (1988) indicated that, in comparison to conventional methods, the soil solution provides a more accurate means of assessing Al toxicity and various methods have been used to extract soil solutions. Soil-water extracts have been used in a number of studies to assess soil toxicity in lands contaminated by acidic deposition from base-metal smelters (Freedman and Hutchinson, 1980; Whitby and Hutchinson, 1974). Soil solution Al^{3+} concentration has been found to be related to plant growth in a number of studies (Adams and Moore, 1983; Evans and Kamprath, 1970; Ragland and Coleman, 1959).

Symptoms of aluminium toxicity

Aluminium toxicity is a complex disorder (Foy 1992) which can manifest as P, Ca, Mg, or Fe deficiency or drought stress (Alam and Adams, 1980; Foy, 1984, 1988; Kamprath and Foy, 1985). In some plants the foliar symptoms of Al toxicity resemble that of P deficiency, in others Ca deficiency (Foy, 1992) or even Fe deficiency (Clark et al., 1981; Foy and Fleming, 1982). Symptoms of Al toxicity in the whole plant tend to occur as stunting of shoots and roots. Aluminium is not known to interfere with seed germination but seedlings may be more susceptible to aluminium toxicity than older plants and this affects establishment by impairing root development (Nosko et al., 1988). The physiological effects of Al on plants have been reviewed by Foy (1974, 1984). The toxicity of aluminium to vascular plants has been reviewed by Andersson (1988).

2.53 Copper

Copper is one of the essential plant nutrient microelements. However, particulate deposition, or decreases in soil pH, may result in excess soluble Cu. At elevated levels, Cu is known to be toxic to both terrestrial and aquatic life (Flemming and Trevors, 1989). Never the less, the toxicity of Cu may be greatly influenced by environmental factors (McBride, 1989).

Many environmental factors affect the toxicity of the cupric cation because it is reactive and susceptible to modification by complexation, and is subject to precipitation and adsorption processes (Flemming and Trevors, 1989; McBride, 1989). Complexation of dissolved Cu controls speciation and concentration. In soils and sediments, Cu adsorption occurs on the surface of Fe and Al hydrous oxides and hydrous Mn oxides (Thornton, 1979), on clay minerals in proportion to CEC, and on organic matter (Elliot *et al.*, 1986, Thornton, 1979). The majority of the total Cu in soils has been found associated with the organic fraction (Thornton, 1979). This fraction is believed to control the mobility and bioavailability of Cu (Elliot *et al.*, 1986). Divalent Cu, for example, has one of the greatest affinities for soil organic matter (Stevenson and Ardakani, 1972). Existing evidence suggests that dissolved Cu^{2+} is almost completely in organically complexed form (McBride and Blasiak, 1979). Consequently, soluble organically complexed metals, such as Cu, are likely to control availability for plant uptake (McBride, 1989).

In pot and solution culture trials, the addition of Cu has stunted growth in trees (Heale and Ormrod, 1982; Fessenden and Sutherland, 1979), grasses (Symeonidis *et al.*, 1985; Rauser and Winterhalder, 1985; Hogan *et al.*, 1977b) and crop species (Wong and Bradshaw, 1982; Toivonen and Hofstra, 1979; Walsh *et al.*, 1972). Rauser and Winterhalder (1985), found that Cu was more toxic than both Ni and Zn. Usually, root growth is more affected than shoot growth, resulting in reduced root-shoot ratios. Critical leaf tissue concentrations for most species are between 15 and 25 µg Cu g^{-1} (Balsberg Påhlsson, 1989). The sensitivity of various organisms to elevated levels of copper in soils has been reviewed by Flemming and Trevors (1989).

Sediments may act as a sink for metal pollutants. Because of this, metal toxicity can be a major problem in lakes and estuarine environments (Flemming and Trevors, 1989). Copper, for example, is known to be extremely toxic to aquatic biota, including fish, invertebrates and algae (Flemming and Trevors, 1989).

However, considerable variation in tolerance is apparent, for example, in laboratory studies Hodson *et al.* (1979) found that 30 μg L^{-1} was lethal for salmonids (96 hr-LC50) whereas for blue gills equivalent levels of mortality were not reached until concentrations reached 6000 μg L^{-1}.

2.6 Migration of soil contaminants

Soil contaminants may be eroded by either water or wind. Erosion may lead to contaminant redistribution locally or throughout catchments. As metal contamination is known to be associated with particular soil fractions, notably clay and organic matter (Ghadiri and Rose, 1991a), particle selectivity may result in the concentration of soil metals, resulting in enrichment. The most common form of enrichment occurs as erosion removes soil materials from catchments and redeposits them as sediments in lakes and estuaries. Eroded material may have contaminant concentrations ten times that of the original soil, although twofold increases are more common (Sheppard *et al.*, 1992).

Soil contaminants may also be redistributed within a soil profile. Redistribution mechanisms within soils include mass flow, fracture flow, the diffusion of dissolved or gaseous species, particle or colloid migration and mechanical mixing or bioturbation of sorbed particles (Sheppard *et al.*, 1992). However, these mechanisms are not necessarily of equal importance in the redistribution of metallic soil contaminants. For example, Sheppard *et al.* (1992) considered it rare for contaminants, such as copper and zinc, that are adsorbed onto soil particles, or are of colloidal size, to migrate through a soil profile. Nair *et al.* (1990) considered mass flow the dominant mechanism of contaminant redistribution within a soil profile.

2.7 Metal tolerance

The evolution of metal tolerance by the non-random reproduction of genes in organisms exposed to high-metal substrates or contaminated environments falls within the theory of natural selection and can occur given sufficient generations. Examples of differential tolerance to elevated concentrations of heavy metals are known for individual species representing a wide range of organisms as diverse as microorganisms, algae, fungi and vascular plants.

A variety or population that displays fewer growth abnormalities or less inhibition for a given contaminant concentration is generally termed metal-tolerant. Tolerance has been established experimentally by the use of solution culture experiments and clonal or varietal comparisons. Many researchers have derived the tolerance indices from root inhibition measurements at various contaminant concentrations (Rauser and Winterhalder, 1985; Fox, 1984, Nicholls and McNeilly, 1979; Hogan and Rauser, 1979; Hogan, *et al.*, 1977b; Gregory and Bradshaw, 1965; Wilkins, 1957).

Tolerance to elevated levels of one or more metallic elements has been recognised in populations of grass species recolonising contaminated soils (Jowett, 1958; Bradshaw, 1952). Despite very high levels of soil contamination, tolerant varieties have been known to evolve and establish apparently healthy populations (Shaw, 1989). Of these metal tolerant populations members of the genus *Agrostis* are perhaps the most well known. For example, tolerance to copper has been identified in genotypes of *Agrostis gigantea* (Rauser and Winterhalder, 1985; Hogan and Rauser, 1979;), *A. capillaris* (Nicholls and McNeilly, 1979; Wainwright and Woolhouse, 1977; Gregory and Bradshaw, 1965), and *A. stolonifera* (Wu and Antonovics, 1975). In a solution culture comparison of root growth in *A. gigantea* clones, Hogan *et al.* (1977b) identified copper tolerance in tillers removed from an acid (pH 4.7 - 5.1), nickel-copper roast bed soil at Sudbury with very high concentrations of copper (317 - 699 ppm ammonium acetate extractable). The authors found that a high metal concentration must be present for a degree of tolerance to be expressed, but that high soil-metal concentrations do not necessarily imply that the plants growing thereon will be metal tolerant. The explanation for this appeared to be the degree of pH-determined, metal availability. Interestingly, although reflecting the metal composition of the soils from which they originated, root and shoot metal concentrations were not considered a reliable indicator of metal tolererance. Non-tolerant clones were understood to be restricted to discrete populations in areas of lower metal availability (Hogan *et al.*, 1977a).

There is some evidence of metal tolerance occurring in vascular plants growing on ultramafic substrates. These are parent materials and low nutrient soils that are characteristically rich in ferromagnesium minerals and their products; magnesium, nickel, chromium and cobalt (Lee *et al.*, 1983; Proctor, 1971). The vegetation on some of these substrates has been reported to exhibit peculiarities, such as distinct but sparse vegetation, and these have been attributed to magnesium and nickel related toxicity. On some ultramafic substrates, distinct physiognomic differences in vegetation, and associated endemism, have been recognised in contrast to surrounding areas on dissimilar substrates (Gibson *et al.*, 1992; Proctor and Woodell, 1975). Morphological differences, such as reduced or altered leaf size, have also been reported (Gibson, *et al.*, 1992; Lee *et al.*, 1983). In Australia, a small number of the species found on these substrates have been termed metal-accumulating species due to the presence of high levels of nickel in their foliage (eg. Batianoff *et al.*, 1990; Severne and Brooks, 1972).

Lee *et al.*, (1983) used the root tolerance test of Wilkins (1957) to study intraspecific differences in the growth of *Agrostis capillaris* derived from ultramafic and non-ultramafic populations. An ultramafic population, originating from New Zealand soils with typically high concentrations of Mg, Ni, Cr and Co, was found to be tolerant of cobalt. However, the authors could not demonstrate tolerance to elevated concentrations of either Mg, Ni or Cr. In contrast, Procter *et al.* (1971), using the solution culture test of Gregory and Bradshaw (1965), reported the existence of nickel tolerant ecotypes of *Agrostis* spp. from an ultramafic area in Britain. It appears that, in contrast to total metal concentrations, such disparate results may be explained by site factors that influence the availability of metals to plants. Nickel availability has been related to soil pH and organic matter content (Halsted et al., 1969), and interactions with Ca, Mg, K, Fe and PO4 are known (Procter and Woodell, 1975).

Tasmanian ultramafic areas, however, appear to contrast with other areas in displaying no consistent expression of physiognomic differentiation and very limited endemism (Gibson *et al.*, 1992). Kirkpatrick (pers. comm.) notes that *Epacris glabella* and *Micrantheum serpentinum* appear restricted to ultramafics in western Tasmania. However, in a survey of the vegetation of Tasmanian ultramafic areas, Gibson *et al.* (1992) were unable to identify indicator species in the higher rainfall areas of the west of the State. These authors suggested that inconsistent physiognomic differentiation in Tasmanian ultramafic vegetation may be a result of interactions between edaphic factors and past fire histories. Interestingly, in New Zealand, Lee *et al.* (1983) found no evidence of intraspecific differences between ultramafic and non-ultramafic populations of *Leptospermum scoparium* when both were grown on ultramafic soils. These authors concluded that tolerance in this species is a result of genetic plasticity in the species rather than the more usual, development of distinctive genotypes. This species is native and common in western Tasmania.

To date, attempts to breed and select metal tolerant plants have been largely confined to cereals (Scott and Fisher, 1989).

2.8 The response of ecosystems to acidic smelter emissions

2.81 Smelters worldwide

The impacts of industrial emissions on ecosystems are often particularly evident near large point sources of air pollution such as steelworks, power stations and coal, coke or fuel-oil fired base-metal smelters. For example, large areas of degraded land or contaminated vegetation and soils have been reported in the vicinity of copper smelters based on sulphide ores at Ashio, Tochigi-ken (The Daily Yomiuri, 1993), Ducktown, Tenessee (Quinn, 1992), Sulitjelma, Norway (Løbersli and Stinnes, 1988), Flin Flon, Manitoba (Hogan and Wotton, 1984), Superior, Arizona (Wood and Nash, 1976), Sudbury, Ontario (Rutherford and Bray, 1979; Hutchinson and Whitby, 1974) and Copper Hill, Tennessee (Hedgecock, 1914). Typically, a zone of severe alteration or impact forms, distended in the direction of flow of the prevailing wind (Hogan and Wottan, 1984; Merry *et al.*, 1981; Little and Martin, 1972). This zone is usually surrounded by zones of less severe impact (Smith, 1974). In terms of severe impact, the smallest of the above examples represented an area of some 6 900 hectares. The reported effects on vegetation of prolonged exposure to high sulphur content emissions from copper smelters range from the elimination of sensitive species to complete deforestation. In many cases deforestation has lead to extensive soil loss and desertification. At the nickel/copper smelter at Sudbury, Ontario, Hutchinson and Whitby (1974) reported the elimination of SO_2-sensitive species, such as the Eastern White Pine (*Pinus strobus*), over vast areas subjected to fumigation dating back to the 1880's. Similarly, Hogan and Wotton (1984) reported that 50 years of SO_2 emissions had resulted in extensive tree mortality and reduced species diversity in a mixed conifer-deciduous forest adjacent the copper/zinc smelter at Flin Flon, Manitoba. In northern Norway, Løbersli and Steinnes (1988) reported heavy reductions in species diversity and density in the vicinity of a Cu smelter emitting approximately 20 000 tonnes of SO_2 annually. In upland Arizona, Wood and Nash (1976) correlated reduced species diversity and density with distance from a recently decommissioned, copper smelter at Superior. Annual species, completely absent at 0.4 km, were more affected than perennial species.

The exposure of vegetation to elevated levels of sulphur, however, does not necessarily result in a loss of species diversity. Preston (1988) reported that, in comparison to relatively unpolluted areas, species diversity in a sage scrub community close to an oil refinery established for 25 years increased with an associated shift in favour of annual plants. Preston likened the shift in relative abundance to that occurring after fire and considered the overall effects of chronic SO_2 stress as retrogression; stress resulting in altered species diversity and structure suggesting a reversal of succession (Whittaker and Woodwell, 1978).

2.82 Soil acidification

Although the deposition of sulphur dioxide and sulfate particulates from base metal smelters has been linked to soil acidification (Wood and Nash, 1976; Hutchinson and Whitby, 1974; Whitby and Hutchinson, 1974), the acidification of a soil system appears to be site-dependent. For example, a number of authors have found no direct relationship between soil acidity, as measured by pH, and displacement in the vicinity of Cu smelters despite a minimum of 50 years of operation (Hogan and Wottan, 1984; Freedman and Hutchinson, 1980). Freedman and Hutchinson (1980) suggested that impact of acidic smelter emissions on soils was largely dependent upon the buffering capacity of natural soils and base cation inputs from litter. The contribution of pollutant cations to acid-buffering capacity, and significant sulfate leaching losses were also proposed (Freedman and Hutchinson, 1980).

2.83 Metal contamination of soils and vegetation

Many studies undertaken in the vicinity of base-metal smelters have indicated that metal contamination of soils has occurred (Gabriel, 1994; Løbersli and Steinnes, 1988; Freedman and Hutchinson, 1980; Hogan and Wottan, 1984; Hazlett. *et al.*, 1983; Martin *et al.*, 1982; Freedman and Hutchinson, 1980; Hutchinson and Whitby, 1974; Little and Martin, 1972). In some instances contamination has been recognised at distances exceeding 10 kilometres from a source (eg. Freedman and Hutchinson, 1980). The majority of these studies also showed that soil metal concentrations declined markedly with increasing distance from a source (Løbersli and Steinnes, 1988; Hogan and Wottan, 1984; Hazlett. *et al.*, 1983; Freedman and Hutchinson, 1980; Hutchinson and Whitby, 1974).

A number of these studies have found similar negative correlations between metal concentrations in foliage and increasing distance from a source (Løbersli and Steinnes, 1988; Freedman and Hutchinson, 1980; Hutchinson and Whitby, 1974). However, metal accumulation in the foliage of plants exposed to smelter emissions appears to be species-dependent. For example, Hazlett *et al.* (1983), found correlation between metal concentrations in foliage and distance from the source in *Agrostis scabra* but not in *Betula pubescens*.

2.9 Assessment of smelter-polluted terrestrial ecosystems

2.91 Classification

Van Haut (1970) and Smith (1974) classified the effects of smelter pollution on ecosystems around a point-source of emission into successional or impact zones. Smith (1974) formed three classes: Class 1 (low pollution load), Class 2 (intermediate pollution load) and Class 3 (high pollution load). In Class 1 the ecosystem acted as a sink for contaminants and nutrients, with individuals showing little or no physical or physiological alteration. In Class 2 reduced growth, reduced reproduction and increased morbidity might occur. The outcome for the ecosystem might be reduced productivity, altered species composition and increased susceptibility to insect and pathogens. In Class 3 individuals suffered acute morbidity or death. Species loss might be accompanied by nutrient and soil impoverishment, altered microclimate and hydrology, and result in severe ecosystem simplification.

Rutherford and Bray (1979) broadly classified the damaged and eroded soils at Coniston in terms of topographical and morphological differences.

2.92 Plant and soil chemical analysis

Most studies of terrestrial ecosystems exposed to smelter emissions over a prolonged period have been concerned with the extent and degree of soil contamination by heavy metals (e.g. Løbersli and Steinnes, 1988; Hogan and Wottan, 1984; Hazlett. *et al.*, 1983; Martin *et al.*, 1982; Freedman and Hutchinson, 1980; Hutchinson and Whitby, 1974). This approach has been adopted due to the persistence displayed by most metallic elements in soils in comparison to their residence times in vegetation (Bowen, 1975). Martin *et al.* (1982) considered soil the ultimate sink for heavy metals in an ecosystem.

Contamination in the vicinity of operational smelters has been also investigated by the analysis of metal concentrations in litter (Martin *et al.*, 1982), in leaf tissue, (Løbersli and Steinnes, 1988; Hogan and Wottan, 1984; Hazlett. *et al.*, 1983; Martin *et al.*, 1982; Freedman and Hutchinson, 1980; Hutchinson and Whitby, 1974; Little and Martin, 1972) and in seed (Merry *et al.*, 1981), usually along transects or downwind of a source. Some researchers have used correlation analyses as a means of identifying trends and sources of contamination (eg. pollutant x distance and pollutant x pollutant correlations; Hogan and Wottan, 1984; Freedman and Hutchinson, 1980). Relationships have also been identified between metal concentrations in foliage and their corresponding levels in soils, although most researchers point out that such relationship may not necessarily be causal (Løbersli and Steinnes, 1988; Hazlett. *et al.*, 1983; Hogan and Wotton, 1984; Little and Martin, 1972).

2.93 Assessment of metal contamination

Researchers have sought biologically relevant measures of heavy-metal toxicity. In general, the assessment of the effects of metal toxicants on biota has been approached in two ways:

1) by field sampling soil or organisms that have been exposed to contamination and examining the concentration or distribution of contaminants in and between soil and organisms. This approach is usually accompanied by correlation analysis or calculation of a concentration ratio.

2) by the adding of specific compounds, extracts or whole soils of known contaminant concentration to test organisms and measuring the responses of those organisms. These experiments are usually performed in pot or laboratory bioassay trials, as the artificial environment minimises the risk of uncontrolled influences masking the relationship between a contaminant and a test organism.

Neither approach is ideal. The former suffers from a lack of demonstrated causality while the latter is potentially misleading if the principles are applied to a whole environment without qualification. The two approaches are considered in more detail below.

Early work on soil contamination relied on the total concentration of metals in the soil. However, it was realised that these may have little direct biological significance, as a large proportion of a total determination may be in a form unavailable to organisms. In order to overcome this limitation, researchers investigating metal contamination have used alternate soil extractants to the strong acids used for total digests.

Examples of commonly used extractants for metals are DTPA (Hogan and Wottan, 1984), ammonium acetate (Freedman and Hutchinson, 1980), water (Freedman and Hutchinson, 1980; Whitby and Hutchinson, 1974). DTPA, a chelating agent, is a weak acetic acid that is understood to remove micronutrient cations and water-soluble constituents adsorded on solid phases and may simulate the action of plant roots (Rayment and Higginson, 1992). DTPA extraction has been used to assess the micronutrient status of soils including Cu and Zn (Lindsay and Norvell, 1978). Ammonium acetate and water are weak extractants. These extractants are considered to better represent plant-available metals. In comparison to total determinations, the use of weak extractants has seen improvements in concentration-response correlations. However, interpretation remains limited to the degree to which these extractants mimic the chemical environment experienced by organisms.

Toxicity assessment by soil and plant chemical analysis has seen the derivation of the concentration ratio. This is based on the recognition that plants may either exclude or accumulate metals from soils (Beckett *et al.*, 1977). Sheppard *et al.* (1992) provide an example of a concentration ratio: the metal concentration of the edible portion of a plant to the concentration in the plough layer. However, as with attempts to correlate metal concentrations between foliage and soil, this model of contaminant transfer via the root system assumes linearity between soil and plant concentrations. The assumption may not be reasonable (Sheppard and Evenden, 1988a).

Laboratory bioassay studies have been used to examine alterations in the response of sensitive organisms when exposed to a range of metal concentrations. Under controlled conditions, this form of bioassay has permitted the identification of upper critical levels: the minimum concentration of essential or non-essential elements, either in solution or in plant tissue, at which toxic effects become apparent (Beckett and Davis, 1977). Van Assche and Clijsters (1990) and Whitby and Hutchinson (1974) provide two examples of the use of bioassays in order to assess soil contamination originating from smelters. For example, Van Assche (1990), found a strong negative correlation between water-soluble Zn in a contaminated soil and shoot length in two week old *Phaseolus vulgaris*. However, while numerous organisms have been selected for soil ecotoxicological work, such as work on the growth and reproduction of earthworms (van Gestal et al., 1988), Sheppard (1992) states that the aspect of soil and aquatic bioassay research that meets with most agreement is that no single test is sufficient to express toxicity. This is because the concentrations at which organisms express toxicity symptoms vary between species. For example, in a review of aluminium toxicity to vascular plants, Andersson (1988) records the great differences in Al tolerance found both between and within species. This has led to attempts to specify the amount of data required to set useful values of soil quality using soil toxicology data for a number of different taxonomic groups (van Straalen and Denneman, 1989), and, in some instances, the advocacy of a bank of soil toxicity tests (Wang and Bartha, 1990; Sheppard *et al.*, 1992a).

In theory, upper critical levels derived from bioassays under controlled conditions could be identified for any number of the biota within an ecosystem. However, as the influence exerted by a toxic element is highly dependent upon its environment, the environmental relevance of laboratory-based, critical levels may be extremely questionable. Beckett and Davis (1977) made considerable progress in the search for an index of toxicity for higher plants that was independent of the environmental conditions under which the organism was grown. These authors considered yield alone a poor index of toxicity as the height of the yield-concentration plateau depended on many factors such as the growth conditions. Moreover, they reasoned that, as the toxic effects of a given concentration of an element in the soil solution also depended on numerous factors, this too is likely to be a poor index of toxicity. Conversely, they showed that the tissue concentrations of metals in barley seedlings at the upper critical level were almost independent of yield under various growing conditions and devised a measure of elemental accumulation relative to the onset of toxicity. Although little use appears to have been made of this measure, foliar analysis has been used in attempts to diagnose levels of contaminants and has generally met with support among researchers (Balsberg Påhlsson, 1989).

While soil and plant chemical analysis, concentration ratios and laboratory-based, critical levels provide useful guidelines, especially where the response of a range of taxa are examined, it is impossible to stipulate field-threshold values above which metal-phytotoxicity appears (Balsberg Påhlsson, 1989; Andersson, 1988). This is because the influence exerted by heavy metals on organisms may be controlled by genetic, environmental, growth stage and toxicological factors (Sheppard *et al.*, 1992). Schuster (1991) discusses the dilemma of extrapolation of laboratory toxicity results to the field. Sheppard *et al.* (1992) conclude that the effects of soil properties on the toxicity of contaminants are poorly known and that judgement is needed to define unacceptable levels of natural toxic elements.

2.10 Definitions in reclamation

Restoration, rehabilitation and replacement

Irrespective of the nature of a degraded environment three general approaches to vegetation management are recognised: restoration, rehabilitation and replacement (Bradshaw, 1992). These terms require some clarification.

Restoration implies a return to pre-existing conditions; that is, a return to the original species composition of a community accompanied by the re-establishment of ecosystem structure and function. At the very least, restoration requires a detailed knowledge of species autecology, inter-relationships and their contribution to ecosystem development. This type of information is not always known for community dominants, let alone a plant community as a whole. Although Bradshaw (1992) believed that restoration is possible, it remains an ultra-specialized goal, and is rarely, if ever achieved in drastically disturbed land (Fox, 1984).

Rehabilitation is generally understood to require a reversal of degenerative processes. Although not always acknowledged, the longer term objectives must involve the re-establishment of ecosystem function, as this will determine the durability of the new community. Understandably, a universal component of a successful rehabilitation strategy is a return to soil stability. This is commonly achieved by a combination of mechanical and vegetative means. There is, however, no requirement that a newly established community be similar to the original.

Rehabilitation does not require the return of a pre-existing diversity, nor does it necessarily imply the use of indigenous species. The strategy may involve the partial return of an indigenous community, or may rely exclusively on the use of exotic species, such as cultivated legumes and grasses. This is known as rehabilitation with replacement. In some situations provision of a cover crop has been beneficial. Since the complexity of the original ecosystem is not required, replacement should be the easiest option (Bradshaw and Chadwick, 1980). The adoption of a strategy of rehabilitation does, however, imply the return of productivity to a site. In certain circumstances, this may exceed that of the original community. The term reclamation is generally understood to include both restoration and rehabilitation.

Reclamation strategies and neglect

It is possible to adopt two broad strategies in regard to degraded land: reclamation and neglect. Reclamation includes any type of physical, chemical or biological ameliorative treatment in order to achieve an acceptable level of site productivity, species richness or ecosystem function. The objectives of reclamation may include restoration, rehabilitation or replacement. Neglect relies on the natural redevelopment of site stability and complexity under the influence of weathering, nutrient accumulation and successional processes.

Neglect may be a viable option on sites previously occupied by well-developed communities, or where nutrient and environmental conditions remain favourable (Bradshaw, 1992). Bradshaw and Chadwick (1980) concluded that in the temperate, northern hemisphere the complete return of ecosystem complexity may occur within 100 years, but observed that, in other situations, where special limiting factors such as acidity and metal toxicity occur, there may be almost no colonisation after the elapse of 50 or 100 years. The strategy of neglect can also lead to further site degradation. Some examples of situations where neglect may lead to further degradation are sites exposed to increased fire frequency, wind or water erosion, or subject to weed infestation.

Under certain circumstances, neglect may result in outcomes similar to those achieved by reclamation. Consequently, either strategy may be a viable option. The selection of an appropriate strategy for a degraded site can only be made following a consideration of the desired objectives of reclamation in relation to existing knowledge of that environment and proven methods of reclamation.

2.11 The amelioration of metal toxicity in acid soils

Organic matter

The amount of organic matter in a soil has been found to be a major factor in determining the toxicity of metals to plant growth (Thomas and Hardgrove, 1984; Hardgrove and Thomas, 1981; Elliot *et al.*, 1979; McLean, 1965). This is due to the high affinity between organic matter and some metals (Livens, 1991; McBride, 1989). This suggests that adding organic matter to contaminated soils should reduce metal toxicity by reducing availability. Hue *et al.* (1986), for example, found that various organic acids were differentially able to ameliorate Al toxicity in soil solutions.

Organic matter amendments, such as plant residues, wood chips, composted tree bark and papermill sludge, have been used to reclaim degraded soils (Logan 1992) and sewage sludge has been applied to Cu and Zn smelter contaminated soils (Berry, 1986; Berry 1985, Franks *et al.*, 1982) with some success. In the case of sewage sludge, care must be exercised not to introduce additional contaminants with the ameliorant.

Neutralising amendments

The fraction of an organically complexed metal, such as Cu^{2+}, relative to the free form increases at higher pH (McBride, 1989). Similarly, Evans and Kamprath (1970) found that as the organic matter content of soils increased, less Al^{3+} was present in the soil solution for a given pH. This type of evidence suggests that pH control of metal sorption/desorption and precipitation/dissolution mechanisms may ameliorate metal toxicity by reducing availability.

Field applications of calcium-based materials or limes, such as calcium carbonate, to acid, metal contaminated lands have reportedly reduced toxicity (Clements *et al.*, 1968), initiated natural plant colonisation (Winterhalder, 1991; 1981a; 1981b), ameliorated nutrient deficiencies (Rodinkirchen, 1992) and aided colonisation by acid-intolerant vascular species (Rodenkirchen, 1992). Winterhalder (1991; 1981a; 1981b) considered the addition of lime a trigger factor for the initiation of natural plant colonisation by birch, aspen and willow. In disturbed lands affected by strip mining, the addition of lime has resulted in the improved survival of N_2 -fixing, woody species on highly acidic spoils (Carpenter and Hendsley, 1979). The application of lime has been found to moderate the effects of acid rain on the understory species of European coniferous forests subjected to simulated H_2SO_4 rain (Rodenkirchen, 1992). Lime application is understood to ameliorate metal toxicity by enhancing sorption and precipitation (Logan, 1992; Sopher and Baird, 1982)

Some studies have shown that gypsum ($CaSO_4.2H_2O$) can neutralise acid subsoils (Oates and Caldwell, 1985; Pavan *et al.*, 1984; Reeve and Sumner, 1972) and stimulate growth by reducing metal toxicty (Adams and Hathcock, 1984; Pavan *et al.*, 1984) or increasing Ca levels (Adams and Hathcock, 1984).

Phosphorus

Reductions in Al toxicity with added P have been reported in nutrient solution studies (Alva *et al.*, 1986; Blamey *et al.*, 1983) and with soil-grown plants (Bache and Crooke, 1981; Awad *et al.*, 1976). In the vicinity of phosphate fertilizer particles, local low pH conditions and high phosphate concentration may cause the dissolution of clays and the precipitation of Al phosphate (Huang, 1988). However, in highly weathered soils with low P availability the usefulness of P applications as a means of overcomming Al toxicity can depend on P-fixing capacities (Fox and Searle, 1978).

2.12 Site histories: the rehabilitation of acid, metal-contaminated terrestrial ecosystems

The methods used to reclaim lands altered by prolonged exposure to emissions from copper smelters have tended to be site specific. The reasons for this are manifold. Firstly, it is apparent from the above discussion of the literature that the significance of smelter emissions on an environment are likely to vary not only with plume composition and deposition patterns but with the chemical and physical characteristics of the environment in which the deposition occurs, and with the tolerance of a local biota to the altered conditions of growth. Secondly, reclamation methods must be scale-appropriate. Factors of scale, for example, might include the practical limitations presented by a local topography to a mechanised method of ground preparation or plant re-introduction. Thirdly, a newly established community may be similar or otherwise to the vegetation that preceded it. However, as similarity does not necessarily confer acceptability, some recourse to subjective judgement may be required. For example, the introduction of a non-native species might be judged acceptable only after consideration of the site within a local context. At sites where reclamation has taken place over many years shifts in the cultural perception of acceptability may occur - alongside technological advances. Even without cost as a consideration, all of these factors are likely to influence strategic choices in reclamation.

In a few cases reclamation works have evolved over decades of trial and error. The following are examples of reclamation works in areas denuded by acidic pollution from copper smelters. All have reclamation histories spanning at least twenty years. The examples are geographically and topographically distinct and demonstrate how the environment of deposition can affect the approach taken toward reclamation. They are introduced in order of increasing topographic severity.

The best known and most studied example of a denuded and highly contaminated environment due to prolonged exposure to emissions from a nickel/copper smelter occurs on the relatively featureless Laurentian Shield at Sudbury, Ontario. A century of SO_2 fumigation, particulate deposition, lumbering and fire resulted in 10 000 ha of barren land and 36 000 ha of stunted open birch-maple woodland (Winterhalder, 1988). Contamination occurred initially from nickel-copper roast beds operating before the turn of the century, and later, from smelters exhausted through stacks and superstacks. Amiro and Courtin (1981) divided the residual communities into two groups, namely: the barren, birch transition and maple transition group and the hemlock, white pine and northern hardwood forest.

In the early 1970's 6 000 tree, shrub and grass seedlings were planted using student labour (Lautenbach and Winterhalder, 1979). Experimental trials examined the application of lime, phosphorus and nitrogen (Balsille and Winterhalder, 1978). Mulches and composts were also tested as soil ameliorants. Liming, fertilizing and grass seeding was undertaken (Hume, 1983). Winterhalder (1991; 1981a; 1981b) considered the addition of lime a trigger factor for the initiation of natural plant colonisation by birch, aspen and willow. Between 1978 and 1990, 3 000 of the 10 000 pollution-affected hectares at Sudbury were reseeded with grass and planted to 1.2 million Jack and Red Pine seedlings by Inco Limited at a cost of Can.$14 million (Canadian Geographic, 1991). Broadacre liming was undertaken from the air.

At Ducktown, Tennessee, copper roasting, smelting and timbergetting that began during the latter part of last century resulted in 6 000 to 9 000 ha of seriously eroded or degraded land (Quinn, 1992; Tyre and Barton, 1986). When reclamation planting began in the 1930's Hursh (1948) estimated that 2 833 ha were bare land and 6 880 ha deforested grasslands. Some early works required replanting three times (Tyre and Barton, 1986). The revegetation trials of Berry

(1979) and Berry (1982; 1985) used deep subsoiling (60+ cm) in combination with municipal sewage or fertilizer tablets to improve the survival and growth of hand planted *Pinus taeda* (Loblolly pine) seedlings. *Pinus virginiana, Robinia pseudoacacia* and other hardwood species were also planted experimentally (Quinn, 1992; Berry, 1983). In subsequent broadacre plantings ground preparation specified the use of contour subsoiling, 90 cm deep to 1.2 m centers, followed after settling, by planting and pellet fertlizer (Tyre and Barton, 1986). However, these methods, could not be used where the topography was steep or gullied. By 1986 the denuded area had only been reduced in size by about 400 ha. (Tyre and Barton, 1986). Aerial sowing of grasses was also tried in limited areas. Plantings of Loblolly pine and sowings of *Eragrostis curvula* (Weeping Love grass) were considered a fairly successful combination (Quinn, 1992).

At Ashio, Tochigi-ken (Japan), 12 000 hectares in a mountainous region, that became known as the "Japanese Grand Canyon", were deforested as a result of pollution from the Furukawa Co. copper smelter and associated timber-getting (The Daily Yomiuri, 1993). Reclamation began in 1956. In contrast to the Canadian and American approaches, the current methods involve either 9 pegging bags filled with seed, fertilizer and soil along narrow, purpose-constructed terraces or, in the steepest terrain, broadcast-sowing grass and tree species by helicopter with dilute asphalt as a stabilizer. To date (1993) reclamation costs have amounted to Y$2 billion.

Chapter 3

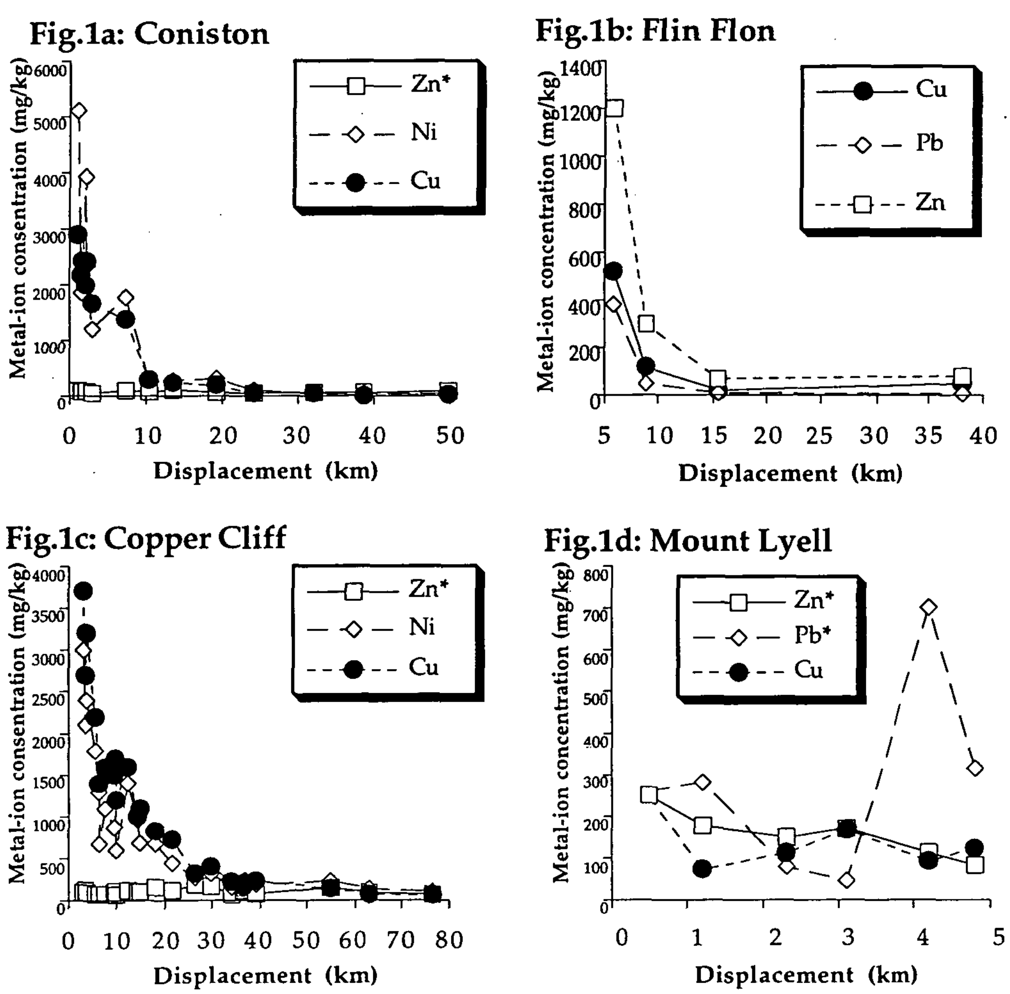
Soil characteristics along transects of increasing displacement from the Mount Lyell smelter ruins

3.1 Introduction

Soils in the vicinity of acid producing, base-metal smelters have been studied along transects originating at a point-source of emission (Hogan and Wotton, 1984; Freedman and Hutchinson, 1980; Rutherford and Bray, 1979; Wood and Nash, 1976; Hutchinson and Whitby, 1974). Many of these studies describe concentration-displacement patterns that indicate particulate metal contamination and its source (Fig.1). In some soils, acid-deposition has led indirectly to profound chemical alteration, principally as a result of acid-neutralising reactions in the exchange complex. These reactions are known to lead to cation leaching (Wookey and Ineson, 1991) and a depletion of the base cation reserve (Dahlgren, 1990). Acid-deposition may result in pH-dependant shifts in the solubility, abundance and speciation of metal-ions (Malmer, 1976), causing alterations in the availability of soil metals to plants. Alterations to metal-ion availability can result in phytotoxicity (eg. Kramer, 1969).

Despite apparent similarities between terrestrial environments exposed to emissions from base-metal smelters, impacts on soils and soil chemistry resulting from the deposition of metals and other smelter contaminants are likely to be unique. For example, some studies have

Figure 1: Surface soil, metal-ion concentrations (total) along transects originating at the smelter stacks of four base-metal smelter sites. At the time of sampling each of the sites had been exposed to base-metal smelter emissions over a duration of at least 50 years. Non-target metals can be considered controls and are identified by an asterisk. The data for Figures 1a-1d have been extracted from Hutchinson and Whitby (1974), Hogan and Wotton (1984), Freedman and Hutchinson (1980) and Kirkpatrick (unpublished), respectively.



identifed soil acidification (Wood and Nash, 1976; Hutchinson and Whitby, 1974; Whitby and Hutchinson, 1974), but others have found no direct relationship between soil acidity and displacement, despite many years of deposition (Hogan and Wottan, 1984; Freedman and Hutchinson, 1980). Discrepancies have been attributed to interactions between contaminants and the receiving environment. Freedman and Hutchinson (1980) suggested that impact of acidic smelter emissions on soils was largely dependant upon the buffering capacity of natural soils and base cation inputs from litter. Other authors have also indicated that soils vary in ability to adsorb acid deposition (Reuss *et al.*, 1987; Krug and Fink, 1983; Richie and Posner, 1982; Heute and McColl, 1984; Malmer, 1976).

The history of environmental impacts at Mount Lyell, however, has not been confined to the effects of smelter emissions on soil types. In the period between smelter establishment and closure, whole soil profiles were eroded to subsoils over much of the mountain. Amounts of 10 M tonnes of lost top soil have been calculated (Locher, 1995). Erosion on this scale, much of which is thought to have occurred in the early years of the development, resulted in significant off-site redistribution of both soils to the lower reaches of the King River and Macquarie Harbour. Soil contaminants are likely to have been similarly redistributed. Against this background of soil erosion, the exposed, but residual subsoils were continuously exposed to acidic, Cu smelter emissions until smelter closure in 1969. Consequently, the status of soil contamination at Mount Lyell remained in question, but was likely to have been greatly influenced by contaminant redistribution and the capacity of residual soils to buffer the deposition of acidic, Cu smelter emissions.

This chapter describes some chemical and physical characteristics of the Mount Lyell soils along two radial transects originating at the smelter ruins. It is concerned with the identification of metallic contaminants surrounding the smelters, their concentrations, distribution and source. The investigations were confined to the surface soils as these have been shown to contain the highest concentrations of metals in contaminated profiles (Freedman and Hutchinson, 1980; Rutherfofd and Bray, 1979; Wood and Nash, 1976; Tyler, 1975). Soil chemical characteristics, such as pH, total exchangeable bases and available metals, provided a means of assessing soil alteration due to the deposition of acid-forming and particulate smelter emissions.

In the absence of universally-accepted diagnostic methods, the extractant used in the determination of available metal-ion concentrations was conservative, and chosen to best reflect bio-availability. The soils were analysed for soluble concentrations of both target (i.e. Cu) and non-target (i.e. Al and Zn) metals. McCray and Sumner (1990) considered the concentration of the soil solution to provide a direct measure of the conditions experienced by plant roots. Baker *et al.* (1988) and Bruce (1988) indicated that, in comparison to conventional methods, the soil solution provides a more accurate means of assessing metal-ion toxicity. Freedman and Hutchinson (1980) and Whitby and Hutchinson (1974) used soil-water extracts to assess soil toxicity in lands contaminated by acid-producing, base-metal smelters.

