CHAPTER 1: INTRODUCTION

1.1 Open–cut Mining in Australia and the Impacts on the Environment

Australia is rich in bauxite, coal, diamonds, gold, iron ore, nickel, uranium, and other minerals. As the third largest mining country of the world, Australian income from the mining industry has been more than $500 billion in the past 20 years (Mineral Council of Australia 2002). Australia is the fifth largest producer of coal and the largest exporter in the world. Black and brown coals provide around 85 percent of Australian electricity and 40 percent of total energy needs. Over 400 medium to large mining operations are under way across the country and provide nearly 50,000 direct full time jobs (Mineral Council of Australia, 2002).

There are two major types of mining: underground and open-cut. Underground mining in Australia is done by the bord and pillar or longwall method. Its application is limited in Australia, particularly for minerals with large quantities and close to the surface. Open-cut mining is an operation that removes the top cover above the mineral layer to win the resources then transport to processing or export ports. However, the impact of open cut mining on the environment is usually greater than underground mining, principally because of the area of surface disturbance.

Open-cut mining degrades the ecosystems by the removal of all soil, plants and animals (Cooke, 2002). Waste production and its disposal may cause long-lasting disturbances to the land and generate consequent impacts on the atmospheric, soil, aquatic and social environment (Farrell & Kratzing 1996). Open-cut mining usually disturbs large areas that are generally used for the disposal of wastes, and this could result in alteration in topography, variation in soil chemistry and the creation of barren landscapes. Mining activities can also pollute surface and ground waters (Ferguson & Erickson 1988; Smith & Barton Bridges 1991; Governor & Secretary 1998).

Open-cut mining disturbances are generally temporary and the influences are generally at a local to regional scale. Although the scale is relatively small on a regional base, the effect is usually large and often serious. The entire removal of topsoil, plants and animals completely destroys the natural ecosystem, and has long-lasting effects on future ecosystem development (Fox, 1990). It completely arrests both physical and biological processes on the land surface and below ground and resets the ecosystem 'clock' so that all soil processes and all flora and fauna required for ecosystem functioning have to make a new start.

Early mining activities had little regard for the environment (Farrell & Kratzing 1996), and this practice resulted in many abandoned mine sites through out Australia needing rehabilitation (DEST, 1996). Apart from ecological considerations of mining impacts on land degradation, society intends to make tradeoffs between sustainable use of natural resources and the economics of ecosystem rehabilitation. This may require consideration of:

1) What is the desired ecosystem after mining land use and how might it be achieved?

2) Will the desired end land use be ecological sustainable?

3) What kinds of rehabilitation methods are reliable to achieve the final goal?

4) Can we predict and model the potential rehabilitation trajectory to that goal?

Unfortunately, there are few long-term rehabilitation studies that demonstrate the development of mature sustainable ecosystem, particularly natural systems. Consequently feasible methods need to be developed to validate them.

Some general vegetation rehabilitation models derived from natural successions can be helpful to represent the initial floristic composition as a result of immediate regeneration after disturbance (Collins *et al.* 1995), and to explain the role of vital attributes within plant life histories to predict the eventual dominant composition (Walker & Moral 2003). However, there is also evidence that the results of post-mining processing are site specific in response to previous mining conditions and the environment in which mining occurs (Morin & Hutt 1999). Consequently the main problem is the difficulty of finding a model that can be widely used for different situations in mining rehabilitation. Traditional analytical tools such as statistical models are commonly used in studies of ecological dynamics to describe stepwise situations. The disadvantage of this approach is that it depends on the sampling of factor states, rather than factor behaviour that drives the process, thereby limiting the extrapolation of these models to other cases. Statistical models, however, are satisfactory to describe rehabilitation states and transitions compared with undisturbed ecosystems nearby.

In contrast to traditional methods, process based and simulation models tend to be more sophisticated and perhaps more appropriate methods. They try to represent the rehabilitation process as a result of interdependence and interaction of major members involved in the ecosystem. As a result, these models have a capability of predicting the future development as well as reproducing the past processes. An Agent-based Model (Franklin & Graesser 1997) is a spatial process-based object oriented simulation model that attempts to derive the future condition from the behaviour of individuals and describes the system dynamics based on the contribution of lower level activities. The potential of this method is to simulate complex system processes more reasonably, which is often more difficult to achieve by applying traditional methods.

Measurements and observations of rehabilitation and recolonisation after open-cut mining have been undertaken at Boggabri, New South Wales, Australia, over the last 20 years where a trial box-cut was made in 1979. The results from this series of studies are used to form the basis of predicting future ecosystem development using an Agent-based simulation model with the model outcomes being compared to reference or analogue sites in the surrounding forest.

1.2 Objectives of the Thesis

To address the questions raised above and to solve associated methodological problems, the objectives of this thesis can be stated as follows:

- To investigate the natural colonisation of native species and the dynamic interactions between individuals in relation to the developing ecosystems from 1981 to 2002 at the Boggabri, mine site in NSW.

- To develop an Agent-based model and present the results spatially by linking the Agent-based Model with a Geographic Information System (GIS).

- To simulate the natural colonisation of native species and ecosystem development using the developed Agent-based Model.

- To demonstrate the potential application of the model in rehabilitation management and how it might be used to provide management advice for decision-making on appropriate rehabilitation strategies.

1.3 Outline of the Thesis

The contents of each chapter are listed below.

Chapter 2 provides a general review of rehabilitation and vegetation colonisation.

Chapter 3 describes the Boggabri mine site (soil, climate, and vegetation) and the experimental design and methodologies used in previous studies to generate the results used in this thesis.

Chapter 4 provides the results of earlier studies over the last 20 years and explains the statistical and GIS spatial analyses used to describe the seed banks and its impact on the initial floristic composition, the role of the surrounding community on rehabilitation states and transition, and the environmental effects on rehabilitation.

Chapter 5 provides a review of models, and describes an Agent-based Model outlining why it was an appropriate option to be used, how it worked and what it could provide.

Chapter 6 explains how the Agent-based Model was developed and the parameters used for Boggabri.

Chapter 7 discusses the results of natural regeneration at Boggabri by using the model. Results are presented on sensitivity analyses of the model by varying parameter dimensions using data collected from the last 20 years of field observations.

Chapter 8 presents the potential application of the model to mine rehabilitation by assisting natural regeneration, using different management strategies (variation in seeds mixes and different types of soil).

Chapter 9 presents the overall conclusions of this thesis.

CHAPTER 2: MINE REHABILITATION AND PLANT COMMUNITY DEVELOPMENT – A REVIEW

2.1 Introduction

An ecosystem is a construct that is formed by biotic and abiotic components and their interactions, and refers primarily to processes and functions (Vogt 1997). Ecosystems change spatially and temporally as a result of natural processes, e.g. geographic variation, climate change and particularly human activity during the last century (SER, 2002). The unreasonable overuse of natural resources causes ecosystem degradation that may become unsuitable for human activity as well as other biological communities. As summarised by Daily (Daily, 1995), more than 20×10^8 ha land (approximately 17% of the terrestrial vegetated surface of the earth) are under a process of degradation as a result of overgrazing, deforestation, over-exploitation and industrial use. Amongst the many different types of causes leading to degradation, surface mining activities always result in complete land degradation by the combined effects of devegetation and soil destruction. Although land disturbed by mining is relatively small in proportion to the globally degraded land, it can cause considerable ecological problems locally where mining is a major industry (Cooke, 2002).

As a result of land degradation ecosystems suffer from a reduction in biodiversity and productivity, a decline in soil and habitat quality, and changes in abiotic and biotic interactions (Chapman, 1992; Daily, 1995). The fact that people have to face is that all of these changes, mostly caused by human disturbances, will ultimately affect human life. Currently, the concept of sustainable ecosystem management is well understood and the importance of ecosystem rehabilitation has been widely realised and accepted.

2.2 Mine Land Degradation and Rehabilitation

2.2.1 Mine Land Degradation

Many natural and artificial factors can cause land and ecosystem degradation, among them mining is a relatively small proportion of globally degraded land, but it still covers a considerable area in some countries, such as 3.7 million ha in U.S.A. and 2 million ha in China (Cooke 2002). In Australia, thousands of mined sites are abandoned (DEST, 1996) and are waiting to be restored.

Australia has large deposits of bauxite, coal, diamonds, gold, iron ore, nickel, uranium, and other minerals. Since it is the major steaming coal export country (O' Reilly 1994) and the third largest mineral producer in the world, the mining industry holds an important position in the national economy (Mineral Council of Australia 2002). Mining occurs throughout the country and across a wide range of ecosystems from the tropics to temperate and arid environments (Fig. 2.1).

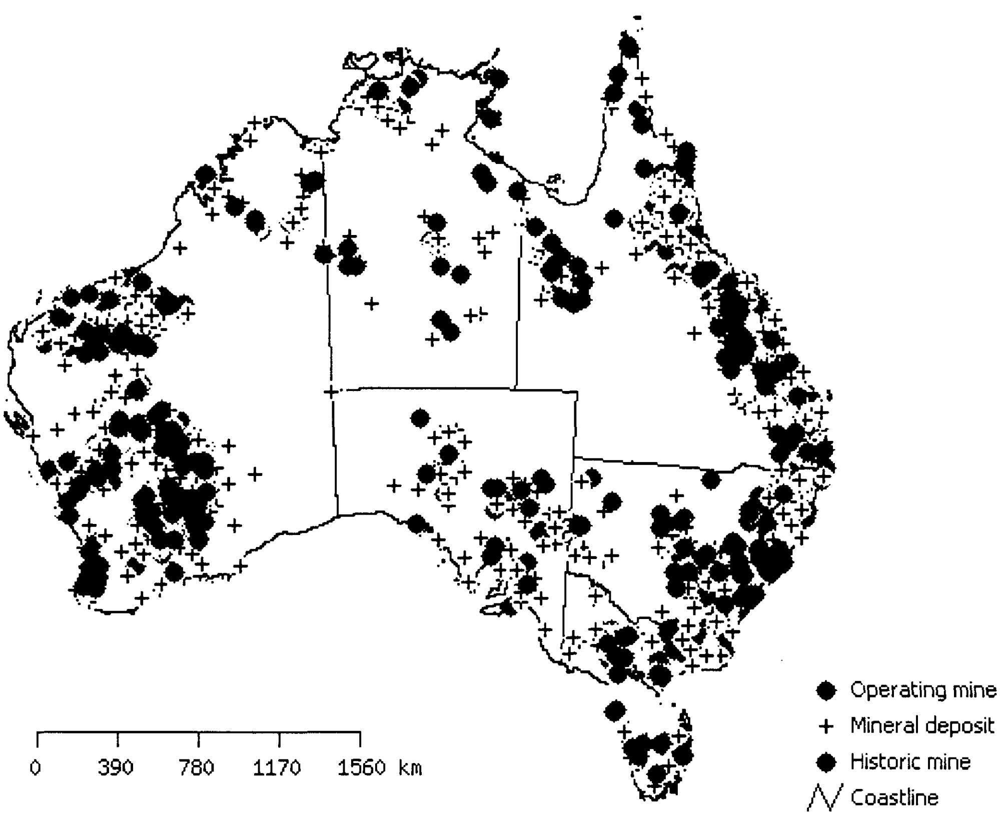


Fig. 2.1 Major mineral locations in Australia (Geoscience Australia 2006)

Although only 0.01% of landmass has been affected by the mining industry in Australia, compared to over 70% of landmass being impacted by agricultural activities (Mineral Council of Australia 2002), the destruction of the existing vegetation and soil profiles at mine sites has a significant effect on the local environment.

There are two major kinds of mining methods: underground and open-cut. Underground mining is mainly used for mineral resources deep under the earth surface. Open-cut mining operates on the earth surface, removing the cover over the mineral seams. Open-cut mining is usually economical in large-scale operations where the resource is close to the surface. However, the environmental impact of surface mining is generally more significant than underground mining. The removal of land cover destroys ecosystem structure and function (Cooke 2002). Waste production and its disposal, including overburden cause long-lasting disturbances by changing the topography, presenting potentially toxic materials at the surface, thereby polluting surface or ground water (Ferguson & Erickson 1988). Therefore, governments and the public require sites to be rehabilitated to environmentally sustainable ecosystems that serve a purpose to society (Mineral Council of Australia 2002).

Ecosystem degradation varies according to the type of disturbance and its intensity, and the location in which it occurs. Therefore, a variety of objectives and techniques are required for rehabilitation that focuses on recovering ecosystem structure and function (Bradshaw *et al.* 1982, Bradshaw 1984, 1993; Pratt & Stevens 1992).

2.2.2 Mine Sites Rehabilitation

Rehabilitation is the process of repairing a damaged ecosystem towards either the original state or an alternative ecosystem associated with the original (Bradshaw 1984, 1990). Thus, rehabilitation is considered by Clewell *et al.* (SER 2002) to emphasise processes, productivity and services (i.e. ecosystem functions). Cook and Johnson (2002) regarded rehabilitation after mining more as ecosystem reconstruction that requires the initiation of primary succession (van Andel *et al.* 1993). In this thesis, the concept of rehabilitation means to repair both the structure and function of an ecosystem, and to bring about a “desirable ecosystem” to ensure that it can be self-sustaining.

The mining industry in Australia is committed to a safe, clean operation that maintains sustainable environments both during and after mining (Minerals Council of Australia, 2003). The Australia Minerals Industry Code for Environmental Management was launched in December 1996, to set standards for the management of the environment, and ensure ecological sustainable development. Consequently, mining companies have calculated the cost of rehabilitation and pollution control into operating costs and, therefore, are interested in any possible approach to achieve rehabilitation goals with lower inputs.

Cooke (2002) and Johnson *et al.* (1994) outlined a general procedure for mining rehabilitation (Fig.2.2). A common rehabilitation method used on mine sites is to reshape topography and to cover the dumped materials with topsoil and sow plant seeds. It is expected that this method will promote vegetation regeneration quickly to meet the requirements of ecosystem rehabilitation at a relative early stage. Revegetation is essential and monitoring of the rehabilitating and developing ecosystem is needed to ensure the final end point is achieved.