3.2 Methods

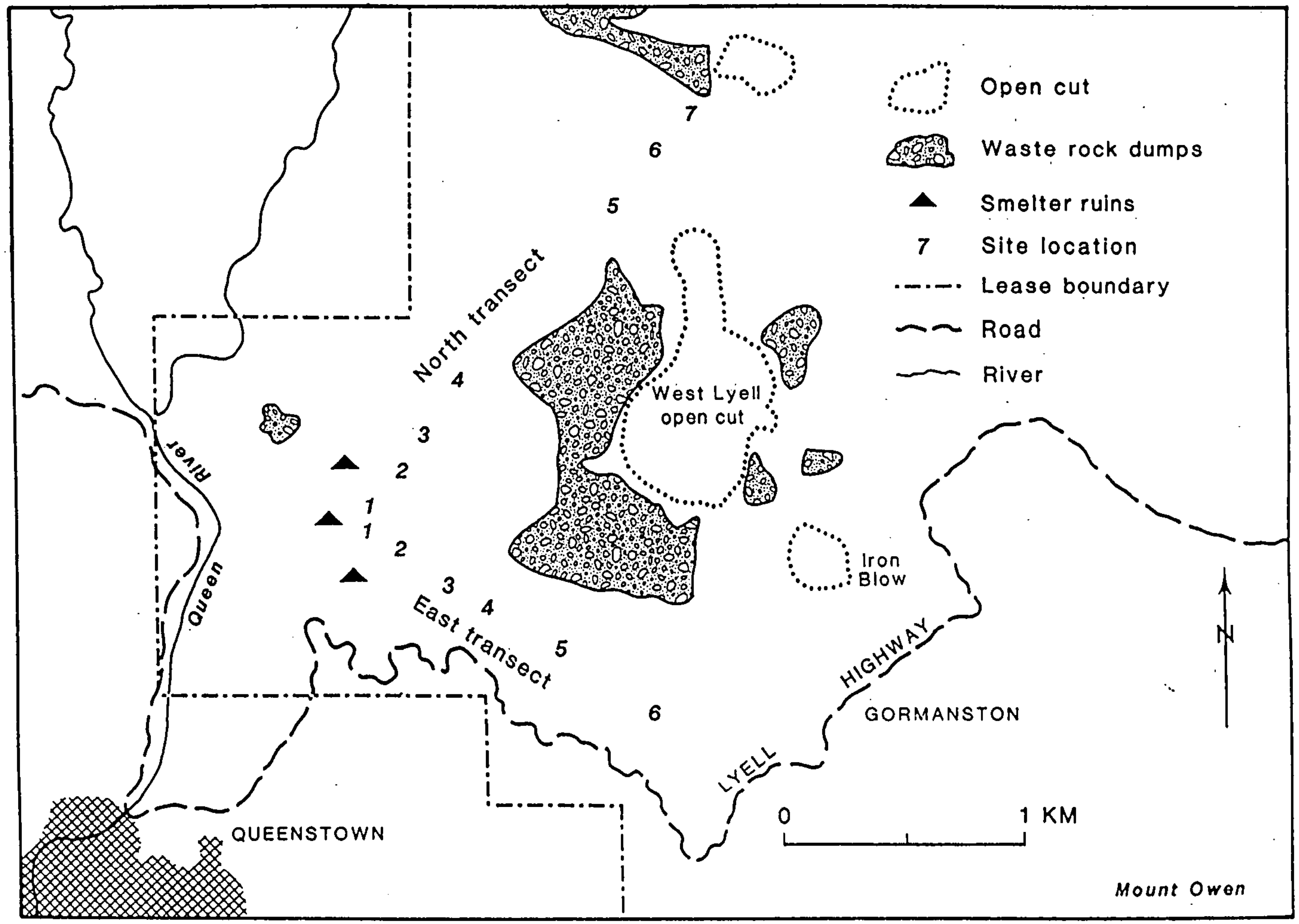
3.21 Soil sampling

Soil samples were taken along two radial transects originating at the the Mount Lyell smelters (Fig. 2). The transects were confined to the Mount Read volcanic sequence. An easterly transect, following the direction of the prevailing winds, rose from an initial elevation of 230 m close to the smelter site, to 480 m at a distance of 1.71 km. A similar, northerly transect, rose from 220 m, to an elevation of 530 m at a distance 2.32 km. The sample sites were located on ridgetops or slopes in direct line-of-sight of the smelters.

At each site, a composite soil sample was prepared from 10 to 15 randomly located subsamples. The subsamples were excavated to a depth of 2 cm. Sufficient soil was removed at each location to provide a subsample of approximately 2 kg. The samples were packed into 20 litre plastic buckets with close-fitting lids for transport.

Samples of exposed Mount Read Volcanic subsoils (C horizon) were collected similarly from several sites at least 5 km removed from the smelters. These were considered uncontaminated and used as an indication of background levels. Composite samples were collected using the same methods as at the transect sites.

Fig.2 Soil sample sites along two radial transects originating at the the Mount Lyell smelters



3.22 Sample preparation, soil analysis

The laboratory preparation of the soil samples followed the recommendations suggested by Rayment and Higginson (1992). The soil samples were initially broken up by crushing between sheets of polyethylene. The samples were then coarse sieved (1 cm nylon mesh), thoroughly mixed and transferred to clean, 20 litre plastic buckets for cold storage (2-3°C) and use in subsequent pot trials (Chapter 5). A 2 kg portion of each sample was reserved for laboratory determinations.

The laboratory samples were divided in a sample divider into two halves of approximately equal mass and allowed to air dry. One portion was sieved (6 mm brass mesh) and a subsample subjected to pH measurement. A soil/water suspension (1:1 soil/distilled water by weight) was prepared and, after shaking, pH measurement performed with the aid of a WTW electronic pH meter (model pH6) fitted with a Type E 50 electrode. Prior to pH measurement the meter was standardised to the manufacturer's specifications. After additional sieving (2 mm mesh), another subsample was reserved in sealed plastic bags for total nitrogen determination using the Kjeldahl procedure (Bremer and Mulvaney, 1982). Total nitrogen was determined by the Mount Pleasant Laboratories, Kings Meadows, Tasmania. Soil texture was determined using a rod and ribbon method and a key based on Northcote (1979). The remaining soil was stored at 2-3°C in labelled plastic bags as reference material.

The other half of each soil sample was used to determine selected soil chemical attributes. As the samples were to be subjected to metal-ion determinations, two precautions were taken in order to minimise external contamination during sample preparation; prior to use all the utensils were rinsed in a solution of 5% nitric acid (14M analytical grade) and plastic-bodied, nylon-mesh sieves were used throughout. The samples were sieved (6 mm nylon mesh) and air-dried to constant weight on polyethylene lined aluminium trays at 40 °C in a fan-forced oven. Drying periods varied with the composition of each sample and in some cases extended over several days. After drying, each sample was sieved (2 mm nylon mesh) and sealed in labelled, plastic bags.

The air-dry samples were used to estimate soil moisture (ODM), organic matter content (LOI), total exchangeable bases (TEB) and metal-ion concentrations. Samples weighing approximately 4 grams were transferred to clean crucibles of known mass and the air-dry mass of each calculated. The samples were then placed in an oven for a period of 6 hours at 105°C. After drying to constant weight and reweighing at room temperature, the moisture content of each sample was calculated using the percentage method of Rayment and Higginson (1992).

Organic matter content was determined by loss-on-ignition (LOI). Oven-dry (105°C) samples of known weight (approximately 2 g) were placed in pre-weighed, high-temperature crucibles and heated to 450 °C in an high temperature oven for a period of 6 hours. Each sample was reweighed at room temperature and the difference in sample mass used to estimate percentage organic matter content.

Total exchangeable bases (TEB) were determined by the Mount Pleasant

Laboratories, Kings Meadows, Tasmania.

A weak extractant, distilled water, was used to determine the soluble metal-ion concentrations. This extractant was chosen in preference to stronger extractants as many, such as the strong acid digests, are believed to have little direct biological significance (eg. Baker *et al.*, 1988 ; Bruce, 1988; Prevan *et al.*, 1982). The method used was similar to that of Freedman and Hutchinson (1980) and Whitby and Hutchinson (1974) during investigation of lands contaminated by acidic deposition from base-metal smelters.

Soil-water extracts were prepared from 66 grams of each sample, diluted 3 to 1 by mass with distilled water, and mechanically shaken in acid-rinsed, high-density polyethylene bottles for 3 hours. After standing for 24 hours, the extracts were decanted into clean polypropylene bottles. The extracts were sent to the Mount Pleasant Laboratories, Kings Meadows where they were preserved by acidification with 2 mL L^{-1} HNO_3 (ASTM, 1989b) and analysed by atomic absorption spectrometry (Varian 1275) for the metals Cu, Zn, and Al. Soluble metal concentrations were obtained in mg L^{-1}.

3.23 Data exploration

The physical and chemical attributes of the soils along each transect were tabulated and contrasted. Soil pH and metal-ion data was explored with the aid of regression analysis. Comparisons were drawn between the metal-ion concentrations recorded at Mount Lyell and those reported for soils from similar near-smelter sample sites. The concentrations at which metal-ions have reportedly resulted in growth abnormalities in the higher plants were documented.

3.3 Results

3.31 Soil characterisation

Two soil types, derived from similar volcanic parent materials, were distinguished by texture analysis in the survey area. The two types were broadly representative of the dominant volcanic geology on the western face of Mount Lyell and differed mainly with respect to quartz composition. On both transects, exposed fine-grained, chlorite-sericitic schists were located at sampling sites nearest the smelters (Table 1a and Table 1b). These formed exposed, clay-dominated subsoils, described as intermediate-mafic volcanic in origin (Mount Lyell Technical Review, 1993). Soils at the remaining sampling sites were sericite-chlorite and siderite schistose pyroclastics. These formed sandy loams, described herein as felsic volcanics. Small lenses of buried soil horizons or buried colluvial material were also noted in the survey area. Soil samples were not taken where these profiles were encountered.

All samples had an acidic reaction. The pH of samples on the easterly transect lay within half a unit (Table 1a). The lowest value, 4.3, was recorded 230 m from the smelter site, at Site 1 and the highest, 4.8, at 850 m (Site 4). There was, however, no clear pH-displacement relationship (R=0.26; Fig. 3a). For example, at 1150 m (Site 5), the most distant sampling

Table 1a and 1b: Surface soil characteristics along two radial transects originating at the Mount Lyell smelters.

Table 1a: Easterly transect

Sample sites	**1**	**2**	**3**	**4**	**5**	**6**
Displacement from smelters (m)	230	400	650	850	1150	1712
Soil texture[1]	medium clay	sandy clay	light sandy loam	sandy loam	sandy clay loam	sandy loam
Soil pH[2]	4.3	4.5	4.7	4.8	4.5	4.5
Loss on ignition (%)	3.9	3.7	3.1	5.3	7.0	5.6
% air dry moisture	0.6	0.7	1.5	2.0	2.6	1.5
N(%Total)[3]	0.78	0.90	0.37	0.47	0.35	0.76
TEB (meq $100g^{-1}$)[4]	1.25	0.99	0.73	0.76	0.95	0.76
Ca (meq $100g^{-1}$)[5]	0.36	0.29	0.16	0.18	0.20	0.19

Table 1b: Northerly transect

Sample sites	**1**	**2**	**3**	**4**	**5**	**6**	**7**
Displacement from smelters (m)	225	435	575	850	1837	2075	2325
Texture class[1]	medium clay	medium clay	sandy clay	light sandy loam	sandy clay loam	sandy clay loam	sandy clay loam
Soil pH[2]	3.8	4.1	4.1	4.5	5.7	5.2	5.2
Loss on ignition (%)	4.6	5.3	2.3	3.6	1.5	2.6	6.0
% air dry moisture[6]	0.6	0.7	0.6	1.1	0.2	0.7	0.7
N(%Total)[3]	0.78	0.65	0.62	0.47	0.35	0.62	0.84
TEB (meq $100g^{-1}$)[4]	1.11	1.02	0.88	0.76	0.95	0.78	0.79
Ca (meq $100g^{-1}$)[5]	0.29	0.34	0.21	0.20	0.28	0.31	0.29

1) Soil texture: Soil texture determination based on ANU Forestry texture evaluation guidelines derived from Northcote (1979).
2) Soil pH was measured in a 1:1 soil/distilled water mix using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode.
3)Kjeldahl procedure (Bremer and Mulvaney, 1982)
4) TEB = Total NH_4/CL exachangable bases
5) NH_4Cl extractable calcium
6) Air dry moisture (ODM); Rayment and Higginson (1992).

Figure 3: Mean soil pH along easterly (3a) and northerly (3b) transects originating at the Mount Lyell smelters. Composite samples were formed. Each sample was comprised of 10 to 15 subsamples (0-2cm depth). Soil pH was measured in a 1:1 soil/distilled water mix using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode. Regression lines are shown.

Fig. 3a: easterly transect

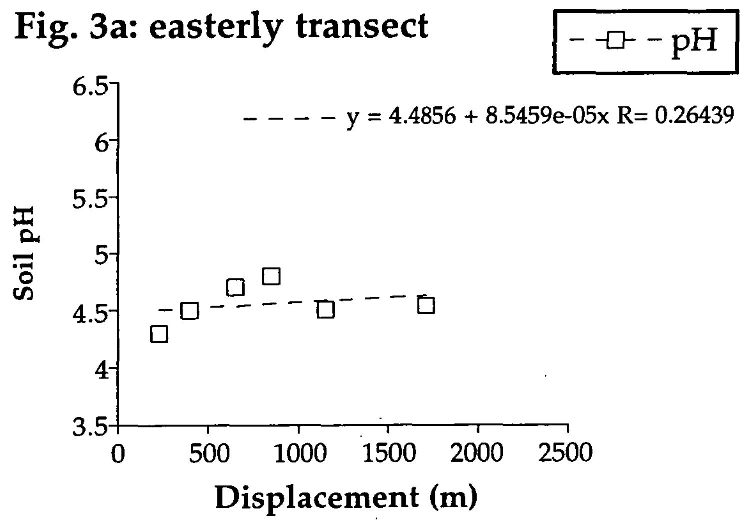
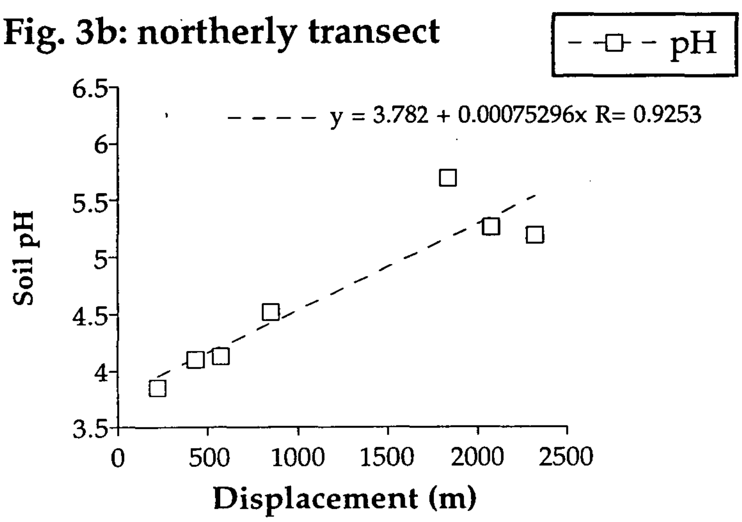


Fig. 3b: northerly transect



point on the transect, soil pH was similar to that recorded at 400 m. The mean soil pH for the transect was 4.6±0.2.

Samples from the northeasterly transect were similarly, moderately acidic, with a mean soil pH of 4.7±0.7 (Table 1b). The lowest pH recorded on this transect was 3.85 at Site 1. A linear correlation indicated that soil pH increased with increasing displacement (R=0.92; Fig. 3b). For example, between 850 m and 1837 m, soil pH increased approximately one unit to a maximum of 5.7 at 1837 m (Site 5). The mean soil pH for the three most distant sites on this transect was 5.4±0.3.

Sample sites along both transects recorded low or very low TEB, LOI and total nitrogen (Table 1a and Table 1b).

Concentrations of the target metal Cu were highest at sites nearest the smelters. The highest concentrations recorded were 10.41 and 9.60 mg L^{-1}, on the easterly and northerly transects, respectively. Concentrations declined rapidly with increasing displacement along both transects (Fig. 4a and Fig. 4b). The concentration-displacement patterns for Cu were strongly negative-exponential (R=0.95 and R=0.97 for the easterly and northerly transects, respectively) and background concentrations appeared to be reached between 1000-1500 m displacement. The mean Cu concentrations for the easterly and northerly transects (3.51±4.1 and 2.77±3.6, respectively) exceeded those of the background volcanics by factors of 12 and 9, respectively (Table 2).

Figure 4: Soluble copper and zinc concentrations for surface soils along easterly (Fig. 4a) and northerly (Fig. 4b) transects originating at the Mount Lyell smelters. Composite samples were formed. Each sample was comprised of 10 to 15 subsamples (0-2cm depth). Soil-water extracts (acidified) were made up with distilled water at a dilution rate of 3 to 1 by mass. Exponential regressions are shown.

Fig.4a: easterly transect

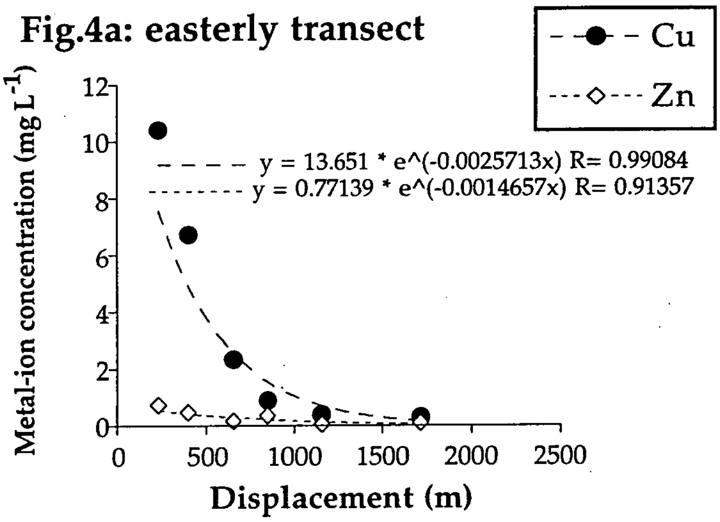


Fig. 4b: northerly transect

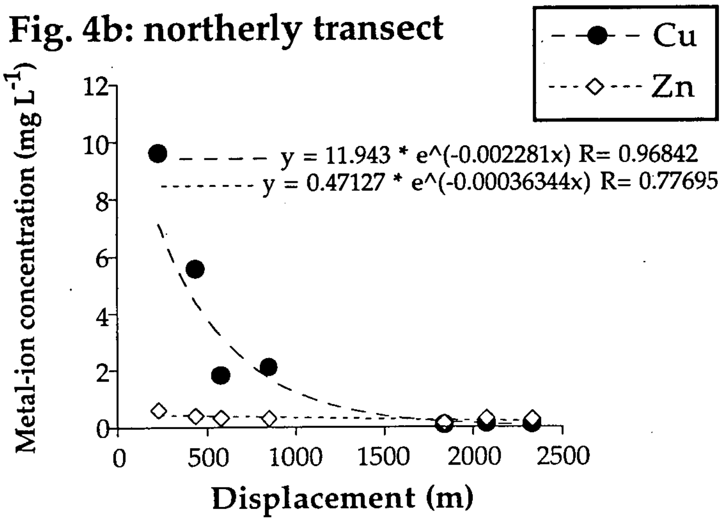


Table 2: A comparison of near-smelter transect data for soil-metal concentrations at Mount Lyell and similar soil data for Coniston[5] (Ontario) and Superior[6] (Arizona). Background data for the Mount Read Volcanics are shown.

	Mount Lyell (East)	Mount Lyell (North)	Mount Read Vol.	Coniston	Superior
Displacement from smelter (m)	230-1 700	225-2 325	~5 000	800-1 500	400-1 400
Sample depth (cm)	0-2	0-2	0-2	surface	surface
Soil pH	4.6 ±0.2[1]	4.7±0.7	4.2[2]	3.2 ±0.2[5]	~5.0[6]
Acid extractable total digest (ppm)					
Cu	160.5±123[4]	na	62[4]	2 527.0 ±516[5]	5 778.5±5 430[6]
Al	na	na	na	24 100.0 ±18 51[5]	na
Ni	na	na	na	3 480.0 ±2296[5]	na
Zn	215.0±52.3[4]	na	62[4]	86.4 ±14.0[5]	207±130[6]
Pb	272.5±13.4[4]	na	53[4]	60.5 ±31.2[5]	218±148[6]
Water extractable metals (mgL^{-1})					
Cu	3.51 ±4.1[1]	2.77±3.6[1]	0.3 ±0.2[1]	50.7 ±12.3[3]	na
Al	6.37±7.5[1]	8.10±9.8[1]	1.4 ±0.4[1]	63.8 ±18.7[3]	na
Zn	0.32±0.3[1]	0.33±0.1[1]	0.6 ±0.8[1]	0.9 ±0.1[3]	na
Pb	<0.6[1]	<0.6[1]	<0.6[1]	na	na
Nutrients and others					
LOI (% organics)	4.6 ±1.6[1]	3.7±1.7	15.6 ±2.5[1]	7.6 ±1.56[3]	na
TEB ($mEq100g^{-1}$)	0.9 ±0.2[1]	0.9±0.1	na	na	na
Total N (%)	0.06±0.02[1]	0.08±0.02[2]	na	na	na

1 This study: eastern transect, n=6 ; northern transect, n=7; Mount Read Vol., n=5
2 Kirkpatrick (1984); n=2
3 Whitby and Hutchinson (1974); n=2
4 Kirkpatrick (unpbl.) n=2
5 Hutchinson and Whitby (1974); n=2
6 Wood and Nash (1976); n=2

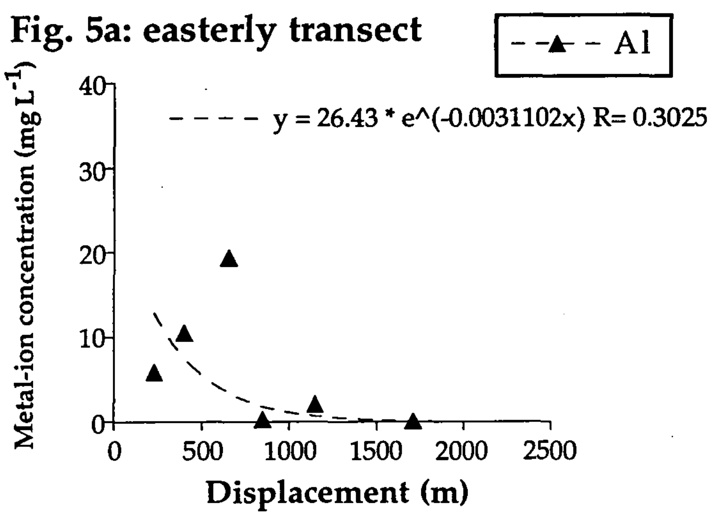
na = not available

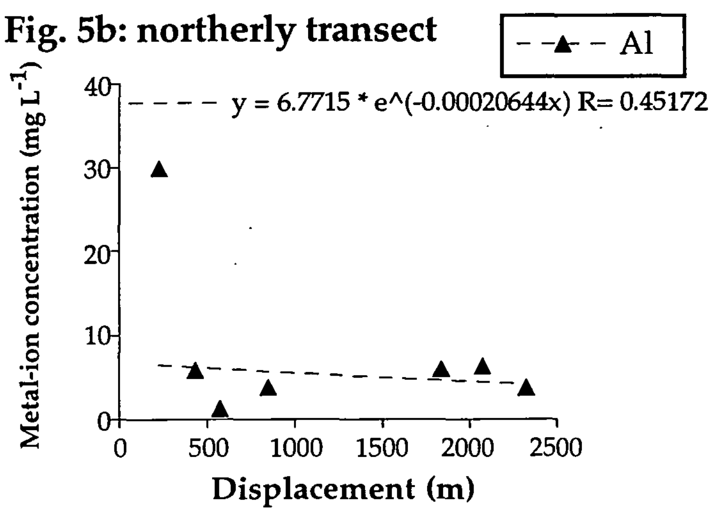
The non-target metal Zn displayed a similar concentration-displacement pattern to that of Cu along both transects (Fig. 4a and Fig. 4b). The highest concentrations occurred nearest the smelter site (0.75 and 0.60 mg L^{-1} for the easterly and northerly transects, respectively). Thereafter, Zn concentrations declined exponentially with increasing displacement (R=0.91 and R=0.78, respectively). Background concentrations were reached at displacements of between 500-600 m. The mean Zn concentrations for the easterly and northerly transects (0.32±0.3 and 0.33±0.1, respectively) did not exceed those of the background volcanics (Table 2).

Concentrations of the metal Al were highest at site 3 (19.3 mg L^{-1}) on the eastern transect and site 1 (29.9 mg L^{-1}) on the northern transect (Fig. 5a and 5b). Al concentrations on the easterly transect were not exponentially related to displacement (R= 0.30) and appeared to peak at a displacement of 650 m before falling to background levels. The concentration pattern for the northerly transect was also unrelated to displacement (R=0.45). An elevated concentration appeared to be confined to the site nearest the smelter. The mean Al concentrations for the easterly and northerly transects (6.37±7.5 and 8.10±9.8, respectively) exceeded those of the background volcanics by factors of 4.5 and 5.8, respectively (Table 2).

Concentrations of the non-target metal Pb approached detection limits (0.1 mg L^{-1}) and did not vary with displacement on either transect.

Figure 5: Soluble aluminium concentrations for surface soils along easterly (5a) and northerly (5b) transects originating at the Mount Lyell smelters. The samples were formed from composites, each comprised of 10 to 15 subsamples (0-2cm depth). Soil-water extracts (acidified) were made up with distilled water at a dilution rate of 3 to 1 by mass. Exponential regressions are shown.





3.4 Discussion

Soil acidity

Soil acidification has been reported in the vicinity of some, but not all, base-metal smelters. For example, at sampling sites located between 0.8 and 1.9 km east of the INCO smelter at Conistion, Ontario, Whitby and Hutchinson (1974) reported mean surface-soil pH 0.6 to 1.6 units below that of the native podzolic soils of the region (pH 3.8 to 4.8). In contrast, soils sampled downwind of the Mount Lyell smelters were on average no more acidic than the native top-soils of the region. The mean soil pH along the two transects were approximately an half unit above those reported for soils on similar parent volcanics (Kirkpatrick, 1984). This was unexpected given the area's history of acid-deposition and raised the question of whether acidification had taken place, or was undergoing a process of reversal. On the Mount Lyell transects, below average soil pH only occurred at sample sites in close proximity to the smelters. Presumably, these near-smelter sites were in receipt of the highest acid-loadings.

Several explanations regarding acidification at Mount Lyell are possible. Severe and extensive soil erosion in the survey area has resulted in the loss of the region's typically acidic, organic-rich surface horizon. This loss is likely to have removed much residual acidity in mass transport of soil and organic material. It could thus be argued that the moderately-acid, soils recorded in the transects bare little resemblance to the acid-depositional soils of the early smelting years. Alternatively, soil acidification may have been counteracted by the release of base cations from subsoils exposed to weathering by erosion. At the time of sampling, 21 years after the cessation of smelting at Mount Lyell, base accumulation may have been sufficient to permit a reversal of the equilibrium between the soil solution and exchange sites. Such reversal processes have been recognised in column-leaching, laboratory experiments (Dahlgren *et al.* ;1990). Under a regime of reduced acid loadings, Dahlgren *et al.* (1990) demonstrated the retention of base cations and the release of SO_4^{2-}. Desorption of SO_4^{2-} led to a temporary lowering of leachate pH but acidification ceased when SO_4^{2-} leachate concentrations approached the input concentrations. Upon the cessation of operation, acidification could therefore be expected to cease, or reverse, as desorbed SO_4^{2-} concentrations approach input concentrations. Consequently, it is plausible that soil pH at Mount Lyell, is today higher than at smelter closure, as a of result of greatly reduced acid-deposition, base cation inputs and attendant equilibrium reversal.

Soil pH along the northerly transect, appeared to increase with increasing displacement. Although this pattern was consistent with the pH-displacement relationships recognised at some other sites of prolonged acid-deposition from Cu smelters (Hutchinson and Whitby, 1974; Whitby and Hutchinson, 1974), a similar, displacement-dependant relationship appeared to be absent to the east of the smelters. This may have been due to an abrupt change in soil type, from clay to loam, along that transect. Alternatively, limited sampling relative to the extent of acid-deposition may have masked any pH-displacement relationship in this direction.

The low TEB, LOI and total nitrogen levels recorded were consistent with those reported at other base-metal smelter sites where significant site degradation had occurred (Hutchinson and Whitby, 1974).

Soil metals: copper

Copper concentrations along both transects at Mount Lyell were elevated in the vicinity of the smelter and declined exponentially with increasing displacement. The concentration-displacement patterns were comparable in form to those of similar pollutant metals found in the vicinity of other point-source, base metal smelters (e.g. Løbersli and Steinnes, 1988; Hogan and Wotton, 1984; Freedman and Hutchinson, 1980; Wood and Nash, 1976; Hutchinson and Whitby, 1974: Fig 1). The displacement patterns provided strong, if circumstantial, evidence of the source of the contamination.

The mean Cu concentrations recorded for Mount Lyell, however, were an order of magnitude lower than those of other world renowned smelter sites (Table 2). For example, Hutchinson and Whitby (1974) and Whitby and Hutchinson (1974) reported mean Cu concentrations of 2 527.0/50.7 ppm (total/water extractable) at downwind sites between 800 and 1500 m from the Coniston smelter. In contrast, Kirkpatrick (unpublished) and the current work record mean Cu concentrations of 160.5/3.5 ppm (total/water extract) at similarly located sites between 230 and 1 700 m from the Mount Lyell smelter. Differences in ore grades, tonnages, smelting processes, the local environment and the period over which smelting occurred are likely to account for these disparities. Contaminant redistribution, due to on-going soil erosion, is also likely to have had a strong influence on the present-day Cu levels at Mount Lyell.

The concentration-displacement patterns along the two transects also suggested that Cu contamination at Mount Lyell was considerably less extensive than that found under comparable circumstances at many of the world-renowned sites. Fitted negative-exponential equations indicated that background concentrations for Cu were regained at displacements of between 1000-1500 metres downwind of the smelter. In comparison, at Coniston, Ontario, background concentrations of the target metals Ni and Cu were reached at downwind-displacements exceeding 13.5 km (Whitby and Hutchinson, 1974). It is likely that elevated topography, climatic influences (eg. temperature inversions) and a relatively low stack height combined to restrict the dispersion of Cu emissions from the Mount Lyell smelters.

Soil metals: zinc

Although Zn was a non-target metal at Mount Lyell, the concentration-displacement pattern for the metal was similar in form to that of the target metal Cu (Fig 4). This was probably a by-product of Cu production and the result of flue-gas losses. Nevertheless, Zn concentrations were an order of magnitude lower than those of Cu, and with the exception of the near-smelter sites, below that of the background volcanics.

Soil metals: aluminium

Elevated Al concentrations were recorded on both transects. Unlike Cu and Zn, however, the concentrations did not describe strong negative-exponential relationships with increasing displacement. This was probably due to the origin of the metal. The presence of significant concentrations of soluble Al was understood to be an indirect consequence of sulphate deposition and occur due to within-soil, chemical reactions. As soils vary in their capacity to buffer acidic inputs, notably in sulphate adsorption capacity (Reuss *et al.*, 1987; Heute and McColl, 1984; Malmer, 1976), Al dissolution could be expected to vary between soil types. Thus, Al concentrations in the soil solutions are likely to be a function of both sulphate loadings and soil type, and are not as likely as the depositional metals to form recognisable concentration-displacement relationships.

The influence of a sulphate-buffering capacity on Al concentration may, in part, provide an explanation of the departure of the Al concentration-displacement patterns from those of the other metals. On the easterly transect, the concentration-displacement pattern for Al was notable for a concentration maximum at some distance from the smelters. A possible explanation was that, in this downwind direction, the mid-sites were in receipt of the highest acid loadings. This might occur as SO_2 gases are relatively light and tend to travel before intercepting the ground (*sensu* Wood and Nash, 1976). This would have consequences for Al dissolution. This explanation could not be confirmed in this study.

Phytotoxicity

The concentrations at which metals are considered phytotoxic are environmentally dependent and difficult to set, even when determined under similar assay conditions, as great differences exist between and within species (Balsberg Påhlsson, 1989; Andersson, 1988). Despite these limitations, the results of solution culture experiments provide an indication of the concentrations at which growth impairments might be expected (Table 3). These data suggest that abnormalities in tree species, due to Zn, Cu and Al toxicity, can be expected at concentrations in the range of 1.0-2.0, 4.0-20.0 and 2.0-200 mg L^{-1}, respectively. In grasses and herbs, toxicity appears to occur at somewhat lower concentrations, with growth abnormalities, for the three aforementioned metals, generally evident at the concentrations 0.1-1.8, 0.02-0.06 and 0.2-25 mg L^{-1}.

Soluble Cu concentrations, on both the Mount Lyell transects, exceeded those expected to impair growth in northern hemisphere, woody species (~4 mg L^{-1}) at 31% of the sampling sites and at all sites located between the smelters and approximately 500 m (Table 3). The maximum Cu concentration recorded, 10.4 mg L^{-1} at Site 1 on the easterly transect, was approximately twice the minimum concentration known to cause growth abnormalities in woody species (Table 3). Aluminium concentrations exceeded those known to impair growth (~2 mg L^{-1}) at 77% of the sampling sites (eg. site 3 easterly transect and site 1 northerly transect). The maximum Al concentration recorded, 29.9 mg L^{-1} at site 1 on the northerly transect, was 15 times that known to cause impairment in woody species (Table 3). In contrast, Zn concentrations were below the known impairment level in tree seedlings. They were, however, generally above those known to be phytotoxic to some grasses and herbs at sites close to the smelters.

Table 3: Reported minimum Zn, Cu and Al concentrations resulting in growth abnormalities in seedlings of various higher plants in metal-salt or nutrient solution.

Metal species	Concentration (mg L^{-1})	Species	Effects*	References
Zn^{2+}	0.1	*Festuca rubra*	rrl	Powell *et al.*, 1986a, b
Zn^{2+}	1.1	*Picea abies*	rr/rs	Godbold and HÜttermann, 1985
Zn^{2+}	1.8	*Lolium perenne*	rr/rs	Wong and Bradshaw, 1982
Zn^{2+}	2.0	*Picea abies*	rrl	Godbold and HÜttermann, 1985
Cu^{2+}	0.02	*Lolium perenne*	rr/rs	Wong and Bradshaw, 1982
Cu^{2+}	0.06	*Agrostis capillaris*	rrl	Wainwright and Woolhouse, 1977; Symeonidis *et al.*, 1985
Cu^{2+}	4.0	*Pinus resinosa*	rr/rs	Heale and Ormrod, 1982
Cu^{2+}	5.0	*Picea sitchensis*	rr/rs	Burton *et al.*, 1986
Cu^{2+}	20.0	*Alnus crispa*	bd	Fessenden and Sutherland, 1979
Cu^{2+}	20.0	*Acer rubrum*	rsd	Heale and Ormrod, 1982
Al^{3+}	0.2	*Glycine max*	rrl	Alva *et al.*, 1986
Al^{3+}	0.7	*Trifolium repens*	rrl	Jarvis and Hatch, 1985
Al^{3+}	2.0	*Picea abies*	rrl	Rost-Siebert, 1984
Al^{3+}	5.4	*Agrostis stononifera*	rrl	Clarkson, 1966a
Al^{3+}	25.0	*Lolium perenne*	rrw	Hackett, 1964

Table 3 continued:

Al^{3+}	45.0	*Pinus sitchensis*	rrl	McCormick and Steiner, 1978
Al^{3+}	120.0	*Betula populifolia*	rrl	McCormick and Steiner, 1978
Al^{3+}	200.0	*Quercus rubra*	rrl	McCormick
	200.0	*Q.palustris*	rrl	and Steiner, 1978

*rrl =reduced root length; rr/rs= reduced root and shoot; rrw=reduced root mass; rsd = reduced stem diameter; db= decreased biomass

3.5 Conclusion

The surface soils at Mount Lyell have been severely eroded leaving extensive, exposed subsoils. Despite soil loss, the chemical characteristics of these subsoils reflect a history of acid and metal-particulate deposition. These influences have resulted in surface soils with chemical characteristics in common with soils in the vicinity of other long-established, base-metal smelters. The exposed subsoils along two transects originating at the Mount Lyell smelters were moderately acidic, deficient in organic matter, low in exchangeable bases and lacking in nitrogen. There was evidence of Cu and Zn deposition and Al mobilization.

Evidence of metal contamination was provided by the distribution patterns of soluble metals in soils. Copper concentrations were elevated near the smelter and declined exponentially with increasing displacement. The contaminant concentration patterns were comparable to those of pollutant metals in the vicinity of other base-metal smelters. However, in comparison to smelters of world renown, Cu contamination at Mount Lyell was an order of magnitude lower and far less extensive. Nevertheless, some soluble Cu concentrations exceeded those known to cause growth abnormalities in seedlings of woody plants.

Soluble Zn was similarly distributed to Cu. The concentrations recorded were an order of magnitude lower than those of Cu and less extensive. Although none of the concentrations exceeded those known to cause growth abnormalities in seedlings of woody plants, they were above those known to be toxic to some grasses and herbs. Contamination by zinc appeared to be a byproduct of Cu production by smelting.

Displacement from the smelters at Mount Lyell did not appear to be an adequate guide to soil Al concentrations. Elevated concentrations of soluble Al were recorded, but unlike the other metals, Al concentrations did not appear to be directly related to displacement. As the sulphate adsorption capacity of a soil type is understood to influence Al dissolution, it is likely that soluble Al concentrations at Mount Lyell are the product of location-specific, depositional concentrations and the inherent buffering capacity of the soils. Some of the Al concentrations recorded exceed those known to cause growth abnormalities in seedlings of woody plants.

Severe soil erosion at Mount Lyell is likely to have reduced metal-ion contamination by off-site redistribution, and concentrations of the metals Cu, Zn and Al are likely to be lower today than at any time during the operation of the smelter. Weathering, and related base-accumulation, may have also contributed to a partial reversal of the process of acidification and rising soil pH may have contributed to reduced metal-ion dissolution. Moderated levels of soil contamination are likely to have significant ramifications for plant colonisation.

Chapter 4

Colonising communities and the environmental factors which influence their occurrence and abundance at Mount Lyell

4.1 Introduction

The area surrounding the Mount Lyell smelters falls into two readily recognizable zones: one with higher plant vegetation and the other without. Aerial photographs taken in the period since the cessation of smelting indicate that the vegetated zone is a result of natural recolonisation. The slow rate and progressive pattern of natural colonisation suggested a response to soil-metal contamination. Accordingly, identification of the edaphic conditions associated with colonising vegetation might add to understanding of the process of natural colonisation in a metal-contaminated area and assist revegetation works by providing a guide to the site conditions that local species are able to tolerate.

The objective of this chapter was to explore the relationships between compositional trends within the colonising community and edaphic environmental variables. A survey of the vascular flora, and multivariate analysis, were used to to describe the composition of the vegetation and trends were related to edaphic factors by vector fitting. The edaphic variables selected were those considered to exert either direct, or indirect, influence on phytotoxicity. Species presence was used as a measure of the differential species performance as this is considered useful where vegetation is developing following a major disturbance. Numerical classification was used to group sites by characterising species coincidences. Species richness, considered in circumstances where a restricted number of environmental gradients dominate a vegetation pattern as a means of interpreting a region-specific multi-dimensional response (Peet, 1992), was used to assist the description of the groups. The classification was used to describe and map colonising communities and was related to contamination patterns.

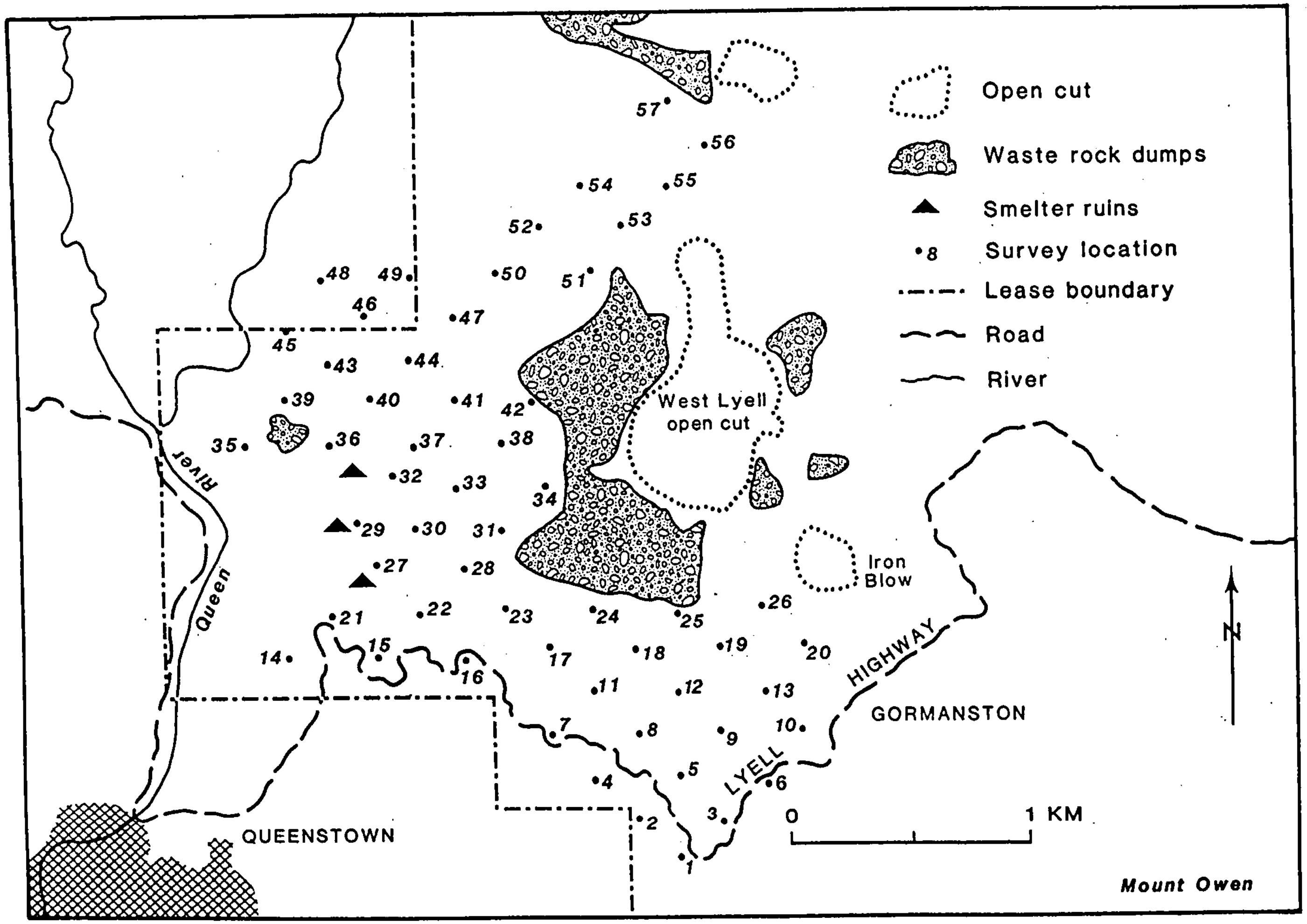
4.2 Methods

Site selection and vegetation survey

Floristic and environmental data were collected at both vegetated and unvegetated sites located to the east of the Mount Lyell smelters. Data were recorded at a total of 57 sites in a survey area of approximately 360 hectares (Fig. 1). The sites were selected systematically using a grid overlay (cell size of 250 m) on 1:25000 map (Gormanston 3834) of the survey area. Mechanically disturbed areas, including mine workings such as roads, open cuts and dumps, were avoided.