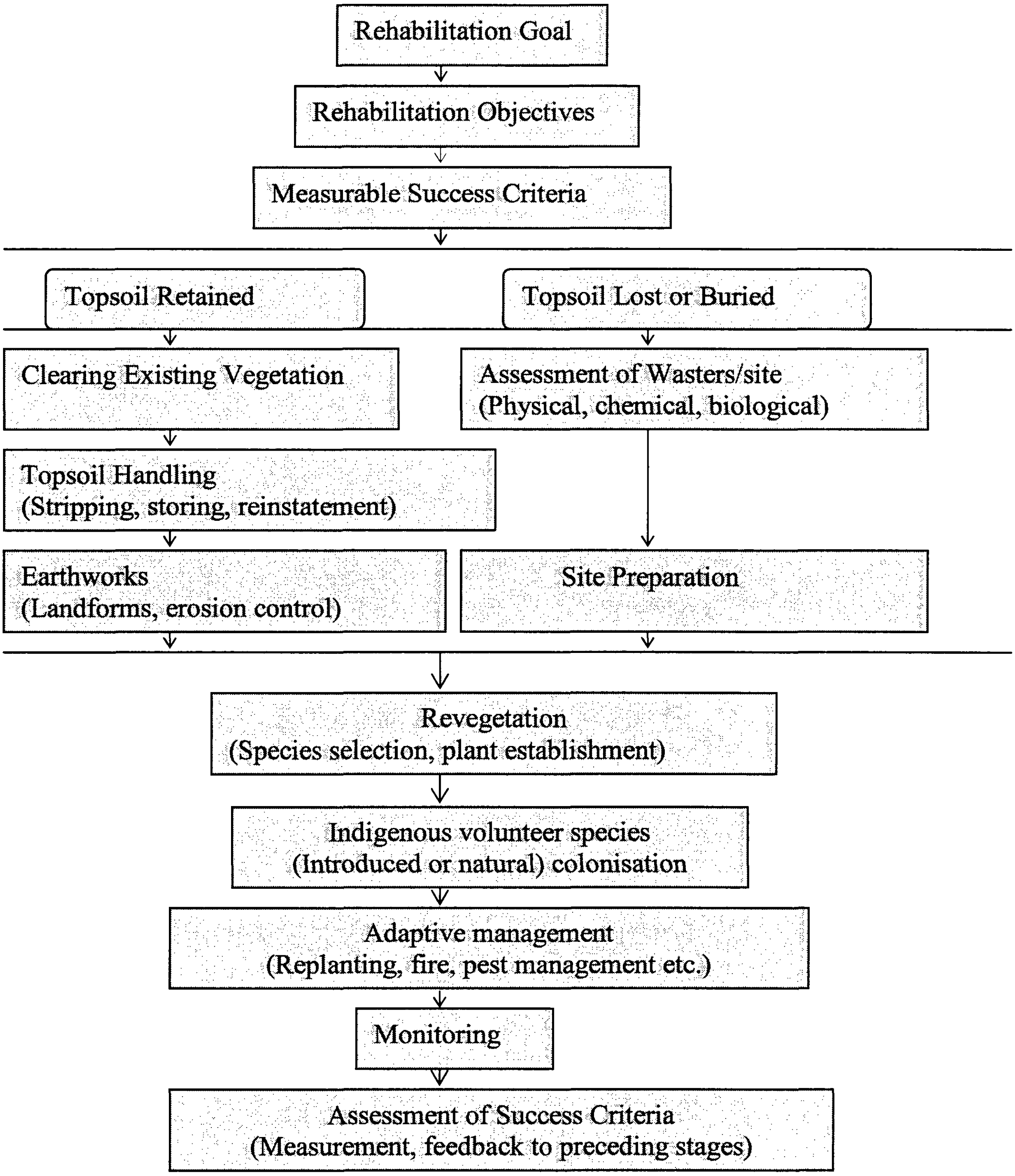


Fig. 2.2 Stages of ecological rehabilitation of mined land (after Cooke 2002, Johnson *et al.* 1994)

The process of mine rehabilitation always takes a considerably long period of time. In addition, rehabilitation is often affected by many other ecological, economical and social factors, which results in uncertainties in the process. It is feasible to set up reference ecosystems in nearby undisturbed ecosystems against which to assess the revegetation progress.

2.3 Species Colonisation and Community Succession

2.3.1 Colonisation

The process of revegetating bare land involves the dynamics of species colonisation and vegetation succession.

Colonisation is used to describe the process of species shift from a distance place to an area and settle there where their species is bare or not exist yet. It describes the ability and establishment of different species on a site from invasion to establishment and persistence (Schmid M. 2006; Tabor *et al.* 2007). Colonisation is based on the theory of island biogeography (MacArthur & Wilson 1967, 2001) and predicts the number of species that occur in an area (an island) on the basis of the distance from the source (mainland) and the size of the area (island).

When land is disturbed and vegetation removed, the area can be rapidly colonised by species that then modify one or more environmental factors (Krebs 1972, Schnitzler & Closset 2003).

This modification provides the chance for other species to colonise with the community moving along a development trajectory. Propagule dispersal characteristics, species life history and reproduction, and species-specific environmental requirements determine the success and the patterns of colonisation (Tabor *et al.* 2007; Snäll *et al.* 2003; Bampfylde *et al.* 2005).

2.3.2 Community Succession

Colonisation and succession are two important processes in vegetation development. Krebs (1972) considered succession to be a directional change in vegetation communities over time, which describes the interactions of species within the community and the development of a community trajectory. Luken (1990) proposed that disturbance, colonisation and species performance are the principal factors for succession. Species performance is related to propagule establishment, persistence, reproduction and reaction to environmental factors.

Rehabilitation can be considered to be the practice of ecosystem repair and a part of vegetation succession (Hobbs & Harris 2001; Walker 2003), while succession pathway in undisturbed systems can be used as a reference for rehabilitation. Therefore, understanding the dynamics of succession is a major theoretical basis for rehabilitation. Understanding succession can be based on the mechanisms involved, the observed pattern of ecosystem change over time or from models (combining both mechanisms and changing patterns in ecosystems). McIntosh (1981) and Glen-Lewin *et al.* (1992) prepared reviews of the theories and mechanisms of vegetation succession. The earliest and widespread vegetation succession theory was by

Clements (1916, 1936) dealing with monoclimax or climatic-climax theory. Predictability, order, convergence and equilibrium of vegetation association development were the basis of this concept. Clements believed every region had only one climax community, and climate determined the final community. Tansley (1935) disagreed that climate was the only function explanation and considered there were many climax communities for a particular region. These different communities were controlled by other factors such as soil, moisture, topography etc., but, given enough time a single climax could be achieved. Tansley's polyclimax theory was an extension of the Clementsian paradigm. Whittaker (1953) proposed another succession hypothesis which was termed pattern climax. He recognized that the landscape contains environmental gradients, and that the stable type of vegetation that developed tended to change along those gradients. Species are replaced independently and the climax vegetation will have a spatial pattern and form a continuum. He supported Gleason's (1917) individualistic concept which was the community development depended completely on the phenomena of the individual species.

The pathways or pattern of ecosystem change over time, and models of succession have been studied for decades. Although Margalef (1958, 1963, and 1968) and Odum (1969) applied ecosystem parameters to explain the successional change towards maximum biomass and diversity, they still considered that succession was the result of interactions within the community. The external factor of disturbance, such as climate change and species invasion were assumed to be stable or played minor roles (Glenn-Lewin 1992), in contrast to the current belief that outside disturbance is an important and major factor in succession.

Competition and species replacement are major factors used to describe successional pathways. Comparing Clements' unique pathway of succession, Egler (1954) suggested that the 'initial floristic composition' following disturbance determined the future species that would dominate. Connell and Slatyer (1977) developed a model and described succession on the basis of three major pathways of "facilitation", "tolerance" and "inhibition". "Facilitation" (Choler *et al.* 2001) means that earlier colonisation by species made the environment more suitable for later species. "Tolerance" recognises that earlier associations have little impact on later colonising species, and that later species can colonise successfully, whether or not they invade later in time. The last pathway, "inhibition", describes the situation that species which have not successfully colonised the area in the early stages of succession will fail to establish later during succession due to stress generated from species which are already present. Sánchez – Velasquez (2003) simulate this mechanisms by models. In contrast, Osawa (1992) suggested another parallel pathway that succession was not caused by the difference between the earlier and following species, but due to the difference in density and growth. Watt (1947) also proposed a cyclical model of species replacement during succession after disturbance. Recent theories all consider succession as a complex process driven by both internal competition and external disturbances (Glenn-Lewin *et al.*1992), and is a non-equilibrium processes with multiple pathways (Miles 1987; Cattelino *et al.* 1979; Botkin 1979).

Noble and Slatyer (1981) summarised some special characteristics commonly existing in successional processes following disturbance as outlined below:

1) The initial floristic composition following disturbance is determined by species surviving the disturbance or from the soil seed bank, or seed rain from plants surrounding the disturbed site;

2) There is a high rate of immediate regeneration after disturbance;

3) After the initial species have established and occupied the area, recruitment rates tend to slow down after an initial phase of rapid germination;

4) The effect of pioneer colonisers may be favourable or unfavourable for succeeding species growth; and

5) Long-lived species which are able to grow to maturity will finally dominate the disturbed site.

2.4 From the Individual to the Community

Community succession consists of establishment, colonisation and persistence that take place at the population level (Arnold *et al.* 1992), while, colonisation occurs at the individual species level and drives community succession. Gross (1987) considers that, given the right conditions all species can be viewed as potential colonisers. Succession is considered as population–based phenomena (Glen-Lewin 1980; Glen-Lewin *et al.* 1992) that is influenced by population density, age distribution, sex ratio, spatial distribution, population growth, mortality etc. Gleason (1926) proposed population dynamics can be driven by individual competition for niches in a temporal and spatial scale, and introduced the 'individualistic concept' into the study of vegetation dynamics, in that individual species were suggested to completely determine the phenomena of vegetation dynamics (Gleason 1917). His concept provided new ideas for subsequent researchers in the study of models for succession dynamics.

The study of individual species and plants can be traced to the term of "gap" as defined by Watt (1947) in his book "Pattern and Process in the Plant Community". "Gap" refers to a position created by a dead individual for providing the recruitment probability of other individuals. This concept led to research on gap and patch dynamics and the modelling of vegetation succession (Botkin *et al.* 1972; Shugart & West 1977). For these studies, competition and disturbance were considered as internal and external causes and played key roles in understanding succession (Pickett *et al.* 1985; Whitemore 1989). Individual-based dynamics, such as the life span, individual growth, development and regeneration, inter-specific and intra-specific competition, and individual and environment interactions have formed the basis of individual-based models (Lömnicki 1999) (see Chapters 5 and 6).

2.5 Ecosystem Resilience and Rehabilitation

Resilience is the capability of an ecosystem to cope with disturbance without collapsing (Holling 1973). An ecosystem can be considered as a system with a complex structure which is capable of self-organization and self-generation to keep it in a metastable state than systems with a simpler structure (Holling 1996). Due to its more complex organization, ecosystems are able to maintain their structure and function to resist stress and disturbance, and also have the resilience to recover following disturbance (SER 2002; Johnson *et al.* 1996). Once disturbance intensity exceeds the tolerance limits of an ecosystem, recovery will become impossible without outside interference. The principle of resilience can be used for degradation assessment. In most cases, the ecosystem might move between different states as a balance between degradation and recovery progresses (Westoby *et al.* 1989; Drake 1990; Hobbs 1996). Hobbs (1996) proposed that there was a threshold between the transitions of different ecosystem states in relation to the recovery function of the ecosystem. When the intensity and frequency of disturbance is within the threshold, the system could shift between resilient states. When disturbance is removed, the ecosystem can recover and move towards the pre-disturbed state, otherwise, if the pressure of disturbance crosses the threshold, it would be difficult to recover due to the collapse of the original structure and function. Obviously, the ecosystem is always between the dynamic processes of degradation and recovery at certain times and under certain disturbance intensities, so that resilience is a measure of the ability to recover under different levels of pressure. The purpose of rehabilitation is to "develop practical and cost-effective methods to force those transitions, so that the degraded ecosystem can be forced towards the desirable state" (SER 2002). Therefore, resilience is a practical indicator to describe the response of ecosystems to various natural and human disturbances (Hobbs 1999). Moreover, it can identify the rehabilitation processes for different states and is able to indicate the transition between different states (such as grassland, shrubland, woodland, forest).

2.6 Research on Vegetation Colonisation and Succession on Mine Sites

The rehabilitation process on mine sites is a process of vegetation colonisation and succession. The main factors which have an impact on the success of colonisation and succession have been widely researched from the investigation of both experiments and monitoring. Bellairs and Bell (1993) compared regeneration patterns after mining and natural disturbance. They found that the major difference between mining and the natural disturbances is a reduction in the abundance of re-sprouting species as a source of plant propagules in the rehabilitated area due to mining removing the entire previous vegetation. Grant *et al.* (2001) developed a state-transition conceptual framework that started from the initial state and moved through a range of conditions to the desired state along the successional pathway. Koch and Ward (1994) showed that the rehabilitation pathway of a jarrah forest in mined areas differed from the forest response to fire using an Initial Floristic Composition Model (Purdie & Slatyer 1976). They found that species similarity between sites prior to mining and nine months after rehabilitation were much lower (20-50%) than reported after fire (89-96%). Meanwhile, there were still larger pre- and post-mining similarities within sites than between sites, which indicated the important role of topsoil seed stock in mining rehabilitation. Dahl and Mulligan (1996) found that at all age stages species richness was greater at rehabilitation than the undisturbed native forest. Indigenous species applied in mine site facilitated the natural succession (Ward *et al.* 1996). A rehabilitation program was undertaken according to the succession trajectory on notch Stradbroke Island in Queensland. After *Spinifex* grass as a natural pioneer species was established on the mine site, brush matting, cover crops and then a large amount of native trees were introduced into the site (Brooks & Yeates 1980). It finally formed a self-sustaining ecosystem.

Reshaping landscape such as internally drained or ponded methods, topsoil covered on the top and direct spreading mix of grass, shrub and tree species would promote plants survival and colonisation, and formed woodland in a short term (McNamara *et al.* 1999; Norman & Koch 2005; Wills & Read 2007). The topsoil introduced a seed bank for the species and density of plants establishing from the topsoil is greatly affected by stockpiling (Hannan, 1981). However, plant species selection is important for the rehabilitation, since different species have different effect on surface soil hydraulic conductivity particularly during the initial period of revegetation (Loch & Orange 1997).

2.7 Summary

Rehabilitation ecology, as a branch of applied ecology, is still a developing area. The rehabilitation goal can generally be achieved more efficiently using the basic principles developed for natural succession of vegetation. Rehabilitation following mining provides unique conditions characterised by the entire destruction of the local ecosystem. There is a wide range of choice for rehabilitation following mining, from low-cost natural re-colonisation to more intensive artificial promotion of species establishment.

Degradation and rehabilitation are reciprocal events taking place at the ecosystem level. Rehabilitation at mine sites is a dynamic process that requires different methods and flexible rehabilitation goals to match different sets of circumstances (Ehrenfeld 2000).

The severity of degradation and subsequent rehabilitation success can be measured as the differences between disturbed and undisturbed reference ecosystems in practice. For this reason the undisturbed reference community can be used as a baseline for rehabilitation but also to improve the understanding of both ecosystem and rehabilitation mechanisms.

CHAPTER 3: STUDY SITE

3.1 Study Site

The study site is located in Leard State Forest, on the North-Western Slopes of NSW, 18 km northeast of Boggabri (30°45′ S and 150°05′ E) and 40 km north of Gunnedah. It lies within the Gunnedah Basin (Fig. 3.1)

3.1.1 Geology and Topography of Leard State Forest

The geology of Leard State Forest consists of early Permian acid flows of Boggabri Volcanics as the basement overlain by similar-aged conglomerates, shales and sandstone of the Vickery Formation (Croft 1979). The Boggabri Volcanics are located in the southwest of the main forest area, but the Vickery Formation covers most of the area.

The Nandewar Range is located to the east of Leard State Forest. The main ridgeline in Leard State Forest forms a ‘u’ shape. West of the main ridge is steep, but other sections are flat to undulating. The study area occurs on the gentle lower slopes from the main ridge.

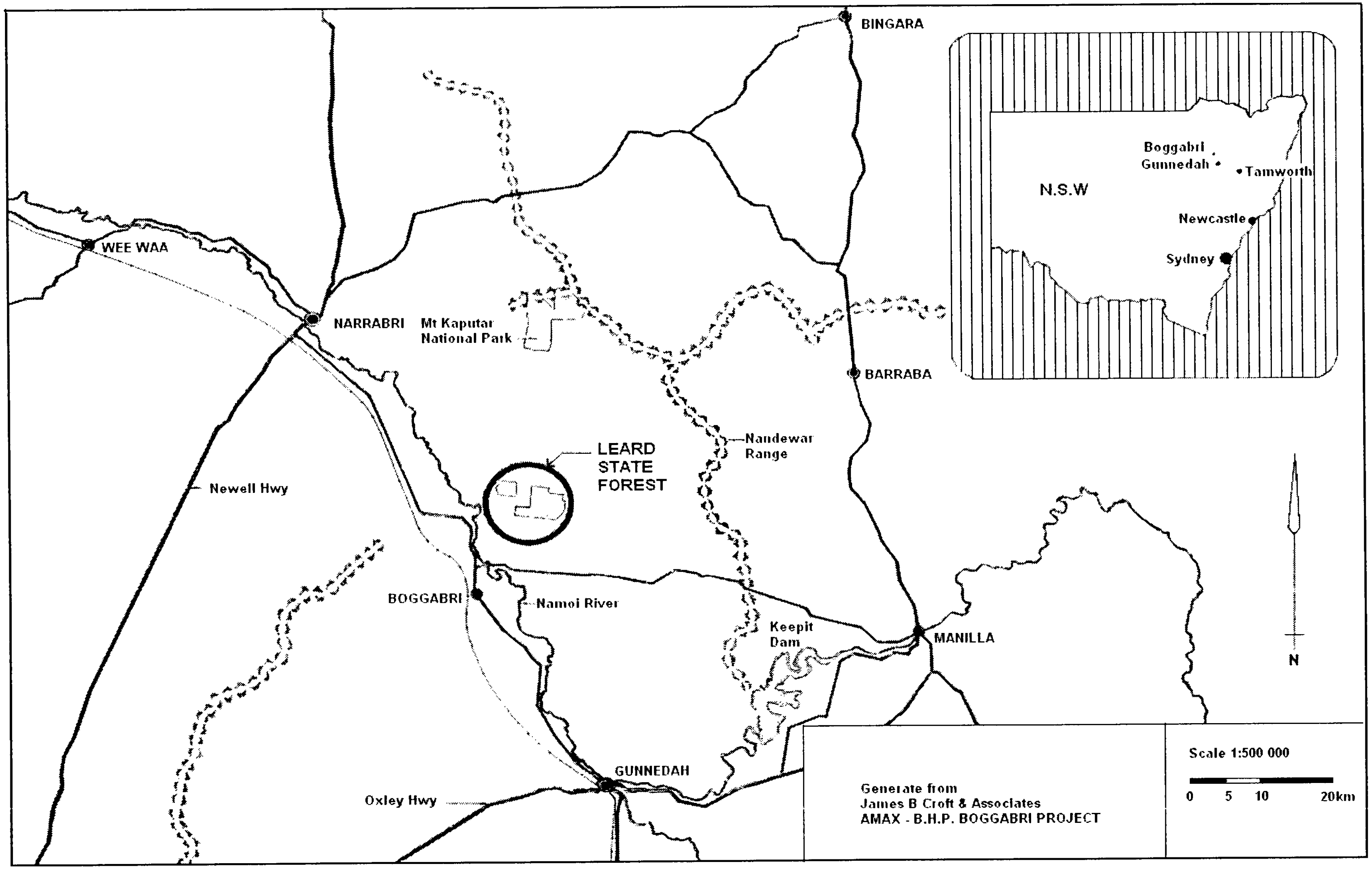


Fig. 3.1 Study site location, Boggabri, N.S.W. Australia

3.1.2 Climate

The study area falls within the subtropical climate with a summer rainfall regime (Bureau of Meteorology, Australia). The mean annual rainfall from 1973 to 2003 is 657 mm (Fig. 3.2). Rainfall occurs at any time of year, but mainly from October to February. January tends to be the hottest month with a mean monthly temperature of 14.6°C, and July is the coldest with mean monthly temperature of 0.3°C. Annual evaporation exceeds annual rainfall by almost 1,000 mm.

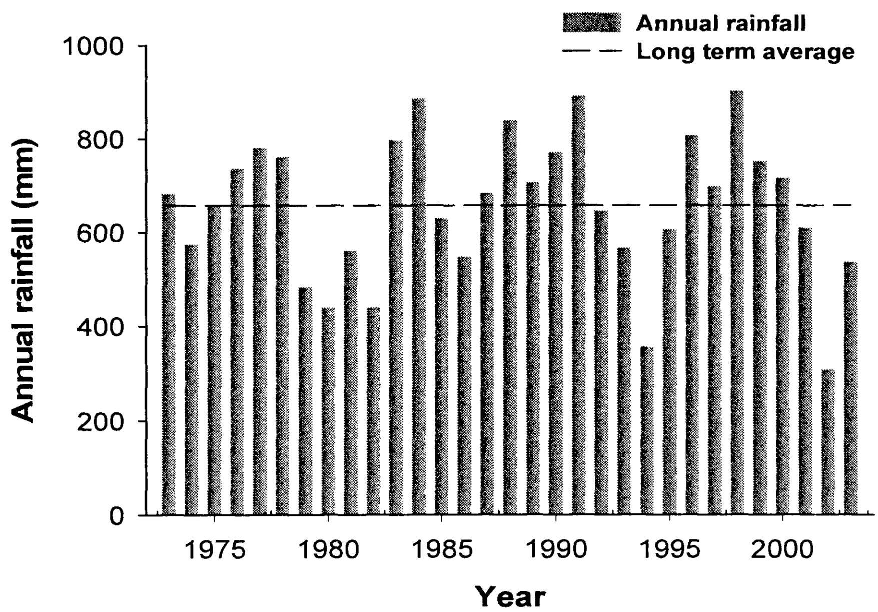


Fig.3.2 Annual rainfall and average rainfall from 1973 to 2003 at Boggabri, NSW, Australia

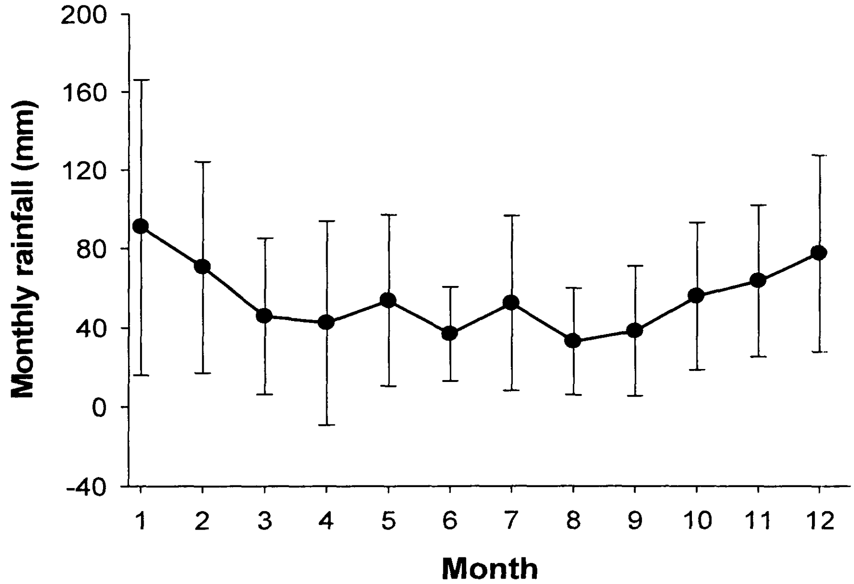


Fig. 3.3 Average monthly rainfall with error bars from 1973 to 2003 at Boggabri, NSW, Australia

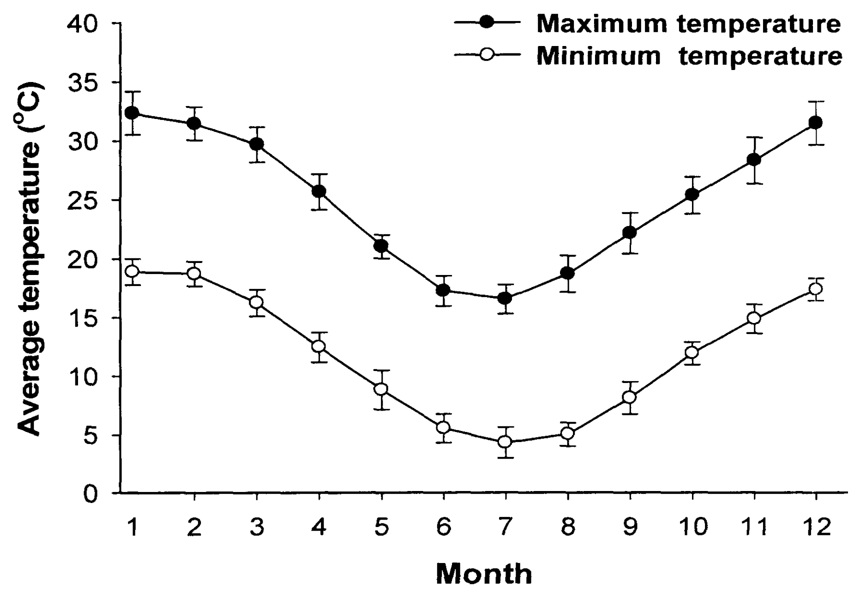


Fig. 3.4 Average monthly maximum and minimum temperature with error bars from 1973 to 2003 at Boggabri, NSW, Australia

3.1.3 Vegetation of Leard State Forest

The vegetation of Leard State Forest comprises dry sclerophyll open forest to woodland, dominated by mixed eucalypt/cypress pine communities. The dominant tree height varies from 10 to 30 meters, with a shrub layer of generally 2 to 3 m in height. The shrub layer in many areas of the forest is weak to absent. Dead timber is common on the ground layer (Croft 1979). The most dominant tree species are *Eucalyptus crebra* (Narrow-leaved Ironbark), *E. fibrosa* (Blue-leaf Ironbark), *E. albens* (White Box), *E. populnea* (Bimble Box), *E. pilligaensis* (Pilliga Box), *E. melanophloia* (Silver-leaved Ironbark), *Casuarina cristata*, *Callitris glaucaphylla* (White Cypress Pine) and *C. endlicheri* (Black Cypress Pine). Eucalypts have the greatest overall diversity in the area. *Acacia deanei*, *A. decora, Dodonaea viscosa*, *D. boroniifolia*, and *Senna nemophila* are common shrubs throughout the forest.

3.1.4 Mining History at the Study Site

The Gunnedah Basin is likely to develop as a major coal producing site in Australia. Mining has been undertaken within the Gunnedah Basin for a long time from the late 1800s initially using underground and but more recently open-cut extraction has been used. At the end of 1970s, a joint exploration program was undertaken by Amax Iron Ore Corporation and B.H.P. Ltd., which indicated the feasibility of an open-cut mine within Leard State Forest. A sample of about 100 tons of coal was extracted from a trial box-cut for market development purposes, and formed the case study area. Two spoil heaps covering 3.1 ha were formed from disposal of overburden (Wiram 1979; Grigg 1987; Duggin 1992) (Fig. 3.5). In 1996, a new road was established across the southern side, which divided the area into three sections.

Topsoil was spread onto the spoil heaps at an average depth of 10 cm, although, the actual depth varied from 0 to 20 cm and two patches were left with bare overburden. A fence was constructed around the research site to exclude kangaroos and wallabies from the area (Plates 3.1- 3.4 and Fig. 3.5).

The Department of Ecosystem Management at the University of New England, Australia, was commissioned by the Coal Division of Amax Iron Ore Corporation in 1980 to undertake studies on mine rehabilitation using these two spoil heaps (Grigg 1987).

Plate 3.1 Box-cut at Boggabri, NSW (Source: J.A.Duggin, 1981)

Plate 3.2 Study site established at Boggabri. NSW, photo showing from south to north (Source: J.A.Duggin, 1983)

Plate 3.3 Spoil heaps with bare overburden in South, photo showing from north. (Source: J.A.Duggin, 1981)

Plate 3.4 Spoil heaps with topsoil in North (Source: J.A.Duggin, 1981)

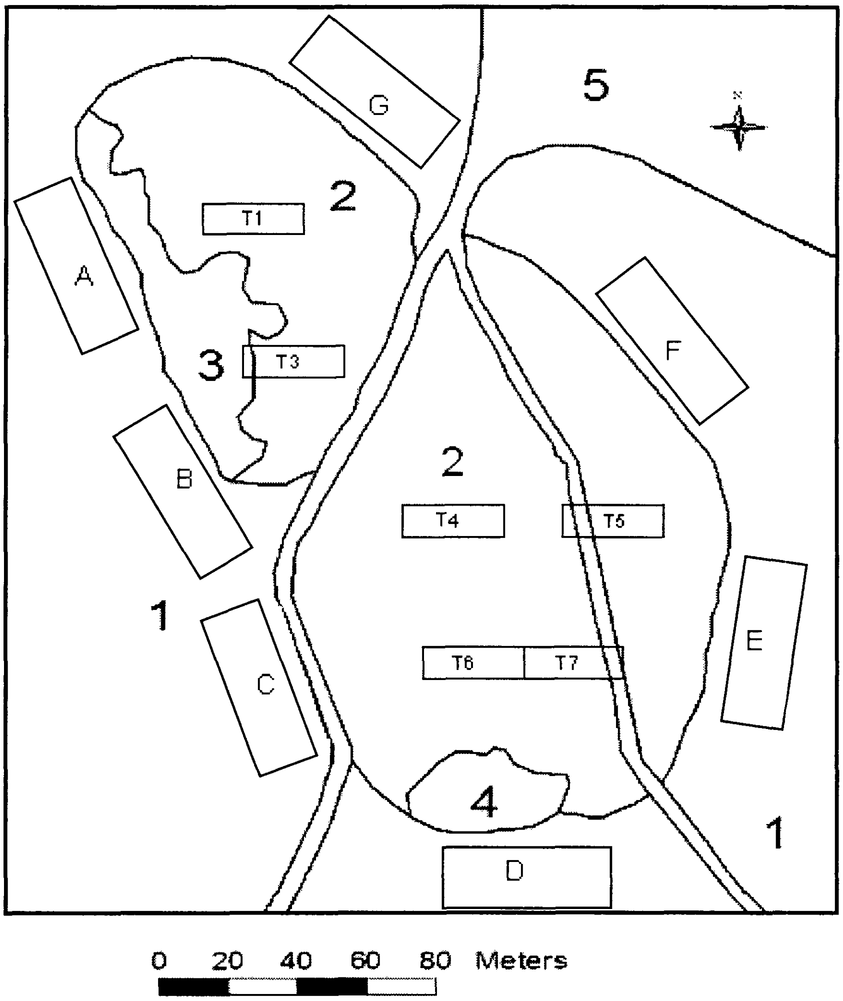
3.2 Experimental Design and Data Collection

3.2.1 Experimental Design

In April 1981, seven plots (10 × 30 m) were established to record the individual plant growth by species after the spoil heaps had been established for 20 months (Fig. 3.5). Unfortunately, later in 1982, plot 2 was destroyed due to the establishment of a eucalypt species trial by the Forestry Commission of NSW on behalf of Amax-B.H.P. to assess the relative suitability of different eucalypts for rehabilitation. The trial was withdrawn and all the species were cleared three years later. In 1996, plots 5 and 7 were partly destroyed due to the establishment of a new road across the spoil heaps, so these plots were only partly remeasured in 2002. The marker pegs for plot 1 and 3 were destroyed sometime after 1996, so the plots were omitted from the analysis after 1996.

An additional seven plots (20 × 50 m) were established in the natural forest surrounding the spoil heaps in 2002, and data were collected to describe forest structure and composition. Height, diameter, and stratum of individual trees and shrubs were recorded by species (Fig. 3.5).

Ecosystem process occurs in space and time. An appropriate scale is fundamental to address problems of succession in ecological research whether it is observation, experiment or modeling (Schneider 2001; Jenkins *et al.* 2007). The researches cover from fine scale to large scale (Getzin *et al* 2006; Larson & Franklin 2006), from 'logs to landscapes' (Lindenmayer & Franklin 2002). The selection of the scale size is determined by the structure and function of the ecosystem and objectives of the study. The experiment in this study area was over 20 years with a fine spatial scale. Dynamics of individual plants can be observed at the fine scale for the purpose of studying the colonisation process in such a small site.