At each survey site, a 20 x 20 m quadrat was marked out. In each quadrat, the presence of vascular plant species were recorded. Species identification was achieved with the aid of Curtis (1963), Curtis (1967), Curtis and Morris (1975), Hyde-Wyatt and Morris (1989), Willis (1973,) and Duncan and Isaac (1986). Nomenclature follows Buchanan (1995). In addition, the basal area of woody species at each site was calculated from stem circumference measurements ($\geq$10 cm) taken at a stem height of 5 cm. This height was used to minimise the inconvenience caused by the multi-stemmed form of many individuals.

Fig.1: Map of soil and vegetation survey locations



Soil sampling and laboratory preparation

A composite, surface soil sample was taken at each site. Each sample was prepared from 10 to 15 subsamples excavated to a depth of 2 cm. Sufficient soil was removed at each location to provide a subsample of approximately 0.4 kg. The samples were placed in sealed plastic bags for transport.

The laboratory preparation of the soil samples followed the recommendations suggested by Rayment and Higginson (1992). The samples were initially broken up by crushing between sheets of polyethylene, coarse sieved (1 cm nylon mesh) and thoroughly mixed. The samples were divided in a sample divider into two halves of approximately equal mass and allowed to air dry.

Soil analyses and other site data

One half of the sample was sieved (6 mm brass mesh) and a subsample subjected to pH measurement. A soil/water suspension (1:1 soil/distilled water by weight) was prepared and, after shaking, pH measurement performed with the aid of a WTW electronic pH meter (model pH6) fitted with a Type E 50 electrode. Prior to pH measurement the meter was standardised to the manufacturer's instructions. After additional sieving (2 mm mesh), another subsample was reserved in sealed plastic bags for total nitrogen determination using the Kjeldahl procedure (Bremer and Mulvaney, 1982). Total nitrogen was determined by the Mount Pleasant Laboratories, Kings Meadows, Tasmania. Soil texture was determined using a rod and ribbon method and a key based on Northcote (1979). The remaining soil was stored at 2-3°C in labelled plastic bags as reference material.

The other half of each soil sample was used to determine selected soil and chemical attributes. As the samples were to be subjected to metal-ion determinations, two precautions were taken in order to minimise external contamination during sample preparation; all the utensils were rinsed in a solution of 5% nitric acid (14M analytical grade) prior to use and plastic-bodied, nylon-mesh sieves were used throughout. The samples were sieved (6 mm nylon mesh) and air-dried to constant weight on polyethylene-lined aluminium trays at 40 °C in a fan-forced oven. Drying periods varied with the composition of each sample and in some cases extended over several days. After drying, each sample was sieved (2 mm nylon mesh) and sealed in labelled, plastic bags.

The air-dry samples were used to measure soil moisture (ODM), organic matter content (LOI), total exchangeable bases (TEB) and metal-ion concentrations. Samples weighing approximately 4 g were transferred to clean crucibles of known mass and the air-dry mass of each calculated. The samples were then placed in an oven a for a period of 6 hours at 105°C. After drying to constant weight and reweighing at room temperature, the moisture content of each sample was calculated using the percentage method of Rayment and Higginson (1992).

Organic matter content was determined by loss-on-ignition (LOI). Oven-dry (105°C) samples of known weight (approximately 2 g) were placed in pre-weighed, high-temperature crucibles and heated to 450 °C in an high temperature oven for a period of 6 hours. Each sample was reweighed at room temperature and the difference in sample mass used to estimate percentage organic matter content.

Total exchangeable bases (TEB) were determined by the Mount Pleasant Laboratories, Kings Meadows, Tasmania.

Distilled water, a weak extractant, was used for soluble metal-ion analyses. This extractant was chosen in preference to stronger extractants as many, such as strong acid digests, are believed to have limited biological significance (McCray and Sumner, 1990; Wright, 1989; Baker, 1988; Adams and Lund, 1966). Soil-water extracts were prepared from 66 grams of each sample, diluted 3 to 1 by mass with distilled water, and mechanically shaken in acid-rinsed, high-density polyethylene bottles for 3 hours. After standing for 24 hours, the extracts were decanted into clean polypropylene bottles. The extracts were sent to the Mount Pleasant Laboratories, Kings Meadows where they were preserved by acidification with 2 mL L^{-1} HNO_3 (ASTM, 1989b) and analysed by atomic absorption spectrometry (Varian 1275) for the metals Cu, Zn, and Al. Soluble metal concentrations were obtained in mg L^{-1}.

The water extractable, metal-ion analyses were complemented by determinations for DTPA extractable Cu (Lindsay and Norvell, 1978), and Zn (Lindsay and Norvell, 1978) and $CaCl_2$ extractable Al. The extractions were performed and measured (AAS) by Mount Pleasant Laboratories, Kings Meadows, Tasmania on sieved (1 mm mesh), air-dry soil samples. Funding constraints restricted the total number of chemical analyses available. For example, water extractable Cu data were available for 38 of the 57 sites. The exact number of analyses available for each soil attribute can be found in Table 5.

In addition to the above chemical attributes, measurements of the depth of the exposed sub-soils and slope were made at each site. Depth was recorded after digging a hole in the subsoils until compacted parent material was reached. The slope at each site was recorded with the aid of a clinometer. Lastly, the sites were described by their linear and perpendicular displacements. These were defined as (1) the linear displacement from the smelters to a sample site (displacement) and (2), the displacement from a line indicating the direction of the prevailing wind to a sample site (perpendicular).

The above community and soil data were collated on a spreadsheet in preparation for subsequent computer-based, numerical analyses.

Numerical analyses - ordination

Multi-dimensional scaling (MDS), a form of indirect gradient analysis, was used to model variation in the distribution of the colonising taxa. In comparison with other ordination methods, MDS is considered robust for ecological data (Minchin, 1987a/b). The method can be used to produce an ordination diagram, formed such that the dissimilarity of sites in terms of composition is reflected in their separation (Faith and Norris, 1989). In the MDS procedure, species composition data are first replaced by a matrix of dissimilarity values between sites. MDS derives an ordination space wherein the distances between all pairs of site points, adjusted over successive iterations, best match the corresponding dissimilarities. A stress function indicates how well, or badly, site separation in the ordination diagram matches the dissimilarity values. In non-metric MDS, the stress function is defined by a monotonic (rank-order) relationship between dissimilarities and distances (Bowman and Minchin, 1987; Kruskal and Wish, 1978).

Species and their occurrences (presence-absence community data) were entered into the ecological data management program DECODA (Minchin, 1990). This program prepares data for ordination. A matrix of compositional dissimilarities was calculated using the Czekanowski (Bray-Curtis) coefficient. The matrix was subjected to ordinations by non-metric MDS from one to four dimensions. The MDS program's default options were followed. An ordered, condensed matrix of the community data was printed. DECODA was then used to calculate the percentage frequencies of species that occurred at three or more of the sample sites.

The vegetation ordination was related to the measured edaphic and derived variables by vector fitting. This procedure has been used to find the directions of maximum correlation for environmental variables within an ordination (Minchin, 1990). The method finds the vector (rotated axis) in the ordination space such that the projections (scores) of the sites on this vector are maximally correlated with the values of a given variable (Bowman and Minchin, 1987).

Vectors of maximum correlation were calculated between the species configuration and each of the environmental variables using an option provided by DECODA (Minchin, 1990). The Monte-Carlo approach, a test of the hypothesis that a correlation could have been found even if the environmental variable were randomly assigned to the sites (Faith and Norris, 1989), was used to test the significance of the maximised correlation. This permitted the identification of variables with significant monotonic trends across the ordination and their directions. DECODA was used to calculate percentile data for the significant variables. Vectors of maximum correlation were presented diagrammatically.

Numerical analyses - classification

TWINSPAN (Hill, 1979b), a program widely used in community ecology, was used to classify the samples. This is a form of divisive cluster analysis or dichotomized ordination analysis (Hill 1979b). The phytosociological concept behind the program TWINSPAN is that a group of sites can be characterised by a group of differential species; species that appear to prevail on one, or other, sides of a dichotomy (Jongman *et al.* , 1987). The method employed by the TWINSPAN program to classify sites and species has been explained by Jongman *et al.* (1987). In TWINSPAN, a crude dichotomy is made by ordinating the sites by a method of correspondence analysis. The centroid of the first ordination axis provides an initial division. In an iterative process, further divisions are created by using the frequencies of the species on either side (known as positive and negative) of the first division. Differential species are identified by computing a preference score. The divisions provide an hierachical classification of sites based on species' preferences.

The program TWINSPAN (Hill, 1979b) was used to classify the sites and to construct an ordered two-way, sites by species table. The site classification identified both preferential and non-preferential species. Preferential species were species not in common between two clusters. A dendrogram was used to indicate the preferential species used in the divisions of sites.

Mapping

The classification of vegetation by TWINSPAN allowed the sites to be mapped. Vegetation mapping was complemented by a spatial representation of soil-Cu concentration classes. Three classes were selected after consideration of the following information: 1) the results of seedling trials in Mount Lyell soils (CH. 3), 2) the mean and range of soil Cu concentrations over which tolerant and non-tolerant species were present in the survey of the colonising community and 3) reference to levels of Cu tolerance identified in the literature (Ch. 3). Environmental data for each of the mapping units were contrasted.

4.3 Results

Species composition, origins and frequency

Forty-four vascular plant species, representing 23 families, were identified from 56 of the 57 sites sampled (Table 1). Among the species were 26 Dicotyledonae, 8 Monocotyledonae, 3 Gymnospermae and 7 Pteridophyta. None of the species were annuals. Fifteen of the species (34 %) were known to occur naturally in West Coast rainforest (Table 2); these included a single occurrence of *Athrotaxis selaginoides* (King Billy pine). Another 14 species were normally confined to eucalypt forest and woodland and/or scrub, heath, sedgeland and herbland. None of the species were recognised as exclusively alpine. Five species were exotic (*Agrostis capillaris, Cortadaria selloana, Cupressus macrocarpa, Cytisus scoparius* and *Hypochaeris radicata*). Two species, *Acacia dealbata* and *A. sophorae,* were believed to have been imported with soil. A single *Cupressus macrocarpa* was believed to be the progeny of a small stand of conifers planted near the smelter.

Twenty-five of the Mount Lyell species occurred in the sampling sites at a frequency of 5% or more. These species were ranked in descending order of percentage frequency (Table 3). Only four species occurred at more than 50% of all sites. The most commonly occurring species was *Restio tetraphyllus,* the

Table 1: Alphabetical list of vascular species represented in colonised zone at Mount Lyell.

Species	Family
Dicotyledonae	
Acacia dealbata Link	Mimosaceae
Acacia melanoxylon R.Br.	Mimosaceae
Acacia mucronata Wild. ex Wendl.f. var. *mucronata*	Mimosaceae
Acacia sophorae (Labill.) R.Br.	Mimosaceae
Atherosperma moschatum Labill.	Monimiaceae
Baeckea leptocaulis Hook.f.	Myrtaceae
Comesperma retusum Labill.	Polygalaceae
Cyathodes juniperina (Forst.f.) Druce	Epacridaceae
Cytisus scoparius (L) Link	Fabaceae
Epacris impressa Labill.	Epacridaceae
Epacris heteronema Labill.	Epacridaceae
Gaultheria hispida R.Br.	Ericaceae
Hypochoeris radicata L.	Asteraceae
Leptospermum glaucescens S. Schauer	Myrtaceae
Leptospermum nitidum Hook. f.	Myrtaceae
Leptospermum scoparium Forst & Forst.f. var. *scoparium*	Myrtaceae
Melaleuca squamea Labill.	Myrtaceae
Monotoca scoparia (Sm) R.Br.	Epacridaceae
Olearia erubescens (DC.) Dippel	Asteraceae
Oxylobium arborescens R.Br.	Fabaceae
Persoonia gunnii Hook. f. var. *gunnii*	Proteaceae
Phebalium squameum (Labill.) Engl. subsp. *squameum*	Rutaceae
Pimelea linifolia R.Br.	Thymelaeaceae
Sprengelia incarnata Sm.	Epacridaceae
Telopea truncata (Labill.) R.Br.	Proteaceae
Trochocarpa gunnii (Hook. f.) Benth.	Epacridaceae
Monocotyledonae	
Agrostis capillaris L.	Poaceae

Blandfordia punicea (Labill.) Sweet	Liliaceae
Cortadaria selloana (Schults& Schltes.f.) Asch. Graebner	Poaceae
Empodisma minus (Hook.f.) L.Johnson & Cutler	Restionaceae
Gahnia grandis (Labill.) S.T. Blake	Cyperaceae
Restio monocephalus R.Br.	Restionaceae
Restio tetraphyllus Labill.	Restionaceae
Isolepis aucklandica (Hook f.) Boeck.	Cyperaceae

Gymnospermae

Athrotaxis selaginoides D. Don	Taxiodiaceae
Cupressus macrocarpa Gord.	Cupressaceae
Phyllocladus aspleniifolius (Labill.) Hook.f.	Podocarpaceae

Pteridophyta

Blechnum nudum (Labill.) Mett ex Luerss.	Blechnaceae
Blechnum wattsii Tind.	Blechnaceae
Gleichenia microphylla R.Br.	Gleicheniaceae
Histiopteris incisa (Thunb) J.Smith	Dennstaedtiaceae
Lastreopsis acuminata	Dryopteridaceae
Lycopodium deuterodensum Herter (Houlston) Morton	Lycopodiaceae
Pteridium esculentum (Forst.f.) Cockayne	Dennstaedtiaceae

Table 2: Community origins of the species occurring at Mount Lyell. The communities of the native species follow Kirkpatrick (1977).

Species	1	2	3	4	5	6
Dicotyledonae						
Acacia dealbata			x			
Acacia melanoxylon			x	x		
Acacia mucronata			x	x		
Acacia sophorae					x	
Atherosperma moschatum		x	x	x		
Baeckea leptocaulis				x		
Comesperma retusum				x		
Cyathodes juniperina		x	x			
Cytisus scoparius						x
Epacris impressa						
Epacris heteronema			x	x		
Gaultheria hispida	x	x				
Hypochaeris radicata						x
Leptospermum glaucescens			x	x		
Leptospermum nitidum			x	x		
Leptospermum scoparium			x	x		
Melaleuca squamea				x		
Monotoca submutica		x	x	x		
Olearia erubescens						
Oxylobium arborescens			x	x		
Persoonia gunnii	x					
Phebalium squameum			x			
Pimelea linifolia						
Sprengelia incarnata	x		x	x		
Telopea truncata		x	x	x		
Trochocarpa gunnii		x				
Monocotyledonae						
Agrostis capillaris						x
Blandfordia punicea	x	x	x	x		
Cortadaria selloana						x
Empodisma minus			x	x		

Table 2 continued	1	2	3	4	5	6
Monocotyledonae						
Gahnia grandis		*x*	*x*	*x*		
Restio monocephalus				*x*		
Restio tetraphyllus				*x*		
Isolepis aucklandica		*x*		*x*		
Gymnospermae						
Athrotaxis selaginoides	*x*	*x*				
Cypressus macrocarpa						*x*
Phyllocladus aspleniifolius	*x*	*x*	*x*			
Pteridophyta						
Blechnum nudum			*x*			
Blechnum wattsii		*x*	*x*			
Gleichenia microphylla						
Histopteris incisa		*x*	*x*			
Lycopodium deuterodensum			*x*			
Lastreopsis acuminata				*x*		
Pteridium esculentum			*x*	*x*		

1 - alpine
2 - rainforest
3 - eucalypt forest and woodland
4 - scrub, heath, sedgeland and herbland
5 - coastal
6 - naturalised exotics or weed species

Table 3: Species frequency table for those species occurring at more than 5% of the sampling sites. The species are sorted in descending order of frequency.

Species	Frequency	% frequency
Restio tetraphyllus	41	71.93
Agrostis capillaris	40	70.18
Gaultheria hispida	32	56.14
Acacia mucronata	29	50.88
Acacia melanoxylon	21	36.84
Cyathodes juniperina	21	36.84
Leptospermum scoparium	15	26.32
Gleichenia microphylla	12	21.05
Persoonia gunnii	12	21.05
Isolepis aucklandica	12	21.05
Oxylobium arborescens	8	14.04
Phebalium squameum	8	14.04
Blechnum wattsii	7	12.28
Epacris heteronema	7	12.28
Gahnia grandis	7	12.28
Epacris impressa	5	8.77
Sprengelia incarnata	5	8.77
Acacia dealbata	4	7.02
Atherosperma moschatum	4	7.02
Monotoca submutica	4	7.02
Comesperma retusum	3	5.26
Hypochaeris radicata	3	5.26
Pimelea linifolia	3	5.26

Cord rush, which occurred at 71.9% of the sample sites. This species normally occurs in acid wetlands. The next most commonly occurring species were the naturalised, exotic grass *Agrostis capillaris* (70.2%), the Snow Berry, *Gaultheria hispida* (56.1%), and the Willow-leaved wattle,*Acacia mucronata,* (50.9%). Of these species, only *G. hispida* was considered a rainforest species. *A. mucronata* normally occurs in eucalypt forest, woodland or scrub.

With the exception of *A. capillaris,* weed species were not prominent among the colonising community. Of the remaining 3 weed species recorded, only *H. radicata* occurred at a frequency exceeding 5%. The species *C. scoparius* was recorded at low frequencies. This species, however, was present in greater numbers on the verges of some local roads and tracks. These locations were excluded from the survey due to the influence of roadside contamination by soil and vegetative material of unknown origin. Consequently, the species frequency recorded in this survey may not accurately reflect the invasive potential of *C. scoparius.* A single occurrence of *C. selloana* was recorded.

Community ordination

DECODA provided a plot of minimum stress verses the number of scaling dimensions. The plot showed minimum stress reduction above two dimensions. At the same time, the mean residuals for the two and three dimensional configurations were consistent and effectively identical. The minimum stress configuration in two-dimensions was accepted as an adequate representation of the compositional variation among the 57 sites (stress = 0.2115). The two-dimensional MDS ordination was then added to the program's master file. A description of the DECODA master file is provided (Table 4).

Table 4: A description of the master file created in DECODA using community and environmental data for 57 sampling sites near Mount Lyell

Number of species = 46
Number of samples = 57
Number of species variables = 4
Number of sample variables = 23
Number of abundance measures = 1

Reading species labels from species index...

Name of MDS input file:
ly.mds

Description of MDS input:
community

Multidimensional scaling type: Global non-metric

Number of starting configurations = 10
Minimum no. of dimensions = 1
Maximum no. of dimensions = 4
Solution scaling option = 2 (Half-Changes)
Seed for random numbers = 333333
Maximum no. of iterations = 100
Stress ratio stopping value = 0.999000
Small stress stopping value = 0.010000 Data type: community data

Presence-absence (binary) data are used.

Species which do not occur in any samples are excluded.

Dissimilarity coefficient: Czekanowski (Bray-Curtis)

No. of samples included = 57
No. of species included = 42

Using the Monte-Carlo approach, vector fitting indicated that 3 of the 13 edaphic environmental variables were significantly correlated with the vectors of maximum correlation in the species' configuration. These were Cu(w), soil depth and loss-on-ignition (LOI). Correlations were also found for the community variables, total basal area and species richness, and the derived variables, perpendicular and displacement (Table 5).

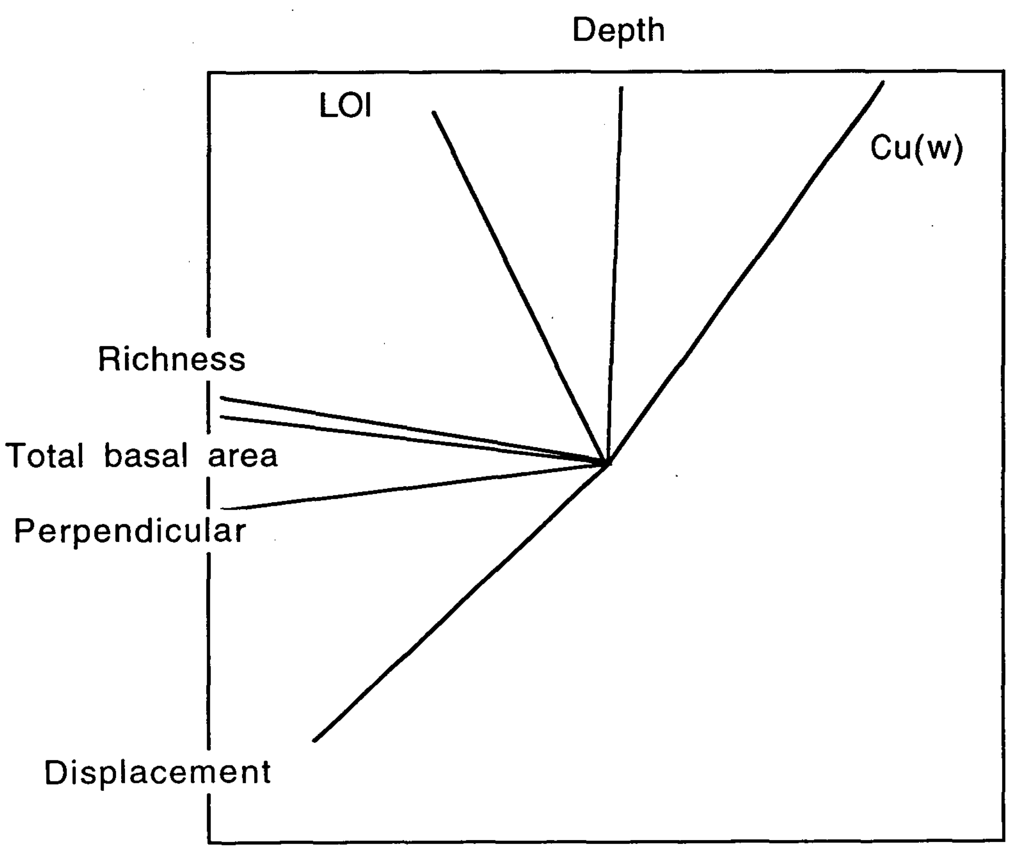
Vectors of equal and arbitrary length, corresponding to the significant environmental variables, were used to indicate the directions of maximum correlation (Fig. 2). A strong, inverse relationship between soil Cu(w) and displacement from the smelter is evident along a vector from the right-hand top corner to the left-hand bottom corner. The productivity and community variables, total basal area and species richness, were closely associated. The directions of these vectors were more closely oriented toward that of increasing displacement than that of increasing Cu(w). Soil depth appeared to be largely unrelated to Cu(w) and the other variables.

The representation of compositional variation offered by the two-dimensional MDS ordination was initially examined in an ordered site by species matrix. Sites were sorted in ascending order of the variable Cu(w). In this matrix, separations in the distributions of the most frequently occurring species were indistinct. This was considered unlikely considering 1) the environmental gradient exhibited by soil Cu(w), and 2) an accepted differential Cu resistance

Table 5: Results of vector fitting

SAMPLE	VARIABLE	N	MAX R	Prob	DIRECTION COSINES AND ANGLES OF FITTED VECTORS WITH CONFIGURATION AXES 1	2
1	Total basal area	57	0.4136	0.000***	-0.9806 168.7	0.1959 78.7
2	Richness	57	0.8133	0.000***	-0.9821 169.1	0.1884 79.1
3	pH	57	0.2679	0.150	-0.5968 126.6	-0.8024 143.4
4	Displ	57	0.7286	0.000***	-0.7048 134.8	-0.7094 135.2
5	ODM	56	0.2490	0.250	0.2478 75.7	0.9688 14.3
6	LOI	55	0.3101	0.050*	-0.4263 115.2	0.9046 25.2
7	Slope	56	0.2060	0.270	0.5341 57.7	0.8454 32.3
8	Depth	55	0.3821	0.020*	0.0635 86.4	0.9980 3.6
9	Perpendicular	56	0.7046	0.000***	-0.9901 171.9	-0.1402 98.1
10	Cu(w)	38	0.5098	0.000***	0.6232 51.5	0.7821 38.5
11	Al(w)	36	0.2920	0.260	0.3108 71.9	0.9505 18.1
12	Zn(w)	36	0.2646	0.320	0.9909 7.7	0.1345 82.3
13	TEB	21	0.1117	0.910	0.8730 29.2	-0.4877 119.2
14	Nitrogen	16	0.1965	0.810	0.4537 63.0	-0.8911 153.0
15	Cu(DTPA)	17	0.4569	0.230	-0.1031 95.9	0.9947 5.9
16	Al(CaCl2)	14	0.4218	0.420	0.9797 11.6	0.2005 78.4
17	Zn(DTPA)	14	0.4768	0.240	-0.3553 110.8	0.9347 20.8

Fig. 2: Vectors of maximum correlation for physico-chemical and derived variables using the Monte-Carlo approach. Vector length does not indicate correlation strength. Significance level p<0.05.



between species. (e.g. *Agrostis* spp. and various tree species). The limited size of the Cu(w) data set may have been partly responsible for the indistinct species separations.

Species' distributions along the Cu gradient were believed to be better represented by the fitted vector for Cu(w). The scores for the sample sites along the fitted vector for Cu(w) were saved in the master file as a new sample variable. Sorted in ascending order, this variable reflected the Cu(w) gradient. The site by species matrix for the fitted vector provided a compositional gradient in which distinct separations between the dominant species were evident (Table 6).

At the upper-end of the Cu(w) gradient, the absence of almost all species, with the exception of *A. capillaris,* and the occasional *A. melanoxylon,* was notable. At a somewhat lower position on the gradient, the species *R. tetraphyllus, G. hispida* and *A. mucronata* are 'sudden' introductions to the new community. Separations of the less frequent species are not as easily interpretable. However, richness appears to increase toward the lower-end of the gradient. Interestingly, *A. capillaris* and *A. melanoxylon* disappear from the sample sites toward the lower-end of the gradient.

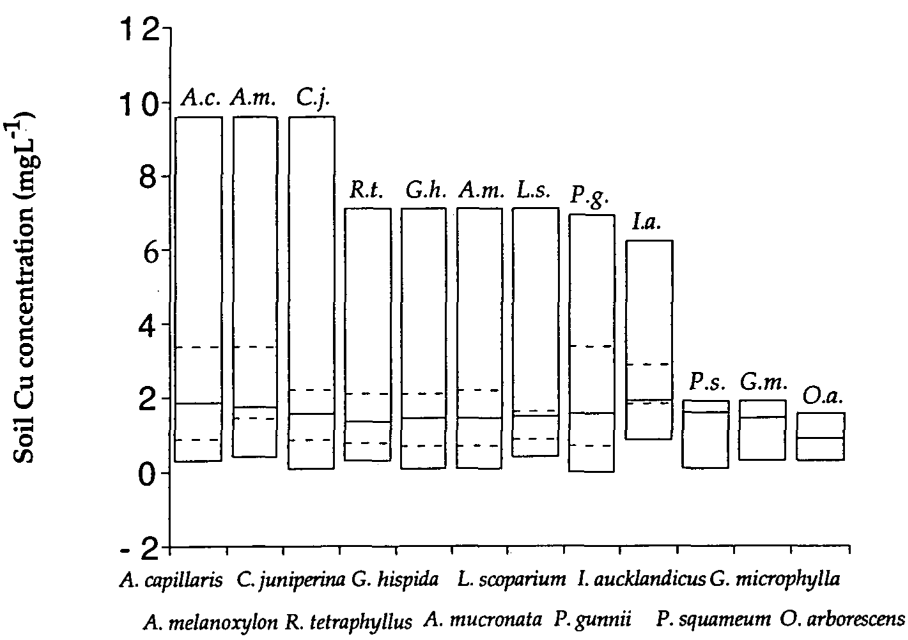
The percentile plots confirmed that species in the colonising community occurred over discrete ranges of soil Cu(w) concentration (Fig. 3). Three distinct concentration ranges were evident. Three species, *A. capillaris, A. melanoxylon* and *C. juniperina,* occurred over the broadest range of Cu(w) concentrations. All three species were present at sites at which soil Cu(w) concentrations exceeded 10 mgL^{-1}. The plots suggested that a second species' group occurred over a somewhat smaller range of soil Cu(w) concentrations. This grouping

Table 6: The scores for the fitted vector, along which species' scores displayed maximum correlation with the variable Cu(w), were used to sort the sampling sites to provide a compositional gradient. The community data was then printed in the form of an ordered, condensed matrix. In this matrix, the columns represent the sampling sites and the rows represent species occurrence. Species occurrence at a sample site was sorted in ascending order of median occurrence and represented by the printed character 1. The sampling sites exhibit a range of Cu(w) concentrations from low (at left) to high (at right). Species occurring at frequencies of less than 5% were excluded.

```
Fitted Cu(w)
                               1  255155451    5 55142121343434432212  4314312234134323
                               23962436775056823410713159945082622084718065347849573196

Hypochaeris radicata           --------------------------------------1--11-------------
Agrostis capillaris            -----------11--1111111111-11111111111111111111-111111-
Acacia melanoxylon             ---------1---1---1-1------1111-11-----1-11-11--111--11--
Acacia dealbata                -------------------1------------1-------1-1-------------
Leptospermum scoparium         ----1--11----1----1-------1-1--11---1---11-1-----1--1---
Persoonia gunnii               -----1-11-1--1------------1---1-1-------1--1-----1----1-
Isolepis aucklandica           ------1--1----1--1----11------1------11-----------11-1--
Acacia mucronata               -1--1111111--1-11-111111----1--1111-111-1111------------
Restio tetraphyllus            1111111111111111111111111-111111-111111-1--1---1-------
Epacris impressa               -----1-------------1------------1-------1--1------------
Phebalium squameum             -----1--1-------11-1-------1-1-------------1------------
Gaultheria hispida             111-11111111111111111111111111--1--------1--------------
Oxylobium arborescens          -1---1--1------1---1------------1---1------1------------
Cyathodes juniperina           11--11-1111--1-11111------1111-11-----1-----------------
Blechnum wattsii               -----1-1-----1--111------------1------------------------
Epacris heteronema             ------1----1-1-----1------------1-------1--1------------
Gleichenia micropylla          ----1111-1-111--11-1-----------1------------------------
Gahnia grandis                 ----11--1-------1---------1-------------11--------------
Sprengelia incarnata           ------11---1--------------------1----------------1------
Monotoca scoparia              -------11----------1-----------------------------1---
Atherosperma moschatum         ----11-----------------------------1---1---------------
Comesperma retusum             -----1----------------------1--1------------------------
Pimelea lindleyana             -1----------1----1--------------------------------------
```

Fig. 3: Percentile plots for the twelve most frequently occurring vascular species occurring at Mount Lyell. The plots represent the range of soil Cu concentrations (water extract) over which the species occur. Each species box encloses 90% of the data; the bottom and top line of each box represent 5% and 95%. The middle line represents the median value of the data (50%), while the lower and upper lines (dotted) represent 25% and 75% of the data, respectively.



included the species *R. tetraphyllus, G. hispida, A. melanoxylon, L. scoparium, P. gunnii* and *I. aucklandica*. These species were present at sites exhibiting soil Cu(w) concentrations to a maximum of between 7-8 mgL^{-1}. A third grouping, comprised of the species *P. squameum, G. microphylla* and *O. arborescens*, occurred over the narrowest soil Cu(w) concentration range. These species were present to a maximum soil Cu(w) concentration of approximately 2 mgL^{-1}.

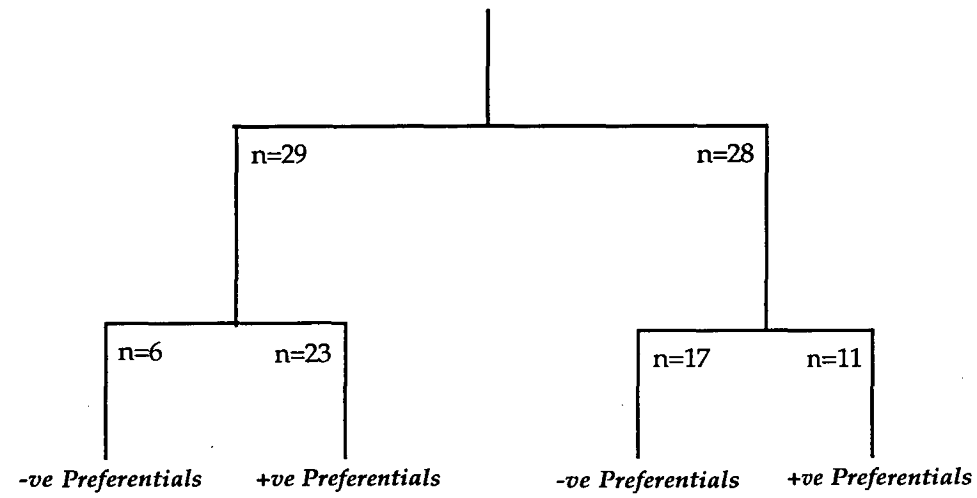
Community classification

The sites by species table provided by TWINSPAN showed that the classification procedure divided the sites 14 times before failing (Table 7). However, three divisions were considered adequate to classify the sites. After three divisions, four distinct sample site groups were identified and these groups were selected as the basis of a 'community' classification (Fig. 4). The first division separated *I. aucklandica* from 12 other preferential species. At this division, three non-preferential species were identified (*A. capillaris, G. hispida,* and *R. tetraphyllus*). The next two divisions separated the species *A. melanoxylon* from *A. mucronata/G. hispida/R.tetraphyllus*, and *R. tetraphyllus/C. juniperina/G. hispida* from *A. melanoxylon/A. dealbata, E. impressa/E. heteronema/P. gunnii/S. incarnata* and *T. truncata*. The species' groupings were termed Groups 1, 2, 3 and 4 respectively. For the purposes of easy recognition, the groups were then described with reference to most characteristic species. The descriptive groups were *Agrostis* grassland (Group 1), Restionaceae/ *Acacia mucronata* tall shrubland (Group 2), Restionaceae low shrubland (Group 3) and *Acacia melanoxylon* shrubland (Group 4).*Agrostis* was selected as characteristic species for Group 1, despite being a non-preferential species. The species, however, did describe an easily recognisable grouping where all other

Table 7: TWINSPAN species by sites table. The order of the site groups was determined by comparison of the two site groups formed at any level with site groups at two higher hierarchical levels. The species classification was based on the degree to which species are confined to particular groups of species (fidelity). Zeros and ones on the right-hand side and bottom indicate the dichotomies.

```
                 344444 5  155553355 445411413 12 23112224 3 12 3312221233
                 639568662382470591540432037609265021723418819573441785917
P. gunnii        1-111111----11--1-1-----------------------1--------------  0000
A. dealbata      ----11--------1------------1-----------------------------  0001
E. impressa      ---111------1-1------------------------------------------  0001
L. acuminata     -----1---------------------------------------------------  0001
M. squamea       ----1-1--------------------------------------------------  0001
P  aspleniifolius---1-----------------------------------------------------  0001
T. truncata      ---1-1---------------------------------------------------  0001
E. heteronema    ---1111-------1---------11-------------------------------  0001
S. incarnata     --1-1--1----------------11-------------------------------  0001
C. retusum       ----1-------1-------1------------------------------------  0001
L. scoparium     -1111111--11-1-111-----1----1----------------------------  0001
A. moschatum     -----1----111--------------------------------------------  001
O. arborescens   ---11---111-111------------------------------------------  001
G. grandis       -----1-----111--1-----1-----1----------------------------  0100
M. scoparia      -1-----1-----11------------------------------------------  0100
P. squameum      ---1--------111----1111----------------------------------  0100
A. sophorae      -----------------------1---------------------------------  0101
B. wattsii       ------11----1----1-1--11---------------------------------  0101
A. selaginoides  -------1-------------------------------------------------  0101
L. nitidum       ------1--------------------------------------------------  0101
L. deuterodensum -------1-------------------------------------------------  0101
P. esculentum    ------1--------------------------------------------------  0101
B. leptocaulis   ------------------------1--------------------------------  0101
B. nudum         -----------------------1---1-----------------------------  0101
C. macrocarpa    ----------------------------1----------------------------  0101
C. scoparius     ---------------------------11----------------------------  0101
C. juniperina    ----1-1111-1111111111111--1---1---------------1----------  011
G. microphylla   ------11---11-1----1--11111-----1------------------------  011
A. melanoxylon   1111111-------111--111-1--1-1-----------------111---1--1-  100
H. radicata      ---------------------------11-----------------1----------  100
A. mucronata     ---1111111111111-11---11-1111-----1111111-----1----------  101
G. hispida       ------1111-111111111111-1111-11-1-111-1-111-11-----------  101
P. lindleyana    ---------1---------1------------1------------------------  101
R. tetraphyllus  ----1-1111111111-1111111111-111111-11111111111111--------  110
A. capillaris    -11111--1-1---11-1-111111--11---111111111-111111-1-111111  1110
H. incisa        -----1---------------------------------------1-----------  1110
E. minus         --------------------------------------------11-----------  1111
O. erubescens    ------------------------------------------1--------------  1111
R. monocephalus  ---------------------------------1-----------------------  1111
I. aucklandica   -------------------1-----11--------1--11-11---1------1-11  1111
                 000000000000000000000000000001111111111111111111111111111
                 000000111111111111111111111110000000000000000011111111111
                 000111000000000000000000111110000001111111111100000001111
                       000000000111111111000110001100000000011100011111
                       001111111000011111          000000011
                         0000001    00001          0111111
```

Fig. 4: TWINSPAN dendrogram. The numbers at each branch indicate the number of sampling sites at each division. Preferential and non-preferential species are listed in decreasing order of frequency for the second division.



Group 4	Group 3	Group 2	Group 1
A. melanoxylon	*R. tetraphyllus*	*R. tetraphyllus*	*A. melanoxylon*
P. gunnii	*G. hispida*	*G. hispida*	
E. heteronema	*C. juniperina*	*A. mucronata*	
S. incarnarta	*G. microphylla*		
A. dealbata	*B. wattsii*		
T .truncata			

Non preferential (Groups 4 and 3)

A. capillaris
A. mucronata
L. scoparium
O. arborescens
P. squameum
G. grandis

Non preferential (Groups 2 and 1)

A. capillaris
I. aucklandicus

woody species, with the exception of remnant *A. melanoxylon* individuals, were absent. An alternative name for this grouping might equally have been '*A. melanoxylon* remnant barrenlands', in reference to the regrowth of a very restricted number of *A. melanoxylon* individuals from root stocks and the low level of occurrence of a limited number of other species.

The *Agrostis* grassland (Group 1) was characterised by the presence of *A. capillaris* and the absence of most other species. Accordingly, with a total of 7 species recorded, the mean species richness was the lowest of the four groups (Fig. 5). Grassland occurred at the highest mean Cu(w) concentrations (4.42±3.81 $mg L^{-1}$; Table 8). This group represented 19% of the sample sites. *A. melanoxylon* tended to occur either as widely-spaced individuals or in discrete clumps. The presence of this species could be best explained as a result of vegetative regrowth. The species is known to sucker (Kirkpatrick, pers. comm.). In the Mount Lyell area, the author has observed root suckers near established trees with damaged root systems.

The second group, the Restionaceae/ *Acacia mucronata* tall shrubland (Group 2), was characterised by the species *R. tetraphyllus* in the presence of the tall shrub *A. mucronata*. The group was differentiated from Group 1 by the presence of *R. tetraphyllus* and *G. hispida*. The total number of species recorded was 13. This group recorded the second highest, mean soil Cu(w) concentration (1.97±1.37 $mg L^{-1}$) and represented 30% of the sample sites.

The third group, the Restionaceae low shrubland (Group 3), was characterised by *R. tetraphyllus* in the presence of *C. juniperina*, *G. hispida* and two fern species: *B. wattsii* and *G. microphylla*. Thirty-three species were recorded and mean species richness was markedly higher than those of Groups 1 and 2

Fig. 5: Mean species richness for sample sites classified by TWINSPAN groups and Cu (w) concentration. The Cu(w) concentrations were grouped into five categories ranging from 0 to >8 mg L^{-1}. The lack of richness data in some concentration categories indicated the absence of a soil category within a group.

Fig. 4

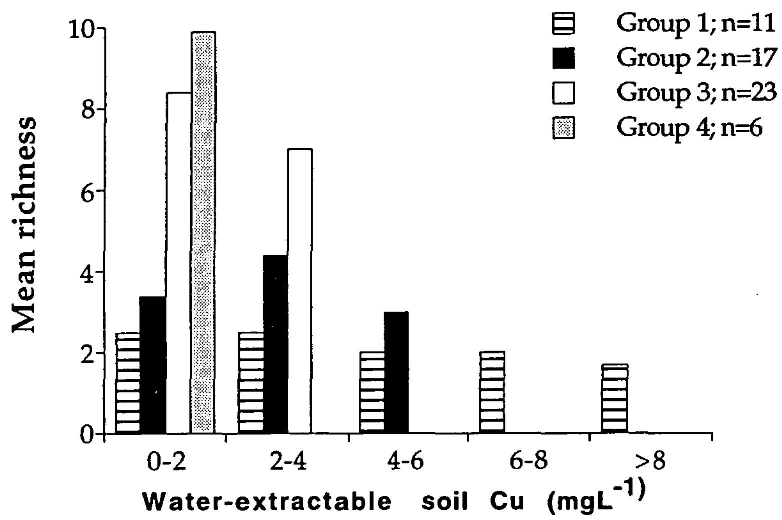


Table 8: Community and environmental data for the four TWINSPAN groups. Means and standard deviations (in brackets) are provided.

Twinspan group:	Group 4 *Acacia melanoxylon* shrubland	Group 3 *Restio/-* low shrubland	Group 2 *Restio/- A. mucronata* tall shrubland	Group 1 *Agrostis* grassland
Area				
(ha)	40	120	120	80
Community data				
Richness	22	33	13	7
Environmental data				
pH[8]	4.2 (0.6)	4.8 (0.5)	4.6 (0.6)	4.4 (0.4)
ODM[1]	1.0 (1.9)	1.1 (1.5)	0.6 (0.4)	1.8 (1.6)
LOI[2]	4.7 (5.1)	4.4 (3.7)	3.2 (2.2)	3.5 (3.0)
Slope[7]	21 (9)	19 (9)	18 (7)	21 (7)
Depth[6]	31 (22)	23 (14)	19 (12)	25 (17)
Cu(w)[5]	1.23 (0.48)	1.65 (1.93)	1.97 (1.37)	4.42 (3.81)
Al (w)[5]	24.31 (35.3)	12.08 (8.88)	12.82 (18.27)	18.84 (24.40)
Zn (w)[5]	1.18 (0.88)	0.56 (0.40)	1.34 (2.11)	1.20 (1.30)
TEB[3]	1.46 (0.18)	0.87 (0.22)	1.99 (2.95)	0.93 (0.23)
N[4]	0.3 (0.1)	0.7 (0.2)	0.6 (0.2)	0.6 (0.3)
n[9]	6	23	17	11

1)%air dry moisture (ODM); Rayment and Higginson (1992).
2) LOI: Loss on ignition (%)
3) TEB: Total exchnageable bases (mEq100g^{-1})
4) Total Kjeldahl nitrogen (%); (Bremer and Mulvaney, 1982)
5) Souluble metals (mg L^{-1})
6) Depth (cm)
7) Slope in degrees
8) Soil pH was measured in a 1:1 soil/distilled water mix using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode.
9) number of sample sites in twinspan group

(Fig. 5). The group had the second lowest mean soil Cu(w) concentration (1.65 ± 1.93 mg L^{-1}). It represented 40% of the sample sites and formed the largest area of any of the vegetation groups.

The last vegetation group was the *Acacia melanoxylon* shrubland. Characteristic shrubs in this group, as distinct from Group 3, included *E. impressa, E. heteronema* and *P. gunnii* (Fig 3). With a total of 22 species represented, the group recorded the highest mean species richness of the four vegetation groups (Fig. 5), while recording the lowest mean soil Cu(w) concentration (1.23 ± 0.48 mg L^{-1}; Table 9). The group was not widespread, being restricted to a total of 10% of the sample sites.

Species frequencies in the four vegetation groups are shown in Table 10.

Mapping of the vegetation groups

Mapping of the distribution of the TWINSPAN groups revealed four discrete vegetation units that were spatially related (Fig. 6). Immediately to the east of the smelters lay Group 1; the group with the lowest species' richness and the highest mean Cu(w) concentration. Adjacent, and progressively further eastwards from Group 1, lay Groups 2 and 3. The same order of vegetation unit separation was evident to the north-east of the smelters. Group 4 lay to the north of the smelters.

Mapping of the soil Cu data into concentration classes provided a spatial representation of soil contamination (Fig. 7). The classes were spatially related and, in general, described an highly contaminated, near-smelter zone surrounded by zones of progressively lower contamination. The pattern was

Table 9: Frequency of species in the four TWINSPAN groups.

Species	Frequency (%)			
	Group 1	Group 2	Group 3	Group 4
Acacia dealbata	-	-	8.7	33.3
Acacia melanoxylon	45.4	-	43.5	100
Acacia mucronata	9.1	41.2	78.3	50.0
Agrostis capillaris	81.8	76.5	56.5	83.3
Atherosperma moschatum	-	-	13.0	16.7
Blechnum nudum	-	-	8.7	-
Blechnum wattsii	-	-	30.4	-
Comesperma retusum	-	-	8.7	16.6
Cyathodes juniperina	9.1	5.9	78.3	16.7
Epacris impressa	-	-	8.7	50.0
Epacris heteronema	-	-	17.4	50.0
Gahnia grandis	-	-	26.1	16.7
Gaultheria hispida	-	70.1	87.0	-
Gleichenia microphylla	-	5.9	47.8	-
Histiopteris incisa	-	5.9		16.7
Hypochaeris radicata	9.1	-	8.7	-
Isolepis aucklandica	33.4	29.4	13.4	-
Lastreopsis acuminata	-	-	-	16.7
Leptospermum scoparium	-	-	43.5	83.3
Melaleuca squamea	-	-	4.3	16.7
Monotoca scoparia	-	-	13.0	16.7
Olearia erubescens	-	5.9	-	-
Oxylobium arborescens	-	-	26.1	33.3
Persoonia gunnii	-	5.9	26.1	83.3
Phebalium squameum	-	-	30.4	16.7
Phyllocladus aspleniifolius	-	-	-	16.7
Restio tetraphyllus	27.3	94.1	91.3	16.7
Sprengelia incarnata	-	-	13.1	33.3
Telopea truncata	-	-	-	33.3

Fig.6: Vegetation map of natural colonisation based on a classification of species' coincidences

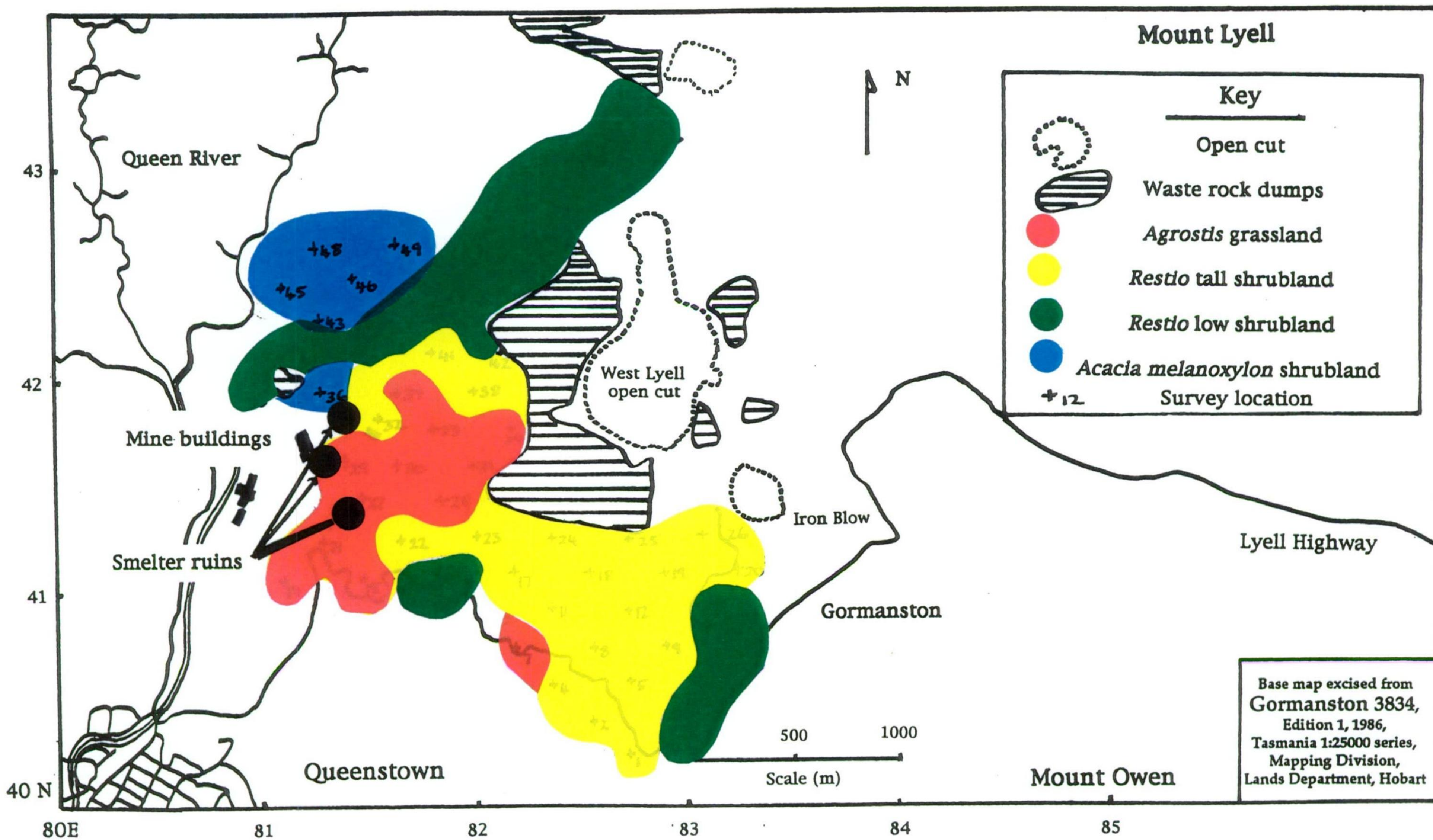
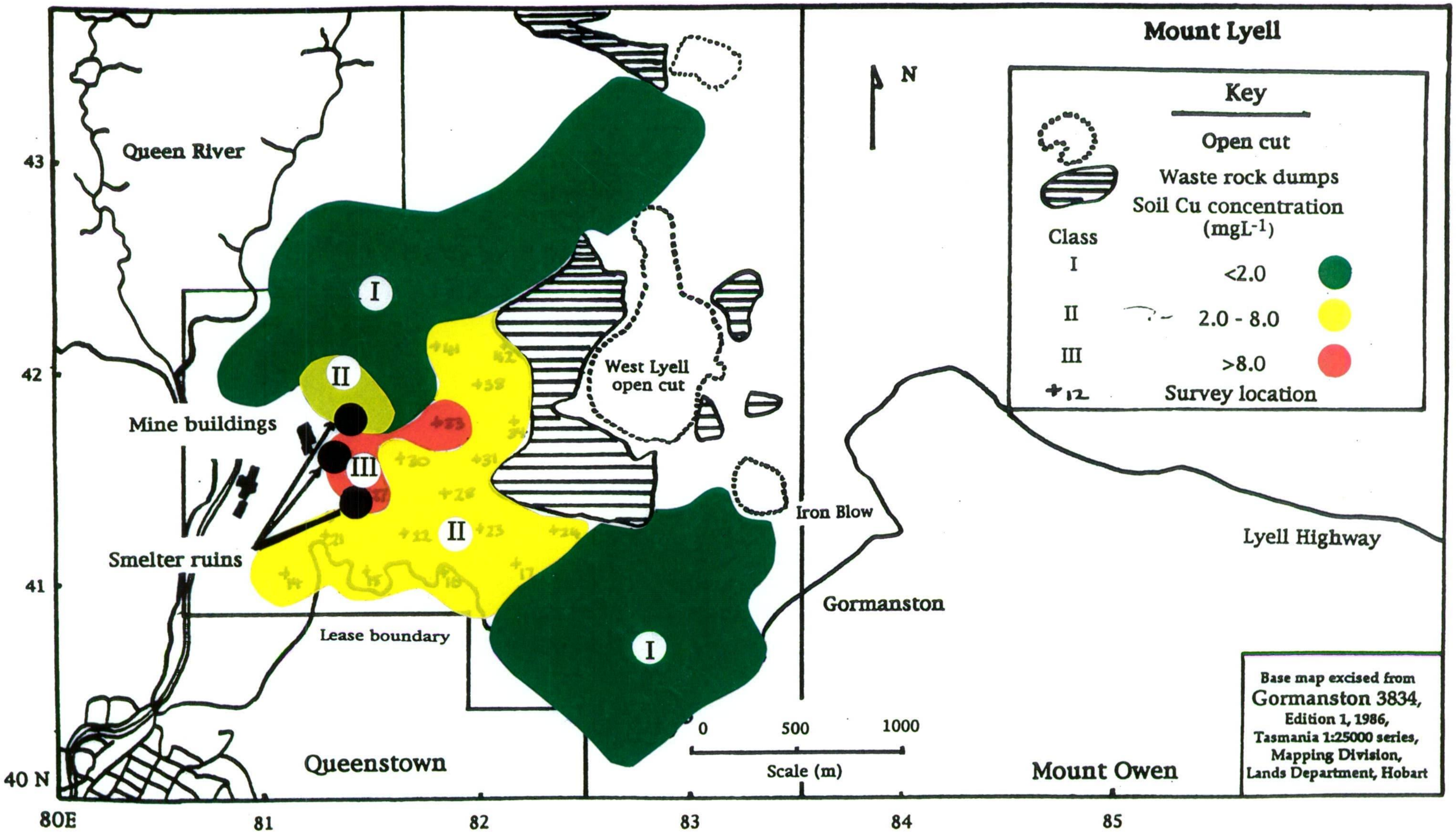


Fig.7: Map of soil Cu concentration classes



consistent with the negative-exponential relationship found previously (Chapter 3).

A strong correspondence was evident between distributions of the Cu concentration classes and the vegetation units. This provided visual evidence of the importance of metal contamination in determining patterns of natural colonisation.

4.4 Discussion

Emissions from base-metal smelters have been known to cause significant vegetation change due to species exclusion (Løbersli and Steinnes, 1988; Wood and Nash, 1976; Hutchinson and Whitby, 1974; Buchauer, 1971; Gorham and Gordon, 1960; Teale, 1951b; Haywood, 1907). During the early operational life of a smelter, species exclusions may occur progressively as a direct result of foliar injury and subsequent defoliation. Early works attributed these impacts to SO_2 fumigation (Blainey, 1993; Linzon, 1972; Gorham and Gordon, 1960). There was some experimental evidence to suggest that foliar injury due to SO_2 fumigation was exacerbated by the presence of metallic dusts (Krause and Kaiser, 1977). Later authors recognised the contribution of elevated concentrations of metallic soil-contaminants to vegetation change (Løbersli and Steinnes, 1988; Hogan and Wotton, 1984; Hazlett *et al.*, 1983; Freedman and Hutchinson, 1980; Wood and Nash, 1976; Hutchinson and Whitby, 1974; Little and Martin, 1972; Buchauer, 1971; Gresta and Godzik, 1969). These authors found phytotoxic soil contaminants capable of causing vegetation change.

The reported effects on vegetation of prolonged exposure to high-sulphur content emissions from Cu smelters range from the elimination of sensitive species to deforestation and denudation (Blainey, 1993; Løbersli and Steinnes, 1988, Hogan and Wotton, 1984; Heale and Ormrod, 1982; Wood and Nash, 1976; Hutchinson and Whitby, 1974). At a community level, these effects usually manifest as reduced species richness. In extreme cases, this type of industrial disturbance can be considered catastrophic; resulting in the almost complete destruction of an ecosystem. Morover, unlike discrete and non-catastrophic, disturbance events, such as fire, flood, landslip or disease, the effects of base-metal smelters on vegetation tend to be long-lasting. This is due to the immobile nature of metallic elements in soils (Wood and Nash, 1976; Jones and belling, 1967).

In circumstances where soil contamination results in phytotoxicity, the effects of residual contamination may outlive the active operation of a smelter by many decades. In consequence, phytotoxic soil contamination is likely to influence the pattern of natural colonisation in the vicinity of decommissioned base-metal smelters. Antonovics *et al.* (1971) recognised that contaminated sites offered an opportunity to study the effect of metals on plant distribution.

Smelter closure, or a reduction in smelter emissions, may provide an opportunity to metal-tolerant colonisers from neighbouring vegetation.
Detailed reports of the patterns of natural colonisation in metal contaminated areas following the closure of a smelter are, however, scant. The Ni/Cu smelter at Coniston, Ontario closed in 1972. Six years later, Cox and Hutcthinson (1980, 1979) reported that the grass *Deschampsia cespitosa* had invaded large areas of the barren area surrounding the smelter and similar nearby complexes. On phytotoxic soils with both low pH and elevated metal concentrations, they attributed colonisation to the development of metal tolerance. Similarly, Rauser and Winterhalder (1985) identified Cu and Ni tolerance in *D. cespitosa* on the

highly contaminated, Coniston roast bed. On another abandoned roast bed in the same Sudbury smelting region, Hogan and Courtin (1977b) identified Cu tolerance in the grass *Agrostis gigantea.* The plants were growing in discrete 'islands' with both high extractable Cu and low pH. Non-tolerant clones were apparently restricted to the more favourable sites (Hogan and Courtin, 1977a).

At Mount Lyell, site disturbance and contamination resulted in site availability. With the closure of the last smelter in 1969, site availability provided an opportunity that is best understood as one of primary colonisation. Unlike secondary colonisation, which begins with a more or less mature soil containing stored seeds and vegetative propagules, colonisation at Mount Lyell began largely from bare rock and inorganic soils. The earliest colonisers were reportedly algae - isolates of algal species from contaminated sites have been elsewhere suspected of copper tolerance (Twiss, 1990) - and subsequently, mosses. More than twenty years later, higher plants have established only a limited presence on the mountain.

In this study, the presence of vascular plant species was understood to be a function of both successful dispersal (or, in the case of *A. melanoxylon*, persistence) and differential species performance. Species performance was measured by occurrence. These factors resulted in a strong, selective gradient. The strong relationship between species coincidence and soil Cu concentrations indicated that differential occurrence was largely a result of varying Cu tolerance.

Varying tolerance to soil Cu was the only environmental variable examined capable of substantially explaining the species performance component of the selective gradient. The role of the variable soil depth is likely to have been minor. Soil depth may have contributed to species coincidence through an effect on the physical favourability of the skeletal soils to germination. In general, the least favourable of the soils were found closest to the smelters. Alternatively, depth may have contributed to species coincidence by chance, due to a gradient in soil depth from the smelter, situated in the valley bottom, to the middle slopes of Mount Lyell. This gradient tended to parallel that displayed by soil Cu concentration. Similarly, soil organic matter, represented by loss-on-ignition (LOI), may have contributed to species coincidence in a minor way through an effect on plant-available Cu. There was no relationship between Cu(w) and depth. This was not unexpected given that the surface Cu concentrations were understood to be largely of depositional, rather than of parental, origin.

Varying tolerance to Cu is considered responsible for the formation of the distinct vegetation groups. The spatial relationships revealed by mapping of the vegetation groups suggested vegetation zonation. This was understood to be the result of spatial shifts in the dominance of tolerant or resistant species (*sensu* Smith and Huston, 1989). Winterhalder (1987), similarly, recognised spatial heterogeneity between colonising plants and related them to contamination patterns in the Sudbury region.

The presence of *Agrostis capillaris* at *Agrostis* grassland sites was testimony of the species ability to tolerate high concentrations of available Cu. Copper tolerance in this species accorded with the findings of earlier experimental works (Symeondis *et al., 1985;* Nicholls and McNeilly, 1979; Wainwright and Woolhouse, 1977; Walley *et al.*, 1974; Gregory and Bradshaw, 1965).

The tolerance, invasive ability and durability of members of this genus have been demonstrated at a number of metal contaminated sites throughout the northern hemisphere. The invasive nature of the local, native, *A. capillaris,* was inadvertently demonstrated on a metal contaminated site in Ammerberg, Sweden (Bergholm and Steen, 1989). Over a 10 year period the species was able to invade plots on zinc sand wastes, amended with sewage sludge or topsoil and sown to pasture grasses, attaining a cover of 20-50%. In the Sudbury region, *A. scabra* and *A. gigantea* populations were able to partially colonise, albeit with a cover of less than 1%, a highly contaminated, abandoned roast bed that had lain undisturbed since 1929 (Hogan *et al.*, 1977b). In the same region, Hazlett *et al.*, (1983) reported that falling smelter emissions in the 1970's permitted partial recolonisation of denuded areas by *A. scabra.* This was the only species recorded to tolerate the least favourable, near-smelter sites.

4.5 Conclusion

The industrial disturbance seen at Mount Lyell was catastrophic in the almost complete destruction of an ecosystem. Subsequent closure of the smelters presented an opportunity to colonising species. However, residual, and spatially discrete, soil contamination has dominated the pattern of natural colonisation by providing a strong, selective gradient. In this study, the differential occurrence of colonising species was found to be a result of varying Cu tolerance. In conjunction with the ability to invade, varying species tolerance was responsible for the formation of distinct, zonal, vegetation groups. This was reflected in the richness of the groups. The vegetation groups were distinct from the present-day environs of the region; typified by the fire-disclimax communities of Eucalypt forest, Blackwood forest, Tea-tree scrub and Buttongrass sedgeland. At Mount Lyell, further unaided species introductions appear to be dependent upon reductions in Cu-related soil phytotoxicity.

Chapter 5

The reclamation of metal-contaminated soils with phytotoxic characteristics: toxicity amelioration with a chemical neutralising agent

5.0 General introduction

Atmospheric emissions from base-metal smelters have contaminated soils by metal deposition, or acid-induced soil alterations (Nelson and Cambell, 1991; Reuss *et al.*, 1987, Malmer, 1976), and have resulted in soils with phytotoxic characteristics. In plants, the symptoms of metal phytotoxicity can be expressed as either foliar or root abnormalities. Stunted growth is a common abnormality in both roots and shoots (Foy, 1992) and foliar discolouration may occur (Foy, 1992). Metal toxicity resulting in root abnormalities may also be expressed indirectly as macronutrient nutrient imbalances or deficiencies (Foy, 1984; Smith and Bradshaw, 1972) and can render plants susceptible to water stress (Foy, 1988). In cases of extreme phytotoxicity, increased mortality is likely. At the community level, a typical symptom of metal phytotoxicity in the vicinity of decommissioned base-metal smelters is low species richness (eg. Wood and Nash, 1976). Low species richness is generally the result of the exclusion of non-tolerant species.

Two general approaches to have been used to reclaim chemically contaminated soils; engineering and ecological approaches (Logan, 1992). Engineering approaches have been used in cases of extreme contamination and include various froms of removal and subsequent treatment, chemical or heat immobilization/mobilization and biotic/abiotic degradation.

Ecological approaches to soil reclamation involve the manipulation of inherent soil processes to immobilize, mobilize, transform or degrade contaminants. Ecological reclamation may include landscape stabilization, neutralisation of acid or alkaline soils, the addition of organic matter, fertilization and the establishment of vegetative cover. Specific site requirements may necessitate a combination of engineering and ecological approaches to soil reclamation.

Steep topography, high rainfall, limited accessibility and the chemical characteristics of the contaminants at the Mount Lyell site restricted the choice of appropriate soil reclamation methods. As is often the case in rocky soils and mine spoils (Logan, 1992), engineering approaches such as deep ripping in order to dilute soil contaminants were impractical at Mount Lyell due to the topography, numerous rocky outcrops, poor accessibility and a risk of severe erosion. Similarly, the importation of large volumes of organic matter in order to complex soluble metallic contaminants was impracticable without a plausible means of incorporation and reasonable chance of bio-degradation.

In situ immobilization methods appeared to offer a means of reducing the solubility of soil contaminants at Mount Lyell. Reference to the literature suggested that lime might be a suitable ameliorant (eg. Alva *et al.*, 1993; Logan, 1992; Logan and Cassler, 1989; Winterhalder, 1988; Winterhalder, 1983a; Hume and Winterhalder, 1983; Winterhalder, 1981a/b). Where metal toxicity occurs, lime application has the potential to ameliorate toxicity by enhancing sorption and precipitation (Logan, 1992) and by enhancing metal complexation with organic matter (Logan, 1992). In circumstances where excess acidity prevails, lime is recognised for its positive contribution to soil structure, to nutrient bioavailability (Logan, 1992), and consequently to plant growth and productivity (Logan, 1992; Atlas and Bartha, 1981).

In this chapter, contaminated and potentially phytotoxic soils were removed at intervals along a metal concentration gradient originating at the Mount Lyell smelter ruins and examined in pot trials. The growth of seedlings of local colonising species was used as a measure of phytotoxicity and lime requirements were determined directly following treatment applications. Field trials examined the ameliorative effects of lime when applied as surface treatments and the potential for broadacre lime application.

Part A: Pot trials

5.1 Introduction

The growth of two local colonising species in soils taken along an easterly transect originating at the Mount Lyell smelters was compared in a pot trial. Along this transect, soils ranged in soluble Cu and Al from concentrations above those reported as phytotoxic to tree species to background levels. Calcitic lime ($CaCO_3$) was selected as an appropriate neutralising agent due to its low cost, availability and reputation as a soil improver on both agricultural lands and acid mine spoils. The sample soils were treated with a calcitic lime at application rates intended to raise soil pH to between 5 and neutral. Lime was added to the upper layer of soil of each pot as the treatment of this layer has resulted in superior growth in pot trials of acid, metal-contaminated soils (Pinkerton and Simpson, 1977). The response to treatments was assessed by observation and measurement.

The local tree/shrub species *Acacia melanoxylon* and *Leptospermum scoparium* were chosen for the trial due to the relatively high frequency of their occurrence in recolonised areas and the potential of their contribution to revegetation. Local provenance seed of *Acacia mucronata,* another species occurring in these areas, was unavailable at the time of the trial.

5.2 Methods

5.21 Soil sampling

Soil samples were taken at five sampling sites located in an easterly direction from the original Mount Lyell. The sites were the same as those selected in Chapter 3 with the exception that site 6 was excluded. The sampling locations are shown in figure 2 of the same chapter.

Soil sampling and sample preparation for soil chemical analysis followed the procedures described in Chapter 3, sections 3.21 and 3.22, respectively. Soluble metal-ion concentrations for the soils were obtained in mg L^{-1} for the metals Cu, Zn, and Al using soil-water extracts. The extraction proceedure for the metal-ion analyses followed was given in Chapter 3 (sect. 3.22). The water soluble, metal-ion analyses were complemented by determinations for DTPA extractable Cu (Lindsay and Norvell, 1978), and Zn (Lindsay and Norvell, 1978) and $CaCl_2$ extractable Al. The extractions were performed and measured (AAS) by Mount Pleasant Laboratories, Kings Meadows, Tasmania on sieved (1 mm mesh), air-dry soil samples.

5.22 Soil and glasshouse preparation

Acacia melanoxylon

In the glasshouse, two hundred tapered, plastic seedling pots (5x5x12 cm) were evenly divided into five blocks. These were filled with soil from one of each of the five sampling sites to a depth of one half their volume. In each block, the remaining volume was made up by allocating soil treated with one of four levels of lime. There were 10 replications.

To minimise variation, the treatments were pre-prepared in bulk by thoroughly mixing the quantity of lime and soil required. The treatments were ground calcitic lime (mesh size 1.0 mm) at the rate of 0, 200, 400 and 600 g m^{-2}. The application rates were selected following consideration of the lime requirements generally experienced with clay-loams (4 000 to 5000 kg ha^{-1} every three to four years; Sopher and Baird, 1982), the origin of the soils and the soil reaction. The treated pots were labelled, watered and allowed to stand for two weeks.

Locally collected, Mount Lyell provenance *A. melanoxylon* seeds were heat stratified in near boiling water for 45 seconds. Directly following stratification the seeds were immersed in cold water and allowed to stand overnight. The seeds were then placed on open plastic trays and incubated at 20°C in a constant temperature cabinet. Following germination seedlings of equivalent radical length (5±1 mm) were selected, transferred to the glasshouse and planted bare-rooted in the pre-prepared tubes. Two seedlings were planted per tube. The tubes were randomised on the glasshouse bench and watered daily.

The seedlings were harvested after 53 days. This period was selected following visual inspection of a preliminary pot trial which indicated a lime-treatment response after 40 plus days. Each seedling was carefully removed from its tube and the root system was thoroughly washed. Root nodulation was noted. The seedlings were then oven dried at 60 °C for 6 hours and weighed, prior to bagging and labelling.

At harvest, four pots from each treatment in each block were selected at random for pH measurement. Only the upper half of the soil contained in each pot was sampled (0-5cm). Soil pH was measured in a 1:1 soil/ water suspension using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode.

Leptospermum scoparium

A similar pot trial was performed using the species *L. scoparium* and soil from the same 5 soil sampling sites. The procedure used was as for *A. melanoxylon* with the following three exceptions; an extra treatment level was added (100 gm^{-2}), the seedlings were transplanted as 5-week old seedlings and one seedling was planted per pot.

5.23 Data analysis

A. melanoxylon

The *A. melanoxylon* experiment formed a single factor, randomised complete block with treatments allocated to pots within blocks. There were four lime treatments, five soil sampling sites and two subsamples per pot. The treatments were replicated 10 times.

The seedling dry weight data were subjected to a two-way analysis of variance (ANOVA) using the generalised linear modelling (GLM) procedure in SAS/STAT® edition 6.03 (SAS Institute Inc., 1988). The program treated the data for each pair of seedlings in the pots (experimental units) as subsample measurements. Plots of residuals versus fitted values indicated that data conformed to the model requirements of normality and homogeneity. The between treatments effect was tested against a block times treatment residual.

Least square means were calculated and ranked. The means were compared using the probability difference (pDiff) option of the SAS/STAT® program. This permitted a pairwise comparison of means.

Soil-metal concentrations were correlated with seedling dry weight.

L. scoparium

The *L. scoparium* experiment formed a single factor, randomised complete block with treatments allocated to pots within blocks. There were five lime treatments and five soil sampling sites. The treatments were replicated eight times.

Statistical handling of the *L. scoparium* data was similar to that of the a *A. melanoxylon* experiment. Soil-metal concentrations were correlated with seedling dry weight.

5.4 Results

5.41 Chemical characteristics of the pot trial soils

The chemical characteristics of the soils taken along the transect and used in the pot trials are summarised in Table 1a.

Table 1a: Surface soil characteristics for five composite soil samples representing sites along a radial transect originating at the Mount Lyell smelters.

Sample sites	1	2	3	4	5
Displacement from smelters (m)	230	400	650	850	1150
Soil texture[1]	medium clay	sandy clay	light sandy loam	sandy loam	sandy clay loam
Soil pH[2]	4.3	4.5	4.7	4.8	4.5
Loss on ignition (%)	3.9	3.7	3.1	5.3	7.0
% air dry moisture	0.6	0.7	1.5	2.0	2.6
N(%Total)[3]	0.78	0.90	0.37	0.47	0.35
TEB (meq $100g^{-1}$)[4]	1.25	0.99	0.73	0.76	0.95
Ca (meq $100g^{-1}$)[5]	0.36	0.29	0.16	0.18	0.20
Water extractable metal ion concentrations[7]					
Cu (mg L^{-1})	10.41	6.72	2.34	0.90	0.39
Zn (mg L^{-1})	0.75	0.48	0.18	0.36	0.06
Al (mg L^{-1})	5.94	10.5	19.32	0.27	0.09
Extractable metal concentrations					
Cu (DTPA)	51.70	222.0	70.7	10.5	31.5
Zn (DTPA)	1.63	17.7	16.30	0.30	1.98
Al($CaCl_2$)	41.40	27.7	17.6	8.28	2.30

1) Soil texture determination based on ANU Forestry texture evaluation guideline derived from Northcote (1979).
2) Soil pH was measured in a 1:1 soil/distilled water mix using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode. n=4.
3)Kjeldahl procedure (Bremer and Mulvaney, 1982)
4) TEB = Total NH_4/CL exachangable bases
5) NH_4Cl extractable calcium
6) Air dry moisture (ODM) Rayment and Higginson (1992).
7) Extraction proceedure as per Ch. 3 (sect. 3.22)

5.42 *A. melanoxylon*

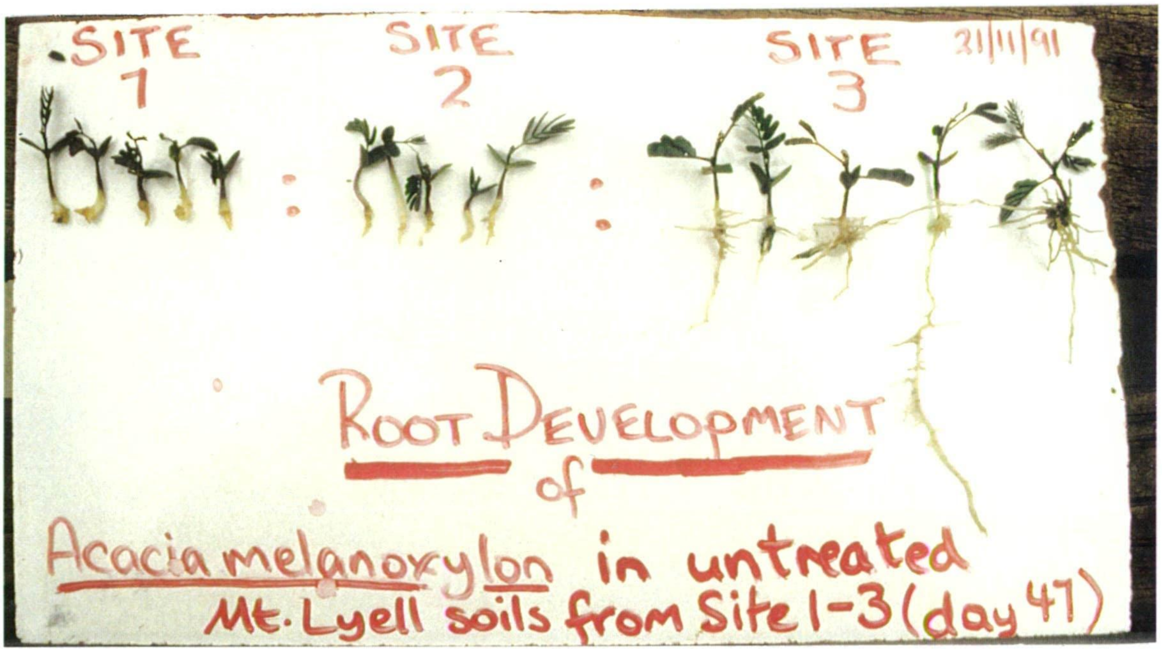
Seedling growth in untreated site 1 and site 2 soils stagnated after the first couple of weeks. Most shoots failed to develop beyond the seed leaves, and root development was severely inhibited. Typically, the root systems were severely stunted, thickened, and lacked branching and root hairs (Fig. 1a). The tips of the primary roots were commonly stained a watery-brown or blackened. This was understood to be necrotic, cell collapse. The worst affected systems lacked any lateral development and were reduced to a stubby and thickened primary root. This resulted in mechanical instability and sometimes collapse. Nodulation did not occur. While none of the seedlings died within the period of the trial, survival appeared unlikely.

The severity of root abnormalities decreased markedly with increasing displacement from the smelter. The root systems in untreated sites 3, 4 and 5 soils were progressively longer, less thickened and more branched (Fig. 1a and 1b). Roots tended to remain cream-white in colour throughout their length but nodulation did not occur. Shoots developed secondary leaves and seedling survival did not appear to be in question.

The appearance of the *A. melanoxylon* root systems improved dramatically with the application of lime at all levels and resulted in predictable rise in soil pH (Table 1b). Nodulation occurred on seedlings grown in soils from sites 3, 4 and 5. The improvement in root appearance with liming was particularly evident in soils from sites 1 and 2. Typically, the root systems gained length, were more branched and less thickened (Fig. 2). Necrosis of the root tip did not occur. Root hairs, however, were commonly absent.

Analysis of variance of *A. melanoxylon* seedling dry weights indicated

Figure 1: *A. melanoxylon* root development in untreated Mount Lyell soils in a preliminary pot trial. Fig. 1a: seedlings grown in site 1, 2 and 3 soils. Fig.1b: Seedlings grown in site 4 and 5 soils. The photographs were taken on day 47 of a preliminary trial. The lowercaption in Fig.1b should read sites 1-5.



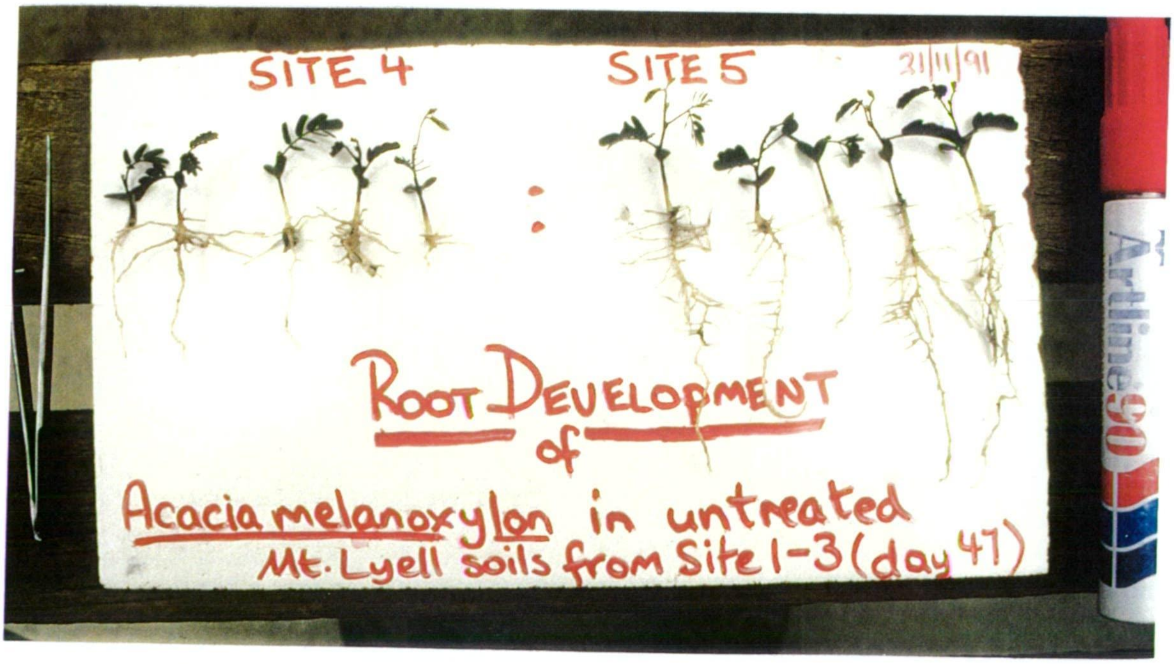
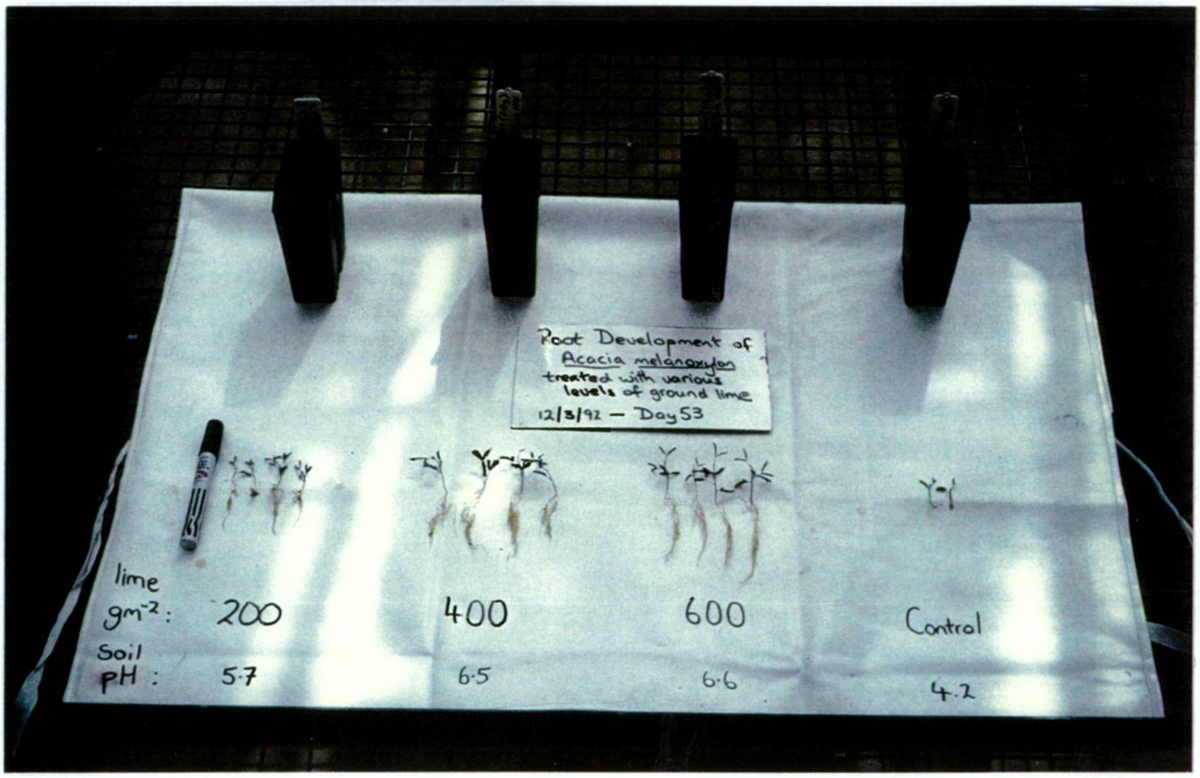


Table 1b: Soil pH[1] for five soils sampled along an easterly transect originating at the Mount Lyell smelters and treated with ground lime in a pot trial. The measurements were taken on the day of harvest.