Number 1	Natural forest
Numbers 2-4	Spoil heaps
	2: topsoil on overburden
	3: northern bare overburden
	4: southern bare overburden
Number 5	road and part of the box cut site to the north
T1 – T7,	Plots in spoil heaps for vegetation observation
A - G	Plots in natural forest

Fig. 3.5 Experimental design for data collection at Boggabri, NSW

3.2.2 Data Collection

Plants Data Collection

Cohorts, which were defined as a group of seedlings that have emerged in the one season (Falińska 1998), were used to record the emergence, growth, and mortality of all individuals for each species in the six plots from 1981 to 2002. From 1981 to 1983, the continuous observation of individual plants provided information on germination and early growth of each species. From 1983 to1996, data were collected periodically by different researchers. In 2002, height and diameter of all individual trees and shrubs in the plots were recorded by the author. The distribution of each individual plant in each plot was mapped at a scale of 1:100. Although the data has limitations due to the irregularity of surveys, it provided the basic information used to describe ecosystem development.

In addition to the plot data, all individual trees across the two spoil heaps were recorded by species, and their growth (height and diameter) were measured in 1987, 1988 and 1990 by different researchers, and again in 2002 by the author. The tree distribution was mapped in the earlier stages at a scale of 1:100 and later recorded by Global Positioning System (GPS) and then transferred into a Geographical Information System (GIS) for mapping in 2002.

In addition, seven plots were established in the natural forest in 2002 and the height of individual plants were measured by species, and diameter breast height was recorded when the plant diameter was greater than 10 cm.

Soil Data Collection

Soil samples were taken from both overburden areas and the surrounding natural forest in 2002. Samples were collected from 0-10 cm and 10-20 cm layer from each plot and were immediately air-dried after collection. Soil moisture was measured. Soil Electrical Conductivity (EC) was measured using a YSI 30 conductivity meter and pH was measured in 1:5 water suspensions using a Metrohm 744 pH meter. Soil organic matter was estimated using the loss-on-ignition technique (Howard & Howard 1990).

Climate Data

Monthly rainfall from 1902 to 2002 was obtained from the Australian Bureau of Meteorology.

Literature Data Collection

Data on the dominant trees and shrubs was collected to provide information on life histories necessary for understanding the colonisation process and model development. In addition, earlier study on the mine site in Leard State Forest were used to provide additional information. Wiram (1979) undertook a preliminary soils-overburden study at Leard State Forest including the study site. The soil profiles are duplex in nature and consist of a sandy loam A horizon with a sub-soil ranging from sandy clay loam to sand clays. The B horizon is potentially sodic.

Grigg (1987) studied the community structure and composition that has regenerated on the spoils heaps. Changes in the community over time were also investigated regularly. The data indicated that shrub species dominated all areas covered by topsoil, while tree species tended to be restricted to overburden areas. Grigg considered that shrubs and trees might follow different but interacting recolonisation pathways. A successional sequence was also suggested by Grigg that shrubs on topsoil areas might inhibit the return to dominance of trees, but the bare overburden areas provide the chance for tree establishment. In addition, Croft (1979) considered that *C. glaucophylla* was overall the most widespread species in Leard State Forest, and that the forest tended to be dominated by *C. glaucophylla* based on qualitative and statistical analyses, although regeneration is relatively slow.

CHAPTER 4: NATURAL COLONISATION AND SUCCESSION ON OVERBURDEN HEAPS

4.1 Introduction

Mining operations have a significant impact on the local environment (Bell 2001), particularly the process of surface mining and subsequent overburden heap development resulting from removal of all original biotic material and redestruction of soil profiles (Hatton & West 1987). This often results in relatively homogenous regeneration conditions on which primary succession may be initiated (van Andel *et al.* 1993). It is required by legislation and society that operators rehabilitate mine sites during operations and certainly before relinquishment (Mineral Council of Australia 2002). However, rehabilitation is a complex process and there are many problems that need to be considered, such as interaction between species and the newly-created environment, and competition among and between species particularly related to vegetation succession to achieve the desired endpoint (Connell *et al.* 1987; Connell & Slatyer 1977; Egler 1954; McCook 1994). Practical considerations, such as species selection, seed mixes, topographic landscaping, and topsoil management, are mostly addressed by case studies and professional experience, often supported by limited levels of applied trials. Rehabilitation results are often not satisfactory due to the choice of methods and the difficulty of predicting the potential vegetation succession trajectory. Even when the same management methods are applied in similar sites or on the same sites under different weather conditions the outcomes can be very different. When successful establishment of vegetation has been achieved, uncertainties still remain in regard to the future development of the community. Research and management tools need to be developed to address this question.

This chapter describes the natural colonisation process comprising establishment and persistence observed at Boggabri from 1980 to 2002. The composition and structural change of the vegetation community during the process of natural recovery, the impact of the topsoil on the revegetation, primary floristic composition are presented to provide a baseline for establishing a rehabilitation model (Chapters 5 and 6), which can then be used to simulate mining rehabilitation according to a range of different management strategies.

4.2 Methods

The population dynamics for each species over space and time were analysed using STATISTICA6.0 (StatSoft 2001). A vegetation stratum was used to analyse the height class structure.

ARC/INFO (ESRI 2006) was used to digitise the spoil heap maps. Road, natural forest, spoil heaps and different soil types were also mapped. Individual tree location was recorded using GPS in 2002 and then converted to GIS format. Zones were used (10 m intervals) from the edge of spoil heaps inward to the centre of the study site to display and calculate seedling emergence rates. Spatial analysis was used to summarise the species distribution. A raster GIS was used to describe the height class of tree species populations.

4.3 Vegetation Colonisation

4.3.1 Species Invasion and Establishment

Seed Banks, Seed Rain and Root Suckers

The topsoil coverage introduced a seed bank and stimulated the successful shrub invasion in the study site. Most shrub seeds, such as *Acacia deanei, Dodonaea viscosa,* and *Senna nemophila* were maintained in the topsoil and germinated in the first three years after the site was established in 1979.

In the first survey in April of 1981, 20 months after spoil heap formation, the total density of shrubs was 656 plants/ha. Three species, *A. deanei*, *S. nemophila* and *D. viscosa,* made up almost 80% of the total number of plants. *Senna nemophila* tended to cluster in several locations whereas the other species were widely distributed across the site. The immediate recruitment of shrubs after disturbance in 1981 was high, and dropped slightly in 1982 and 1983. Such a pulse of recruitment immediately after disturbance rehabilitation is not unusual (Noble & Slatyer 1981). From 1987, the newly established seedlings of shrubs increased in number gradually over the years. *Senna nemophila* is given as an example with *A. deanei* and *D. viscosa* showing a similar recruitment pattern (Fig. 4.1).

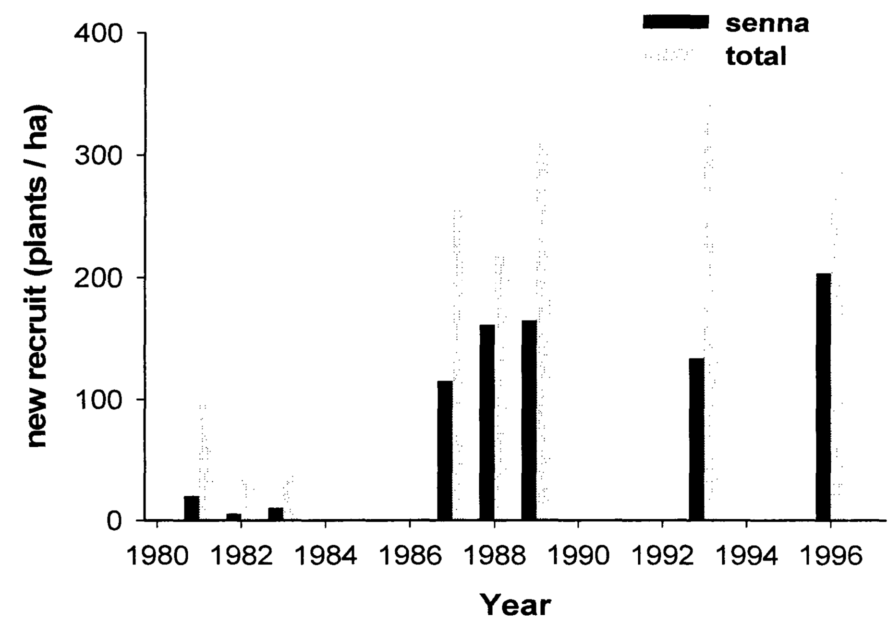


Fig. 4.1 Number of new shrubs recruit at each measurement year at Boggabri, NSW (SENNA = *S. nemophila,* TOTAL = total new shrubs)

Compared with the recruitment of shrubs, the emergence of tree species was very slow. In 1987, the first tree species were recorded with a few eucalypts and *C. cristata* seedlings noted close to the edge of the spoil heaps near the natural forest but the density declined apparently with increasing distance from the edge. Since most tree seeds lose viability after being buried in the soil for a period of time (Luzuriaga *et al.* 2004), together with a higher density of seedlings along the edge than inside of emergence pattern, it was judged that the seeds were derived from nearby natural forest.

In addition to the seed bank and seed rain, some species can recruit as root or stem suckers. In the field, some plants of *Senna* bush were found to be suckering from roots, as they were observed to grow in lines radiating from older plants (Fig. 4.2). *Dodonaea viscosa* also produced suckers, but clustered around the stem base rather than from roots. Boland *et al.* (1984) and Cunningham *et al.* (1992) suggested that recruitment of *C. cristata* was mainly from root suckers. However, in the spoil heaps according to our observations, *C. cristata* established mainly as seedlings and were distributed randomly around the edge of the spoil heaps, which was similar to that found by Chesterfield and Parsons (1985). In addition, as a pioneer species after disturbance, *C. cristata* did not show a significant population at Boggabri. Most of the *C. cristata* parent trees were at a distance from the edge of spoil heaps so seed dispersal may be the reason. Chesterfield and Parsons (1985) also found that *Casuarina* seeds germinated readily in the laboratory but suggested that ant harvesting accounts for the loss of a significant proportion of seeds in the field.

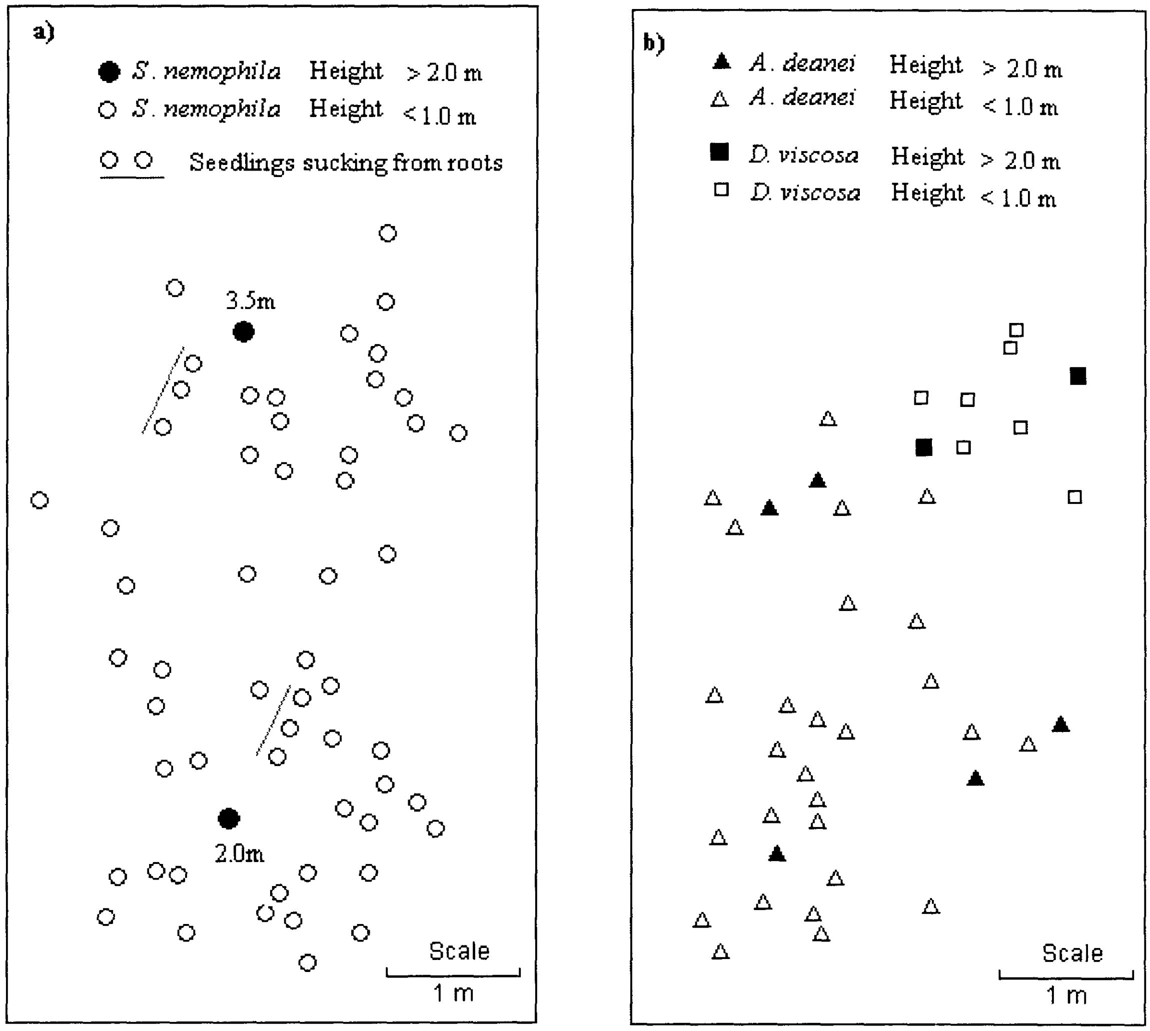


Fig. 4.2 Seedlings distribution around the seeding shrubs in selected plots in 2002. (a) *S. nemophila* in plot 6 (b) *A. deanei* and *D. viscosa* in plot 4 (after Grigg 1987).

4.3.2 Community Development and Persistence

Species population dynamics forms and shapes community composition and structure and drives successional processes. Population abundance (N) at a particular point in time is determined by four demographical processes: birth (B), death (D), immigration (I), and emigration (E) (Falińska 1998)

$$N_{t+1} = N_t + B - D + I - E$$

Where t is time and $t+1$ is a time interval.

However, changes in the population of one species influence the abundance of other populations, and together they describe the compositional changes of a community. Species richness is expressed as the number of species and diversity is an index combining richness and abundance of each species. Both are likely to affect community stability and resilience.

Seed Germination and Soil Types

Soil provides moisture and nutrients for plants, and different soil types have various impacts on species germination and plant growth.

In a laboratory experiment, Grigg (1987) showed that 76% of *E. crebra* germinated on northern bare overburden but only 35% on southern bare overburden, while a 61% germination occurred when overburden was covered with topsoil. Germination of *A. deanei* was 15% on southern bare overburden, 49% in topsoil area and 55% on northern bare overburden. However, *C. cristata* had nearly 37% germination rate on both bare northern overburden and topsoil areas but with only 19.8% survival on southern bare overburden. Denham (1989) observed *C. glaucophylla* germination was not affected by soil types, and the survival rate of *C. glaucophylla* seedlings was high at about 86% with temperatures of 5-

15°C, and 90% with temperatures of 10-20°C. Denham also found that *C. glaucophylla* was not inhibited by the presence of topsoil from both the result of germination and early growth.

Gouvernet (1981) studied the overburden profile and carried out a lysimeter experiment to measure sodium, potassium, calcium and magnesium from different types of soil in the study site and surrounding nature forest. He indicated that bare overburden, particularly the southern bare overburden, released higher quantities of calcium than undisturbed and stockpiled topsoil, the stockpiled topsoil initially released higher quantities of calcium than the natural forest site, but it decreased with time and was the same as undisturbed topsoil.