Site	Displacement from smelter (m)	Lime treatment (gm^{-2})				
		0	100	200	400	600
1	230	4.22	5.14	5.75	6.53	6.58
2	400	4.51	5.25	5.67	6.28	6.53
3	650	4.73	5.39	5.43	6.31	6.49
4	850	4.84	5.46	5.68	6.74	7.65
5	1150	4.49	5.05	5.55	6.36	7.15
Mean		4.6	5.3	5.6	6.4	6.9

1) Soil pH was measured in a 1:1 soil/distilled water mix using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode. n=4.

Figure 2: Photo of *A. melanoxylon* root development in lime treated and untreated (control) Mount Lyell soils from site 1 in pot trial. The photo was taken on the day of harvest.



that a site by treatment interaction existed ($F_{12,19}$ = 6.84, p = 0.0001; Table 2). The interaction between treatments and sites can readily be recognised in the plot of mean seedling dry weight against displacement of each of the sites (Fig. 3). Subsequent analysis, using site by treatment mean square as an error term, showed that the means differed between treatments ($F_{3,19}$ = 8.26, p = 0.0030), but not significantly between sites ($F_{4,19}$ = 1.09, p = 0.4046).

Pairwise comparisons of differences between treatment means indicated that, in comparison to the controls, the addition of ground calcitic lime increased the dry weight of *A.melanoxylon* seedlings at most, but not all, of the Mount Lyell sites (Table 3). The exceptions were the 400 and 600 g m^{-2} treatments at site 5. However, although all of the lime treatments improved dry weight at sites 1 to 4 inclusive, differences between the responses to treatment levels at these sites were not statistically distinguishable. At site 5, only the 200 gm^{-2} treatment improved seedling dry weight. Higher lime application rates at this site produced no measurable benefits over that of the control treatment.

Correlation of soil-metal concentrations and seedling dry weights

Regressions of soluble soil-metal concentration and the seedling dry weight data displayed distinct differences between the metals. For *A. melanoxylon*, seedling dry weight was negatively and linearly related to the metals Cu (R = -0.91) and Zn (R = -0.89), but appeared unrelated to Al (R = -0.26; Fig. 4a-c). In comparison, *A. melanoxylon* seedling dry weight was negatively, but less strongly related to DTPA-extracted Cu (R = -0.64) and DTPA-extractable Zn; Fig. 4d-f). A strong negative, linear relationship was found between seedling dry weight and $CaCl_2$ extracted Al (R = -0.93).

Table 2: Summary of an analysis of variance of *Acacia melanoxylon* seedling dry weights from a lime-application pot trial. Pre-germinated seedlings of equivalent radical development were transplanted into soils taken from five sites located to the east of the Mount Lyell smelter complex. The soils were treated prior to transplantation with ground lime at the rates of 0, 200, 400 and 600 gm^{-2}. Two seedlings were planted per pot. The seedlings were harvested after 53 days. The data were analysed as a randomised complete block of five sites and four treatments with subsampling .

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p*
Model	19	0.085	0.0045	14.45	0.0001
Error	333	0.1033	0.0003		
Corrected Total	352	0.1884			

Source	*DF*	*Type IIISS*	*MS*	*F Value*	*P>F*
Treatment	3	0.053	0.017	56.52	0.0001
Site	4	0.009	0.002	7.46	0.0001
T'ment* Site	12	0.254	0.002	6.84	0.0001
*Tests of hypotheses using Type 3 MS for Site*Treatment as an error term*					
Treatment	3	0.053	0.017	80.26	0.0030
Site	4	0.009	0.002	1.09	0.4046

Figure 3: Mean total dry weights for *A. melanoxylon* seedlings in soil removed from five Mount Lyell locations and subjected to four lime treatments in a pot trial. The soils samples were taken at increasing eastward displacement from the original Mount Lyell smelters. The treatments (ground lime) were applied at the rates of 0, 200, 400 and 600 gm^{-2} and mixed with soil in the top five centimetres of each pot. The seedlings were transplanted as germinated seed and harvested after 53 days. Error bars represent standard deviations.

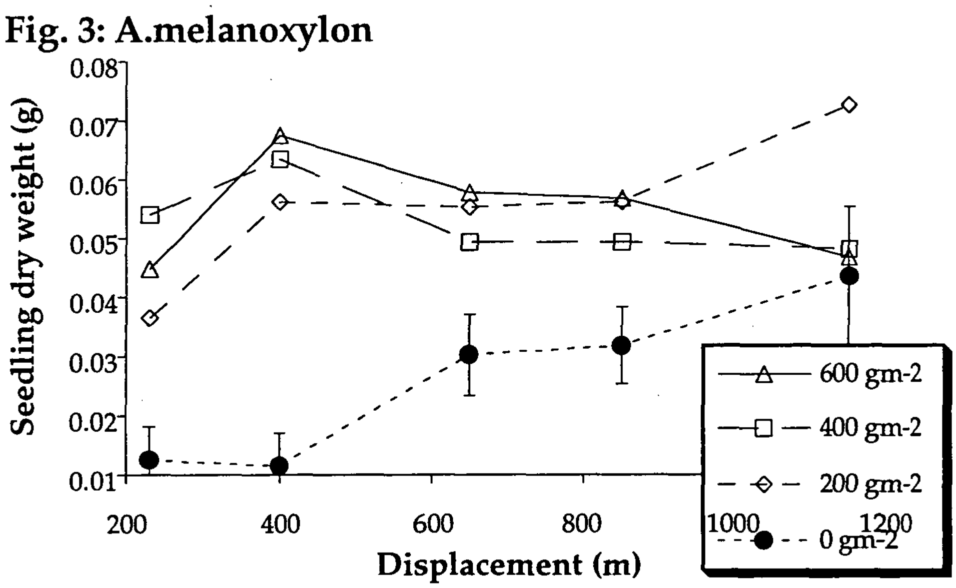


Table 3: Mean total dry weights for *A. melanoxylon* seedlings grown in soils from five Mount Lyell sites and subjected to four lime treatments (0, 200, 400 and 600 gm^{-2}) in a pot trial. Differences between ranked treatment means at each site were extracted from a probability table generated by the 'pdiff' option in the analysis of variance procedure offered in SAS/STAT[R] edition 6.03 (SAS Institute Inc., 1988). Differences are summarised by annotation: any two means annotated with a dissimilar letter, or letter combinations, are significantly different at the 0.05 level or greater.

Site	Lime treatment means				
1	Treatment	0	2000	6000	4000
	Mean	0.012a	0.036b	0.045b	0.054b
2	Treatment	0	2000	4000	6000
	Mean	0.011a	0.056de	0.063df	0.067df
3	Treatment	0	4000	2000	6000
	Mean	0.030c	0.050de	0.055de	0.058de
4	Treatment	0	4000	2000	6000
	Means	0.032c	0.049de	0.056de	0.057de
5	Treatment	0	6000	4000	2000
	Means	0.043be	0.047de	0.048de	0.073f

Figure 4a-c: Regression of soil-metal concentrations and *A. melanoxylon* mean seedling dry weight. Soil-water extracts were made up with distilled water at a dilution rate of 3 to 1. Acidified samples were analysed by AA spectrometry. Seedlings were grown in pot trial in Mount Lyell soils sampled along a transect originating at the smelter installations. n =10.

Fig. 4a: Soil metals/ A. melanoxylon

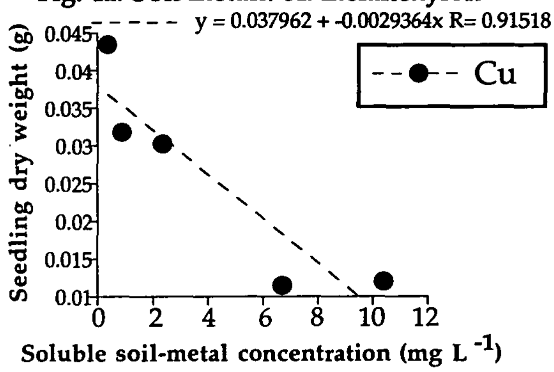


Fig. 4b: Soil metals/ A. melanoxylon

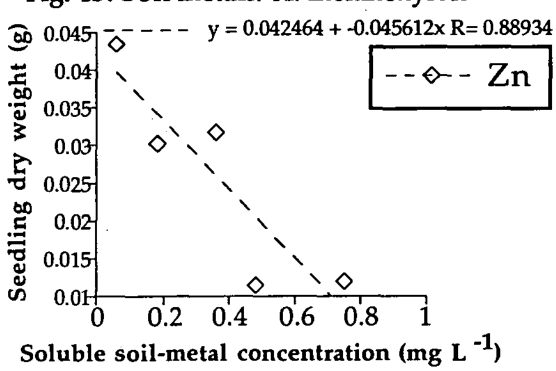


Fig. 4c: Soil metals/A.melanoxylon

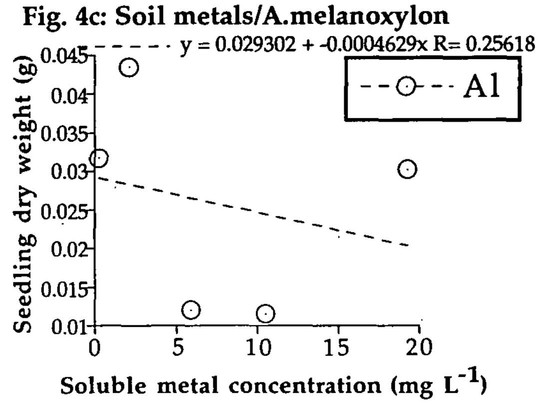
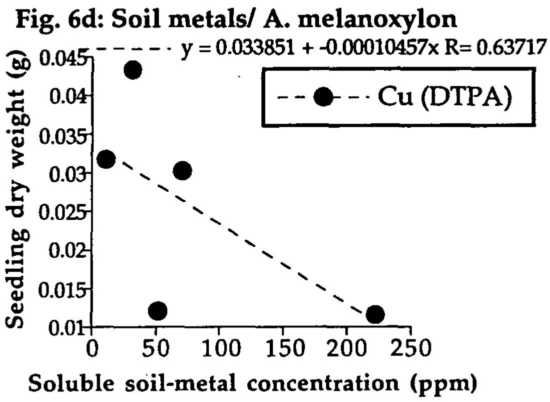
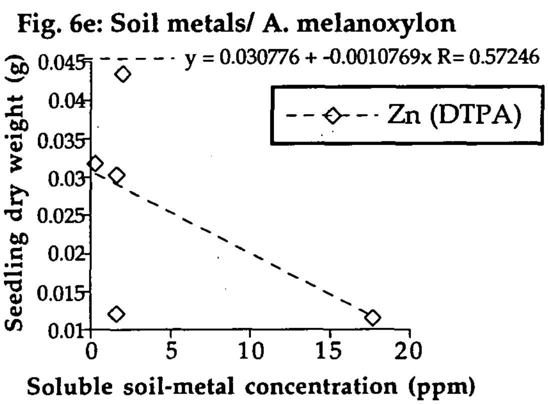
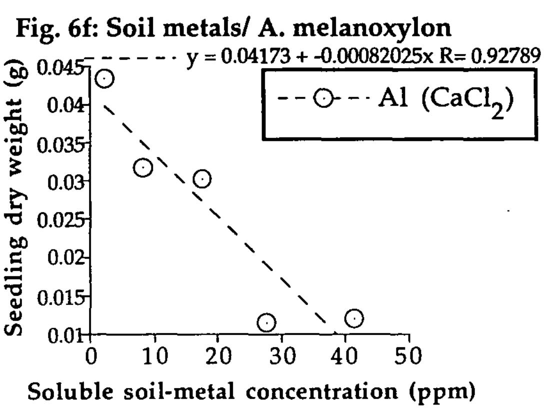


Figure 4d-f: Regression of soil-metal concentrations and *A. melanoxylon* mean seedling dry weight. Soil metals were extracted in DTPA (Cu and Zn) or $CaCl_2$ (Al) and analysed by AA spectrometry. Seedlings were grown in pot trial in Mount Lyell soils sampled along a transect originating at the smelter installations. n=10.







5.43 *L. scoparium*

Although survival did not appear to be in question in any of the untreated soils, seedling growth was slow and resulted in reduced internode lengths and leaf sizes. In untreated site 1 and site 2 soils, pallid leaf tones indicated chlorosis. Above-ground growth rates appeared to increase with the displacement of the sampling sites from the smelters. Root abnormalities in *L. scoparium* appeared less severe than in *A. melanoxylon*. In soils from sites 1 and 2, the systems were poorly developed, lacking both length and branching. Root tip necrosis and thickening were, however, generally absent. Like *A. melanoxylon*, the severity of root abnormalities appeared to be related to increasing displacement of the soil sampling sites. In untreated site 3, 4 and 5 soils, *L scoparium* root systems became progressively longer and more branched.

The appearance of the *L. scoparium* root systems improved with the application of lime at all levels. Again, the improvement was particularly noticable for sites 1 and 2. The root systems gained length and were more branched.

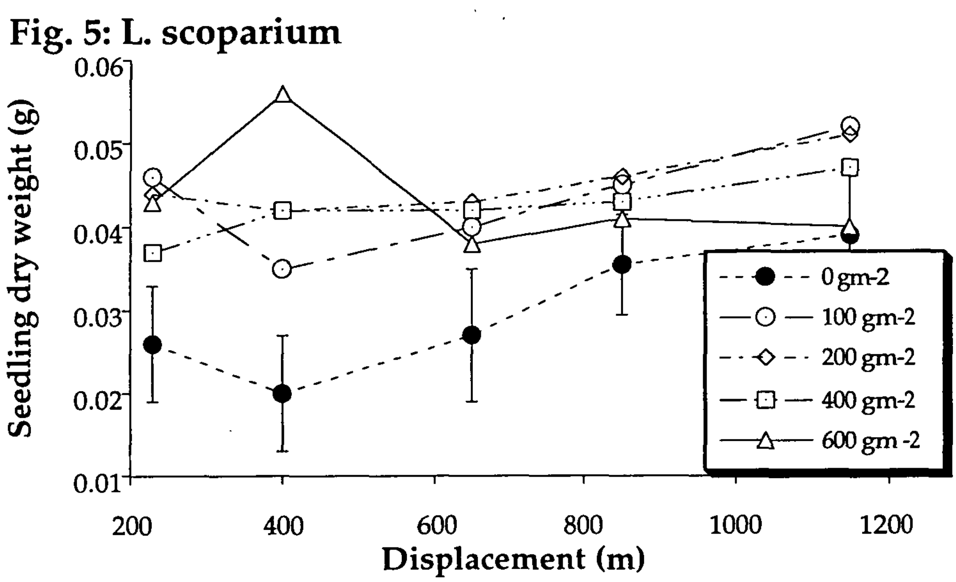
Analysis of variance for *L. scoparium* mean seedling dry weights indicated that a site by treatment interaction existed ($F_{16,168} = 1.73$, $p = 0.0455$; Table 4). The interaction between treatments and sites can be recognised in the plot of mean seedling dry weight against displacement of each of the sites (Fig. 5). Subsequent analysis, using site by treatment mean square as an error term, showed that the means differed between treatments ($F_{4,16} = 5.67$, $p = 0.0049$), but not significantly between sites ($F_{4,16} = 1.68$, $p = 0.2027$; Table 4).

Pairwise comparisons of differences between treatment means indicated

Table 4: Summary of an analysis of variance of *Leptospermum scoparium* seedling dry weights from a lime-application pot trial. Pre-germinated seedlings of equivalent radical development were transplanted into soils taken from three sites located to the east of the Mount Lyell smelter complex. The soils were treated prior to transplantation with ground lime at the rates of 0, 100, 200, 400 and 600 gm^{-2}. One seedling was planted per pot. The seedlings were harvested after 53 days. The data were analysed as a randomised complete block of three sites and five treatments without subsampling.

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p >F*
Model	24	0.0116	0.0005	3.28	0.0001
Error	168	0.0366	0.0001		
Corrected Total	192	0.0366			
Source	*DF*	*Type IIISS*	*MS*	*F value*	*p>F*
Site	4	0.0017	0.0004	2.91	0.0231
Treatment	4	0.0058	0.0014	9.81	0.0001
T'ment* Site	16	0.0041	0.0002	1.73	0.0455
*Tests of hypotheses using Type III MS for Site *Treatment as an error term*					
Treatment	4	0.0058	0.0014	5.67	0.0049
Site	4	0.0017	0.0004	1.68	0.2027

Figure 5: Mean total dry weights for *L. scoparium* seedlings in soil removed from five Mount Lyell locations and subjected to five lime treatments in a pot trial. The soil samples were taken at increasing eastward displacement from the original Mount Lyell smelters. The treatments (ground lime) were applied at the rates of 0, 100, 200, 400 and 600 gm-2 and mixed with soil in the top five centimetres of each pot. The seedlings were transplanted as germinated seed and harvested after 53 days. Error bars represent standard deviations.



that, in comparison to the controls, the addition of ground calcitic lime increased the dry weight of *L. scoparium* seedlings at sites 1, 2 and 3. (Table 5). With one exception (Site 2, 600 gm^{-2}), there was nothing to distinguish between the four levels of lime treatment at these sites. In contrast, there was no improvement as a result of lime treatment at site 4. At site 5, only the treatments 100 and 200 gm^{-2} resulted in an increase in dry weight.

Correlation of soil-metal concentrations and seedling dry weights

Regressions of soluble soil-metal and seedling dry weight data for the species *L. scoparium* displayed similar patterns to that found with *A. melanoxylon*. Negative linear relationships were found with the metals Cu (R = -0.73) and Zn (R = -0.60; Fig. 6a-b). A weaker relationship was found with the metal Al (R = -0.63; Fig. 6c). Seedling dry weight was also negatively related to DTPA-extracted Cu (R = -0.81), DTPA-extracted Zn (R = -0.69) and $CaCl_2$ extracted Al (R = -0.80; Fig. 6d-f).

5.5 Discussion

Tolerance

Seedlings of both *A. melanoxylon* and *L. scoparium* were similarly intolerant of the untreated Mount Lyell soils. Intolerance was especially marked in soils taken from the sampling sites closest to the smelters. The visual symptoms of intolerance at these sites were typified by stunted shoots and severe root system abnormalities. The symptoms, and particularly those of the root systems, were consistent with those known to occur as a

Table 5: Mean total dry weights for *L. scoparium* seedlings in soil removed from three Mount Lyell locations and subjected to five lime treatments (0, 100, 200, 400 and 600 gm^{-2}) in a pot trial. Differences between ranked treatment means at each site were extracted from a probability table generated by the 'pdiff' option in the analysis of variance procedure offered in SAS/STAT[R] edition 6.03 (SAS Institute Inc., 1988). Differences are summarised by annotation: any two means annotated with dissimilar letter combinations are significantly different at the 0.05 level or greater.

Site	Lime treatment means					
1	Treatment	0	400	600	200	100
	Mean	0.026a	0.037b	0.043b	0.044b	0.046b
2	Treatment	0	100	200	400	600
	Mean	0.020a	0.034b	0.034b	0.042b	0.056d
3	Treatment	0	600	100	400	200
	Mean	0.027ac	0.037b	0.040b	0.042b	0.043b
4	Treatment	0	600	400	100	200
	Mean	0.035cb	0.041b	0.043b	0.045bd	0.046bd
5	Treatment	0	600	400	100	200
	Mean	0.039cb	0.040cd	0.040cd	0.050d	0.051d

Figure 6a-c: Regression of soil-metal concentrations and *L. scoparium* mean seedling dry weight. Soil-water extracts were made up with distilled water at a dilution rate of 3 to 1. Acidified samples were analysed by AA spectrometry. Seedlings were grown in pot trial in Mount Lyell soils sampled along a transect originating at the smelter installations. n =10

Fig. 6a: Soil metals/ L. scoparium

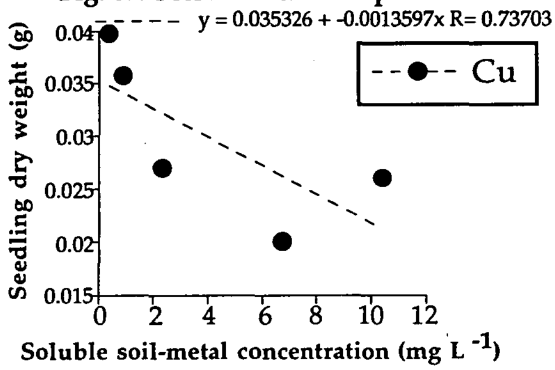


Fig. 6b: Soil metals/ L. scoparium

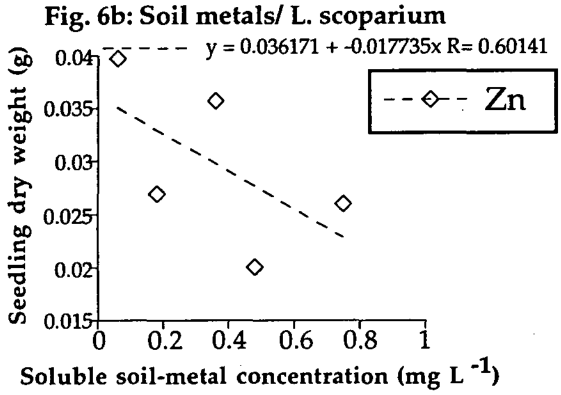


Fig. 6c: Soil metals/L. scoparium

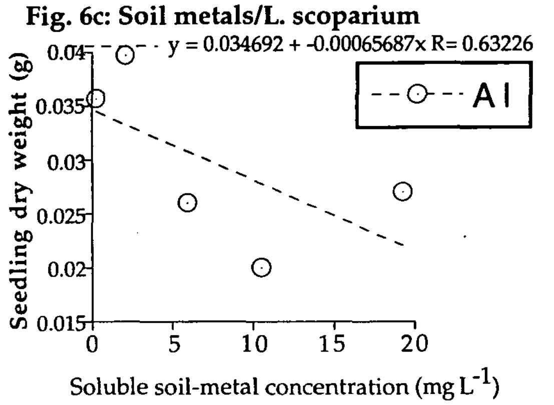


Figure 6d-f: Regression of soil-metal concentrations and *L. scoparium* mean seedling dry weight. Soil metals were extracted in DTPA (Cu and Zn) or $CaCl_2$ (Al) and analysed by AA spectrometry. Seedlings were grown in pot trial in Mount Lyell soils sampled along a transect originating at the smelter installations. n=10.

Fig. 6d: Soil metals/ L.scoparium

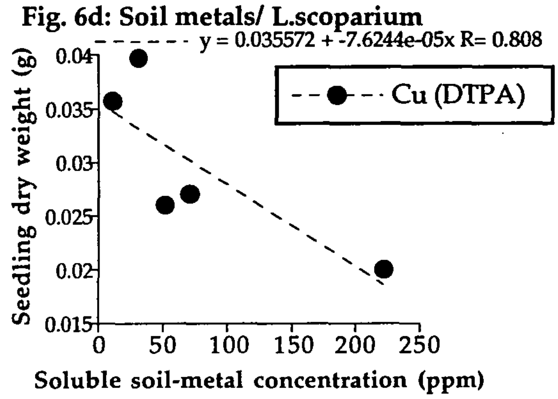


Fig. 6e: Soil metals/ L. scoparium

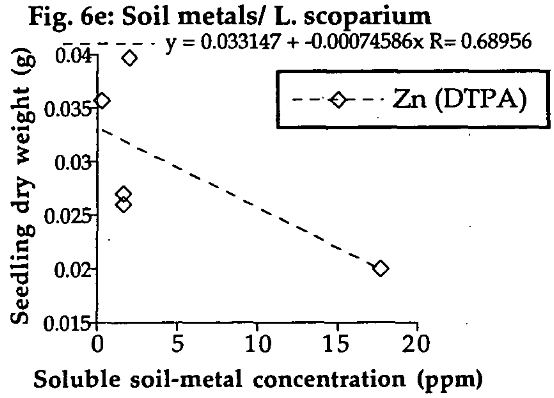
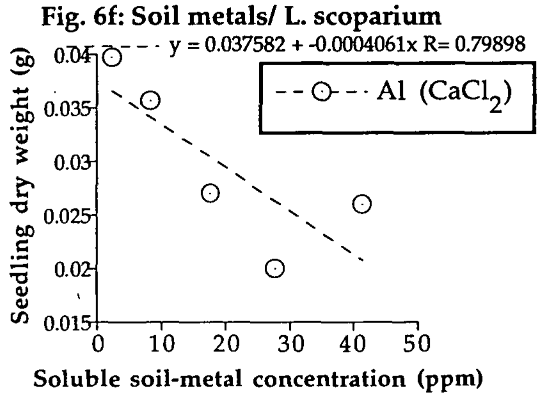


Fig. 6f: Soil metals/ L. scoparium



result of exposure to phytotoxic concentrations of both soil Al (Foy, 1992) and Cu (Symeonidis *et al.*, 1985; Heale and Ormrod, 1982; Wong and Bradshaw, 1982; Fessenden and Sutherland, 1979; Toivonen and Hofstra, 1979). Symptoms of severe phytotoxicity occurred at water-soluble Cu, Al and Zn concentrations at, or above, 2.0, 5 and 0.3 mg L^{-1}, respectively. At these concentrations, both Cu and Al were of a similar magnitude to those known to cause growth disorders in woody species in solution trials (Table 3/chapter 3). Seedlings grown in Mount Lyell soils taken from the more distant sampling sites displayed fewer phytotoxic symptoms.

The visual symptoms of intolerance were reflected by the seedling dry weight measurements. These indicated that seedling dry weight varied positively with the displacement of the sampling site from the smelters. As increasing displacement from the smelter generally represented a decline in soil-metal concentrations, it was likely that seedling tolerance varied as a function of soil-metal concentration. In the case of soil Cu, this hypothesis was supported by strong negative correlations between seedling dry weight and soluble Cu. A similar, relationship existed between seedling dry weight and soluble Zn. It was unlikely, however, that Zn was a cause of toxicity as the concentrations present were below those recognised as causing growth disorders in tree, grass and herb species.

Inverse relationships, between soil Cu and seedling dry weight, have been found for both woody and crop species in pot trials. Fessenden and Sutherland (1979) treated Black Spruce (*Picea mariana*) and Green Alder (*Alnus crispa*) with $CuSO_4$ at Cu concentrations ranging from 20 to 150 ppm. In unlimed soil, they found reductions in dry weight of seedling at all concentrations of applied Cu. The dry weight reductions were especially marked in the case of Green Alder. Applications exceeding 60 ppm, however, did not result in further growth reductions. Toivonen and Hofstra (1979) found that the dry weight of barley cultivars decreased significantly when treated with Cu solutions ranging from 10 to 100 ppm.

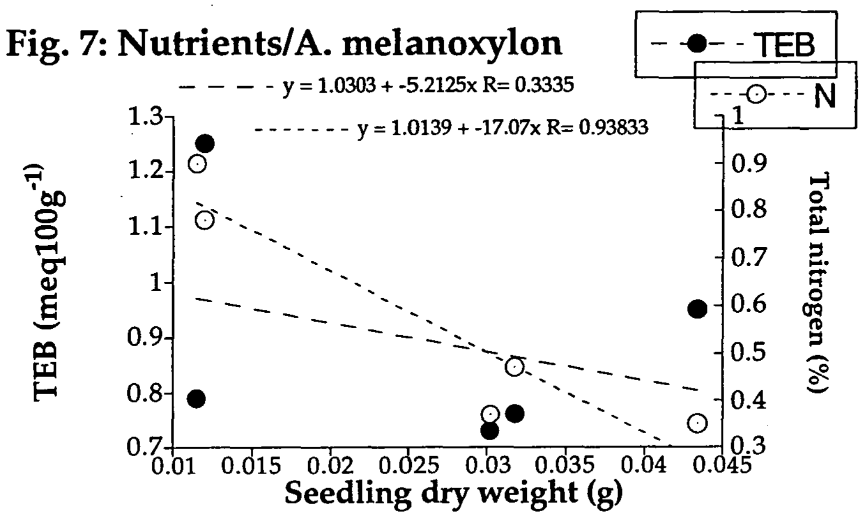
In contrast to the metals Cu and Zn, the regression analyses suggested that the relationship between soluble Al and seedling dry weight was inconsistent between species. This suggested that either 1) the extractant inadequately reflected the presence of toxic Al species, 2) that the species differed in their tolerance of Al or 3) that Al was not a primary cause of phytotoxicity. It appears likely that water extraction reflected the presence of toxic Al species as several authors have indicated that, in comparison to conventional methods, the soil solution provides a more accurate means of assessing Al toxicity (McCray and Sumner, 1990; Baker *et al*, 1988; Bruce, 1988; Pravan *et al.*,1982). It is more likely that the species differed to some degree in their tolerance to soluble Al. This explanation, however, could not be substantiated with the data available. Strong support for the third explanation comes from a survey of the vascular flora at Mount Lyell, and multivariate analysis, which failed to find a relationship between soil Al, both soluble and $CaCl_2$ extracted, and compositional trends (Chapter 4). It is worth noting that the relationships found between $CaCl_2$ extractable Al and seedling dry weight could not be adequately explained. However, as Al is known to form complexes with phosphorus, rendering P less available for plant growth, the regression with the stronger, $CaCl_2$ extractant may be reflecting P availability rather than Al toxicity.

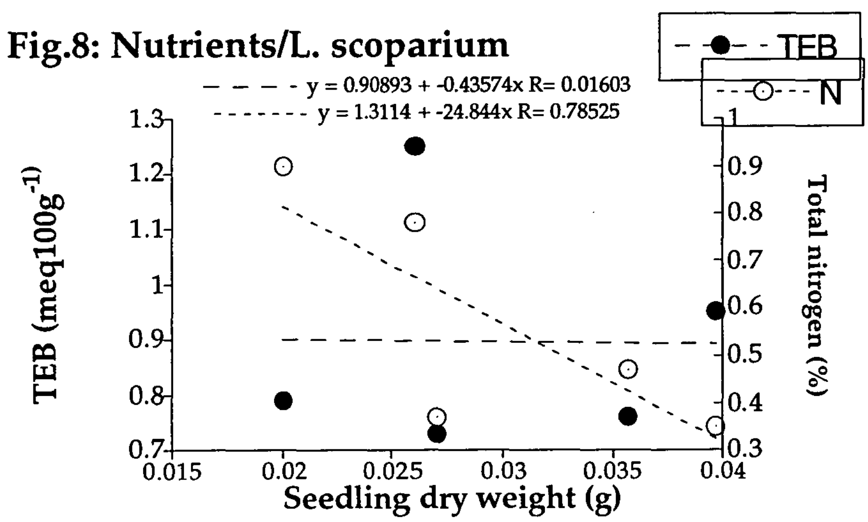
An alternate hypothesis to that of metal intolerance was that the seedlings were responding to a nutrient gradient. This was plausible as acid soils are known to restrict the availability of nutrient resources directly as a result of altered elemental solubilities (Thompson and Troeh, 1973). The hypothesis, however, appeared unlikely as the relationship between total nitrogen and seedling dry weight, while strong, was negative (Fig. 7 and 8). This suggested that, against a background of metal contamination, increasing concentrations of soil nitrogen might prove unfavourable to young seedlings, due to higher, and presumably more toxic, levels of metal assimilation. This explanation could not be examined in this work.

Phytotoxicity amelioration

The addition of lime to Mount Lyell soils improved the appearance and dry weight of seedlings of both species. The improvement was especially noticable in sample soils exhibiting high metal concentrations. This result was consistent with the concept of soil-metal immobilization by sorption or precipitation (Logan, 1992; Logan and Cassler, 1989). Root systems, in particular, benefited, developing greater length and branching. The benefits, however, were not universal. As distance of the soil sampling sites from the smelter installation increased there was a tendency for lime treatment to become less effective or ineffective. This tendency, substantiated by site by treatment interactions, indicated that the response to lime treatments of both species was inconsistent. There was, for example, no benefit from the higher lime applications at the most distant site (site 5). The most likely reason for this was falling Cu concentrations, which at ~0.3 mgL^{-1}, had dropped below those generally known to cause phytotoxicity. At these Cu concentrations, only the lowest rate of lime application benefited growth.

Figure 7 and 8: Regression of total exchangeable bases and total nitrogen with *A. melanoxylon* (Fig. 7) and *L. scoparium* (Fig. 8) mean seedling dry weight. Seedlings were grown in pot trial in untreated Mount Lyell soils sampled along a transect originating at the smelter. n = 10.





This may have been a result of improvements in pH-influenced, nutrient availability. The ineffectiveness of high-lime application, although not deleterious to growth at any of the applied levels, suggested that over-liming was possible and could result in induced deficiencies.

The response of the *A. melanoxylon* and *L. scoparium* seedlings to lime application was essentially similar to that of other woody species treated with various concentrations of $CuSO_4$ and lime in pot trial. Fessenden and Sutherland (1979) found that lime addition at low application rates generally improved the growth of Black Spruce. At high Cu concentrations, they concluded that the effect was mainly due to reduced Cu availability, and therefore reduced Cu toxicity, while at lower concentrations, the response was probably due to raised soil pH and concomitant changes in available nutrients. However, the same authors found that, at low applied Cu concentrations (20 ppm), lime application reduced the growth of Green Alder, while at higher concentrations little or no benefit was realised. They speculated that, at low Cu concentrations, lime application introduced a Cu deficiency, while at higher Cu concentrations (80 to 150 ppm), lime was an ineffective ameliorant.

Nodulation

Reference to the literature suggested that soil pH plays an important role in rhizobial survival and multiplication, and root infection and nodulation (Richardson *et al.*, 1988a,b; Franco and Munns, 1982; Carvalho *et al.*, 1981; Bryan, 1923a, b). In pasture and legume species, soils of low pH may affect rhizobial survival (Bryan, 1923a, b) and nodulation number (Franco and Munns, 1982). In early work, Bryan (1923a, b) reported that the critical soil pH for rhizobial activity varied with species. For example, alfalfa, red clover and soybean rhizobia were killed at pH 5.0, 4.5 and 3.5, respectively. More recent work suggests that the root infection process requires higher soil pH than rhizobial survival (Richardson *et al.*, 1988a,b; Carvalho *et al.*, 1981).

Nodulation did not occur on *A. melanoxylon* seedlings in the untreated Mount Lyell soils. These soils ranged in pH from 4.3 to 4.8. In these soils, it is likely that H ion concentrations reduced the activity of soil rhizobia. Nodulation did occur on seedlings in some of the lime treated soils at soil pH greater than 5. Exceptions occurred at sites 1 and 2 where lime treatment, while improving seedling appearance and dry weight, did not result in nodulation within the period of the trial. At these near-smelter sites, nodulation is likely to have been restricted by metal toxicity.

Part B: Field trials

5.6 Introduction

Field applications of lime in acid, metal contaminated lands have reportedly reduced toxicity (Clements *et al.*, 1968), initiated natural plant colonisation (Winterhalder, 1991; 1981a; 1981b), ameliorated nutrient deficiencies (Rodinkirchen, 1992) and aided colonisation by acid-intolerant vascular species (Rodenkirchen, 1992). Winterhalder (1991; 1981a; 1981b) considered the addition of lime a trigger factor for the initiation of natural plant colonisation by birch, aspen and willow. In disturbed lands affected by strip mining, the addition of lime has resulted in the improved survival of N_2 -fixing, woody species on highly acidic spoils (Carpenter and Hendsley, 1979).

The objective of the lime field trials was to determine whether the benefits of lime application realised under pot trial could be achieved in the field. In most soils, the most effective procedure for promoting root growth is to mix lime throughout the plough layer, although commonly this procedure is not economically feasible (Foy, 1992). Other methods include mixing at deeper soil layers, placement in a band on the plow sole and injection behind subsoiling chisels. As mechanical working of the soils at Mount Lyell was not possible given the steepness and inaccessibility of the terrain, lime applications were restricted to surface spreading or top-dressing.

The trial provided an indication of the response of seedlings to broadacre applications of ground calcitic lime in exposed, acid sub-soils displaying elevated concentrations of soil metals. Single and repeat lime applications were compared.

5.7 Methods

5.71 Experimental design

A. melanoxylon and *L. scoparium* seedlings were raised as tube stock from Mount Lyell provenance seed. The seed provenances were the same as those used in the ground lime pot trial. The seedlings were grown in a standard potting mix with slow release fertiliser. At five months of age, the strongest of the tube stock were selected for planting out. Planting was undertaken in November 1991.

The sites selected for the field trials were a subset of those sampled along the easterly transect in Chapter 3, and later used in the lime pot trial (Part A, this chapter). The sites, each placed on a ridge-line, were numbers 1, 3 and 5. At each site, sixty 0.25 m^2 experimental plots were pegged out in 10 blocks of six treatments. Each plot was separated from its nearest neighbour by 0.5 m. Treatments were allocated at random to plots within blocks and each plot labelled. The trial areas were then fenced to ensure protection from rabbits. One seedling of each species was planted per plot. Water was provided initially but there were no follow-up applications.

Three levels of lime treatment (0, 2000 and 4000 kg ha^{-1}) and two of application (single and repeat) were used. The form of lime used was ground calcitic lime. Lime treatments were made six weeks after planting, in January 1992. Repeat treatments, where prescribed, were applied in December 1992. The treatments were applied to the surfaces of each 0.25 m^2 plot as dry granules. In order to minimise the movement of granules, the soil surface surrounding each seedling was lightly scarified with the aid of a rake-hoe prior to the application of the treatment. A summary of the application rates, together with the relevant treatment codes, is provided (Table 6).

The growth rate of the seedlings was monitored by periodic stem diameter measurements taken at a height of 5.4 cm above the soil surface. Stem measurements were taken at plant-out and in conjunction with treatments. The final measurements were taken in December 1993.

As diameter increase is not linearly related to productivity, diameter measurements were initially converted to basal stem area. Basal-area increments were then calculated as the difference between successive stem area measurements and these were used as the basis for subsequent analyses. The data presented here represent the growth increment for the 24 month period from December 1991 to December 1993. Sites 2 and 3 were destroyed by vandals in early 1992. Consequently, no data were obtained for these

Table 6: Lime application rates and treatment codes

Lime treatment (kg ha^{-1})	0	2000	2000	4000	4000
No. of Applications	na	1	2	1	2
Treatment code	Control	$L_1(1)$	$L_1(2)$	$L_2(1)$	$L_2(2)$

sites. The following analyses refer to data derived from site 1. At a displacement of approximately 220 m, this site was the closest of the three to the Mount Lyell smelters. The surface soil characteristics for the field sites are presented in Table 1a of this chapter.

5.72 Data analysis

The raw data were initially converted from stem diameter measurements to stem area. Growth increments during the first year following the lime applications, the subsequent year, and the total were then calculated by subtraction. Calculation as relative growth rate was not considered necessary as the initial stem diameters of the seedling stock were similar (± 2 mm) and the trial was to run over an extended period. Data transformations were not applied as a plot of residuals against fitted values gave no indication of changing residual variability.

The experiment formed a two-factor, randomised complete block with 6 treatments and 10 blocks. The treatments were structured with 3 levels of lime and 2 levels of application (single and dual) to form a 3 x 2 factorial design. Data for the two species were analysed separately. There was no replication with respect to site.

The basal area data were analysed as a two-factor, randomised complete block and subjected to an ANOVA using the GLM procedure of SAS/STAT^{R} edition 6.03 (SAS Institute Inc., 1988). Least squares means were calculated and compared using the probability difference (pDiff) option of the program. The treatment means for each species were then ranked in order of increasing increment.

5.8 Results

Soil pH

Prior to lime treatment the mean soil pH of the field site was 4.3±0.1. Subsequent pH measurements indicated that the mafic-clay soils responded predictably to the addition of lime (Table 7) and increases in soil pH were recorded in response to each of the treatments. One year after application, the surface soil pH of the L_1 (1) and L_2 (1) treatments were 5.9±0.2 and 7.7±0.3, respectively. These treatments represented application rates of 2000 and 4000 $kgha^{-1}$. Without repeat treatment, these levels appeared to be maintained during the subsequent year. Repeat applications further raised soil pH.

Seedling mortality and appearance

Seedling mortality was very low with only one seedling death per species during the two-year field trial (Fig. 9). The deaths, which occurred within three months of transplantation, resulted in a mortality rate of 1.7%.