In 2002, soil samples were taken (Fig. 3.5) and pH, electrical conductivity (EC) and Organic Matter were measured (Table 4.1). The results were compared with that from Gouvernet (1981). By 2002, except the natural forest, pH had increased on the study site compared with that in 1981. The pH on the southern bare overburden had increased from 8.1 to 9.0. EC was significantly different on bare overburden from topsoiled area, being much higher on the southern bare overburden than the other site. Organic Matter on bare overburden was also lower than the topsoiled area. The results suggest that the nutrients in bare overburden were low and it had salinity problems.

Table 4.1 Changes in Different Soil Type at Boggabri, NSW (1981 data from Gouvernet (1981))

Site	pH		EC (mS/cm)	Organic Matter %
	1981	2002	2002	2002
Natural Forest	5.7	5.7	35.6	4.7
Topsoiled overburden	5.7	6.4	49.2	5.1
Northern bare overburden	6.2	7.9	144	4.1
Southern bare overburden	8.1	9.0	587	3.9

Soil moisture is one of the most important factors for seed germination. Therefore rainfall in combination with soil moisture holding capacity is important. *Casuarina cristata* can germinate after adequate rain at any time of the year. *Callitris glaucophylla* is observed to have a high germination rate in the field after high winter rainfall from February to July in 1989 and 1990, resulting in the establishment of 24 plants/ha and 28 plants/ha emerging respectively.

Seed Production and Dispersal

The quantity of seeds brought in by topsoil was finite, and the time that seeds of different species remain viable in the soil is another limiting factor in providing adequate seeds for rehabilitation. Sowing seeds mixes is the most common method used in rehabilitation

(Hannan 1995). However, natural regeneration at this study site depended on dispersal from the adjacent forest, in addition to seeds persisting in the topsoil and then on seeds produced by mature plants. *Senna* and *Dodonaea* are capable of flowering and setting viable seed within three years after germination (Hodgkinson & Griffin 1982), while for *Acacia* it is between 8 to 10 years (Hall *et al.* 1972; New 1984). Good seed years for *C. glaucophylla* are around 12 years (Lacey 1973), although seed production occurs about every three years. Eucalypts generally produce abundant seed in around 15 years with good production every two years (Jacobs 1955). Some of the earlier invading individuals of eucalypts produced seeds which have then dispersed across the spoil heaps. Seed pods of *Acacia* and *Senna* were found in the field in 2002 (but also seen earlier too, see Fig. 4.2). Some trees also had started to produce seeds. Two *C. glaucophylla* were found with fruits on branches, and there were five to ten *C. glaucophylla* seedlings with height of 20 cm around a larger individual. Some individual eucalypt capsules were observed on branches or on the ground around trees. This clearly indicates that besides the seed bank in the topsoil and seed rain from the natural forest, individual plants established in the spoil heaps were producing seeds, and it is assumed that recruitment can occur without further assistance from an external seed source.

Seed production and dispersal distance directly influenced the revegetation of spoil heaps. For wind-dispersed seeds the distance of dispersal is related to wind speed, height of individual trees, density of cover around the seeding tree, seed size and weight, and even special dispersal structures on seeds. Eucalypt capsules and *C. glaucophylla* cones were found on the ground within close proximity to parent plants.

Most shrubs, such as *S. nemophila, A. deanei and D. viscosa,* drop seeds directly to the ground within 5 m (Hodgkinson *et al.* 1980). Figure 4.2 illustrates the shrub seed dispersals and the seedling distribution pattern of different shrubs in the study site. The shrub seedlings emerged in a cluster with a high density around the parent shrub. For instance, there were more than 20 seedlings within 1 m around a large *Acacia* and more than 20 seedlings within 1 m around one mature *Senna* bush in plot 6, with an average height of 10 cm.

4.3.2.1 Changes in Community Composition

4.3.2.1a Re-colonisation of Shrubs

According to the data recorded in 1981, the study area was covered with grass with isolated and scattered shrub seedlings with an average shrub height of 6 cm. *Acacia deanei*, *D. viscosa*, *S. nemophila* were the earliest species observed at the spoil heaps (Fosdick 1981). They distributed across where topsoil had been spread over the overburden, except for *S. nemophila* which was only found in cluster on the southern side.

Acacia deanei, S. nemophila, and *D. viscosa* were the three main shrubs species occurring on spoil heaps. *Acacia deanei* is a fast growing shrub and early coloniser in secondary succession. It can grow up to 7 m (Harden 2000) and can have a 20 year life span. *Senna nemophila* is a medium shrub of 1-3 m (Cunningham *et al.* 1981), and can germinate at any time of the year but with high seedling mortality and a life span of about ten years. Large populations of *Senna* are mostly observed as dominants in shrub communities on flat areas with sandy soils or on stony hillsides (Randell 1970), and 'tends to grow in inter-tree areas' (Harrington *et al.* 1979). *Dodonaea viscosa* is a tall shrub, reaching a height of up to 8 m

(Harden 2000), and can live for 20 years. In the spoil heaps, it can grow to over 5 m, usually it occupies the open sites between tree canopies in dense to open woodland (Cunningham 1992).

In the early stage of colonisation individual shrubs tended to be randomly distributed but later in clusters around parent plants with a high density in topsoiled overburden areas. By 1993, shrubs colonised most of the topsoiled overburden area. The shrub density increased with time (Fig. 4.3). In 1981, the density of all shrubs seedlings was 567 plants/ha on the northern topsoiled area and 500 plants/ha in the southern topsoiled side, and the average height was 6 cm. In the north, about half of the seedlings were *A. deanei* with *D. viscosa* having the second largest population. Only four plants of *S. nemophila* were found on the northern side and they died by 1982. On the southern spoil heap *A. deanei* also had a larger population, followed by *S. nemophila* and *D. viscosa*. Each population changed over time and increased gradually, peaking in 1993, thereafter declining as recruitment was less than mortality.

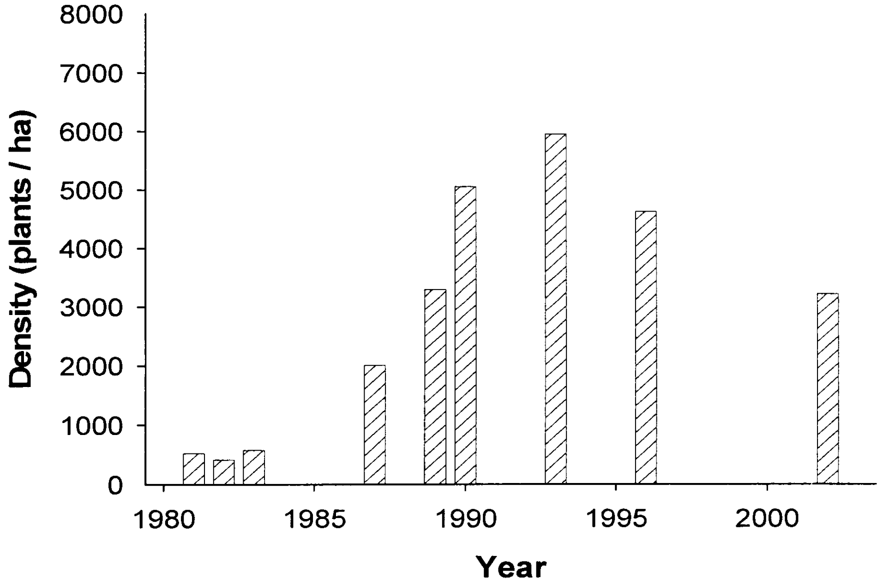


Fig. 4.3 Shrub density (plants/ha) at each measurement year in study site, Boggabri, NSW

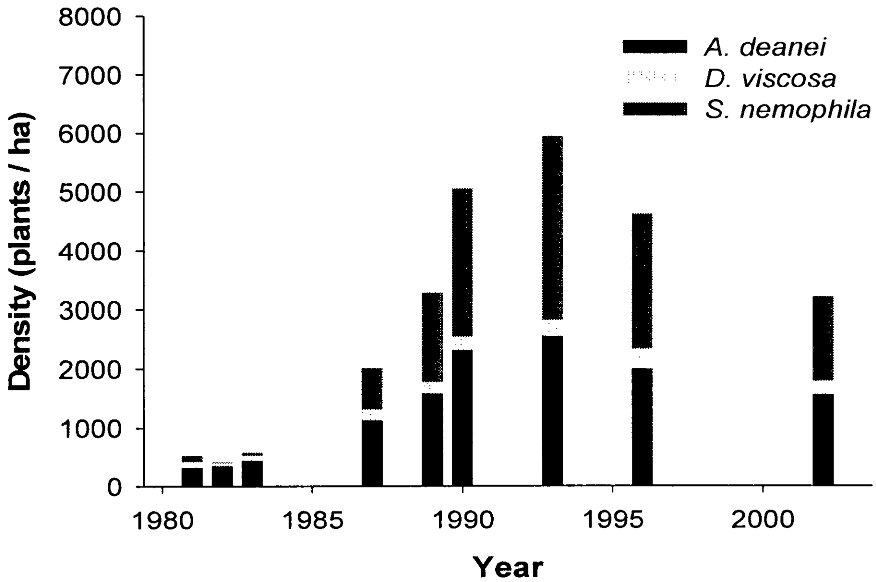


Fig. 4.4 Density (plants/ha) of three major perennial shrubs on spoil heaps for each measurement year in study site, Boggabri, NSW

Fig. 4.4 shows population density changes of the three major shrubs over the time. All species declined after 1993, but it is most significant for *A. deanei*.

The distribution of these three major shrubs in the study site was not uniform and varied according to location as is evidenced by individual plot data (Fig. 4.5). Shrubs did not establish in areas where topsoil was not spread and grew poorly with low survival where the topsoil was thinly spread.

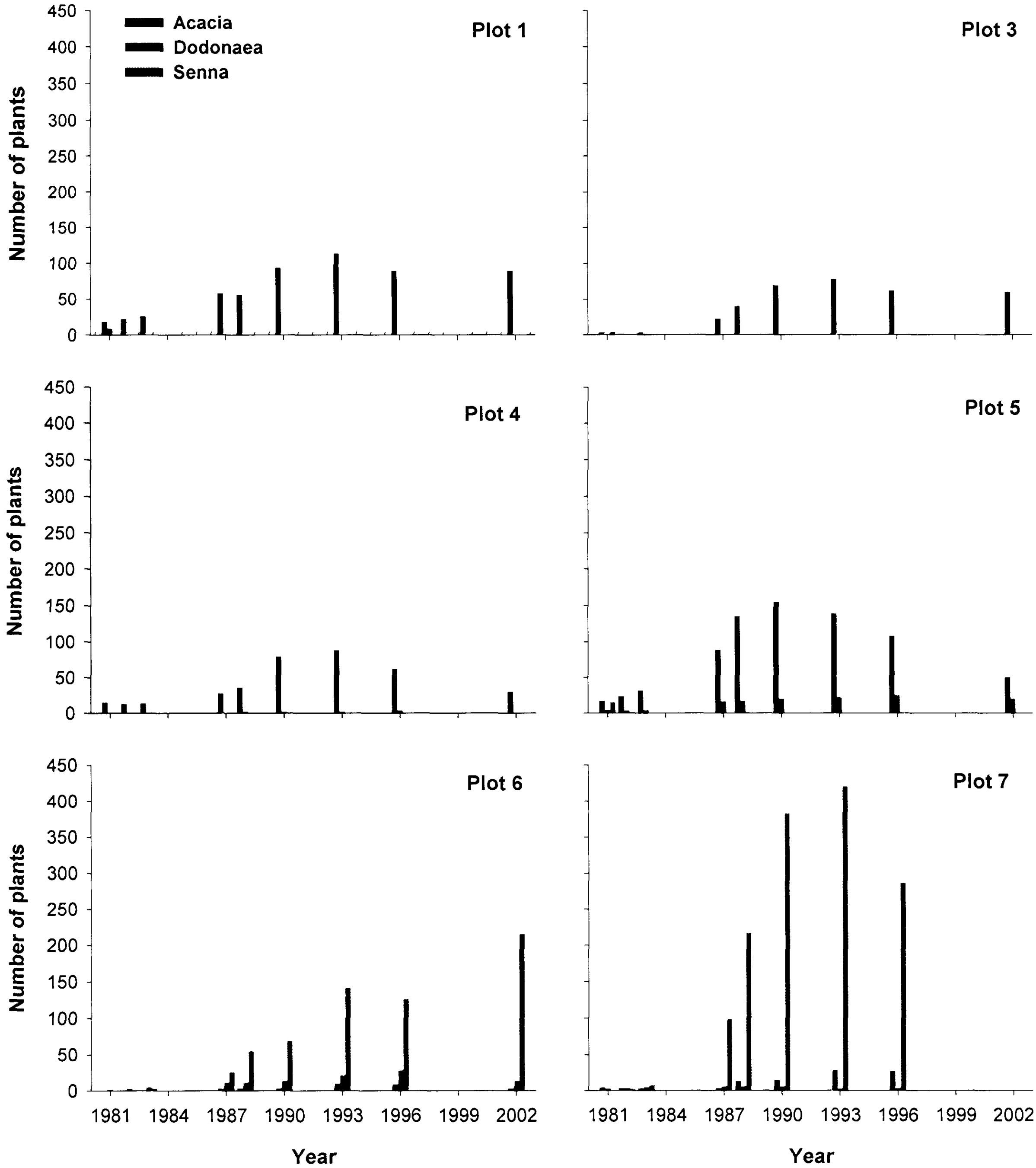


Fig. 4.5 Shrubs number change in responding to different investigation years in different plots (see Fig 3.5). *Acacia: A. deanei; Dodonaea: D. viscose; Senna: S. nemophila.*

4.3.2.1b Re-colonisation of Tree Species

Natural recovery by native tree species on coal-mine overburden is usually slow (Burns *et al.* 1981). Seed source, early growth, survival, mortality, insects, animals, suitable climate, soil moisture and competition can all affect the colonisation rates of trees. The major tree species that established in the study sites were *E. albens, E. pilligaensis, E. crebra, E. populnea, C. glaucophylla,* and *C. cristata*.

Tree Populations Distribution and Variation

Trees were not recorded growing in the study site from 1981 to 1986. In 1987, 105 individual tree seedlings of eucalypt and a few *C. cristata* were found and recorded by Grigg (1987), with a density averaging 35 plants/ha. Since there was a gap in the investigations from 1984 to 1986, and the height of some eucalypts was over 1 m, it was likely that some of them might have emerged during the period from 1984 to 1987. In 1987, five tree species were recorded and they were mainly distributed on the northern bare overburden area (Fig. 4.6 a). In 1988, another tree species, *C. glaucophylla* was found growing on the northern bare overburden. Figure 4.6 illustrates the distribution of each species and Table 4.2 lists changes in number in different soil types over time. The percentage of each tree species as part of the total population of tree is shown in Figure 4.7.

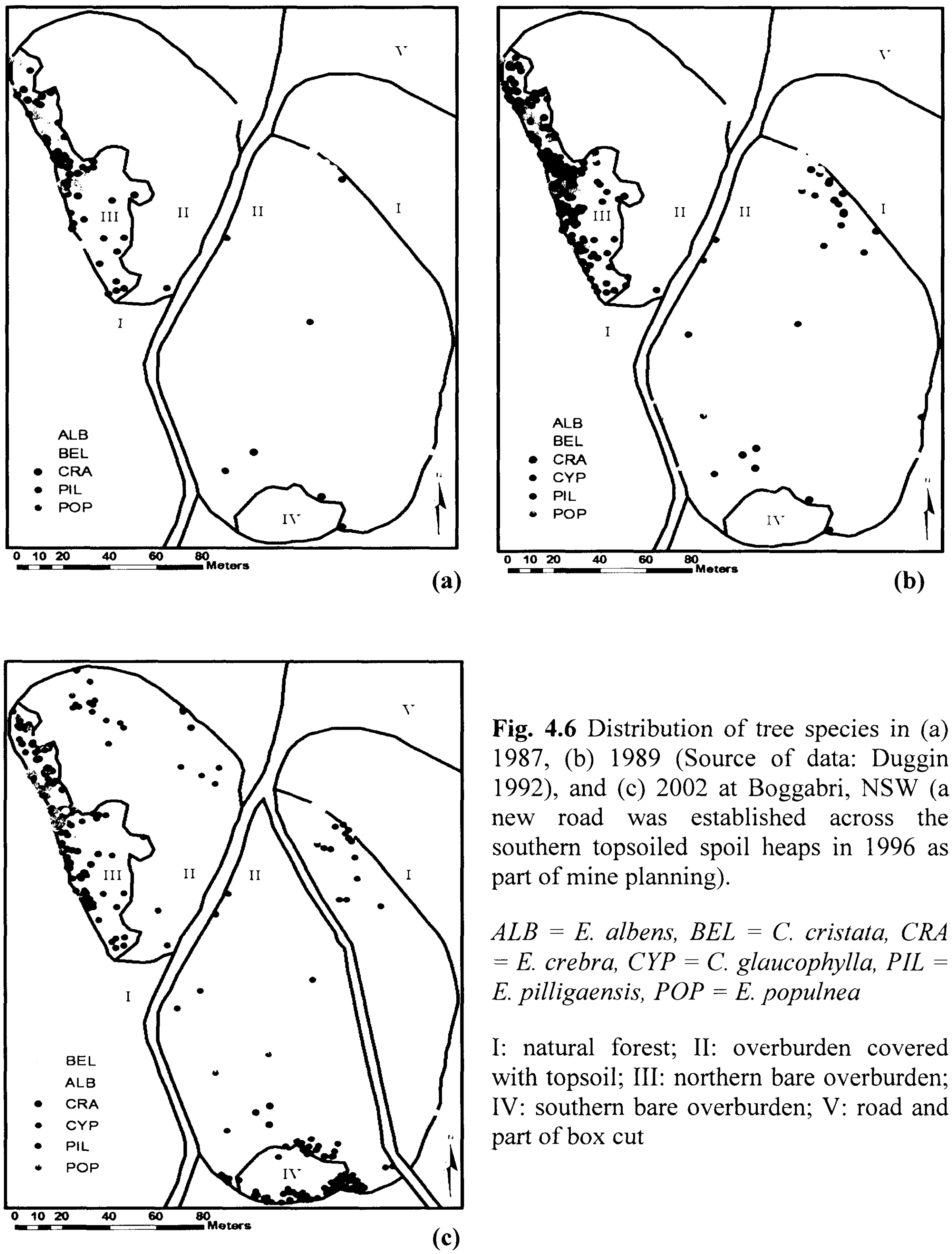


Fig. 4.6 Distribution of tree species in (a) 1987, (b) 1989 (Source of data: Duggin 1992), and (c) 2002 at Boggabri, NSW (a new road was established across the southern topsoiled spoil heaps in 1996 as part of mine planning).

ALB = E. albens, BEL = C. cristata, CRA = E. crebra, CYP = C. glaucophylla, PIL = E. pilligaensis, POP = E. populnea

I: natural forest; II: overburden covered with topsoil; III: northern bare overburden; IV: southern bare overburden; V: road and part of box cut

Table 4.2 Population changes for major tree species over time in Boggabri, NSW (total number of plants). TS: overburden covered with topsoil, NBO: northern bare overburden, SBO: southern bare overburden.

Species	Population Size (total number of plants in the study site)														
	1987			1988			1989			1990			2002		
	NBO	TS	SBO	NBO	TS	SBO	NBO	TS	SBO	NBO	TS	SBO	NBO	TS	SBO
C.glaucophylla				34			86			164			69	50	48
C. cristata	25	4		31	5		38	5		39			23	3	
E. crebra	22	3		49	7		82	12		93			32	4	
E. pilligaensis	27	4		34	4		38	6		40	2		24	28	
E. populnea	7	1		12	1		11	3		10	3		7	6	
E. albens		13			27			33			12		7	22	
Total	81	25	0	160	44	0	255	59		346	17		162	113	48

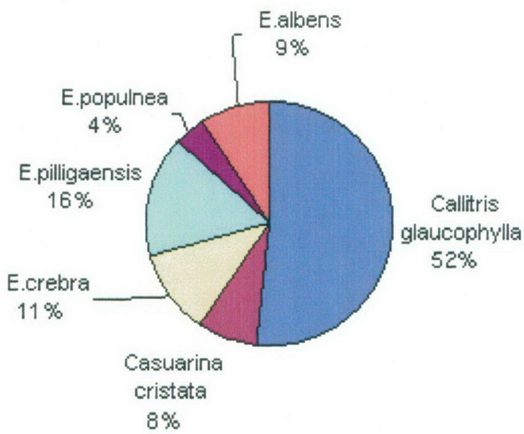


Fig. 4.7 Percentage of each tree species population in 2002 at Boggabri, NSW

Eucalypt, the major tree genus established in the study site, initially established on the northern bare overburden but then extended over the site, although at a much lower density (Table 4.2, Fig. 4.6). Most eucalypts occurred on the northern bare overburden area, and their population size changed over time. In 1987, 56 plants were recorded on the bare overburden northern site with 21 plants in topsoiled area coming from five different eucalypt species. The average height was 40 cm. The numbers of eucalypt seedlings increased dramatically from 1988 to 1990, probably due to high rainfall and suitable temperature (Table 4.2, Fig. 3.2) in competition with adequate seed sources. However, the mortality rate was also high with only 30% of those established in 1987 surviving by 2002. In 2002, 72 eucalypts had grown to become large trees with an average height of more than 10 m, and an average Diameter Breast Height (DBH) of 20.4 cm. These individuals dominated the overstorey community. The tallest tree in the study site was *E. pilligaensis* which had reached over 24 m with 26.9 cm DBH, and had fruited and seed capsules were evident on the ground. The average height growth rate for all eucalypts was 40 cm/year in the study site. *Eucaplytus pilligaensis* was the fastest growing species, and in the earlier period of its life span it grew at 60.5 cm/year.

Casuarina cristata, as a major species in the secondary tree layer in the surrounding natural forest and often seen as a pioneer tree species emerging on disturbed sites, also established in the study site. The population size did not change significantly over 20 years with around 30 individuals being recorded (Table 4.2).

Callitris glaucophylla was one of the major trees in the study site with the largest population (see Fig. 4.7). Figure 4.8 shows changes in the number of *C. glaucophylla* from 1987 to 2002. Before 1987, *C. glaucophylla* was not recorded, but from 1988 it was found on the bare overburden northern site with a total of 34 trees. In 1990, the number of *C. glaucophylla* increased to 164. From 1988 till 1990, *C. glaucophylla* was only recorded on the bare overburden northern site, and was absent from topsoiled sites and the southern bare overburden area. However, in 2002, many *C. glaucophylla* emerged on the southern bare overburden site and a few were scattered in the northern topsoiled overburden site (Fig. 4.6 c, Table 4.2). Figure 4.8 shows changes in the population of *C. glaucophylla* over time. Some *C. glaucophylla* had fruited, one in the south of the southern site and another two at the northern site near the edge of the spoil heaps.

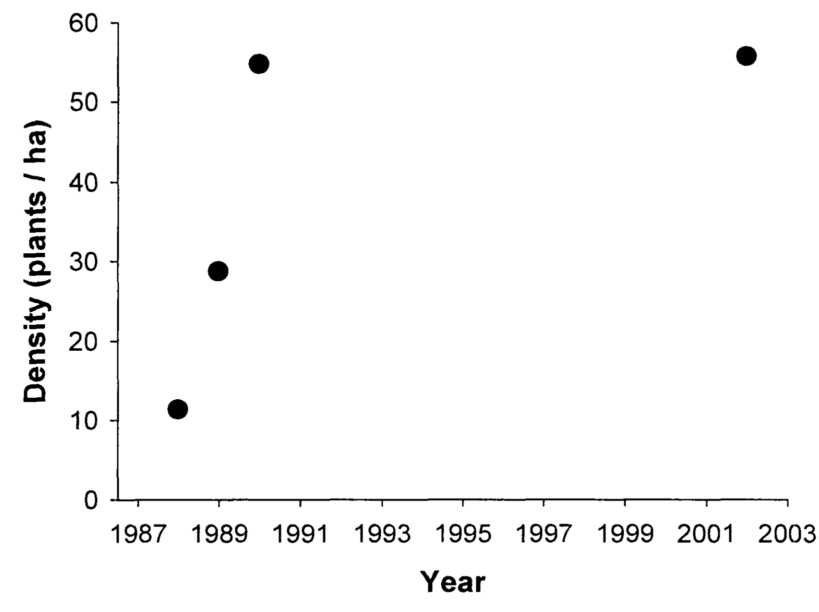


Fig. 4.8 Density of *C. glaucophylla* at each measurement year in study site, Boggabri, NSW

Colonisation Pattern

Trees occurred throughout the study site, but the density of trees decreased as the distance increased from the edge of spoil heaps (Figs. 4.9 and 4.10). Figure 4.9 (a) and (b) indicate the distribution of different species on the two spoil heaps by 10 m intervals. In 1987 (Fig. 4.9 a), five tree species had established (*E. albens, E. pilligaensis, E. crebra, E. populnea,* and *C. cristata*) with most emerging on the northern area which was the bare overburden soil type (see Fig. 3.5). *Eucalyptus pilligaensis* was distributed more widely than other species. No trees were found on the southern bare overburden. In 2002 (Fig. 4.9 b), more trees colonised the spoil heaps and the number of trees in each distance zone increased. *Callitris glaucophylla* established predominately on the bare overburden with only a few individuals in the topsoiled area. *Callitris glaucophylla* was the only species that established on the southern bare overburden. However, no *C. glaucophylla* were observed near the southern edge of the spoil heaps.

Figure 4.10 shows the differences in tree distribution for each zone. This figure clearly shows that five species established between 0-10 m from the edge in 1987, with a total number of plants of 79 in the study site. In the 10-20 m zone, although there were still five species, the number of trees dropped to 17. In the 20-30 m and 30-40 m zones, there were two species with only a few individuals. Further from the edge there was only one *E. albens* in 40-50 m zone and one *E. pilligaensis* in 50-60 m zone and no trees had established in the 60-70 m zone.

Figure 4.10(2) shows the tree distribution in 2002. The number of major tree species had increased from five to six, and all species were observed within 10 m from the edge. *Callitris glaucophylla* had a large population with 130 trees in the zone close to the edge. Both *C. cristata* and *E. populnea* had extended to 40 m, and there were also some individuals of *E. pilligaensis* in the 30-40 m zone. The number of trees in 40-50 m zone had increased from one to three with one individual of *E. populnea*, *C. cristata,* and *E. pilligaensis.*

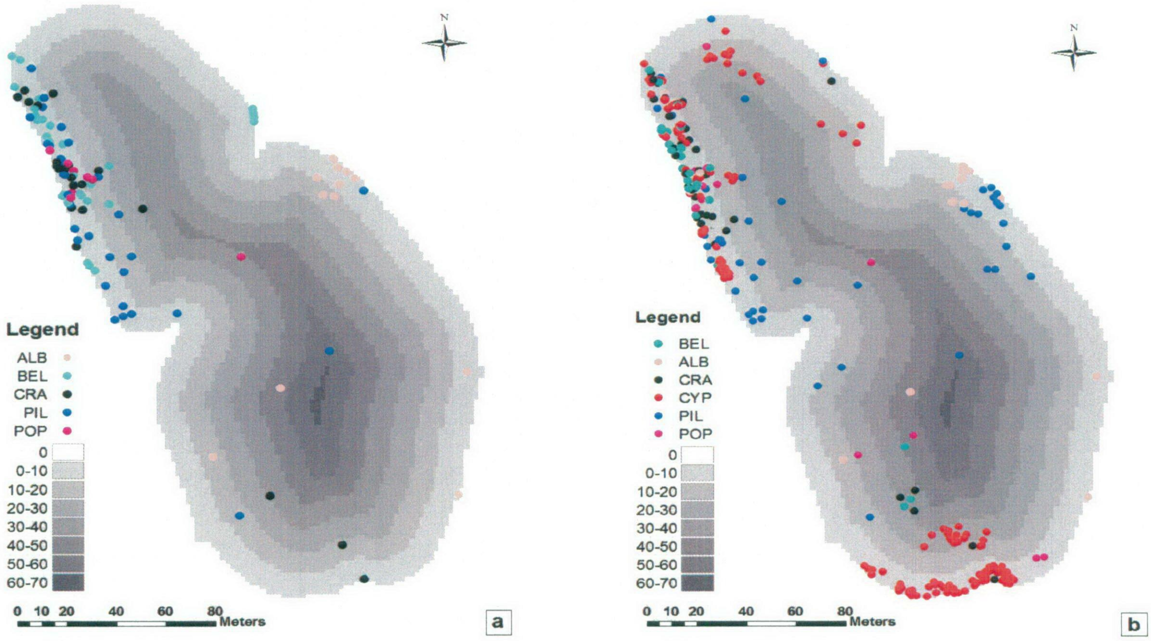


Fig. 4.9 Gray shading shows the distance from the edge of spoil heap inwards divided into 10 m intervals, Coloured dots indicate individuals of different trees species in (a) 1987 (Data source : Grigg 1987) and (b) 2002. *ALB = E. albens, BEL = Casuarina cristata, CRA = E. crebra, PIL = E. pilligaensis, POP = E. populnea, CYP = Callitris glaucophylla*

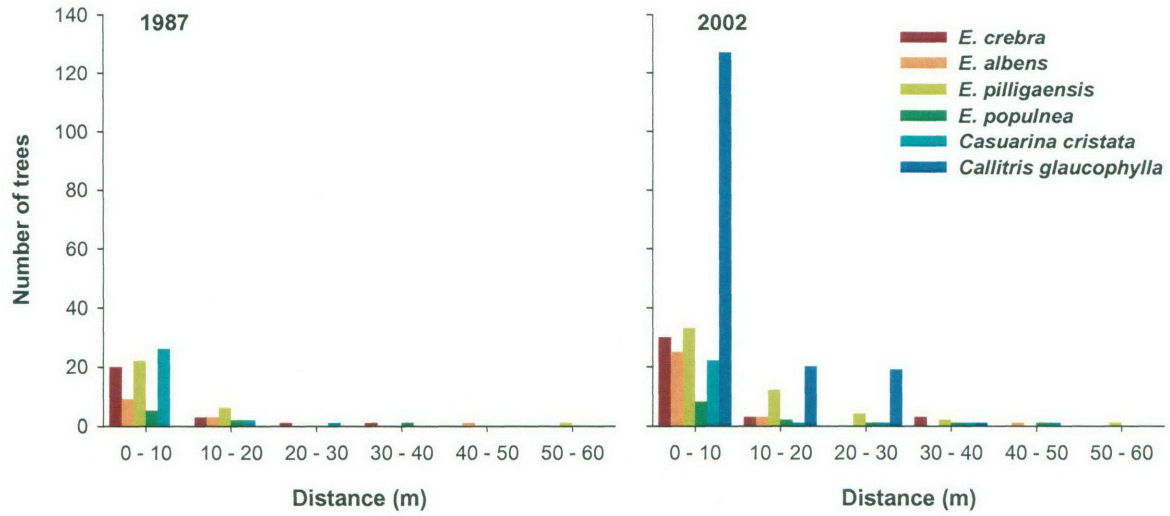


Fig. 4.10 Total number of plants of each tree species distributed in different distance zones from the edge of spoil heaps inward to the spoil heaps in 1987 (Data source: Grigg 1987) and 2002.

4.3.2.2 Changes in Community Composition and Structure over Time

The species composition on the northern and southern parts of the study site was different and varied over time. Composition and development were related to community composition of the adjacent natural forest, the distance from seed trees, and the topsoil seed bank.