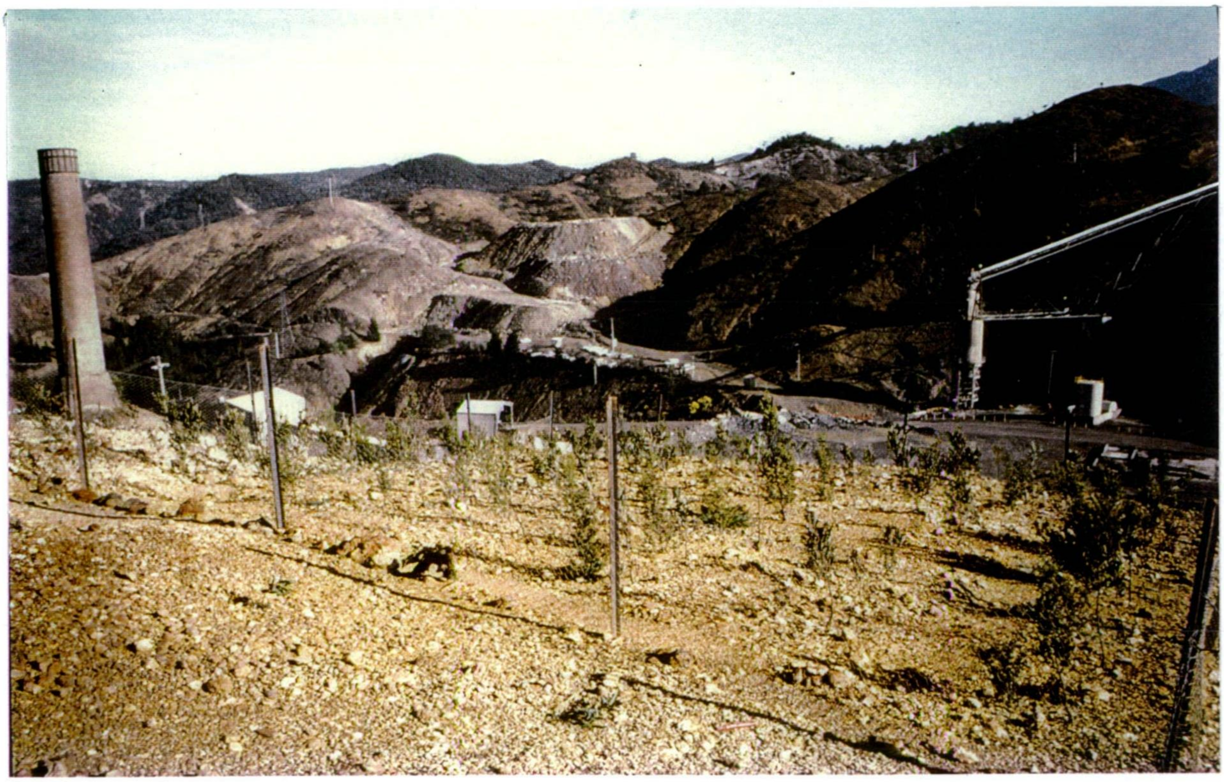
The majority of *A. melanoxylon* seedlings exhibited abnormal growth. Symptoms of stress were visually manifested by leaf discolouration, premature leaf loss, reduced leaf size, stunted stems and, in comparison to uncontaminated lands, markedly reduced growth rates. Leaf chlorosis and fall were particularly noticeable during the first 6 months after transplantation and the worst affected individuals lost most of their foliage. Recovery, invariably poorer in the control treatments, did not occur until the first spring. Stunted growth, witnessed by basally thickened and tapered stems, was noticable during the subsequent years growth. Purple hues

Table 7: Mean soil pH for lime treated site 1 soils in a field trial. Surface lime applications were made six weeks after planting, in January 1992. Repeat treatments were applied in December 1992. The pH measurements were recorded one year after treatment.

Soil pH was measured in a 1:1 soil/distilled water mix using a WTW electronic pH meter (model pH6) fitted with a Type E50 electrode. n=4.

Treatment	C	$L_1(1)$	$L_1(2)$	$L_2(1)$	$L_2(2)$
December '92	4.3	5.9	6.1	7.7	7.6
December '93	4.5	5.1	7.5	7.4	8.6

Figure 9: Photo of *A. melanoxylon* and *L. scoparium* seedlings in a lime application trial (site 1) in December 1992. At this stage, 13 months had elapsed since planting out.



appeared in stems and foliage of the treated seedlings.

Growth abnormalities in *L. scoparium* were not as pronounced as those in *A. melanoxylon*. Leaf discolouration and fall were displayed on some individuals by light-green leaf tones and sparse foliage. Growth rates appeared to be below that expected for the species. The appearance of most of the seedlings improved in the second year although some degree of stunting was evident. Purple hues again appeared in stems and foliage of the treated seedlings. A few seedlings flowered 6 months after transplantation and produced fruits thereafter.

At the conclusion of the trial, excavation of the root systems showed abnormal development. The *A. melanoxylon* seedlings had formed tight root balls in residual potting soils. In this zone, fine roots were numerous and nodulation occurred. Beyond the potting soils, longer, thicker but weakly branched roots formed. Fine roots were infrequent and nodulation absent. The *L. scoparium* seedlings produced similar, although less pronounced, root abnormality symptoms.

The response to lime treatments

All the raw data conformed to the assumptions associated with the ANOVA technique.

The *A. melanoxylon* stem increments for the two year period to December 1993 did not differ significantly between treatments ($F_{2,20} = 2.71$, $p<0.091$) or applications ($F_{1,20} = 2.13$, $p<0.1233$; Table 8). In the same period, the *L. scoparium* stem increments differed between treatments ($F_{2,20} = 6.01$, $p<0.0090$; Table 9), but not between applications ($F_{1,20} = 0.46$, $p<0.5062$).

Table 8 and 9: Analysis of variance of *A. melanoxylon* and *L. scoparium* seedling stem area increments from a lime-application field trial. Three levels of lime treatment (0, 2000 and 4000 kgha^{-1}) and two of application (single and dual) were used. The form of lime used was ground calcitic lime. Lime applications were made to the soil surface six weeks after planting, in December 1991, and again, where prescribed, one year later. Stem area increments for the two year period ending December 1993 were calculated. The data were analysed as a two-factor, randomised complete block of three treatments and two applications.

Table 8 *A. melanoxylon*

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*P*
Model	9	9895.83	1099.54	2.16	0.0728
Error	20	10185.68	509.28		
Corr. total	59	20081.51			

Source	*DF*	*Type IIISS*	*MS*	*F Value*	*P>F*
Treatment	2	2758.04	1379.02	2.71	0.0911
Blocks	4	4336.17	1084.04	2.13	0.1149
Application	1	1318.33	1318.33	2.59	0.1233
T'ment*App.	2	1483.28	741.64	1.46	0.2568

Table 9 *L. scoparium*

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*P*
Model	9	8978.04	997.56	2.31	0.0572
Error	20	8639.76	431.99		
Corr. total	59	17617.80			

Source	*DF*	*Type IIISS*	*MS*	*F Value*	*P>F*
Treatment	2	5194.68	2597.34	6.01	0.0090
Blocks	4	2002.61	500.65	1.16	0.3583
Application	1	197.96	197.96	0.46	0.5062
T'ment*App.	2	1582.78	791.39	1.83	0.1860

The mean stem increments for the *A. melanoxylon* seedlings were plotted (Fig. 10). Although the response to lime treatment was not significant, inspection of the response trends suggested that the ground lime treatments did not adversely affect growth (Fig. 10). As the ANOVA suggested that the response to treatments approached significance, it is foreseeable that a positive lime treatment response might occur if the trial were held over a longer duration.

Pairwise comparisons of differences between *L. scoparium* treatment means indicated that three treatments increased the growth increment. In comparison to the control, each of the treatments $L_2(2)$, $L_1(2)$ and $L_2(1)$ increased the mean growth increment by more than three times (Fig. 11). Of these treatments, the treatment $L_2(1)$ resulted in the highest mean stem area for the site (47.8±22.6 m m^2) during the 1991/93 period.

5.9 Discussion

Growth and mortality

Despite symptoms of metal-related, phytotoxicity throughout the trial, the mortality of the transplanted tube stock, in untreated soils at site 1, was low. This result was in contrast to the pot trial, which demonstrated that site 1 soils were extremely toxic to bare-rooted, seedling transplants and likely to result in early death. Consequently, low seedling mortality in the field trial indicated that soil in the pots influenced survival. Excavation showed abnormal root systems. These were typified by two types of root formation: fine, 'pot-bound' roots forming a tight root-ball in residual potting soils and longer, thicker, hairless but less numerous roots in

Figure 10: Mean stem area increments for *A. melanoxylon* seedlings grown in a field trial at a near-smelter site (Site 1). The data represent the annual area increments for the 2-year period following seedling transplantation. Lime treatments were applied to the soil surface at the rates of 0 (C), 2000 $kgha^{-1}(L_1)$ and 4000 $kgha^{-1}(L_2)$. Repeat applications, one year after the initial treatments, were tested and identified by the designation 2. The means were ranked in ascending order of magnitude.

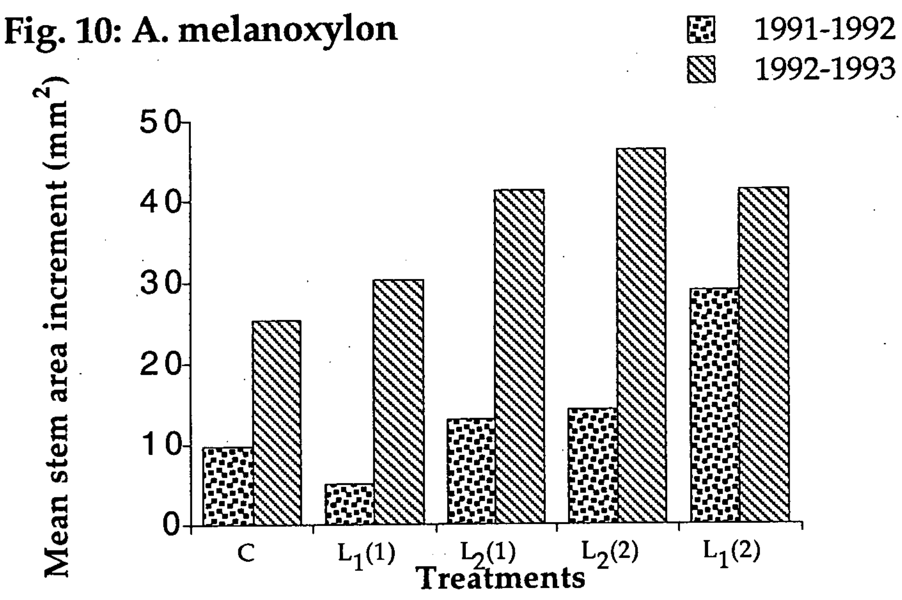
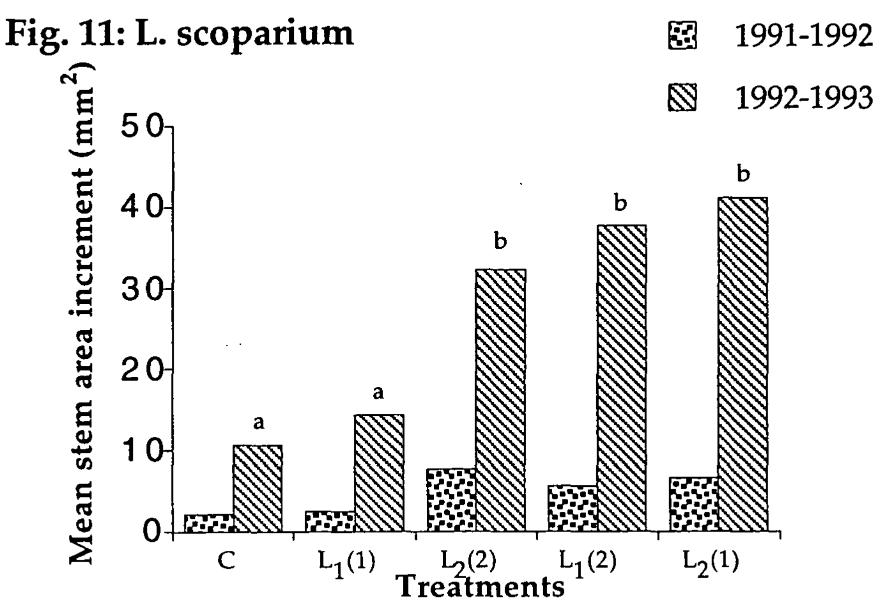


Figure 11: Mean stem area increments for *L. scoparium* seedlings grown in a field trial at near-smelter site (Site 1). The data represent the annual area increments for the 2-year period following seedling transplantation. Lime treatments were applied to the soil surface at the rates of 0 (C), 2000 kgha^{-1}(L_1) and 4000 kgha^{-1}(L_2). Repeat applications, one year after the initial treatments, were tested and identified by the designation 2. The means were ranked in ascending order of magnitude and tested by pairwise comparison using the pDiff option in SAS/STAT^{R} edition 6.03 (SAS Institute Inc., 1988). Differences between treatments for the 2-year period, at significance levels of 0.05 or greater, are indicated by dissimilar annotations.



'native' soils. It appeared that the potting soils enabled the seedlings to persist, while root exploration of less favourable soils proceeded.

Two years after planting out, the mean stem areas of the A. *melanoxylon* and *L.scoparium* seedlings, in untreated soils, were 35.0 and 12.8 mm^2, respectively. In the case of *A. melanoxylon*, 72% of the growth had occurred in the second year. The second year growth of *L. scoparium* represented 83%. This demonstrated that growth was possible in these soils despite evidence of significant metal contamination.

Lime treatment

The amelioration of metal contaminated soils by the surface application of lime depends on the movement of lime to the rooting zone. The low solubility of lime suggests that surface applications may fail to reach the root zone (Logan, 1992). In agricultural systems, Sopher and Baird (1982) reported that lime applied to the surface of clay soils may move downward less than 1 cm in 18 months. Consequently, poor solubility may limit the ability of lime to ameliorate metal contamination. A commonly adopted remedy, the deep incorporation of lime, is usually considered preferable to surface spreading, but is only possible under restricted circumstances. Despite this, surface applications of lime have been judged beneficial to growth in some acid soils. For example, Rechcigl *et al.* (1988) concluded that surface lime application was adequate for the establishment of and productivity of alfalfa on an acid clay loam. In a greenhouse column leaching experiment, Mathews and Joost (1990) found that surface lime application effectively corrected soil Mn and pH problems limiting alfalfa growth, but had no effect on subsoil chemical properties.

At Mount Lyell, topography restricts lime treatment to surface applications. Despite this restriction, surface lime application at the near-smelterfield trial moderated above-ground toxicity symptoms, including slow growth, premature leaf loss, foliar discolouration and chlorosis, stunting, and premature flowering. Surface applied lime was generally beneficial to seedling productivity with both species responding similarly. The largest growth increments over the two year period were achieved by application of lime at rate of 4000 kg ha^{-1}. This application rate was comparable to the lime requirements (3600-3700 kg ha^{-1}) typically considered necessary in order to raise the pH of clay loams from 4 to 6 (Sopher and Baird, 1982). In comparison to the untreated control, the lime treatment $L_1(2)$ increased the mean stem area of *A. melanoxylon* by a factor of 2. Similarly, the lime treatment $L_2(1)$ increased the mean stem area of *L. scoparium* by a factor of 3.7. Higher application rates, however, did not result in further stem area increases for either species. Application rates of 2000 $kgha^{-1}$ resulted in inferior growth rates. There were no measurable benefits when lime treatments, of equivalent total application rate, were applied over two, rather than one, season.

Chemically the net effect of lime addition is understood to be a reduction in the relative number of hydrogen ions in the soil solution resulting in a rise in soil pH. Rising soil pH reduces the solubility, and therefore the availability of many metal-ions (Foy, 1988; Foy, 1984: Foy, 1974). The growth response of seedlings to lime treatment at Mount Lyell was largely attributed to metal-ion immobilization rather than, for example, pH correction or a rectification of Ca deficiency. Some benefit may have been derived from improved organic matter mineralisation and phosphorus availability. However, purple stem and foliage discolouration indicated that liming, in the absence of other macronutrients, induced P deficiency on these soils.

Products other than lime may be suitable ameliorants for acidic, metal contaminated soils such as found at Mount Lyell. Gypsum ($CaSO_4.2H_2O$) is more soluble than lime. Some studies have shown that gypsum ($CaSO_4.2H_2O$) can neutralise acid subsoils (Oates and Caldwell, 1985; Pavan *et al.*, 1984; Reeve and Sumner, 1972). Phospho-gypsum, a by-product of fertilizer production, may also be suitable. Watson and Hoitink (1985) showed that the high free $CaCO_3$ content of of papermill sludge enhanced the ability of sludge to reclaim acidic mine spoil by maintaining pH 7.6 three years after the application of 150 to 300 t ha^{-1} to spoil with an initial pH of 3.4. The fibrous nature of the product also gave it desirable slope stabilizing properties. A recent product, cement kiln dust-stabilized sludge, may also be suitable for the reclamation of highly acidic soils (Logan, 1992).

Neutralising soil amendments such as lime commonly provide an effective short-term means of controlling soil pH and the bioavailability of heavy metals but long-term control must consider the acid-buffering capacity of the soil (Logan, 1990). However, raising the buffering capacity of contaminated soils requires the addition of various organic amendments such as sewage sludge. This approach, and most others involving organic amendments, would be impractical given the area of denudation and topography of Mount Lyell. In comparison, chemical neutralisation offers short-term control of metal-contaminated soils and improvements in the growth rates of colonising species. With time, species proliferation and subsequent *in situ* decomposition are likely to offer increased soil buffering capacity and long-term control of metallic contaminants.

5.10 Conclusions

Seedling growth, in pot and field trials, was used as a measure of plant tolerance to soil metals along a concentration gradient at the Mount Lyell smelter site. In pot trials, tolerance, as measured by seedling dry weight, varied with the displacement of the soil sampling site from the smelter. Acute toxicity symptoms, notably root system abnormalities, were evident at sampling sites nearest the smelters. Survival in these soils was improbable. In field trial, the survival rate of transplanted tube stock at a near-smelter site was high. However, growth remained depressed, and two years after planting out, resulted in stunted individuals with abnormal root systems.

Characteristic toxicity symptoms, elevated water-soluble metal concentrations and strong metal to dry weight correlations linked seedling intolerance to Cu phytotoxicity. In contrast, Al phytotoxicity could not be demonstrated despite sometimes high analytical concentrations. The complexity of the Al chemistry suggested that the tests used here might not be able to adequately discriminate between the toxicities of Al species. The problems of interpretation were compounded by limited data. This was a product of field-site vandalism.

Lime application was used in an attempt to ameliorate metal-related, soil phytotoxicity. In general, seedlings benefited from lime application by exhibiting fewer toxicity symptoms while producing increased biomass. The response was likely to be largely related to increased soil pH and reduced Cu availability. In pot trials, lime application greatly improved the dry weight of seedlings in near-smelter soils exhibiting high Cu concentrations. At contaminated field sites, the application of lime was generally beneficial to growth. This finding was in accordance with those at other metal-contaminated smelter sites where lime application has benefited native plant colonisation (Winterhalder, 1981a/b, 1983a, 1988) At Mount Lyell, however, the effectiveness of the treatments may have been hampered by the limitations of surface application and a low-solubility neutralising agent. Nevertheless, the responses provide a guide to the lime requirements of the colonising species along a toxicity gradient.

Neutralising soil amendments such as lime commonly provide an effective short-term means of controlling soil pH and the bioavailability of heavy metals but long-term control must consider the acid-buffering capacity of a soil. This work indicates that, at Mount Lyell, neutralising amendments will assist the growth and proliferation of colonising species. Long-term control of metallic contaminants, however, is only likely to be provided by improved buffering capacity as a result of the *in situ* decomposition of organic materials derived from colonising species.

Chapter 6

Seedbed preparation and the use of a mechanical roller-aerator to enhance the lodgement, germination and early survival of broadcast sown seed at an eroded, near-smelter location with phytotoxic soil characteristics

6.1 Introduction

For any one species, a large disparity usually exists between the number of viable seeds produced and dispersed into an area and the actual seedling numbers established (Sheldon, 1974). Harper *et al.*(1961; 1965) recognised that any seed or seedling losses result in a widening of the gap between the seedling establishment potential of a species and the actual numbers of seedlings occurring. This difference, between the seedling establishment potential of a species and the seedling population, can be termed the 'seedling gap'. If a receiving surface is sufficiently unfavourable for establishment, the seedling gap may approach, or equal, the seedling establishment potential of a species.

In comparison to a forest floor, which is highly heterogeneous in its physical and chemical make-up, the exposed sub-soils at Mount Lyell offer little diversity, and therefore few establishment opportunities. There are, for example, limited opportunities for seed to lodge and anchor. Establishment opportunities are further restricted by the number of sites capable of supporting growth. Harsh environmental conditions, such as exposure to extreme moisture stress, are likely to result in desiccation and death. Consequently, significant seed and seedling losses could be expected from both a failure to lodge and from exposure to unfavourable growth conditions. Similar factors have limited the success of broadcast sown seed on mine overburden (Koch, 1980).

The aim of this chapter was to quantify seedling establishment following broadcast sowing of a number of colonising species and to maximise establishment through the provision of appropriate seedbed treatments. This approach permitted an assessment of the feasibility of broadcast sowing as a method of rehabilitation within the Mount Lyell environment.

To maximise seedling establishment, two methods of seedbed preparation were compared to control sowings on a machine-accessible, ridge-top site adjacent the original smelters. Seedbeds were prepared with the aid of a multi-tyned offset roller, or roller-aerator. For comparison, a similar seedbed was prepared by conventional deep-rip treatment. The intention of the roller-aerator treatment was the creation of a seedbed of physical complexity sufficient to increase seed lodgement, and germination and growth opportunities, without producing undue surface destabilisation. Soil phytotoxicity was combated by the addition of calcitic lime.

Seedling establishment was calculated from seedling population counts. Establishment percentages permitted the calculation of species sowing rates and cost evaluation. This information was essential for species and seedbed treatment recommendations, particularly where seed was in limited supply.

6.2 Methods

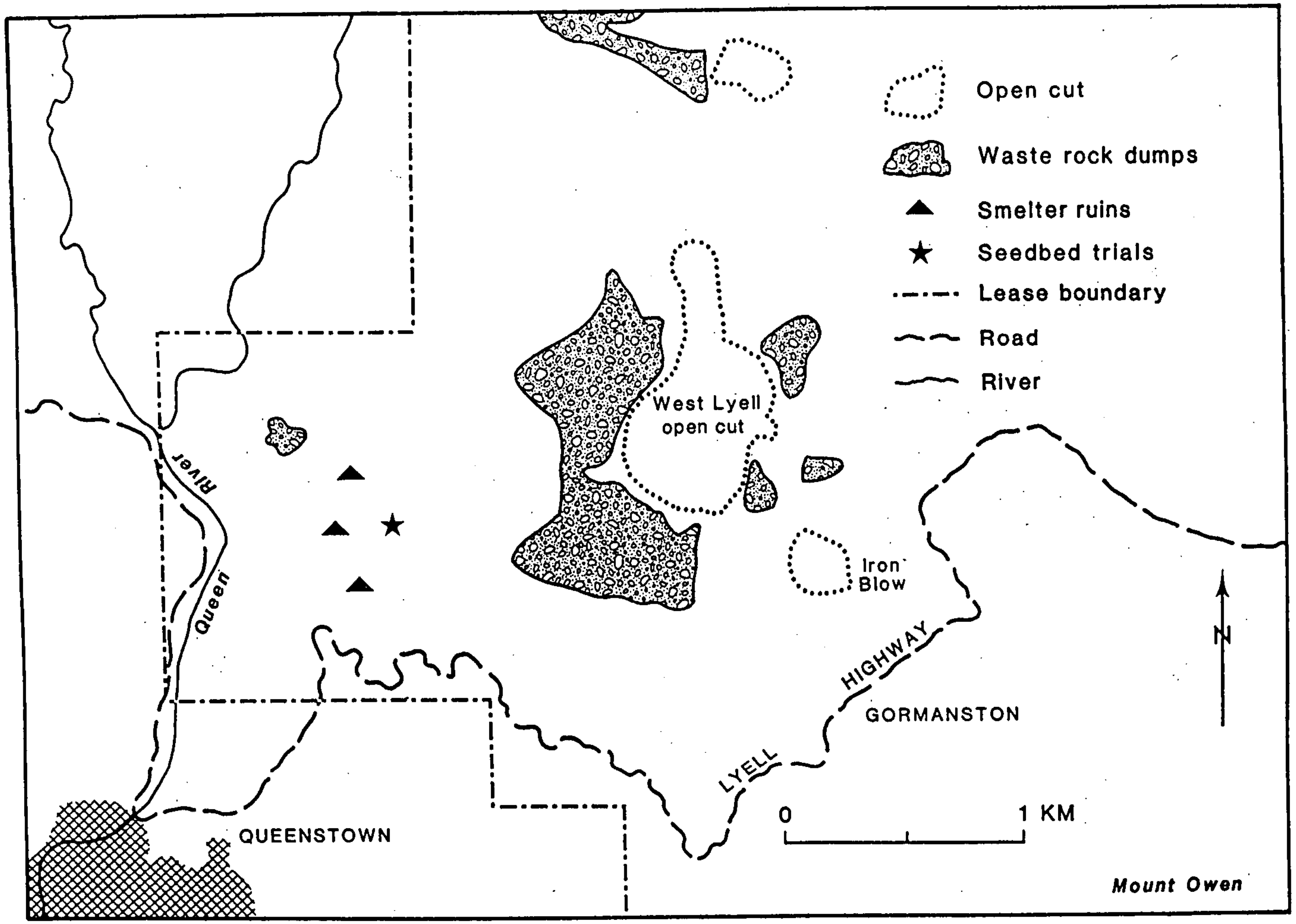
The experimental site was located approximately 150 m east of the original smelters within the *Agrostis* grassland zone (Fig. 1). The site was a ridge-top of westerly aspect, with an average slope of 6°. With the exception of isolated individuals or small communities of *Agrostis capillaris, Restio tetraphyllus* and a single *Acacia melanoxylon,* the area was entirely free of existing vegetation.

The site was located on exposed, mafic-clay subsoils. Prior to treatment, ten equally spaced, surface soil samples (0-5 cm) were removed in an 'S' formation. A composite sample was formed for nutrient and metal-ion analysis. Sample preparation followed that of section 3.22 (Chapter 3).

Two mechanical, pre-sowing seedbed treatments were evaluated against broadcast sowings on unprepared ground. The mechanical seedbed treatments were a double rip-line (tyne length 40cm) and a soil 'aeration' treatment. The latter treatment was created using a 'Superworm Soil Aerator' (Appendix 1). The aerator, two multi-spiked, off-set rollers (tyne length 20cm), was attached to a four wheel drive tractor via a 3-point linkage. The surface of the control treatment was undisturbed.

The mechanical implementation of the treatments demanded a split-plot experimental field plan. Fifteen mainplots, in five blocks of three, were marked out and randomly allocated to one of the two mechanical seedbed treatments or a control. Each mainplot measured approximately 4x35 m with long axis across contour. The rip and the aeration treatments were applied as single passes parallel to the long axes of the plots.

Fig.1 Location of mechanically-prepared seedbed field trial



Following the mechanical treatment of the seedbeds eight 1x2.5m subplots within each mainplot were marked out with corner pegs, with each subplot separated from its nearest neighbour by a 1m buffer. The subplots were randomly allocated one of the test species and labelled with a coded tagging system. Ground calcitic lime was applied to the surface of each subplot at the rate of 4000 kg ha^{-1}. Rabbit-proof fencing surrounded the entire field trial.

In order to minimise inter-plot seed contamination, each subplot was surrounded by an in-ground drainage system, with plastic lined sumps forming seed traps located at appropriate intervals. The sumps were excavated five months after sowing. This procedure allowed the number of seeds lost from each subplot to be counted.

Local provenance seed was used in the trial. The species sown were *Acacia melanoxylon, Acacia mucronata, Acacia verticillata, Acacia dealbata, Leptospermum scoparium, Oxylobium arborescens, Gaultheria hispida and Agrostis capillaris.* All of these species occur on the west coast of Tasmania and, with the exception *A. verticillata,* are present as colonisers within the partially recolonised area of Mount Lyell. The occurrence of *A.dealbata,* unlike the other species, appeared to be related to roading. Its presence at Mount Lyell is likely to be the result of soil spillage or storage during transportation from the nearby river valleys over the life of the mine rather than as a result of unaided, natural recolonisation.

Seed was broadcast during the last week of June 1992. Details of the sowing rates and seed viabilities are provided in Table 1. The germinants were counted during the weeks commencing 5/11/92 (Spring), 11/1/93

(Summer), 14/6/93 (Winter) and 28/10/93 (Spring). Upon identification each germinant was marked with a wire peg. When new germinations occurred further pegs were added. Where deaths occurred, pegs were removed. Recruitment and losses for each subplot were tallied at each survey.

Seedling establishment was calculated from seedling population counts (standardised field germination number minus mortality) as the percentage of the number of viable seeds sown (the seedling potential). Standardisation count data accounted for unequal residual seed numbers due to erosional losses from the subplots. The distribution of each species' seedling potential was characterised by the proportion of the viable seeds sown attributed to seedling establishment, seed loss due to erosion, seedling mortality or unaccounted loss factors.

6.3 Data analysis

Preparation of the germination and mortality count data

The total number of seeds sown to each subplot (Table 2) were estimated from sowings of known mass (Table 1). The number of seeds remaining in each subplot five months after sowing (5/11/92) were then calculated by subtracting the number of seeds captured in the drainage sumps (i.e. (Table 2).

Table 1: Seed lot collection numbers, seed counts, viability and sowing rate data

Species	Collection No.	Estimated[1] no. of seeds kg^{-1} in bulk seedlot	Seed viability (%)	No. viable seeds kg^{-1}	Fieldtrial broadcast sowing rate; $kgha^{-1}$ and rate per subplot (brackets)
A.mucronata	001/02/92	66 666	89.5	59 666	300.0 (0.075)
A.melanoxylon	003/02/91	70 149	81.0	56 821	268.0 (0.067)
A.verticillata	003/02/92	48 640	79.2	38 523	50.0 (0.012)
A.dealbata	001/01/91	53 030	96.5	51 174	197.4 (0.049)
L.scoparium	001/11/90	~10^6	90.0	900 000	10.4 (0.003)

[1] - Estimate based on mean seed counts (n=3)

Table 2: Total number of viable seeds sown, estimated erosional losses and an estimate of the number of residual seeds remaining five months after sowing in trial plots given three pre-sowing soil treatments. n = 5

Species	Collection no.	Treatment	Total no. of seeds sown per subplot	Mean no. of seeds in drainage sumps[2] to 5/11/92; standard deviation (brackets)		Residual[1] no. of seeds per subplot; standard deviation (brackets)	
A.mucronata	001/02/92	rip	5000	15.2	(12.4)	4977.6	(27.0)
A.mucronata	001/02/92	aerate	5000	22.4	(27.0)	4971.2	(30.0)
A.mucronata	001/02/92	control	5000	28.8	(30.2)	4993.4	(6.8)
A.melanoxylon	003/02/91	rip	4700	10.2	(9.8)	4689.8	(9.8)
A.melanoxylon	003/02/91	aerate	4700	25.8	(35.3)	4674.2	(35.3)
A.melanoxylon	003/02/91	control	4700	8.4	(18.4)	4691.6	(18.7)
A.verticillata	003/02/92	rip	608	3.6	(4.3)	604.4	(4.3)
A.verticillata	003/02/92	aerate	608	0.2	(0.5)	607.8	(0.4)
A.verticillata	003/02/92	control	608	12.4	(11.5)	595.6	(11.5)
A.dealbata	001/01/91	rip	2618	6.8	(15.2)	2611.2	(15.2)
A.dealbata	001/01/91	aerate	2618	19.2	(38.6)	2598.8	(38.6)
A.dealbata	001/01/91	control	2618	47.0	(42.0)	2571.0	(42.4)
L.scoparium	001/11/90	rip	2600			ne	
L.scoparium	001/11/90	aerate	2600			ne	
L.scoparium	001/11/90	control	2600			ne	

1 - estimates derived from sieving of runoff sediments in drainage sumps
ne - not estimated.

lost to surface wash) from the number of seeds sown. This was termed the residual number of seeds per subplot. The mean residual number of seeds were calculated for each species/ seedbed treatment combination The germination count data for each subplot were standardized with the number of seeds sown per subplot in proportion to the residual number of seeds per subplot. The intention was to account for unequal residual seed numbers between subplots. In practice, this meant that, in subplots where erosional loss of seed occurred, the number of germinants in a subplot were adjusted upwards. Where deaths occurred, these were standardised and subtracted from the corresponding germination counts. The result of these calculations was the seedling number per subplot (standardized) at four periods of sampling.

With each successive sampling, recruitment and mortality data became cumulative. These data were termed the seedling number per subplot (cumulative). In order to allow inter-specific comparison, the seedling numbers per subplot (cumulative) were divided by number of viable seeds sown per subplot (the seedling potential). These data were termed the seedling establishment percentage per subplot and were used as the basis for statistical analyses.

Analysis of seedling establishment

The experiment formed a split-plot randomised complete block with treatments allocated to mainplots within blocks and species allocated to subplots within mainplots.

The seedling number per subplot (cumulative) and the seedling establishment percentage per subplot data were subjected to an ANOVA using the GLM procedure of SAS/STAT^R edition 6.03 (SAS Institute Inc., 1988) and analysed as a two level, randomised complete block with treatments allocated to mainplots and species to subplots. The between treatments effect was tested against a block times treatment residual. These data conformed to the model requirements of normality and homogeneity and arc-sine transformation was considered unnecessary.

Least squares means were calculated and compared using the probability difference (pDiff) option of the SAS/STAT^R program. Following the ranking of treatment and species means, a multiple-range comparison of the pairwise probability differences between means was created and summarised.

Analysis of erosional seed losses and seedling mortality

Erosional loss counts of *Acacia* seed captured in drainage sumps were calculated as a percentage of the total number of viable seeds of each species sown per subplot. The data were subjected to an ANOVA using the GLM procedure of SAS/STAT^R edition 6.03 (SAS Institute Inc., 1988) and analysed as a two level, randomised complete block. The between treatments effect was tested against a block times treatment residual. Similarly, the standardised seedling mortality counts were calculated as a percentage of the total number of viable seeds of each species sown per subplot and the data subjected to an ANOVA. Both data sets conformed to the model requirements of normality and homogeneity.

Least squares means were calculated and compared using the probability difference (pDiff) option of the SAS/STAT^R program. Following the ranking of treatment and species means for both data sets, multiple-range comparisons of the pairwise probability differences between means were created and summarised.

The relative cost of species establishment by broadcasting

The seedling establishment percentages permitted the calculation of sowing rates and species composition in year-old seedlings. The relative cost of species' establishment were then calculated. No attempt was made to cost, in absolute terms, the implementation of each treatment as the costs of applying each pre-sowing treatment vary with circumstance.

The relative costs of the establishment of a single seedling of each species were calculated, using the seedling establishment percentages for each treatment, at an average cost of $100 per kg bulk seed.

The distribution of seedling potential

The number of viable seeds sown per subplot represented the seedling potential of each species. The distribution of each species' seedling potential was calculated from the mean number of seedlings to establish, the mean seed loss due to surface erosion and the mean number of seedling deaths as a percentage of the number of viable seeds sown. The proportion of the seedling potential unaccounted for by known factors was calculated by subtraction.

6.4 Results

The physical appearance of the treated plots

The multi-tyned, offset rollers of the Superworm Soil Aerator punctured the soil surface with approximately 13 discrete, rip-cavities per meter squared. The implement produced minimal surface disturbance and profile upheaval (Fig. 2). Approximately 10% of the soil surface within the plots was disturbed by the single-pass treatment.

With time, a percentage of the broadcast seed were washed into the cavities and buried by back-fill from surface wash. Accordingly, most of the germinants in this treatment occurred in clusters at the point of tyne-soil contact. Surface wash into cavities may have also assisted in the incorporation of lime.

After initial soil resettling, the roller-aerator treated surfaces did not appear to further contribute to erosion. No rill or gully erosion was initiated by the treatment. Despite the rocky terrain, the Superworm Soil Aerator was reliable during the application of this treatment. There were no mechanical breakages; a function of the 'roll-over, ride-up' feature of the design.

In contrast to the roller-aerator treatment, the rip-line treatment resulted in considerable profile upheaval and brought many large diameter pieces of parent material to the surface. This material offered few opportunities for seed to lodge and germinate. Conversely, depressions formed along

Fig.2: Photo of the soil cavities created by the multi-tyned, offset rollers of the Superworm Soil Aerator.

the rip-lines, filled with eroded material and tended to trap seed. Soil moisture in these depression tended to be retained long after the surrounding area had dried. Most germinants occurred in this zone. Germination also occurred on both sides of the rip-lines in areas traversed by the dozer tracks.

The rip-lines and dozer-track scarification disturbed more than 90% of the soil surface of each plot. In the months following treatment, many of the lines became sites of gully erosion, despite the relatively shallow gradient of the experimental area.

The surface of the control plots was an exposed subsoil clay. The surface was heavily littered with pebble and rock fragments of volcanic origin. The fragments themselves appeared to have accumulated on the surface due to the erosion of the finer soil particles over time. As a result, many relatively recently exposed fragments were raised on short, clay pedestals.

No changes were apparent in the overall appearance of the unworked surface of the control plots during the period of monitoring. Germinants occurred with no obvious spatial pattern other than a tendency to lodge and establish adjacent to rock fragments.

Soil chemical properties of the experimental site

The mean soil pH at the experimental site was 4.40±0.29. The water soluble concentrations of Cu, Al and Zn were 7.11, 29.88 and 0.95 mg L^{-1}, respectively. In the cases of Cu and Al, these concentrations were above the means recorded for the *Agrostis* grassland sites (Table 9, Chapter 4). The Zn concentration was approximately average for the zone. The addition of ground lime raised the mean soil pH to 5.33±0.45.

The seedling populations

Seeds of *A. capillaris, G. hispida* and *O. arborescens* failed to germinate in any of the subplots, irrespective of the seedbed treatment. Sowings of a fourth species, *L. scoparium,* resulted in very few germinants, giving mean seedling establishment percentages of less than 0.5% in both of the mechanical treatments.

A substantial number of seeds of all the remaining species germinated (Fig. 3). The mean germination number and the mean number of deaths for the 16 month period following broadcast sowing are presented by treatment in Table 3.

The germination of *Acacia* began in mid-October 1993 with each of the four species displaying numerically similar population trends during the first year following sowing. Seedling numbers in both of the mechanical seedbed treatments rose rapidly during late spring and early summer and peaked during the first summer following sowing. Thereafter seedling numbers exhibited a gradual decline as losses exceeded recruitment. The mean seedling number (standardised) at four sampling periods during the 16 months following broadcast sowing are presented in Figures 4a-d. The mean seedling numbers recorded for *L. scoparium* contrasted markedly with those of the *Acacia* species (Fig. 4e).

Figure 3: Photo of *Acacia mucronata* seedlings growing in cavities created by a pre-sowing seedbed treatment. The treatment was provided by multi-tyned, offset rollers. The photo records seedlings, indicated by wire markers, 21 months after sowing. The tape delimits the 1 x 2.5 m subplot.

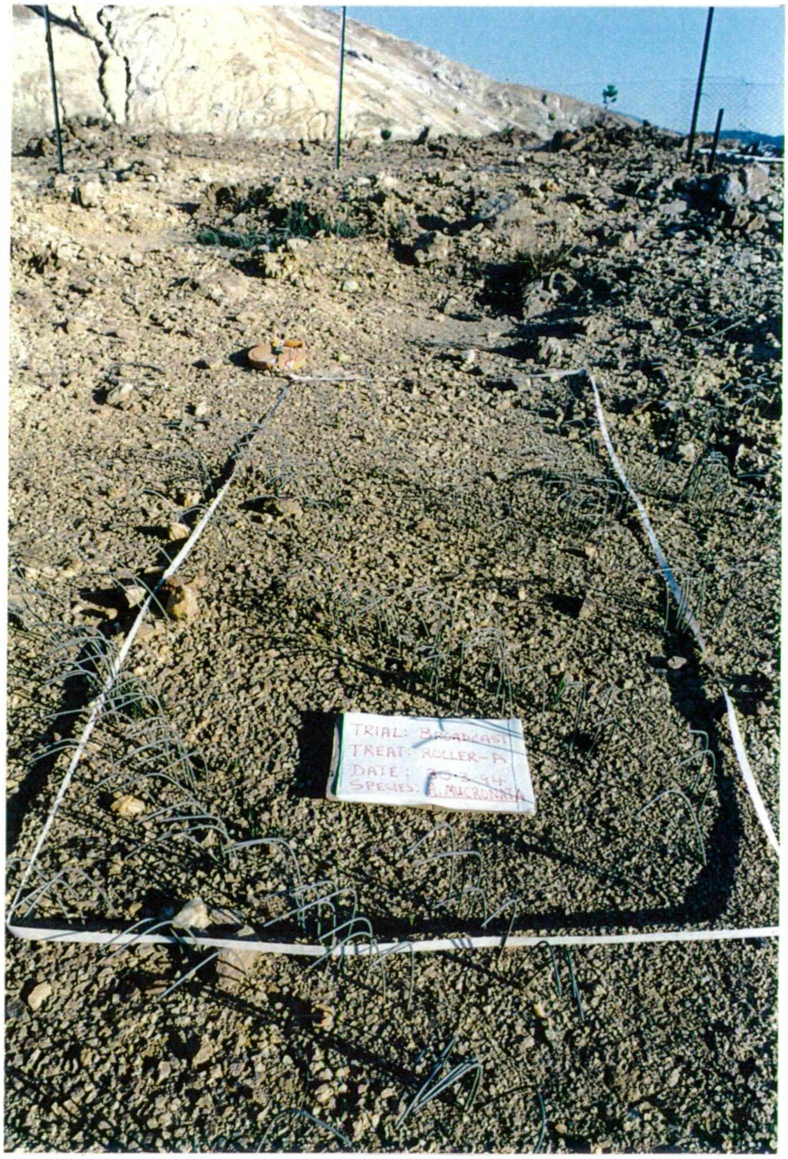


Table 3: Mean number of germinants and seedling deaths for four species of *Acacia* and *L.scoparium* provided with one of two pre-sowing, mechanical seedbed preparation treatments or an unprepared control. The means represent total seedling counts for the 16 month period following broadcast sowing.