At the time of the first survey in April 1981, the topsoiled overburden area in the study site was mainly covered with grass, and some shrubs seedlings (see Plate 3.4 and Fig. 4.11 a) while bare overburden areas were devoid of vegetation. Over the next two years, the shrub population had not significantly increased but individuals increased in size. The vegetation was dominated by grasses with sparse shrubs, and no tree seedling were recorded (Fig. 4.11

a). In 1987, shrub recruitment around individuals was evident and shrubs started to dominate. Consequently most of the topsoiled overburden areas succeeded in structure from grasslands to shrublands. On the northern spoil heap *A. deanei* dominated the topsoiled area along with a few scattered individuals of *D. viscosa*. On the southern site, most of the topsoiled area was occupied by *A. deanei* and *D. viscosa*, while the southern part was occupied by *S. nemophila* along with a few individuals of *A. deanei* and *D. viscosa*. Where tall shrubs dominated, smaller shrubs and grasses formed the understory. Shrubs were still absent from the northern bare overburden area although tree seedlings were present, mainly *C. cristata*, *E. populnea*, *E. crebra* and *E. pilligaensis*. Scattered tree seedlings were also found in topsoiled areas. In contrast, no trees or shrubs were found in the southern bare overburden site. From 1988, *C. glaucophylla* started to establish on the study site (See Fig. 4.11 b).

By 2002, trees still occupied the northern bare overburden without shrubs, to become an open forest land (Fig. 4.11 c). In the topsoiled overburden area, the shrub layer persisted and formed a dense shrubland in most areas but with small patches of grasslands. Although the density of *Acacia* dropped and *Senna* increased compared with that in 1987, the distribution of shrubs did not change. *Acacia* still dominated in the northern topsoiled area, and *Senna* dominated on the southern site. Scattered trees were still evident in the topsoil and had increased from 8 stems/ha in 1987 to 38 stems/ha in 2002. This structure was classified as an open woodland. The species compositions between northern and southern topsoiled sites were similar with *E. pilligeansis*, *C. glaucophylla*, *E. populnea* and *E. crebra* in the north, and *C. glaucophylla, E. pilligeansis, C. cristate, E. albens* in the south. A small patch of forest dominated by *E. albens* and *E. pilligeansis* developed in the northern part of the southern overburden heaps (Fig. 4.11 c).

Callitris glaucophylla developed along the southern edge in bare overburden. As there was a 10 year gap in observations, it is difficult to judge when they established. As *C. glaucophylla* usually starts producing seeds when they are around 12 years (Lacey 1973), at least two of them might have emerged in the early 1990s.

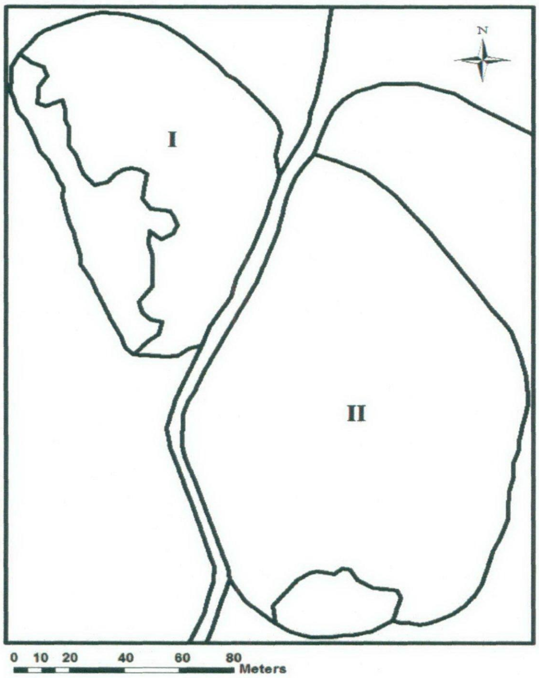


Fig. 4.11 a Community composition in 1981 (Data source: J.A. Duggin).
I: grassland with saplings of *A. deanei*, a few of *D. viscose*
II: grassland with saplings of *A. deanei, S. nemophila*, a few number of *D. viscosa*

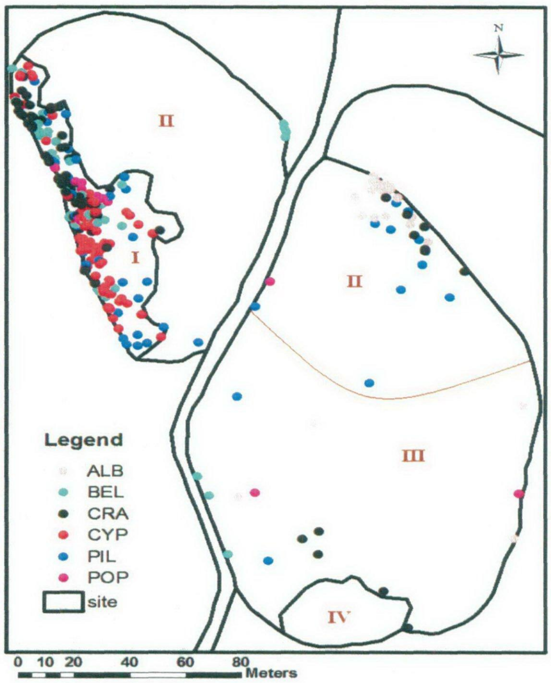


Fig. 4.11 b Community composition in 1989 (Data source: J.A. Duggin).
I: Mixed young forest
II: Shrubland, *A. deanei* dominated, with a few of *D. viscosa*
III: Shrubland, *S. nemophila* dominated, with a few of *D. viscose, A. deanei* and eucalypts seedlings
IV: Bare overburden.

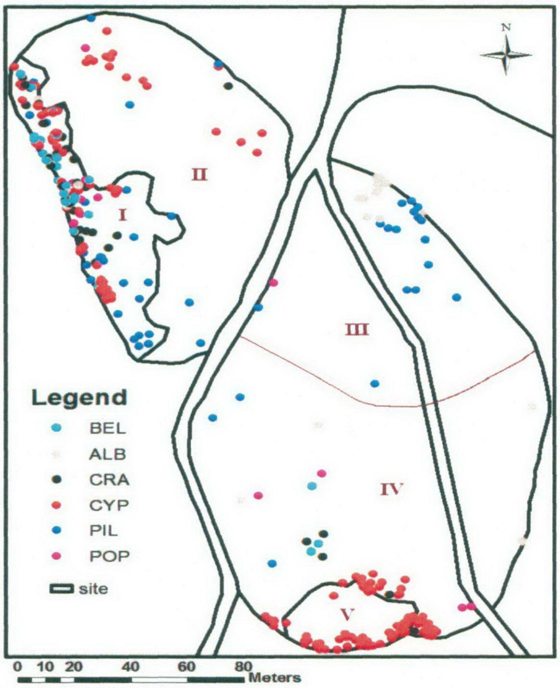


Fig. 4.11 c Community composition in 2002.
I: Mixed young forest
II: Woodland: mixed tree species with a shrubby understorey (*A. deanei* and a few of *D. viscosa*)
III: Woodland: mixed tree species with a shrubby understorey (*A. deanei* and *D. viscosa*)
IV: Woodland: mixed tree species with a shrubby understorey (*S. nemophila, A. deanei* and few of *D. viscosa*)
V: *Callitris glaucophylla* forest on bare overburden area

Height-class is a useful factor to show how community structures vary. The distributions of tree height classed in 1987 and 2002 are presented in Figure 4.12. Figure 4.13 shows the contribution of each species to the different height classes in 2002. In 1987, there were only three trees with a height greater than 5 m out of 106 tree seedlings; two being *E. pilligaensis* and the other *E. populnea*. Forty percent of the total number of trees was less than 1 m. In 2002, four trees were taller than 20 m with and average DBH of 27.8 cm, three being *E. pilligaensis* and the other *E. albens*. Thirteen trees were between 15 m and 20 m with an average DBH of 21.2 cm. Forty-eight trees were between 10-15 m, and most of them established in the northern bare overburden and the topsoiled southern area. There were a large number of trees smaller than 1 m. *Callitris glaucophylla* had the most seedlings compared to the other species, and the height of the more advanced individuals was below 5 m.

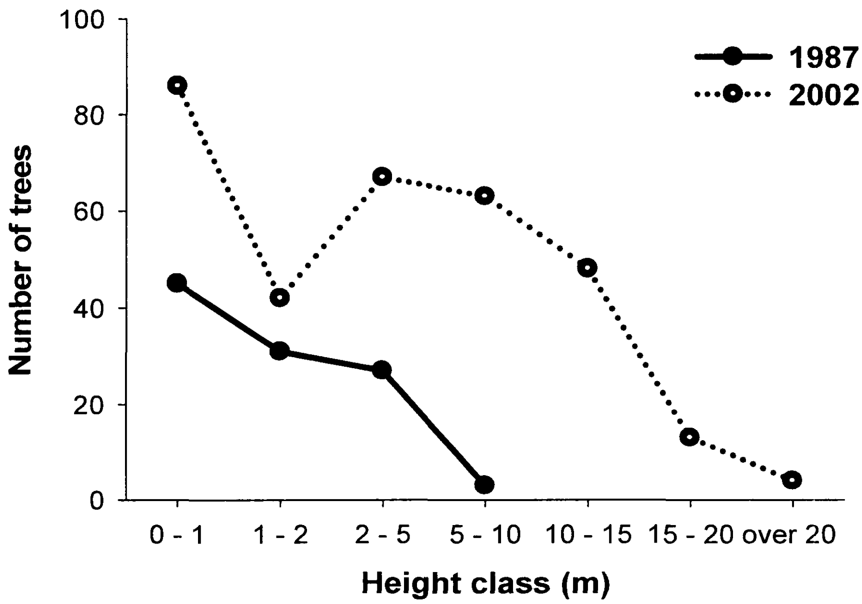


Fig. 4.12 Frequency of trees by height classes in 1987 (Data source: Grigg 1987) and 2002

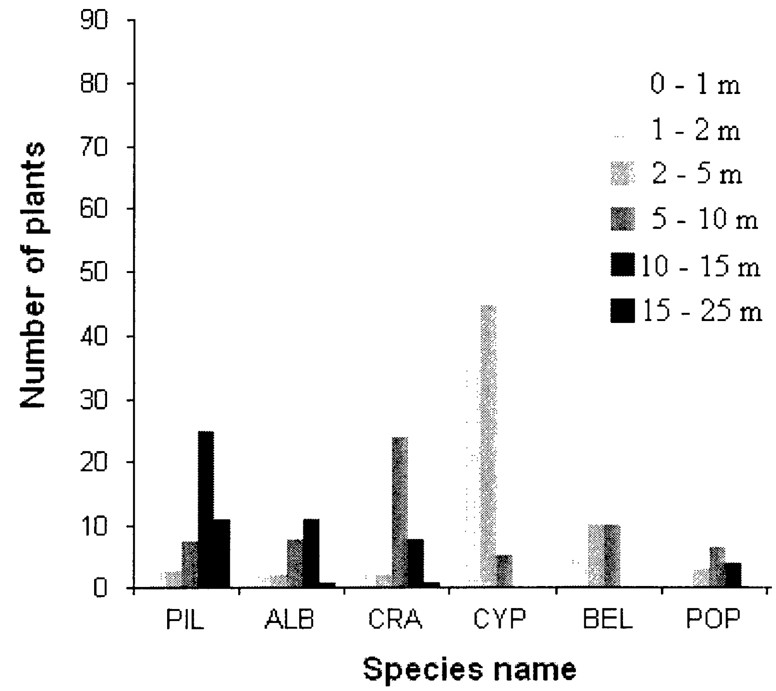


Fig. 4.13 Distribution of individuals by height classes and tree species in 2002. *ALB = E. albens, BEL = C. cristata, CRA = E. crebra, CYP = C. glaucophylla, PIL = E. pilligaensis, POP = E. populnea*

4.3.3 Competition

Competition and disturbance are internal and external factors that form and promote vegetation succession. Competition between individuals influences composition and structure of the community, as well as ecosystem dynamics. The competitive ability of a species for resources of moisture, nutrients and light, and the ability to occupy a site influence the successional development of a community. Soil moisture can be a major limitation for plant growth. Light is another important factor for competition, but it was not a critical factor driving the first ten years in the Boggabri study site.

Dense shrubs inhibited the growth of *C. glaucophylla* (Plate 4.2). *Callitris glaucophylla* is a shade tolerant species, so it is common to find seedlings under other tree crowns, or very close to small shrubs. However, as the growth rate of shrubs is greater than that of *C. glaucophylla*, shrubs trend to outgrow *C. glaucophylla*, and suppress *C. glaucophylla* growth (Plate 4.2).

Moisture stress appears to be the main cause of mortality amongst young plants. Seedlings are sensitive to dry conditions during the first few weeks after germination. Germination of a good quality commercial seed generally has a viability of about 80-90 %, but under field condition 0.1-1.0% survival is more usual (Penfold & Willis 1961).

The competition for moisture between grasses and trees seedling may also contribute to seedlings mortality (Plate 4.1). Where dense swards of grasses developed shrubs generally failed to establish, but when they did, they were suppressed (Plates 4.1 & 4.3)

On the northern part of the bare overburden (Fig. 3.5, Plate 4.4), where the overburden contained a quantity of coal, shrubs were absent by 2002. However, germination tests showed that shrubs can germinate on it but at a low percentage (Grigg 1987). The southern bare overburden area (Fig. 3.5) was white coarse sandy clay texture with a pH of 8.1 in 1981. The only species recorded was *C. glaucophylla*.

Plate 4.1 Competition between grasses and eucalypts seedlings, Nov. 2004

Plate 4.2 *Callitris glaucophylla* was suppressed by *A. deanei*, Nov. 2004

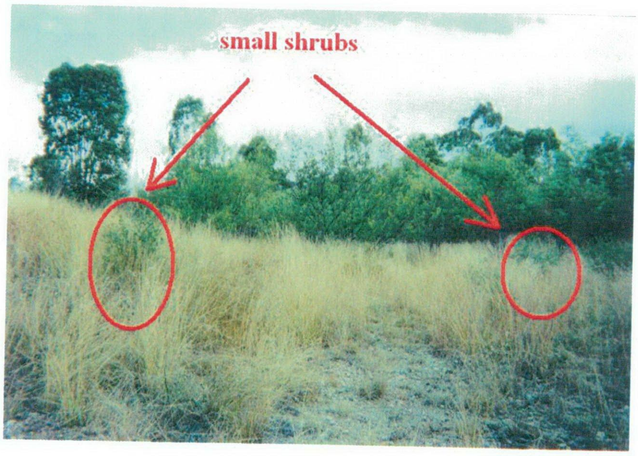


Plate 4.3 Competition between grasses and shrubs, Nov. 2004

Plate 4.4 Eucalypt seedlings established in bare overburden area, Nov. 2004

4.4 Species Richness

Richness is a key factor in evaluating rehabilitation and ecosystem recovery through the comparison of similarity between rehabilitated and reference ecosystems. However, full recovery of an ecosystem and its components may take many years, probably even millennia

(Cooke 1999). In this study, species richness changed over time. In 1981, no tree species were recorded and only 3 shrub species were present (herbaceous species are excluded from this study). In 1987, besides the shrub species, 5 tree species, *E. pilligaensis, E. populnea, E. creba, E. albens,* and *C. cristata* were observed, and in 1988, *C. glaucophylla* established in the northern bare overburden area, and the species number had increased to nine (Fig. 4.14).

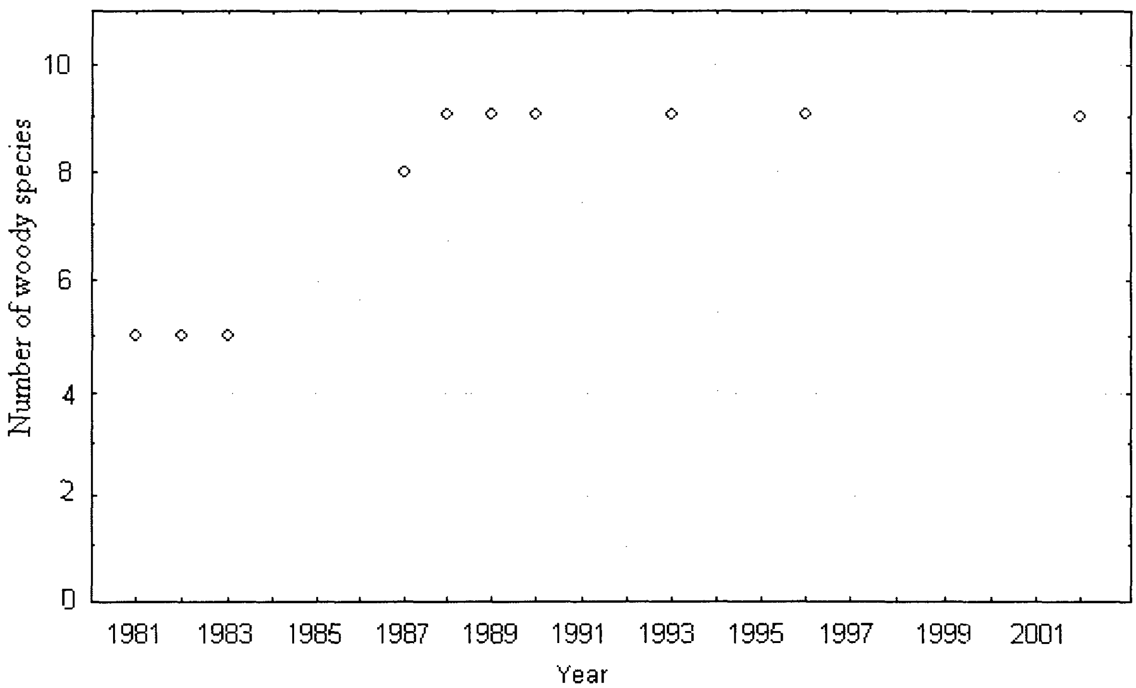


Fig. 4.14 Woody species richness at each measurement year in the study site (Boggabri, NSW).

4.5 Similarity between Rehabilitation and the Reference Ecosystem

The natural forest surrounding the study site was used as a reference ecosystem against which the rehabilitation ecosystem was compared to evaluate whether the rehabilitation was moving towards the end goal.

A comparison of community composition and structure between the rehabilitation and the surrounding natural forest showed that shrub density was higher in the rehabilitation system than that in the natural forest (Table 4.3), whereas trees density was lower (Table 4.4). However, the relative abundances of *Acacia* and *Dodonaea* in rehabilitation area are close to that in natural forest (Table 4.3). The proportion of shrubs to trees in the rehabilitated site was 16.1, which was significantly different from that of 0.17 in the natural forest.

Callitris glaucophylla has the highest density in both systems, but the relative abundances are different. However, the main difference is that in the rehabilitation system, most were seedlings and small trees. Eucalypts were the major species in both stands. The relative abundances of *E. crebra and E. albens* in the rehabilitation site are very similar to that in the natural forest. The distribution of *E. populnea* in the rehabilitation site was low. In the natural forest, *E. populnea* was not recorded from the sample plots, but was found outside the plots and in close proximity to the disturbed area. In addition, *C. cristata* density in rehabilitation area was lower than that in the natural forest.

Table 4.3 Comparison of three major shrubs densities and relative abundances between rehabilitation ecosystem and natural forest in 2002

Species Name	Rehabilitation Ecosystem		Natural Forest	
	Density (plants/ha)	Relative abundance (%)	Density (plants/ha)	Relative abundance (%)
Acacia deanei	922	52.9	150	66.7
Dodonaea viscosa	189	10.8	34	15.1
Senna nemophila	633	36.3	41	18.2
Total	1744		225	

Table 4.4 Comparison of major tree species densities and relative abundances between the rehabilitation ecosystem and natural forest in 2002

Species Name	Rehabilitation Ecosystem		Natural Forest	
	Density (plants /ha)	Relative abundance (%)	Density (plants /ha)	Relative abundance (%)
E. albens	10	9.3	100	7.5
E. crebra	12	11.1	149	11.2
E. pilligaensis	17	15.7	20	1.5
E. populnea	4	3.7	0	0
Callitris glaucophylla	56	51.9	296	22.3
Casuarina cristata	9	8.3	764	57.5
Total	108		1329	

Trees with a DBH greater than 10 cm were recorded in both the rehabilitation ecosystem and natural forest (Table 4.5). Although the average height and diameter of trees in the rehabilitation area were similar to that in the natural forest, the density (34 plants/ha) was much lower than that in the natural forest (411 plants/ha). Figures 4.15 and 4.16 show the percentage of trees with a DBH greater than 10 cm in these two communities respectively. In the natural forest, 32.1% of the trees were *C. glaucophylla*, whereas it was only 1% in the rehabilitation system. The population of *C. cristata* was larger in natural forest (36.6%) than that in the overburden area (2%).

Table 4.5 Average Diameter and Height (The tree diameter breast height higher than 10 cm is counted) in spoil heaps and around natural forest

	Densities (Tree DBH ≥10 cm) (plants/ha)	Average DBH (cm)	Average Height (m)
Rehabilitation Ecosystem	34	18.7	11.5
Natural Forest	411	20.0	12.9

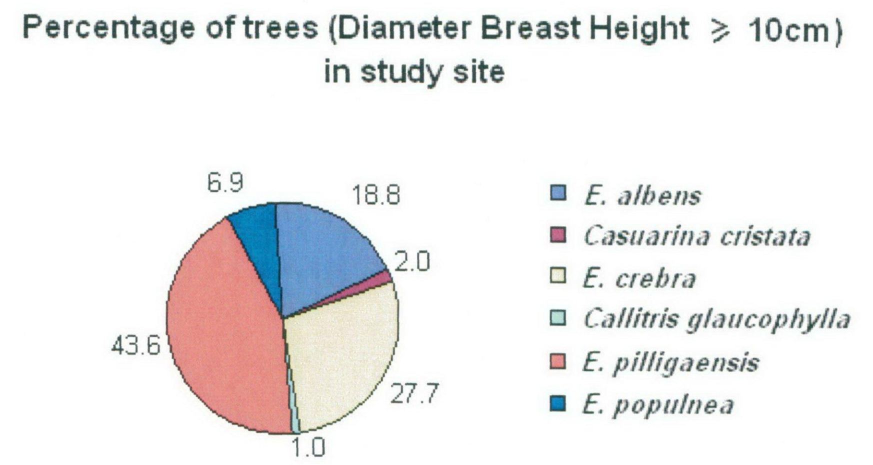


Fig. 4.15 Percentage of large trees (diameter breast height is equal to or greater than 10 cm) distribution in spoil heaps

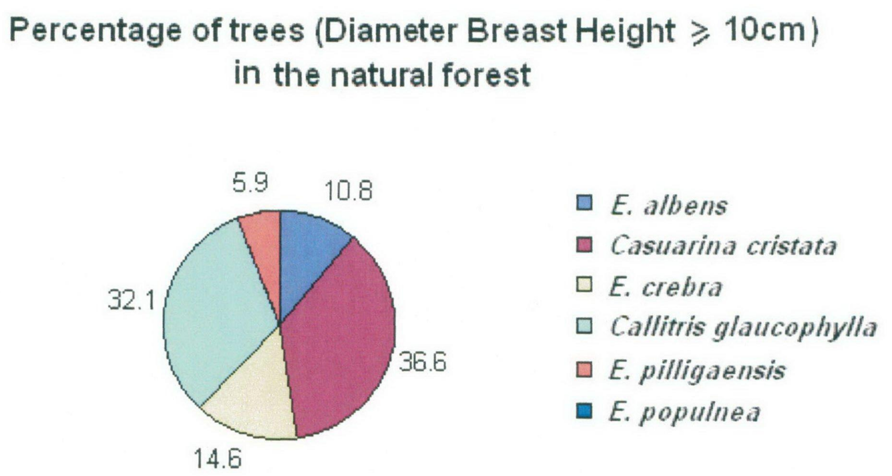


Fig. 4.16 Percentage of trees (diameter breast height is equal to or greater than 10 cm) distribution in natural forest

In the surrounding natural forest, small trees (height < 5 m) occurred in greater densities (Table 4.6). *Callitris glaucophylla* and eucalypts were the dominant species (Table 4.4) and occupied the overstorey (Table 4.7). *Casuarina cristata* formed a secondary layer below the overstorey. The shrub layer was generally sparse and consisted of *S. nemophila*, *D. viscosa* and *A. deanei* (Table 4.4). The canopy cover was about 50 % which indicated it was an open-forest structure (Croft 1979).

The community structure in the rehabilitation ecosystem was significantly different from the reference ecosystem (Tables 4.6). All the densities in each height class were lower with only 5 plants/ha over 15 m high and all of them were eucalypts. *Callitirs glaucophylla* and *C. cristata* were always less than 10 m in disturbed area. The densities of tree saplings (tree height < 5 m) were significantly lower than that in natural forest which suggested that the rehabilitation ecosystem had a lower regeneration capability than the natural forest.

Table 4.6 Comparison of communities structures between the rehabilitation and natural forest

	Height Classes (m)						Total
	<1	1-5	5-10	10-15	15- 20	20 over	
Tree Density in Rehabilitation Ecosystem (plants/ha)	28	37	22	16	4	1	108
Tree Density in Natural Forest (plants/ha)	323	612	124	164	73	33	1329

Table 4.7 Species density according to different height class (plants/ha)

Species Name	Density (plants/ha)					
	Height 10 – 15 m		Height 15 – 20 m		Height 20 m over	
	Rehabilitation Ecosystem	Natural Forest	Rehabilitation Ecosystem	Natural Forest	Rehabilitation Ecosystem	Natural Forest
E. albens	4	11	0	14		13
E. crebra	3	27	0	17		7
E. pilligaensis	8	6	4	3	1	10
E. populnea	1					
Callitris glaucophylla		73		36		3
Casuarina cristata		47		3		

4.6 Discussion and Conclusions

Significant shrub establishment occurred across the study site in 1981 followed by a decline in germination over the next two years (Fig. 4.2), in a similar manner as noted by Noble and Slayter (1981) who indicated that there was a pulse of recruitment after disturbance. Since 1987, the number of new shrub seedlings increased progressively suggesting that some shrubs had matured and started to produce and disperse seeds across the site.

Trees recolonised from the edge of the rehabilitated area to the centre with a decline in density over distance from the edge, indicating that the natural forest was the major seed source. The species composition in each distance zone was different due to variation in dispersal capabilities of each species. Both species richness and density increased in each zone over time.

Some of the first established trees started to produce seeds before 2002 thereby providing another seed source. This, in combination with the ability of shrubs to produce seeds shows that the study site has the capability to revegetate without any further external seed inputs.

The topsoil introduced a shrub seed bank for the species in this study site. However, many factors influence the viability of seeds stored in the topsoil. Density of plants establishing from the topsoil is greatly affected by stockpiling. Fresh topsoil contains a higher proportion of seeds and active micro and macro organisms than the stockpiled soil (Hannan 1981). Stockpiled topsoil loses quality in terms of physical, chemical and biological properties. Seed viability declines, nutrients are leached and micro-organisms decline. Topsoil stockpiled over six months tends to lose structure and some of its fertility (Scullion 1992). Seed germination declines over time, particularly when stored in soil seed banks. Seeds of *Acacia*, for example, can remain in the soil for many years until they are stimulated to germinate (Jacobs 1955) whereas the number of seeds of *S. nemophila* capable of germination is almost zero after two years when buried in the field under normal conditions (Hodgkinson & Oxley 1990). Eucalypt seeds are susceptible to death during stockpiling in sandmining rehabilitation (Burrows 1986). The depth of topsoil stockpiles also has an impact on seed germination (Grant 2001; Cremer 1965; Lacey 1973). Seventy-two percent of germinatable seeds were found in the upper 5 cm of the soil in eucalypt woodlands at Newholme, Armidale (Grant

2001). Cremer (1966) found that most *Eucalyptus regnans* seeds buried deeper than 5 cm died within 14 months. It is suggested that a topsoil depth of 10 cm is adequate to provide a good seedbed and a moderate supply of plant nutrients for plant growth (Bradshaw & Chadwick 1980).

Other factors such as insects also can destroy large quantities of eucalypt and other tree seeds (Withers 1978; Jocobs 1955; Cremer 1966; Ashton 1979). For instance, Chesterfield and Parsons (1985) found that though *Casuarina* seed readily germinated in the laboratory, ant predation accounted for a significant loss of seeds in the field. A decline in eucalypt seeds over a period of 11 months in a *E. populnea* shrub woodland was attributed to ant predation (Hodgekinson *et al.* 1980). In addition, it is well know that eucalypts tend not to have significant soil seed banks. All of these factors may impact on the variability of seeds in the topsoil, and might explain why there was no tree species found in the rehabilitated area before 1987.

Plants respond differently to soil types. Soil physical, chemical and biological characteristics are significantly modified by mining, which is likely to impact on plant germination and growth. Eucalypts significantly respond to different soil types, and they had low germination (Grigg 1987) and could not emerge on southern bare overburden material which had a pH of 9.0 and a high electrical conductivity. However *C. glaucophylla* was not affected by soil type and established on different soil materials, whether covered with topsoil or not.

Intra-specific competition between trees and shrubs was evident. In the topsoiled area, eucalypts grew where shrub densities were low. Koch (1987) found that eucalypts were suppressed by *Acacia* in restored bauxite pits in Western Australia, but could successfully regenerate on bare overburden when seeds were directly sown onto it (Burns 1987). This has been confirmed by this study site. The low number of trees established in topsoiled areas from the seed rain indicated that there may be some competition from the herbaceous layer and/or shrubs. Topsoil provides a good seed bank, a good medium for plant growth and reduces soil erosion, but it also increases the potential competition between shrubs, grass and trees.

Open-cut mining disturbance is very different from other disturbances due to the complete removal of vegetation, soil and the seed source. Koch and Ward (1994) found there was only 20 - 50% similarity in jarrah forest between pre-mining and nine month post-mining rehabilitation in bauxite mined areas in Western Australia. Overburden areas respread with topsoil at Boggabri were first covered by grasses and scattered shrubs. Then, shrubs began to dominate with increasing density. The structure and composition of the developing community were simple with three major shrub species widely distributed in spoil heaps. Eight years later, tree species were evident in the study site while after 15 years, community structure and composition had increased in complexity. Species diversity has increased. Six tree species with three major shrub species cover the whole area. Eucalypts dominated the tallest stratum, *C.cristata* and some *C. glaucophylla* formed in the middle stratum, while shrubs occupied the understorey. The structure of community approached that of the natural forest, except that *C. glaucophylla* had not yet reached the overstorey.

This study has described species colonisation and community succession after mining disturbance. It demonstrated that the developing ecosystem had changed from grassland to shrubland and then woodland, and was trending towards an open forest. One of the main rehabilitation goals is to establish and maintain plant communities that meet desirable land-use objectives. Therefore, it is critical to understand plant colonisation and community succession on a reconstructed landscape. In order to be sure that rehabilitation practices are successful, there is a need to track progress to ensure that the desirable aims of rehabilitation are met (Brooks 1981). In this study, 20 years of monitoring has provided baseline information for rehabilitation, but it is insufficient to judge the ultimate success of achieving a long-term goal. Such a limitation may be overcome by developing a model that simulates colonisation and succession on open-cut mine sites. A model with such predictive functions can also be used to compare different rehabilitation strategies. Therefore, a simulation model with strong predictive functions needs to be developed to meet such requirements.