Species	Collection no.	Treatment	Mean germination[1] number (standardised) to 28/10/93 and standard deviation (in brackets)		Mean no. of deaths[2] (standardised) to 28/10/93 and standard deviation (in brackets)	
A.mucronata	001/02/92	rip	316.8	(190.0)	17.4	(8.5)
A.mucronata	001/02/92	aerate	336.7	(104.5)	13.6	(2.1)
A.mucronata	001/02/92	control	78.8	(44.3)	4.4	(3.4)
A.melanoxylon	003/02/91	rip	267.9	(76.1)	33.2	(36.3)
A.melanoxylon	003/02/91	aerate	203.5	(84.0)	23.4	(10.4)
A.melanoxylon	003/02/91	control	102.4	(73.6)	11.0	(5.7)
A.verticillata	003/02/92	rip	84.5	(25.8)	8.8	(5.2)
A.verticillata	003/02/92	aerate	78.4	(23.7)	7.8	(3.9)
A.verticillata	003/02/92	control	20.2	(12.9)	3.0	(2.0)
A.dealbata	001/01/91	rip	107.2	(168.5)	4.4	(6.1)
A.dealbata	001/01/91	aerate	84.5	(147.9)	4.2	(5.8)
A.dealbata	001/01/91	control	35.4	(72.1)	2.6	(4.3)
L.scoparium	001/11/90	rip	20.4	(16.5)	2.4	(3.7)
L.scoparium	001/11/90	aerate	11.5	(10.4)	1.6	(1.5)
L.scoparium	001/11/90	control	3.0	(2.1)	1.4	(1.5)

1 - germination number; the mean of five replicate counts

2 - mortality; the mean number of deaths in five replicate plots

Fig 4a - 4e: The mean seedling number (standardised) for *Acacia* species and *L. scoparium* for four periods of sampling during the 16 months following broadcast sowing. One of two mechanical seedbed treatments or an unprepared control were provided prior to sowing on a ridgetop location adjacent to the original smelter stacks. The seedbed preparation treatments were a double rip line and a roller-aeration. The seedling number represents standardised subplot germination counts minus seedling deaths for each period of survey. Error bars represent standard deviations. n=5

Fig. 4a

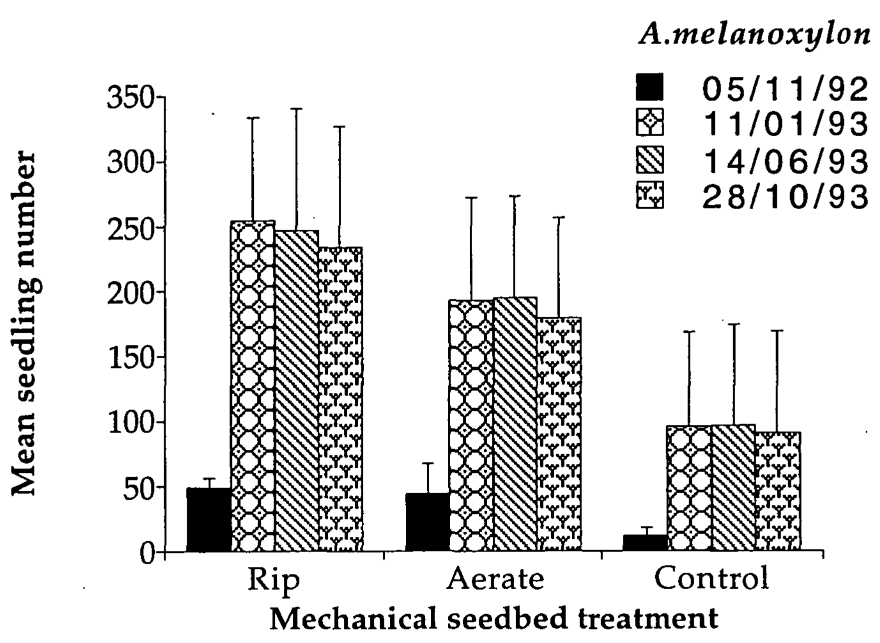


Fig. 4b

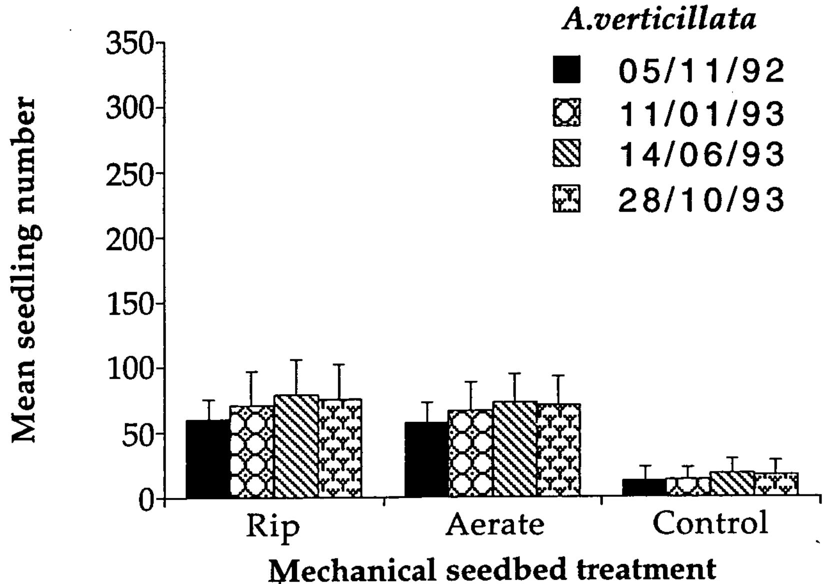

A.verticillata
05/11/92
11/01/93
14/06/93
28/10/93
Mean seedling number
350
300
250
200
150
100
50
0
Rip
Aerate
Control
Mechanical seedbed treatment

Fig. 4c

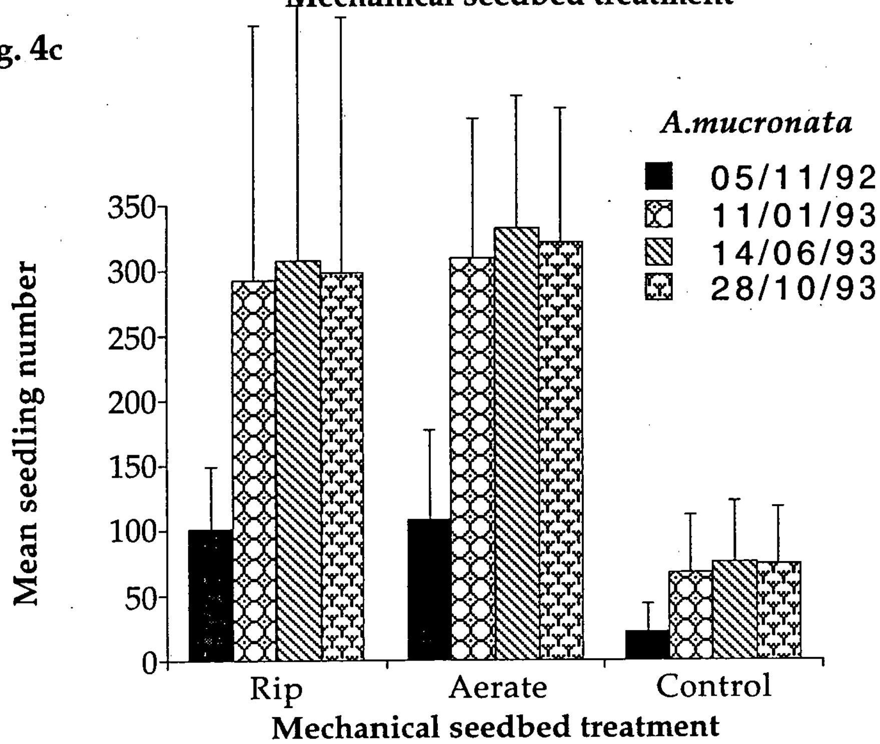

A.mucronata
05/11/92
11/01/93
14/06/93
28/10/93
Mean seedling number
350
300
250
200
150
100
50
0
Rip
Aerate
Control
Mechanical seedbed treatment

Fig. 4d

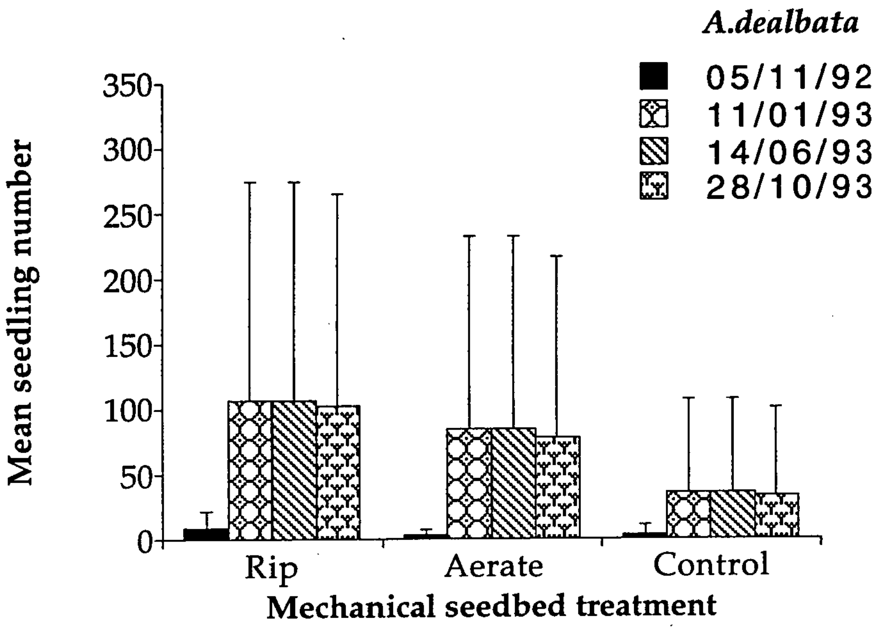
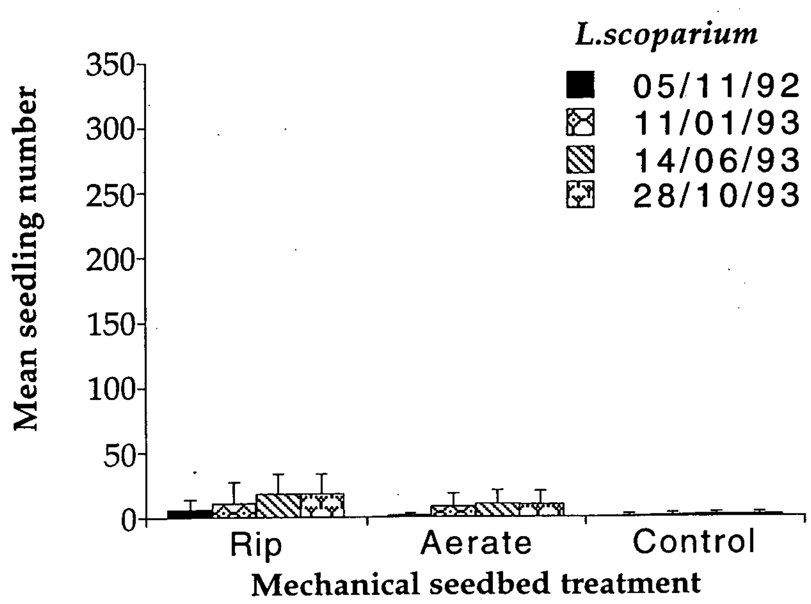

A.dealbata
05/11/92
11/01/93
14/06/93
28/10/93
Mean seedling number
350
300
250
200
150
100
50
0
Rip
Aerate
Control
Mechanical seedbed treatment

Fig. 4e


L.scoparium
05/11/92
11/01/93
14/06/93
28/10/93
Mean seedling number
350
300
250
200
150
100
50
0
Rip
Aerate
Control
Mechanical seedbed treatment

Erosional seed losses

Sowings of *L. scoparium, A. capillaris* and *O. arborescens* resulted in few or no germinants. All three species, and particularly *A.capillaris* and *L. scoparium,* have small, light seed and much of the seed sown may have been washed from the subplots by surface flow following rainfall, despite the soil preparation treatments. Quantitative estimates of the seed losses of these species were not attempted, as grass and tea tree seeds are buoyant when dry, and seed is therefore likely to have escaped collection in a water-trap system.

Analysis of the seed loss data for *Acacia* seed captured in the drainage sumps during the first five months following sowing indicated that the mean erosional seed losses differed between seedbed treatments ($F_{2,36}$ = 4.81, $p < 0.0424$; Table 4a), when tested against a block*treatment error term, but not between species ($F_{3,36} = 1.74$, $p < 0.1752$). Mean seed losses for all three seedbed treatments represented 2% or less of the total number of viable seeds sown per subplot (Table 5a).

The multiple-range comparison of treatment means indicated that both the aerate and the rip seedbed treatments resulted in a significant decrease in the erosional seed losses in comparison to the control. A summary of the multiple range comparison of seed loss treatment means is provided (Table 5a).

The response to seedbed preparation treatments

The seedling number per subplot (standardized) and the seedling

Table 4a: Summary of an analysis of variance of *Acacia* seed loss from a broadcast sowing field trial of split-plot design. The data were derived from counts of seed captured in drainage sumps during the four months following sowing and calculated as a percentage of the total number of seeds of each species sown per subplot. The data were analysed as a randomised complete block of two levels with treatments allocated to mainplots and species allocated to subplots within mainplots. The seedbed treatments were a surface rip, roller-aeration and an undisturbed control.

The field trial was located on a ridge-top of moderate gradient approximately 150 m from the original smelter site.

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p*
Treatment	2	7.77	3.88	4.38	0.0199
Species	3	4.64	1.55	1.74	0.1752
Block	4	2.78	0.69	0.78	0.5436
T'ment* Species	6	10.76	1.79	2.02	0.0881
Error	36	31.93	0.89		
*Tests using Block*Treatment as an error term*					
Treatment	2	7.77	3.88	4.81	0.0424
Block	4	2.78	0.69	0.86	0.5261

Table 5a: Mean percentage seed loss for species of *Acacia* provided with one of three pre-sowing seedbed treatments. The seedbed treatments were a surface rip, roller-aeration and an undisturbed control.

A summary of a multiple range comparison of treatment means based on the pdiff option of the GLM procedure in SAS Institute Inc. (1988) is provided*.

	Mechanical seedbed treatment		
	Control	**Aerate**	**Rip**
Seed loss treatment means	1.15	0.50	0.30
		--------	--------

*Those means not underlined by the same line differ significantly at $p \leq 0.05$.

establishment percentage per subplot data were subjected to the ANOVA procedure. These analyses displayed similar trends in regard to significance effects for the four survey periods. The data selected for further development were the seedling establishment percentages per subplot for the species *L. scoparium*, *A. mucronata*, *A. melanoxylon*, *A. verticillata* and *A. dealbata* derived from seedling count data recorded to 28/10/93.

The mean seedling establishment percentages for five colonising species and each of the seedbed treatments are presented in Figure 5. The lowest non-zero seedling establishment percentage recorded was for the species *L. scoparium* and the control seedbed (0.1%), while the highest was recorded for *A. verticillata* and the rip seedbed treatment (15.7%).

The ANOVA of the seedling establishment percentages indicated that the means differed significantly between seedbed treatments when tested against a block*treatment residual ($F_{2,48} = 13.36$, $p < 0.0028$; Table 4b).

The multiple range comparison of treatment means indicated that, for the species *A. mucronata* and *A. verticillata*, both the aerate and the rip seedbed treatments resulted in significant increases in seedling establishment in comparison to the control treatment. The response of these species to the mechanical treatments, however, did not differ significantly from each other and consequently, were equally effective as a means of providing a seedbed.

Seedling establishment in response to seedbed treatments was, however, not significant for the species *L. scoparium*, *A. melanoxylon* and

Fig.5: Seedling establishment percentages for five colonising species provided with two mechanical, pre-sowing seedbed treatments (Rip, R; Aerate,A) and a control (C) 16 months after broadcast sowing. The mean seedling establishment percentages were calculated from standardised count data termed the seedling number per subplot (cumulative germinations minus deaths for each subplot to the 28/10/93) divided by the total number of viable seeds sown per subplot and represented as percentages. n = 5. Error bars represent standard deviations.

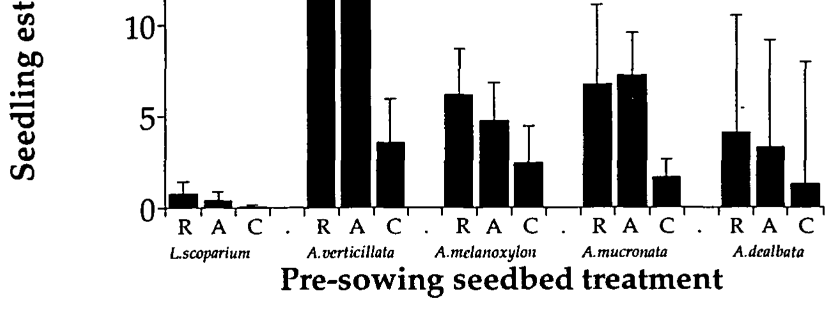


Table 4b: Summary of an analysis of variance of the seedling establishment percentage for a split-plot randomised complete block design with three pre-sowing seedbed treatments and five species (four species of *Acacia* and *L.scoparium*) for data derived from seedling counts during the 16 months following sowing. The seedbed treatments were a surface rip, roller-aeration and an undisturbed control.

The field trial was located on a ridge-top of moderate gradient approximately 150 m from the original smelter site.

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p*
Treatment	2	351.66	175.83	16.65	0.0001
Species	4	975.41	243.85	23.09	0.0001
Block	4	109.58	27.39	2.59	0.0480
T'ment* Species	10	247.42	30.93	2.93	0.0096
Error	48	506.85	910.59		
*Tests using Block*Treatment as an error term*					
Treatment	2	351.66	175.83	13.36	0.0028
Block	4	109.58	27.39	2.08	0.1752

A. dealbata. A summary of the multiple range comparison of treatment means is provided (Table 5b)

The species' response

The seedling establishment means differed significantly between species ($F_{4,48}$ = 23.09, $p < 0.0001$; Table 4b). The mean seedling establishment percentages for the five colonising species without regard to seedbed treatment were calculated and a multiple range comparison of species means provided (Table 6). The mean seedling establishment percentage of *A. verticillata* (11.3%) was significantly higher than all the other species in the trial. Conversely, the establishment percentages of *L. scoparium* (0.4%) and *A. dealbata* (2.88%) were significantly lower than all the other species in the trial.

There was a significant treatment*species interaction ($F_{10,48}$ = 2.93, $p < 0.0096$). This was due to somewhat lower rip treatment means for *A. mucronata* than that recorded for the species in the roller-aerate treatment (Table 5b). A converse trend in the magnitude of the mean response existed between these two treatments for all the other species under trial.

Mortality

Analysis of the mortality data for *Acacia* seedlings for the twelve months following germination indicated that the mean mortality percentage differed between seedbed treatments ($F_{2,36}$ = 8.92, $p < 0.0092$; Table 4c), when tested against a block*treatment error term and between species

Table 5b: Mean seedling establishment percentages for five colonizing species provided with one of three pre-sowing seedbed treatments. The seedbed treatments were a surface rip, roller-aeration and an undisturbed control.

A summary of a multiple range comparison based on the pdiff option of the GLM procedure in SAS/STAT^{R} edition 6.03 (SAS Institute Inc., 1988) is provided*.

	Mechanical seedbed treatment		
	Control	**Aerate**	**Rip**
Species			
A.dealbata	1.30	3.27	4.08
A.melanoxylon	2.41	4.76	6.18
A.mucronata	1.67	7.26	6.71
A.verticillata	3.58	14.56	15.7
L.scoparium	0.07	0.43	0.77

*Those means not underlined by the same line differ significantly at $p \leq 0.05$.

Table 6: Mean seedling establishment percentages for five colonising species given without regard to seedbed treatment.

A summary of a multiple range comparison based on the pDiff option of the GLM procedure in SAS/STAT[R] edition 6.03 (SAS Institute Inc., 1988) is provided*.

Species	*L.scoparium*	*A. dealbata*	*A.melanoxylon*	*A. mucronata*	*A. verticillata*
Species means	0.4	2.88	4.45	5.21	11.27

(A single line underlines the means 4.45 and 5.21.)

*Those means not underlined by the same line differ significantly at $p \leq 0.05$.

Table 4c: Summary of an analysis of variance of *Acacia* seedling mortality from a broadcast sowing field trial of split-plot design. The data were derived from standardised seedling mortality counts calculated as a percentage of the total number of viable seeds of each species sown per subplot for the twelve month period following germination. The data were analysed as a randomised complete block of two levels with treatments allocated to mainplots and species allocated to subplots within mainplots. The seedbed treatments were a surface rip, roller-aeration and an undisturbed control.

The field trial was located on a ridge-top of moderate gradient approximately 150 m from the original smelter site.

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p*
Treatment	2	30.6	1.53	5.55	0.0079
Species	3	13.27	4.42	16.04	0.0001
Block	4	1.32	0.33	1.20	0.3288
T'ment* Species	6	2.06	0.34	1.24	0.3076
Error	36	9.93	0.28		
*Tests using Block*Treatment as an error term*					
Treatment	2	3.06	1.53	8.92	0.0092
Block	4	1.32	0.33	1.96	0.1997

($F_{3,36} = 16.04$, $p < 0.0001$). Mean seedling mortality for four species of *Acacia* ranged from 0.10 (*A.dealbata*) to 1.8 % (*A.verticillata*).

The multiple-range comparison of treatment means indicated that both the Aerate and the Rip seedbed treatments resulted in significantly lower seedling mortality for the species *A.verticillata* in comparison to the control. The differences in the treatment means for the other species were not significant. A summary of the multiple range comparison of seed loss treatment means is provided (Table 5c).

During the summer following the final seedling count most of the *A. dealbata* seedlings died. None had appeared healthy throughout the trial with most displaying premature seedling-leaf loss. With the exception of *A. melanoxylon,* none of the remaining species were as adversely affected with regard to appearance or leaf fall, although the growth rates of most individuals were judged abnormally slow. Leaf discolouration and premature loss affected many of the *A.melanoxylon* seedlings in the period of survey. However, catastrophic mortality rates did not occur.

Phytotoxic symptoms were infrequent or absent in *A. mucronata, A.verticillata* and *L. scoparium* in the survey period.

The cost effectiveness of species/treatment combinations

Calculation of the seedling establishment percentages for each of the species and treatment combinations permitted evaluation of the results on the basis of cost. In order to provide an example of the relative

Table 5c: Mean seedling mortality percentage for four species of *Acacia* provided with one of three pre-sowing seedbed treatments. The seedbed treatments were a surface rip, roller-aeration and an undisturbed control.

A summary of a multiple range comparison of treatment means based on the pdiff option of the GLM in SAS/STAT^{R} edition 6.03 (SAS Institute Inc., 1988) is provided*.

	Mechanical seedbed treatment		
	Control	**Aerate**	**Rip**
Species			
A.dealbata	0.10	0.17	0.17
A.melanoxylon	0.29	0.62	0.87
A.mucronata	0.10	0.30	0.39
A.verticillata	0.63	1.61	1.82

*Those means not underlined by the same line differ significantly at $p \leq 0.05$.

species/treatment costs, the seed quantities required to produce 1000 seedlings given each combination were calculated (Table 7). The number of kilograms of seed required (bulk seedlot) to produce 1000 seedlings in the field was calculated by dividing the number of seedlings required by the number of viable seeds per kilogram multiplied by the seedling establishment percentage for the treatment.

Using an average purchase cost of $100 per kg for each of the species examined, the cost of establishing a single seedling of each species was calculated for each treatment. Four species cost less than $0.05 per seedling to establish using the roller-aerate pre-sowing treatment namely, *A.melanoxylon* ($0.03), *A.mucronata* ($0.02) *L.scoparium* ($0.02) and *A.verticillata* ($0.01). At $0.06 per seedling, *A.dealbata* was comparatively expensive to establish, despite the provision of seedbed preparation.

The distribution of the seedling potential

Seedling establishment, erosional seed loss and seedling mortality were all calculated as a percentage of the number of viable seeds sown per subplot. This permitted an assessment of the distribution of the seedling potential of each species, and by subtraction, an estimation of the seedling potential unaccounted for by seedlings, seed loss or death.

The seedling establishment percentage, the percentage seed loss due to erosion and the percentage seedling mortality accounted for a relatively small portion of the seedling potential of the four species of *Acacia* (Table 8).

Table 7: Mean seedling establishment percentages for broadcast sown seed, an example of the seed quantities required to produce 1000 seedlings in the field and the relative cost of seedling establishment.

The seedling establishment percentages were based on standardised germination count and mortality data for the 16 month period following sowing (termed the seedling number per subplot - cumulative). Standardisation accounted for unequal subplot losses of surface lying seed due to erosion. The seedling establishment percentages were calculated by dividing the cumulative seedling number per subplot by the number of viable seeds sown per subplot. Standard deviations are given in brackets.

Species	Seedlot collection number	Pre-sowing seedbed treatment	Seedling establish/ment %[1]	Seed quantity required to provide 1000 seedlings[2] (kg)	Cost of seedling est/ment[3] ($/seedling)
A.mucronata	001/02/92	rip	6.71 (4.40)	0.249	0.02
A.mucronata	001/02/92	aerate	7.25 (2.33)	0.235	0.02
A.mucronata	001/02/92	control	1.67 (0.98)	1.000	0.10
A.melanoxylon	003/02/91	rip	6.18 (2.48)	0.231	0.02
A.melanoxylon	003/02/91	aerate	4.75 (2.09)	0.300	0.03
A.melanoxylon	003/02/91	control	2.42 (2.04)	0.589	0.06

continued

Table 7 continued

Species	Seedlot collection number	Pre-sowing seedbed treatment	Seedling establish/ment %[1]	Seed quantity required to provide 1000 seedlings[2] (kg)	Cost of seedling est/ment[3] ($/seedling)
A.verticillata	003/02/92	rip	15.70 (5.52)	0.130	0.01
A.verticillata	003/02/92	aerate	14.55 4.44)	0.141	0.01
A.verticillata	003/02/92	control	3.59 (2.37)	0.573	0.06
A.dealbata	001/01/91	rip	4.08 (6.45)	0.462	0.05
A.dealbata	001/01/91	aerate	3.27 (5.87)	0.577	0.06
A.dealbata	001/01/91	control	1.30 (2.69)	1.450	0.14
L.scoparium	001/11/90	rip	0.77 (0.68)	0.130	0.01
L.scoparium	001/11/90	aerate	0.42 (0.45)	0.238	0.02
L.scoparium	001/11/90	control	0.06 (0.01)	1.666	0.16

1 - n=5

2 - Bulk seed quantity(kg) required to provide 1000 seedlings at the treatment establishment rate determined for the species.

3 - Based on an average seed purchase cost of $100 per kilogram.

Table 8: The distribution of the seedling potential for four species of *Acacia* given three pre-sowing, seedbed treatments at a near-smelter site. The seedling potential equalled the number of viable seeds sown of each species to each treatment subplot.
The number of seedlings to establish (seedling establishment), seed loss due so surface erosion (seed loss), and the number of seedling deaths (mortality) were calculated as a percentage of the number of viable seeds sown. The proportion of the seedling potential unaccounted for by seedling establishment, seed loss and mortality was calculated by subtraction.

Species	Treatment	Mean seedling establishment %	Mean seed loss %	Mean mortality %	Unaccounted loss of seedling potential (%)
A.mucronata	rip	6.7	0.3	0.4	92.6
A.mucronata	aerate	7.2	0.5	0.3	92.0
A.mucronata	control	1.7	0.6	0.1	97.6
A.melanoxylon	rip	6.2	0.3	0.9	92.6
A.melanoxylon	aerate	4.7	0.7	0.6	94.0
A.melanoxylon	control	2.4	0.2	0.3	97.1
A.verticillata	rip	15.7	0.7	1.8	81.8
A.verticillata	aerate	14.5	0.0	1.6	83.9
A.verticillata	control	3.6	2.5	0.6	93.3
A.dealbata	rip	4.1	0.3	0.2	95.4
A.dealbata	aerate	3.3	0.8	0.2	95.7
A.dealbata	control	1.3	1.9	0.1	96.7

1 - Seedling potential equalled the number of viable seeds sown
2 - Seedling establishment percentage equalled the germination counts (standardised) minus deaths (standardised) divided by the seedling potential
3 - Seed loss % equals the number of viable seeds lost to surface erosion divided by the seedling potential
4 - % mortality equals the number of deaths (standardised) divided by the seedling potential

Seedling establishment for *A. mucronata* increased from a mean of 1.7% in the control treatment to 7.2% in response to the roller-aeration treatment, however, only a small component (0.1%) of this increase could be attributed to a reduction in seed loss due to the treatment. Conversely, seedling mortality for the species increased marginally (0.2%) in response the roller-aeration treatment. For this treatment/species combination the proportion of the seedling potential remaining unaccounted for was approximately 92%.

The distribution of the *A. verticillata* seedling potential was similar to that of *A. mucronata.* Seedling establishment for the species *A. verticillata* increased from a mean of 3.6% in the control treatment to 14.5% in response to the roller-aeration treatment. However, only 2.5% of this increase could be attributed to a reduction in seed loss due to the treatment. Seedling mortality for the species increased 1.0% in response the roller-aeration treatment. For this treatment/species combination the proportion of the seedling potential remaining unaccounted for was approximately 84%.

6.5 Discussion

The preparation of an appropriate seedbed is commonly cited as a requirement for successful revegetation with Australian native species where broadcast methods are employed (Glossop, 1982; Hinz, 1980). For example, surface ripping has been applied widely, as the initial, and sometimes only, means of broadacre seedbed preparation prior to broadcast sowing in denuded environments as diverse as grazing land, logging coupes and graded, artificial landscapes formed from replaced mine overburden (Hinz, 1995; Lyons, 1995; Gunness and Lawrie, 1988). Successful use of this method of ground preparation has been attributed to factors as diverse as the environments themselves. These include increases in the number of seed-suitable microsites, improved soil aeration, water penetration, drainage, burial and protection from predators.

The majority of Mount Lyell, however, is unsuited to most conventional forms of seedbed preparation due to the steepness of the terrain and the likelihood of major soil disturbance leading to further serious and possibly uncontainable erosion. Nevertheless, the potential efficiencies offered by an appropriate, mechanical means of seedbed preparation remain compelling.

In this chapter, a roller-aeration implement, originally designed to aerate compacted dairy pasture, was evaluated as a means of providing a seedbed suitable for the establishment of broadcast sown, colonising species. The Superworm Soil Aerator, while limited to relatively moderate terrain by the capabilities of the tow vehicle, eliminated concerns regarding soil erosion as the tynes punctured the soil surface creating discrete, rip-cavities. This produced minimal surface disturbance and profile upheaval.

The mean seedling establishment percentages for the *Acacia* species in the control sowings on unprepared ground ranged from a minimum of 1.3±2.7 (*A.dealbata*) to a maximum of 3.6±2.4% (*A.verticillata*). Clemens (1980) suggested that a field seedling establishment of 10% should be considered the upper limit possible for viable, broadcast sown *Acacia* and other legume seed in degraded, but otherwise uncontaminated environments. The same author suggested that a more typical figure might be 5%. Other researchers, working in the area of mine site revegetation, support this estimate (Bellairs, pers. comm.). Consequently, the establishment percentages in response to the control treatment were below that expected for the genus despite the absence of competition from other species and the exclusion of browsing pressures.

In the roller-aeration treatment, the mean seedling establishment percentages for the *Acacia* species were higher than those of the control treatment. The seedling establishment percentages for the treatment approached or exceeded those expected for the genus and ranged from a minimum of 3.3±5.9 (*A. dealbata*) to a maximum of 14.5±4.4% (*A. verticillata*). The increases were significant for *A. mucronata* and *A. verticillata.* The establishment percentages represented a four-fold improvement in seedling establishment for both species.

The conventional, deep-rip seedbed treatment also resulted in higher mean seedling establishment percentages for all of the *Acacia* species in the trial. The increases were significant for *A. mucronata* and *A. verticillata.* Notably, the seedling establishment percentages in response to the rip treatment were statistically indistinguishable from that recorded for the roller-aeration treatment. Consequently, the two disparate mechanical treatments were equally effective as seedbed preparations for these two species.

The increases in seedling establishment recorded in response to the mechanical seedbed treatments were due to both a reduction in the number of seeds lost to surface erosion and to higher levels of germination and establishment. However, the distribution of the seedling potential in the mechanical seedbed treatments indicated that the contribution of seed loss to higher levels of seedling establishment, although significantly less than those of the control treatment, was small in proportion to the improvement in seedling establishment. This suggested that response to mechanical seedbed treatments had, to a large extent, occurred as a result of a moderation of the environmental extremes to which the surface lying seed were exposed during germination and early growth. It appeared that the treatments increased the number of seed-suitable microsites. There was no evidence from the control treatments to suggest that the increases in seedling establishment were caused by a reduction in the number of seedling deaths.

In general, the mechanical seedbed treatments used in these trials were unsuitable for the smaller seeded species. The species *L. scoparium*, *O. arborescens*, *G. hispida* and *A. capillaris* responded poorly, or failed completely, in response to both of the mechanical seedbed preparation treatments. Some seed losses are likely to have occurred due to water, and possibly wind, erosion. Additional losses may be accounted for by an inappropriate depth of cultivation produced by the mechanically treated seedbeds in relation to the size of the seeds.

Based on the average seed price of $100 kg^{-1}$, estimates of the cost of seedling establishment on an unprepared seedbed ranged from $0.06 (*A. verticillata*) to $0.14 (*A. dealbata*) per seedling. Mechanical seedbed preparation reduced the cost of establishment of all the *Acacia* species in the trial. The cost of establishment of the two species exhibiting significant responses to the roller-aeration treatment, *A. mucronata* and *A. verticillata*, was reduced from $0.10 to $0.02 and $0.06 to $0.01 per seedling, respectively. These estimates were well below the average cost of seedling stock at approximately $0.65 per seedling.

Despite a low establishment percentage, the cost of *L. scoparium* establishment was comparable to that of the *Acacia* species. This was a consequence of the high number of seeds per kilogram. Consequently, the inclusion of this species in a broadacre sowing program could well be justified on the basis of comparable establishment costs.

A large proportion of the seedling potential of the broadcast seed was unaccounted for by erosional seed loss, germinant and seedling mortality. Seed dormancy or death are likely to account for the discrepancy. The influence of soil phytotoxicity on seed mortality is not known.

6.6 Conclusions

The complexities of experimental design and the difficulties posed by field trials have apparently hindered research into Australian native seed establishment on degraded and mined lands. They have also resulted in doubtful practices, such as unreplicated trials with poor sowing and monitoring proceedures. Consequently, there are very few reliable, published references regarding field establishment rates for native species on degraded or mined lands. For example, no reliable references to seedling establishment percentages for *Leptospermum* species, or the other non-Mimosaceae species, were found in the literature. Much of the work done in this field remains unpublished in inhouse documents. Field sowings on unprepared seedbeds rarely, if ever, approach the seedling potential of a species and many losses of surface lying seed may occur as a result of an unfavourable seedbed. Seedbed preparation can narrow the seedling gap, the difference between the seedling potential of a species and the actual numbers of seedlings occurring, and result in cost-effective sowings.

The objective of this chapter was to evaluate mechanical seedbed preparation for the establishment of a range of broadcast sown colonising species within the Mount Lyell environment. Broadcast sowings on unworked subsoils resulted in poor seedling establishment and three of the eight species in the trial failed to produce seedlings. Mechanical seedbed treatment did not assist these species. In comparison, a small number of several species of the genus *Acacia* established on unworked ground. In general, these species responded well to two mechanical, pre-sowing seedbed treatments.

The roller-aerator, seedbed treatment provided by the Superworm Soil Aerator resulted in significant increases in seedling establishment in *A. mucronata* and *A. verticillata.* The improvement greatly reduced the establishment cost of a seedling. The seedling establishment percentages recorded for the treatment approached or exceeded the field percentages considered the upper limit possible for sowings of the genus on uncontaminated land. The establishment percentages achieved by the roller-aerator treatment were statistically equivalent to those recorded for a conventional deep-rip treatment. The roller-aerator treatment, however, did not aggravate soil erosion.

Improvements in seedling establishment percentages as a result of mechanical seedbed treatment were attributed to the provision of favourable microsites. These provided a buffer to the environmental extremes to which the surface lying seed were exposed during germination and early growth. Small gains were achieved by reductions in erosional seed losses and seedling mortality. Presumably, this was due to the creation of a larger number of seed-suitable microsites. This was consistent with the observation that desiccation of unprotected seedling radicals was a major cause of seedling death in surface broadcast seed.

The seedling establishment percentages achieved during these trials indicated that three species of *Acacia*: *A. verticillata, A. mucronata* and *A. melanoxylon*, can be economically re-established by broadcast sowing at a highly degraded, contaminated site. The ease of the mechanical seedbed treatment and the improvement in seedling establishment greatly improved the overall cost-effectiveness of broadcasting as a method of revegetation. However, seedbed preparation by these mechanical methods is limited by steepness and accessibility. It is estimated that approximately 10% of Mount Lyell could be considered machine-accessible to conventional or adapted machinery. This includes most ridge-top locations. Although representing only a small proportion of the total area, rehabilitation of these areas could, over time, influence much of the remaining, under-vegetated areas by acting as an on-site source of seed and organic matter.

Chapter 7

An assessment of mininum impact, seedbed preparation and seed application methods for the establishment of colonising species on steep, eroded slopes at Mount Lyell

7.1 Introduction

Attempts have been made to re-establish vegetation on degraded landscapes by surface broadcasting (Winterhalder, 1983b). However, even in moderate terrain, seed broadcast under these circumstances is likely to be lost in surface wash and heavy losses may negate the cost-advantages offered. Mechanical surface contouring, or re-grading, accepted practises within some sectors of the mining industry, provides a stable soil surface for revegetation, but is expensive and not suited to steep, erodable and culturally significant sites.

On a small scale, steep slopes, such as along road batters, have been rehabilitated to introduced pasture species by non-mechanical methods such as hydro-mulching (eg. Hydro Electric Corporation of Tasmania). Largely due to their costs, however, such methods are not generally applied broadacre. In contrast, the sand mining industry provides one of the few examples of the use of a non-mechanical means of broadacre surface stabilization prior to revegetation. On North Stradbroke Island, a bitumen emulsion, sold as Terolas, is sprayed over a drilled cover-crop and native seed mix (Bellairs *et al.*, 1995; Brooks and Bell, 1984). The stabilizer minimises aeolian sand movement during plant establishment and is bio-degradable over 20 plus years (pers. obs.). At Ashio, reclamation of mountainous areas denuded by pollution from the Furukawa Co. copper smelter has been attempted by broadcast-sowing grass and tree species by helicopter with dilute asphalt as a stabilizer (The Daily Yomiuri, 1993).

Over much of the area of Mount Lyell, steep erodable slopes prevent the use of conventional, mechanical surface stabilization methods in a region that receives high rainfall. In addition, it is likely that the proximity of the township to the site would make the use of mechanical methods, such as re-contouring, culturally unacceptable. Never the less, cost-effective revegetation at Mount Lyell is likely to require the use of broadcast sown seed on steep-slopes and these will require adequate stability to enable establishment.

This chapter compares minimum-impact methods of seedbed preparation and seed application for colonising species on steep slopes. Seven non-mechanical treatments suited to broadcast sowing were examined and assessed by seedling establishment. The treatments could be categorised as either low-cost methods based on broadcasting or higher-cost methods based on a proprietary product designed for erosion control. The organic product, J-tac™ (Appendix 2), was selected for its rapid bio-degradability.

The treatments were selected to minimise erosive seed loss prior to germination and to moderate environmental extremes. Four of the treatments specified the application of the organic stabilizer/adherent : one of these treatments included straw as mulch; another a pre-glued, straw/paper-shred 'amalgam'. The adherent treatments were compared to a pre-sowing, cover crop and manual, soil preparation treatments.

Each of the treatments was implemented in a manner relevant to broadacre or 'strip acre' application. None of the treatments required significant working of the soil or relied on the direct placement of seed to depth.

7.2 Methods

The experimental site was located approximately 150 m east of the original smelters within the *Agrostis* grassland zone (Fig.1).

Eighty four 1 m^2 plots were laid out in twelve blocks of seven treatments on a vegetation-free, 20° slope of westerly aspect (Fig.2). Each plot was located in order to preclude overland 'seed contamination' from nearby plots.

All of the blocks were located on exposed, mafic-clay subsoils. The surface of most blocks exhibited heavy scatterings of fractured, quartz scree, probably of vein origin, while some featured erosion-polished clays. The area selected for each block was based on the within-block surface uniformity.

Plots within blocks were randomly allocated to one of each of the seven seedbed preparation and application treatments. The plots were marked with a steel peg and identified using a coded tagging system. Fencing was not provided. Ground calcitic lime was applied at the rate of 4000 kg ha^{-1} to each plot prior to treatment. This resulted in an increase in soil pH from 4.35±0.30 to 5.20±0.40. The water soluble concentrations of Cu, Al and Zn were 2.71, 10.9 and 0.11 mg L^{-1}, respectively. In the cases of Cu and Al, these concentrations were above the means recorded for the *Agrostis* grassland sites (Table 9, Chapter 4). The Zn concentration was approximately average for the zone.

All treatments, with the exception of the cover crop treatment, received a surface sowing of an *Acacia-Leptospermum* seed mix at the time of seedbed

Fig. 1: Location of 'steep-slopes' seedbed field trial

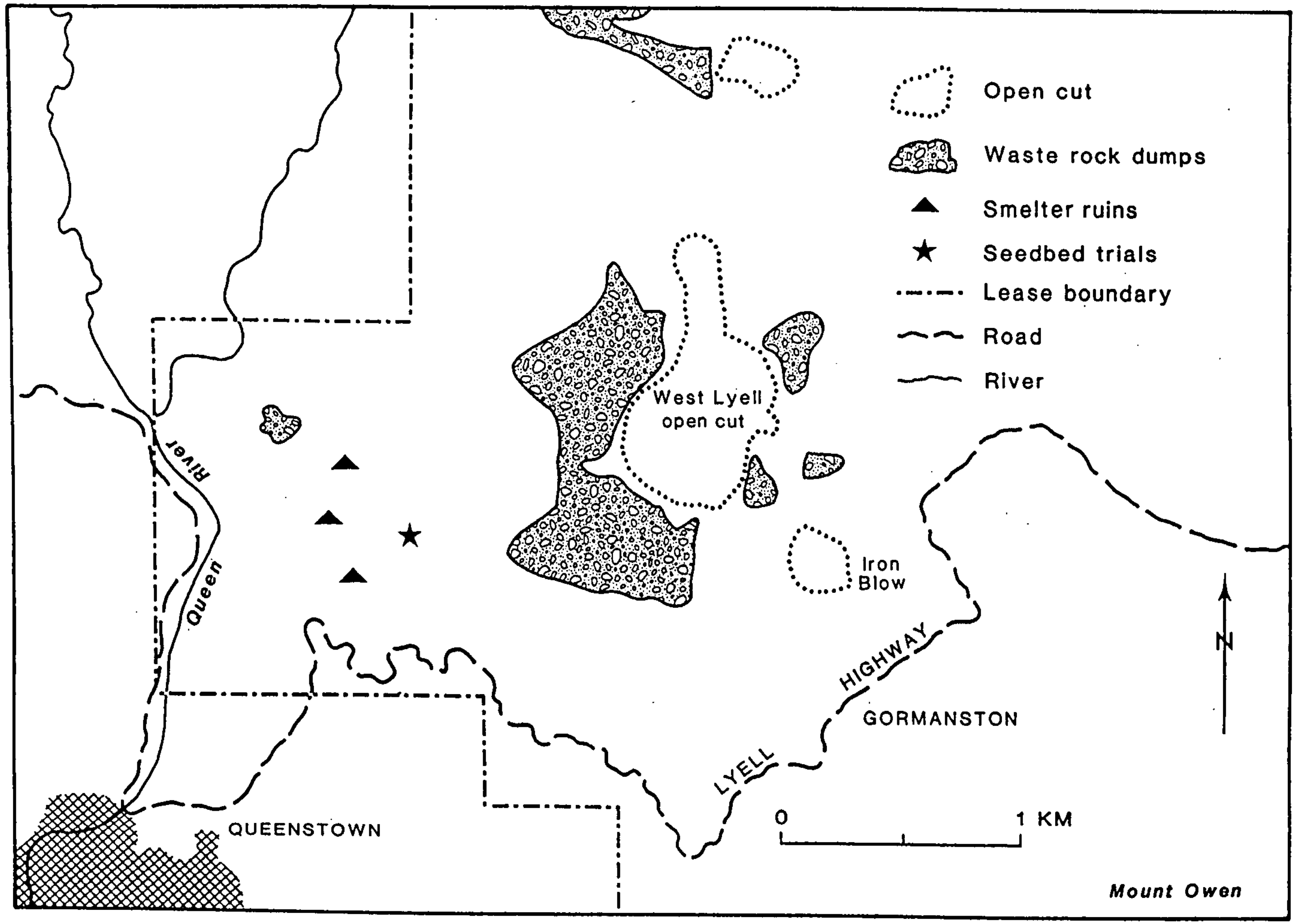


Figure 2: The location of the field trial on an eroded, 20° slope. The operator is depicted applying adherent from a pressure container.

preparation (June 1993; Fig.3). The cover crop was to be oversown with the *Acacia-Leptospermum* seed mix later in the same year.

The seed mix included three species of *Acacia* and *L. scoparium*. The *Acacia* species were *A. melanoxylon*, *A. mucronata* and *A. verticillata*. The seed used was of local provenance. These species are considered colonising species in the region.

The seven seedbed preparation and application treatments were:
1) Adherent 1 (A1): an organic binder;
2) Adherent 2 (A2): a higher concentration of organic binder;
3) Mulch (M): straw plus an organic binder;
4) Paper-glue (Pg): seed, shredded straw and paper sprayed with PVA glue, allowed to dry and applied with an organic binder;
5) Cover crop (CC); a broadcast sown cover crop (two pasture species and a naturalized exotic) and an organic binder;
6) Manual soil perforation with a Leggot's spear (L): 25 x 5 cm deep divots per m^2 (Leggot, 1981).
7) and a control (C): broadcast sown seed-mix only.

The adherent used is described in Appendix 2. Details of the species mixtures, sowing rates and the methods of seedbed preparation and seed application of each treatment are provided in Appendix 3. The germinants were counted during the last weeks of October 1993 and again in March 1994. In the October count, the three species of *Acacia* were recorded collectively as '*Acacia* species' as the seedlings' size precluded positive identification. Each plot was overlain with a quadrat divided into 100 equally sized cells. Nine cells, representing nine percent of the surface of each plot, were then selected at random for seedling counts.

Figure 3: Seed mixes were hand broadcast. The photo depicts the sowing of the paper-glue mix and the application of the adherent.

7.3 Data analysis

The Treatments

To facilitate comparison with seedlots of differing viability and application rate, the germination count data for the late spring (October) and the early autumn (March) survey periods were converted to percentage seedling establishment by division by the total number of viable seeds sown per plot.

As differentiation between the three species of *Acacia* was difficult at the October seedling count the total number of *Acacia* seedlings was summed. Seedling establishment percentages were calculated from the total number of viable *Acacia* seeds sown per plot using an average seedlot viability of 90%. Establishment percentages for the species *L. scoparium* at the October survey were also calculated.

The experiment formed a single factor, randomised complete block with twelve blocks and six non-structural treatments allocated to plots within blocks. The October seedling establishment percentage data for the *Acacia* species and *L.scoparium* were subjected to an ANOVA using the GLM procedure of SAS/STAT^R edition 6.03 (SAS Institute Inc., 1988) and analysed as a randomised complete block with treatments allocated to blocks. These data conformed to the model requirements of normality and homogeneity and arc-sine transformation was considered unnecessary.

The March seedling establishment percentage data for each the four species were subjected to an ANOVA using the GLM procedure of SAS/STAT^R

edition 6.03 (SAS Institute Inc., 1988) and analysed as a randomised complete block with treatments allocated to blocks. These data conformed to the model requirements of normality and homogeneity and arc-sine transformation was considered unnecessary.

Least squares means were calculated and compared using the probability difference (pDiff) option of the SAS/STAT^R program. Following the ranking of treatment and species means, a multiple-range comparison of the pairwise probability differences between means was created and summarised.

Treatment costs

The cost of each treatment was calculated based on the seedling establishment percentages and material costs. The seed costs were based on the nominal target of 5000 seedlings of each species per hectare. The required number of stems per hectare were divided by the number of viable seeds per kilogram and then multiplied by the appropriate seedling establishment percentage. The resultant 'number of kilograms of viable seed required to produce 5000 seedlings in the field for a particular treatment' was costed at the average seed cost of \$100 kg^{-1}. No attempt was made to incorporate labour costs as these will vary with circumstances.

7.4 Results

It was intended that the cover crop (CC) treatment provide a physical barrier to seed loss by erosion and a protected seedbed environment onto which the desired colonising species might be later sown. The introduced pasture species used in the cover crop (CC) treatment did not, however, display sufficient vigour (or density) at any time in the four month period between sowing and initial scoring to provide this function. Despite the addition of a nitrogenous fertilizer (ammonium nitrate; 250 $kgha^{-1}$) growth was restricted to widely spaced plants with abnormally small leaf sizes. Leaf chlorosis was also evident, particularly in the rye grass. With the exception of *A. capillaris* all of the seedlings in the cover crop treatment were dead after nine months. The seedling establishment rate for *A. capillaris* in the cover crop treatment was 0.15%. Consequently, the treatment was considered unworkable and the Cover crop (CC) treatment was discontinued.

No maintenance difficulties were experienced with the remaining six treatments.

Assessment of the Spring seedling counts

An initial assessment of the response to treatments was made for both the *Acaci*a species and *L. scoparium* using the seedling count data for October.

The seedling establishment means for the October seedling counts for the *Acacia* species and *L. scoparium* differed between treatments ($F_{5,45} = 6.88$, $p < 0.0001$ and $F_{5,45} = 4.19$, $p < 0.0033$, respectively; Table 1). Seedling establishment means for the *Acacia* species ranged from a maximum of 9.3% (Leggot's spear treatment) to a minimum of 2.2% (control treatment; Fig. 4a). The equivalent maximum and minimum for *L. scoparium* were 8.6 (mulch) and 2.8% (control), respectively (Fig. 4b).

Table 1: Analysis of variance of the effect of six broadcast-sowing seedbed treatments on seedling establishment in a 'steep slopes' field trial. The data were analysed as seedling establishment percentages for the total number of *Acacia* seedlings arising from sowings of three species and *L .scoparium*. The data represent the seedling counts for October.

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p*
Acacia species					
Block	11	280.99	25.54	2.37	0.0208
Treat	5	371.07	74.21	6.88	0.0001
Error	55	485.18	10.78		
L. scoparium					
Block	11	233.46	21.22	1.73	0.0974
Treat	5	257.09	51.42	4.19	0.0033
Error	55	552.39	12.27		

Figure 4a: Seedling establishment percentages for three species of *Acacia* given six seedbed preparation and application treatments on a 20° slope. The treatments were Adherent (A1), Adherent (A2), Paper glue (Pg), mulch (M), Leggot' spear (L) and control (C). The data were calculated from the sum of the *Acacia* seedling counts (*A. melanoxylon, A. mucronata* and *A. verticillata*) divided by the total number viable *Acacia* seeds sown and represented as a seedling establishment percentage. Mean establishment percentages were calculated for both a spring (October) and an autumn (March) seedling count. The percentages were ranked in order of increasing magnitude for the March count. Error bars represent standard deviation. n = 12

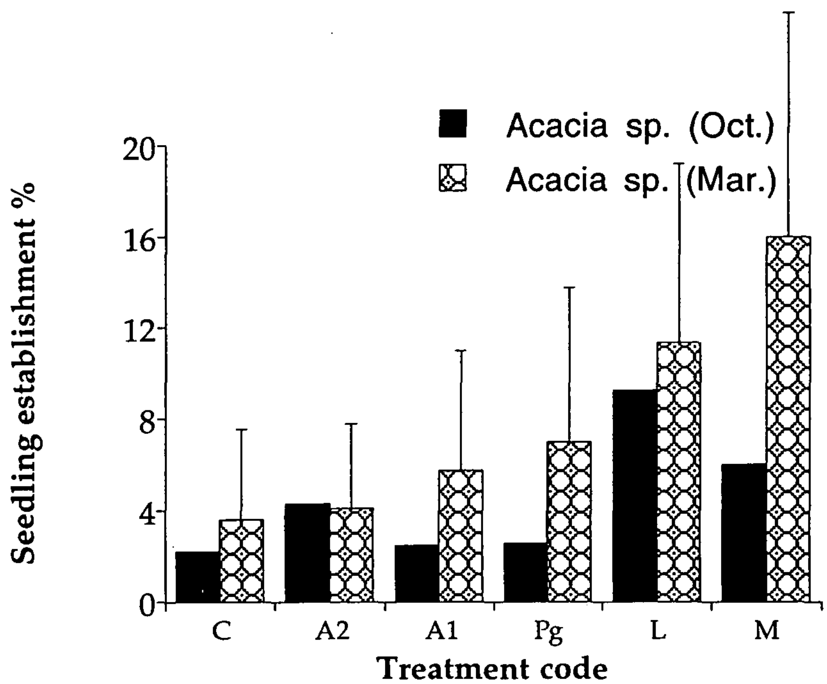
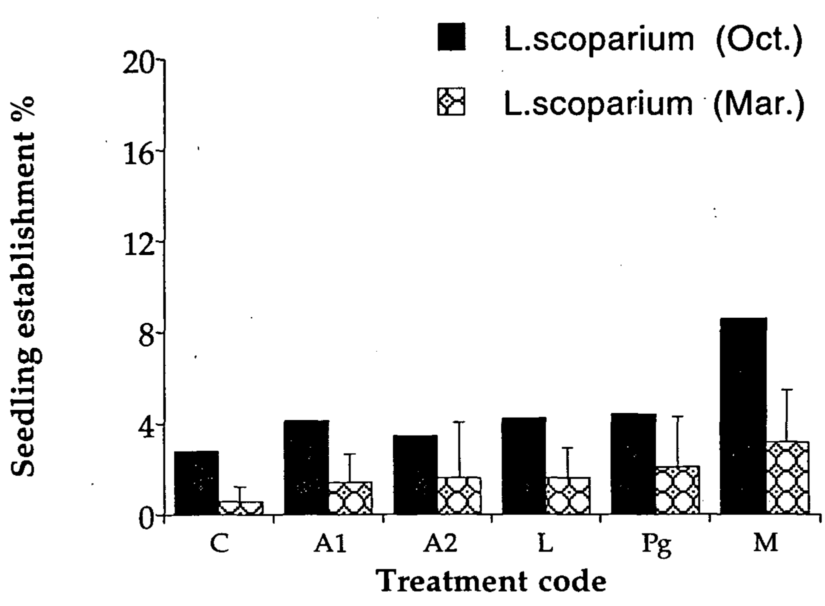


Figure 4b: Seedling establishment percentages for *L. scoparium* given six seedbed preparation and application treatments on a 20° slope. The treatments were Adherent (A1), Adherent (A2), Paper glue (Pg), mulch (M), Leggot's spear (L) and control (C). The data were calculated from seedling counts divided by the total number viable seeds sown and represented as a seedling establishment percentage. Mean establishment percentages were calculated for both spring (October) and autumn (March) seedling counts. The percentages were ranked in order of increasing magnitude for the March count. Error bars represent standard deviation. n =12



The multiple range comparison of the October treatment means indicated that both the mulch and the Leggot's spear treatments resulted in significantly higher *Acacia* seedling establishment in comparison to all the other treatments. For the species *L. scoparium,* only the mulch treatment provided a significantly higher mean seedling establishment response.

Three species of Acacia

The estimated total number of *Acacia* seedlings for all treatments increased from 2 999 to 5 149 in the period between the October and the March counts. This was reflected in higher seedling establishment percentages across all treatments for the species *A. mucronata* and *A. verticillata* in comparison to the other *Acacia* species (Fig. 4a). A converse trend appeared to be true for *A. melanoxylon,* which exhibited declining establishment percentages over the same period.

The March seedling counts permitted the identification of *Acacia* by species. Seedling establishment treatment means for *A. mucronata* and *A. verticillata* ranged from maxima of 21.7±13.1% (mulch) and 29.3±24.5% (mulch) to minima of 4.6±5.5 and 7.6±10.7%, respectively (control treatments; Fig. 5a/5b).

The seedling establishment means for *A. mucronata* and *A. verticillata* differed between treatments ($F_{5,55} = 9.49$, $p < 0.0001$ and $F_{5,55} = 3.27$, $p < 0.0118$, respectively; Table 2). There was no corresponding treatment response for *A. melanoxylon* ($F_{5,55} = 1.86$, $p < 0.1158$).

The multiple range comparison of the March treatment means indicated

Fig. 5a: Seedling establishment percentages for *A. melanoxylon* and *A. mucronata* given six seed application treatments. The treatments were Adherent (A1), Adherent (A2), Paper glue (Pg), mulch (M), Leggot' spear (L) and control (C). The data represented the mean number of germinants nine months after sowing (March '94 count) divided by the total number viable seeds sown and calculated as a percentage. The percentages were ranked in order of increasing magnitude. Error bars represent standard deviation. n = 12

A summary of a multiple-range comparison of treatment means based on the pDiff option of SAS/STAT[R] edition 6.03 (SAS Institute Inc., 1988) is provided. Means annotated with disimilar letters are significantly different ($p< 0.05$).

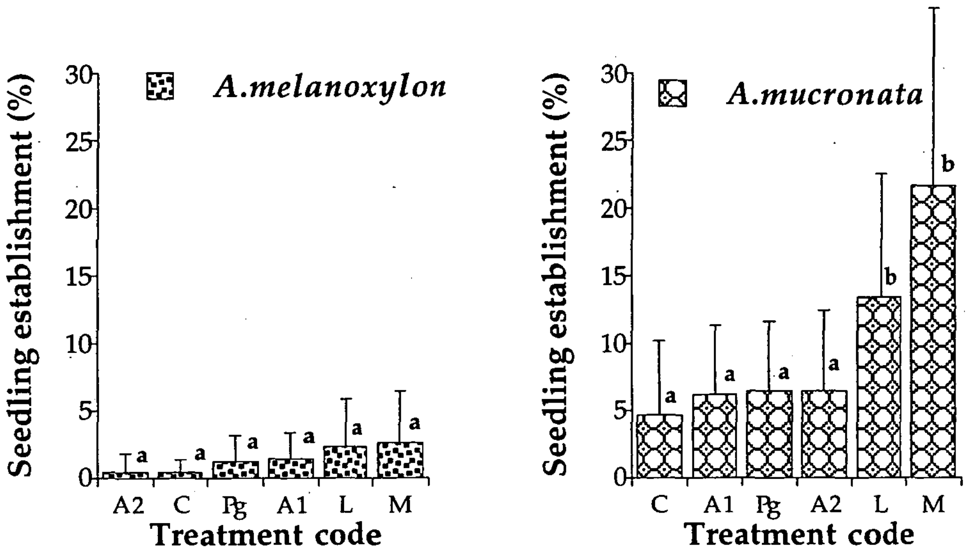


Fig. 5b: Seedling establishment percentages for *A. verticillata* and *L. scoparium* given six seed application treatments. The treatments were Adherent (A1), Adherent (A2), Paper glue (Pg), mulch (M), Leggot' spear (L) and control (C). The data represented the mean number of germinants nine months after sowing (March '94 count) divided by the total number viable seeds sown calculated as a percentage. The percentages were ranked in order of increasing magnitude. Error bars represent standard deviations. n =12

A summary of a multiple-range comparison of treatment means based on the pDiff option of SAS/STAT[R] edition 6.03 (SAS Institute Inc., 1988) is provided. Means annotated with disimilar letters are significantly different ($p < 0.05$).

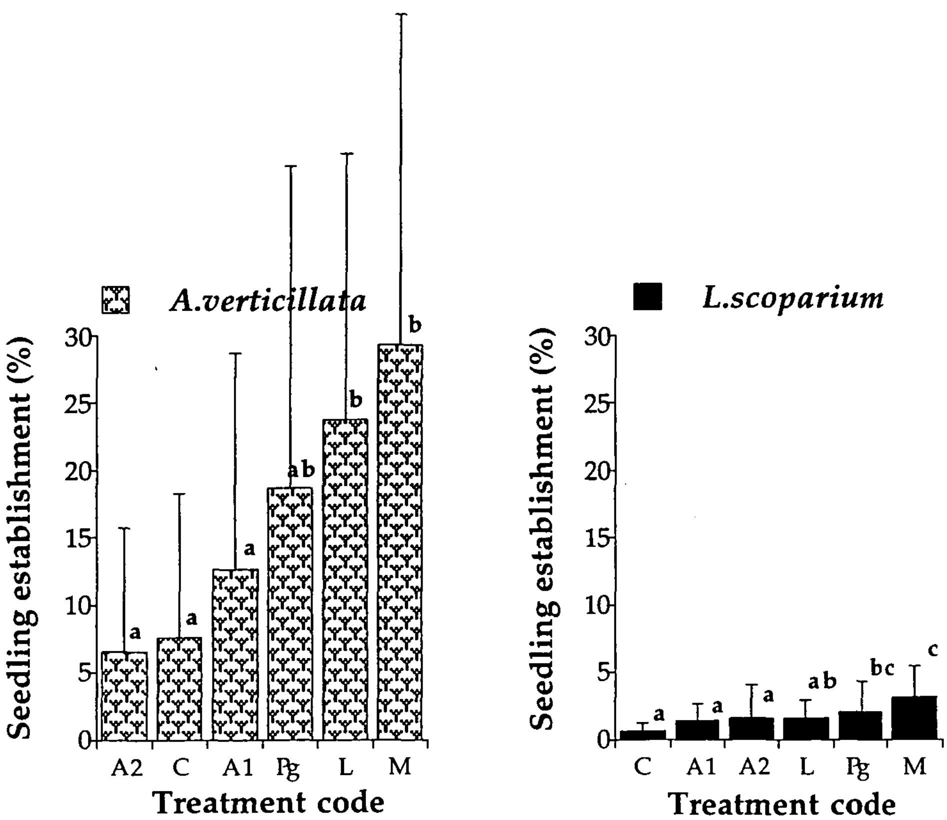


Table 2: Analysis of variance of the effect of six broadcast-sowing seedbed treatments on seedling establishment in a 'steep slopes' field trial. The data were analysed as seedling establishment percentages for three species of *Acacia* and *L. scoparium*. The data represent the seedling counts for March.

Source	*D.F.*	*Sums of squares*	*Mean square*	*F value*	*p*
A. melanoxylon					
Block	11	110.06	10.00	1.83	0.0713
Treatment	5	51.02	10.20	1.86	0.1158
Error	55	301.24	5.48		
A. mucronata					
Block	11	1115.15	101.38	1.85	0.0673
Treat	5	2600.84	520.17	9.49	0.0001
Error	55	3014.16	54.80		
A. verticillata					
Block	11	5446.48	495.15	1.63	0.1173
Treat	5	4977.23	995.44	3.27	0.0118
Error	55	16757.93	304.69		
L. scoparium					
Block	11	72.78	6.61	2.48	0.0132
Treat	5	43.13	8.63	3.24	0.0124
Error	55	146.56	2.66		

that both the Leggot's spear and the mulch treatments resulted in significantly higher *A. mucronata* seedling establishment in comparison to all the other treatments (Fig. 6). The responses represented three-fold and five-fold increases, respectively, in mean seedling establishment in comparison to the control treatment (Fig. 5a).

The response of *A. verticillata* to treatments was similar to that of *A. mucronata*. With the exception of the Paper-glue treatment, the Leggot's spear and the mulch treatments resulted in significantly higher *A. verticillata* seedling establishment in comparison to the all other treatments. For these treatments, the responses represented three-fold and four-fold increases in mean seedling establishment, respectively (Fig. 5b).

L. scoparium

The total number of *L. scoparium* seedlings across all treatments decreased from 21 100 to 9 368 over the October to March sampling interval. The seedling establishment percentages calculated for *L. scoparium* in March were commensurately lower than those of October. The mean March seedling establishment percentages for the species ranged from a maximum of 3.2 +/- 2.3 (mulch) to a minimum of 1.4 +/-1.3% (control).

The establishment means for the species differed between treatments ($F_{5,55} = 3.24$, $p < 0.0124$) and between blocks ($F_{11,55} = 2.48$, $p < 0.0132$; Table 1). Both the paper glue and the mulch treatments provided significantly higher mean seedling establishment responses in comparison to the other treatments (Fig. 5b). The response to the paper glue and the mulch treatment represented three-fold and five-fold increases in mean

Figure 6: Photo of a mulch (M) plot 10 months after treatment

germination in comparison to the control treatment, respectively.

The cost of broadacre treatment

Cost comparisons indicated that the Leggot's spear treatment was the most economical means of establishing *Acacia* and *L. scoparium* (Table 3).

7.5 Discussion

In order to prevent plot to plot contamination by seed due to erosion, the design of this experiment required a field layout in which the plots within blocks were laid out along a contour. This layout may have reduced the spatial correlation normally expected between the soils of plots in blocked, field trial designs (Mulla *et al.*, 1990). In this trial, slope and surface material differences occurred along the contour. Consequently, the comparatively large standard deviations recorded in this trial are likely to be, in part, a consequence of the heterogeneity of the field site. In the mulch treatment, micro-topographic wind-shear differences between the plots may have compounded variability.

Some of the treatments applied in the field trials have been tested elsewhere. For example, Lyons (1995) used deep ripping and the Leggot's spear method to successfully sow approximately 20 hectares of silica scree following surface mining in the Mount Isa area. Mulching has only been applied to relatively small areas. With the exceptions of the examples from the sand mining industry given earlier (Bellairs *et al.*, 1995; Brooks and Bell, 1984) and reclamation works at Ashio (The Daily Yomiuri, 1993), the use of stabilizers/adherents in broadacre work is unusual. The

Table 3: The cost of broadacre application for four species using six broadcast-sowing seedbed treatments based on seedling establishment percentages and material costs. A nominal stem density of 5000 seedlings per hectare is used by way of example. The seedling establishment percentages were obtained in a field trial located at a near-smelter site on an eroded, 20° slope.

Species	Treatment	Mean seed estab. (%)	Seed[1] req. to provide 5000 stems ($ha^{-1})	Materials[2] ($ha^{-1})	Total cost of t'ment ($ha^{-1})
A. mucronata					
	control	4.6	180	n.a.	180
	Ad.1	6.2	135	270	405
	Ad.2	6.4	130	540	670
	Pg	6.4	130	57	187
	Leggots	13.4	62	n.a.	62
	mulch	21.7	38	645	683
A. melanoxylon					
	control	0.41	2150	n.a.	2150
	Ad.1	1.42	619	270	887
	Ad.2	0.41	2150	540	2690
	Pg	1.22	720	57	777
	Leggots	2.34	376	n.a.	376
	mulch	2.64	333	645	978
A. verticillata					
	control	7.58	171	n.a.	171
	Ad.1	12.6	103	270	373
	Ad.2	6.57	197	540	737
	Pg	18.7	69	57	126
	Leggots	23.8	55	n.a.	55
	mulch	29.3	44	645	649

continued

Table 3 continued

Species	Treatment	Seed estab. %	Seed[1] req. to provide 5000 stems (ha^{-1})	Materials[2] (ha^{-1})	Total cost of t'ment (ha^{-1})
L.scoparium					
	control	0.61	91	n.a.	91
	Ad.1	1.41	39	270	309
	Ad.2	1.62	34	540	574
	Pg	2.08	27	57	84
	Leggots	1.62	34	n.a.	34
	mulch	3.18	17	645	662

1 - The required number of stems per hectare were divided by the number of viable seeds per kilogram and multiplied by the seedling establishment percentage. The resultant 'number of kilograms of viable seed of a given provenance required to produce 5000 seedlings in the field for a particular treatment' was costed at the average seed cost of $100 kg^{-1}.

2 - The cost of material requirements per hectare were calculated as follows;

control:	n.a.
Adherent (Ad1):	30kg x $9kg^{-1} = $270 (J - Tac adherent)
Adherent (Ad2):	60kg x $9kg^{-1} = $540 (J - Tac adherent)
Paper glue (Pg) :	$52 (PVA); 2 straw bales ($2.50 bale^{-1}) =$5; newspaper = $0
Leggot's spear (L):	n.a.
mulch (M):	150 straw bales ($2.50 bale^{-1}) = $375; 30kg x $9kg^{-1} = $270 (J - Tac adherent)

author does not know of the use of J-tac™ outside of the landscaping industry. The paper-glue method is, to the author's knowledge, novel.

L. scoparium

Both the mulch (M) and the paper-glue (Pg) treatments resulted in significant increases in *L scoparium* seedling establishment in comparison to the control (C). The response is likely to be due to improved seed lodgement and more favourable seed microsites. After setting, the combination of straw and adherent in the mulch (M) treatment formed a rigid, surface-covering straw 'mat'. The mat provided protection from rain-splash, surface wash and environmental extremes. The artificial organic seedbed may have partially mimicked those supporting seedlings naturally.

In the paper-glue (Pg) treatment, shreds of paper with seed adhered to its surface stuck to the soil surface. This assisted seed lodgement. As the treatment added no bulk organic matter, however, it is less likely that *L. scoparium* seed in the Paper-glue (Pg) treatment benefited from improvements in the microsite environment. The treatment appeared well suited to the exceptionally small seed of the species.

The two treatments were, however, not ideal. Straw from the mulch (M) treatment and paper shreds from the Paper-glue (Pg) treatment were removed from sections of many plots by wind action. A number of the mulch (M) treatment plots lost up to half of their straw within the survey period. The microtopography of each plot appeared to be important in the retention or loss of treatment materials.

None of the remaining treatments provided an improvement in the seedling establishment percentage for this species. This may have been due to a dissipation of the binding agent in the absence of structural material or, in the case of the Leggot's spear treatment, dimensionally inappropriately microsites in relation to the size of the seed.

Acacia species

Both the mulch (M) and the Leggot' spear (L) treatments resulted in improved *A. mucronata* and *A. verticillata* seedling establishment percentages. In contrast, however, none of the treatments resulted in significant increases in seedling establishment for *A. melanoxylon* in comparison to the controls.

In the case of the mulch (M) treatment, the response is likely to be due to improved seed lodgement and more favourable seed microsites. Seedling losses due to low surface humidity and post-germination dessication may have been avoided. The comparatively large seed size of *Acacia* was apparently well suited to the dimensions of the cavities formed by the Leggot's spear (L) treatment. Seed washed into the cavities was back-filled by eroded mineral soil. This resulted in seed burial and provided a suitable germination environment.

The *Acacia* seedling establishment percentages achieved by the Leggot's spear (L) and mulch (M) treatments on steep slopes were comparable to those achieved by the same species in the mechanical rip and roller-aerate seedbed treatments evaluated under similar field conditions in Chapter 6.

Treatment costs

The cost of seed and materials to establish 5000 seedlings per hectare were calculated for each of the treatments under examination. The costing did not, however, include labour, as this varied with the site and resources available on site. The following discussion relates only to those species displaying a significant response to treatments in comparison to the controls. Non-significant treatments for example, such as the Leggot's spear treatment with *L. scoparium* cannot be considered reliable estimates of establishment costs.

The most cost-effective 'steep slopes' treatment for two of the three species of *Acacia* was the Leggot's spear (L) treatment. This was the result of relatively high seedling establishment percentages for each species with negligible material costs. Seed costs for the Leggot's spear treatment ranged from \$55 ha^{-1} to \$62 ha^{-1} for *A. verticillata* and *A. mucronata*, respectively. In comparison, the establishment of 5000 seedlings of each of the above species using the mulch (M) treatment cost \$649 and \$683 ha^{-1}, respectively. These treatments could only be justified in exceptional circumstances. For *A. melanoxylon*, the control treatment (C) resulted in excessively high establishment costs (\$2150 ha^{-1}). The high cost of establishing some species, such as *A. melanoxylon*, may best be overcome with tube stock.

The most cost-effective treatment for *L. scoparium* was the paper glue (Pg) treatment (\$84ha^{-1}). This treatment was much less expensive than the next most cost-effective, the mulch (M) treatment (\$662ha^{-1}). The treatment was, however, only marginally cheaper than control (C) treatment. If labour costs were added, it is doubtful whether this treatment could be justified. Year-round availability, and comparitively low seed collection costs, suggest that this species may be suited to aerial sowing with little or no ground preparation.

7.6 Conclusions

The difficulty of establishing vegetation by broadcast methods at degraded sites with seedbeds offering low surface stability and few germination opportunities is compounded by steep terrain and regions of high annual rainfall. Mechanical soil stabilization methods may be inappropriate. In this chapter, the germination and establishment of local, colonising species on steep, eroded slopes was compared to establishment following minimum-impact methods of seedbed preparation and seed application. The treatments included a nursery crop, mulching, two applications of a stabilizing agent, a seed-paper amalgam and manual working.

Three of the treatments, the paper-glue (Pg), the Leggot's spear (L) and the mulch (M) treatment, were beneficial to the establishment of one or more of the colonising species. The responses were understood to be the result of improved seed lodgement and moderated environmental extremes. Treatments offering the highest seedling establishment were not necessarily the most cost-effective.

Chapter 8

Conclusions

8.1 Research summary

Compositional trends and the edaphic environment

The alterations caused by copper mining and smelting at Mount Lyell have been catastrophic, resulting in a much simplified near-smelter environment that bares little resemblance to the original. Soil and biotic diversity, known and unknown, have been lost leaving exposed subsoils that are unfavourable to colonisation and growth. Although on-site smelting ceased in 1969, recovery appears artifically slow.

Massive and extensive soil losses typify Mount Lyell. Despite these losses, and the likelihood of contaminant redistribution, the present-day chemical characteristics of the Mount Lyell subsoils reflect a history of acid and metal-particulate deposition. Metal distribution patterns suggest Cu and Zn deposition and Al mobilization, with Cu concentrations elevated near the smelters but declining rapidly with increasing displacement. This type of concentration-displacement pattern mirrors those found in the vicinity of other base-metal smelters of world renown. However, similar comparisons suggest that contemporary Cu concentrations at Mount Lyell are an order of magnitude lower and far less extensive than those elsewhere.

Never the less, Cu and Al concentrations that exceeding those known to cause growth abnormalities in seedlings of woody plants were recorded in the vicinity of the smelters.

It has long been understood that plant species, and even genotypes, may differ in their susceptibility to particular forms of environmental stress, and may therefore exercise a different effect upon vegetation composition (Grime, 1979). A consequence of this is that vegetation development may be inhibited by environmental stress (Grime, 1979). While the causes of environmental stress, defined as external constraints which limit plant growth, are commonly complex due to the manifold and inter-active nature of the environment, residual soil contamination resulting from the deposition of smelter emissions creates extra-ordinary stresses upon plants colonising disturbed areas.

At Mount Lyell, the closure of the smelters presented an opportunity to pre-adapted, colonising species. However, residual, and spatially discrete, metal contamination has contributed to environmental stress, and this has dominated the pattern of natural colonisation and vegetation development by providing a strong, selective gradient. This is reflected in the formation of distinct, zonal vegetation groups by colonising species of varying metal tolerance. Typically, the composition of these groups was aberrant and lacking in species richness. Vegetation zonation was understood to be the result of spatial shifts in the dominance of tolerant species. It is believed that shifts in dominance will continue to occur, albeit artificially slowly, with time.

Rehabilitation methods

Seedling growth, in pot and field trials, measured plant tolerance to soil metals along a concentration gradient. In general, tolerance varied with displacement from the smelters. The trials indicated that severe phytotoxic symptoms in colonising species are linked to elevated soil Cu concentrations. Aluminium phytotoxicity, however, is unconfirmed.

Lime amendment was used in an attempt to ameliorate metal-related, soil phytotoxicity. In pot trials, seedlings of colonising species benefited with fewer toxicity symptoms and increased biomass. Field amendments are generally benificial, but hampered by the the limitations of surface applications. At Mount Lyell, neutralising amendments have potential for the control of pH and metal availability in the short-term.

Harsh seedbed conditions as found at Mount Lyell provide unfavourable circumstances for the germination and establishment of broadcast sown seed. Mechanical seedbed preparation improves seedling establishment for hard-seeded species by providing lodgement and favourable microsites. For these species, seedbed preparation assists the cost-effectiveness of broadcasting as a means of species introduction. Other species may not respond to mechanical seedbed preparation.

Steep terrain and low surface stability provides restricted opportunities for mechanical seedbed preparation. Minimum-impact methods of seedbed preparation and application differ in their effectiveness by species.

Treatments offering the highest establishment are not necessarily the most cost-effective as material costs vary markedly.

8.2 The future of restoration and neglect

The pre-European vegetation of Mount Lyell cannot be restored. Unlike many rehabilitation works, soil replacement, perhaps the easiest means of rehabilitation, and regaining diversity, is not feasible due to prohibitive costs.

If Mount Lyell cannot be restored, what is its future? What would be the outcome of neglect? This thesis has provided some of the answers. Natural colonisation over the past 20 years has resulted in the re-invasion of a limited number of higher plant species. The occurrence of vascular plants in the new community appears to be largely a result of varying resistance to soil Cu. Varying resistance to Cu is primarily responsible for the formation of distinct, spatially-related vegetation groups. These floristically-depauperate groups, provide partial cover.

Over time, the presence of the new community offers a potential for improved site quality. Nutrients and organic matter should accumulate to form soils. Soil formation should reduce phytotoxicity and contribute to the re-establishment of ecosystem function. Such improvements in site quality are likely to permit invasion by less tolerant species. Growth rates should increase commensurately. However, significant improvements to site quality, sufficient to markedly increase species richness, abundance and productivity, are unlikely in the near-future. Natural processes, such as the reversal of acidification and improved soil-buffering capacity, may require many tens of decades.

The vegetation zone occupied by the *Agrostis* grassland group is a case in point. Neglected, the residual phytotoxicity is likely to maintain a zone of depauperate vegetation well into the next century.

8.3 A strategy for rehabilitation

Although it has not been the objective of this thesis to explore every known rehabilitation method - though more than reported here have been the subject of preliminary trials - I wish to provide an example of a an appropriate rehabilitation strategy for Mount Lyell. It is not the only suitable strategy. It should also be noted that any prescription must be location appropriate. The zones identified in the vegetation classification provide a starting point. Again, I will refer to the *Agrostis* grassland zone.

Most of the *Agrostis* grassland zone is relatively steep, erodible and phytotoxic. The zone can be subdivided topographically into ridges, mid-slopes and drainage lines. Rehabilitation of the ridge-lines would, with time, provide a source of propagules and organic matter to the mid-slope areas. The rehabilitation of these relatively small areas would contribute to on-going site amelioration and lower-slope colonisation.

Rehabilitation of the ridgelines could be achieved in a number of ways.

Two examples are given. The ridgelines at Mount Lyell permit mechanical access for seedbed preparation and broadcast sowing. Only resistant species with seed available in quantity should be selected. Mechanical seedbed preparation should aim to provide seed micro-sites without sacrificing surface stability. Elsewhere, a low-impact method of seed bed preparation could be used on inaccessible terrain. The application of a soil-neutralising agent would assist the establishment of seedlings of resistant, colonising species. *Acacia mucronata* provides a good example of a nitrogen fixing, resistant species with a life-span measured in decades. Subsequently, the appropriate application of fertilizers might prove beneficial. However, extreme caution must be used in order to avoid the leaching of nutrients and downstream pollution. Aerial applications of fertilizers are not recommended.

Alternatively, the ridgelines could be ripped and planted to stock of resistant species. In conjunction with rabbit control, survival rates could be expected to be high. At planting, nutrients as pellets could be applied sub-aerially. Tree guards and weed mats are not considered necessary.

Other methods of seedling establishment are expected to have limited application. Sub-soiling might provide a means of gaining rapid growth, in especially unfavourable sites. The method, however, would require good machine access and care would be needed to avoid the importation of weeds. The application of sewage should be investigated. The cost of soil importation, even in relatively small quantities, can be expected to be high. Terracing and bagging with a soil/seed mixture might prove appropriate in the steepest and most inaccessible of sites. However, the results of a preliminary, terracing trial at Mount Lyell were not encouraging.

The tolerant species, *Agrostis capillaris*, should be encouraged to further colonise the mid-slopes. Observations of fenced plots suggest that this could be achieved by reducing browsing through rabbit control. Fencing may assist. Further work is needed to establish the conditions required for field germination of this species under harsh or contaminated conditions.
Drainage lines in the zone are being adequately colonised by the rush, *Restio tetraphyllus*. There are some examples where rush colonies have facilitated later introductions. Attempts to germinate and transplant (open-rooted) this species have failed.

In 1993, mismanagement, community attitudes and political gain combined to stop all rehabilitation work at Mount Lyell. I hope this thesis provides inspiration to those that follow.

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Appendices

Appendix 1: The Superworm Soil Aerator

The Superworm Soil Aerator is manufactured by M and M Hedgus Pty. Ltd., General Engineers, 8-10 Forest Street, Colac, Victoria 3250. The manufacturers claims for the Superworm are improved soil aeration, relief of soil compaction, improved pasture root systems and internal drainage, reduced reaction time of applied lime and fertilizer while retaining soil structure; all without creating erosion channels or disturbing roots and rocks. Reconditioned roller-aerators sell for approximately $5 000 to $6 000.

Appendix 2: J - Tac^R

J-Tac^R was supplied by Field Air (Ballarat) Pty.Ltd. under the product name Plantac. It is made by the Reclamare Company, Seattle,WA. as an organic hydrocolloid. Its makers say that it is a "complex formulation of high quality polysaccharides and other linear polymers of high molecular weight". Among its claimed properties are a mulch, seed and soil stabilizer.

Appendix 3: 'Steep slopes' treatment specifications

NB. All treatments received ground calcitic lime at the rate of 4000 $kgha^{-1}$ prior to seedbed preparation and seed application.

1) Cover crop (CC)

Seedbed preparation and seed application

The seed mix was broadcast sown. The soil was surface sprayed with binder (J-Tac[R]) mixed at 3 gL^{-1} and applied at the rate of 1 Lm^{-2}. One half of the binder by volume was applied to the soil surface prior to sowing and the remainder applied immediately after sowing.

Seed mix

species	sowing rate ($kgha^{-1}$)
Medicago sativa cv. alsike	20
Lolium perenne cv. concord	20
Agrostis capillaris cv. Mt.Lyell	1.75

2) Leggot's spear (L)

Seedbed preparation and seed application

Twenty four evenly spaced soil surface 'perforations' (2.5x5 cm deep) per m^2 were made manually using a Leggot's spear. The seed mix was broadcast sown.

Seed mix:

Species	Sowing rate ($kgha^{-1}$)
A. melanoxylon	80
A. mucronata	60
A. verticillata	47.5
L. scoparium	82.5

3) Adherent 1 (A1)

Seedbed preparation and seed application

The soil surface was sprayed with binder mixed at 3 gl^{-1} and applied at the rate of 1 Lm^{-2}. One half of the binder by volume was applied to the soil surface prior to sowing. The seed mix was then broadcast sown. The remaining binder was applied immediately after sowing.

Seed mix: as for #2

4) Adherent 2 (A2)

Seedbed preparation and seed application

The soil surface was sprayed with binder mixed at 3 gL^{-1} and applied at the rate of 2 Lm^{-2}. One half of the binder by volume was applied to the soil surface prior to sowing. The seed mix was then broadcast sown. The remaining binder was applied immediately after sowing.

Seed mix: as for #2

5) Mulch (M)

Seedbed preparation and seed application

The soil surface was sprayed with binder mixed at 3 gL^{-1} and applied at the rate of 1 Lm^{-2}. One half of the binder by volume was applied to the soil surface prior to sowing. The seed mix was then broadcast sown and the soil surface spread with straw (2000 $kgha^{-1}$ or 150 square bales). The remaining binder was applied after mulching.

Seed mix: as for #2

6) Paper glue amalgam (Pg)

Seedbed preparation and seed application

Seed mix and finely chopped straw (2 cm long) sprayed with dilute PVA wood glue (2.5:1 PVA to water) on a backing of newspaper. The seed/straw mix was allowed to dry and lightly broken up to permit even spreading during broadcast sowing. The soil surface was sprayed with binder mixed at 3 gL^{-1} and applied at the rate of 1 Lm^{-2}. One half of the binder by volume was applied to the soil surface prior to sowing. The seed mix was then broadcast sown. The remaining binder was applied immediately after sowing.

Seed mix: as for #2

7) Control (C)

Seedbed preparation and seed application

Seed mix broadcast sown on unprepared ground.

Seed mix : as for #2