Best Practice Environmental Management in Mining Program

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Title: Comparison of regeneration following burning, clearing or mineral sand mining at Tomago, NSW: I. Structure and growth of the vegetation

Reference Number 86
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Authors: Fox B J et al

Issues
Rehabilitation and Revegetation: Flora Succession

Journal: Australian Journal of Ecology (1996), Volume 21, Number 2, p 184-199
Year of Publication: 1996
Abstract This is a long-term study which sets out to examine the question of whether multiple disturbances have an additive or interactive effect on species richness in ecological communities. The area studied is at Tomago (32o52'S, 151o45'E) and it includes forest, woodland, shrubland and swamp on a sand substrate. The vegetated dunes form part of a coastal embayment. It is a protected water catchment area and is affected by fires, sand minding and clearing. A detailed history of such disturbances to the area is available. Forested sites were chosen and classified according to whether they had been disturbed by fire, sand mining or clearing. There were forty-four sites of four age classes studied. The age classes were: less than 1 year since disturbance, nominally 1991; 5 years, nominally 1986; 11 years, nominally 1980; and 17 years, nominally 1974. As a control, responses to disturbances are recorded for a set of burned sites (the time of the last fire is matched to the time of other disturbances).

This present paper describes the area and sites being studied and looks at the effects of each single disturbance on the structure of vegetation. Development of a canopy varied with time and the type of disturbance. Cleared or mined sites took 17 years to develop a canopy equivalent to that of a 11-year old burned site. Understorey development was significantly greater two years after burning than two years after clearing or mining. However, after five years, understorey development was similar for all three site types. The rate of regeneration of mined sites is shown to be considerably slower than that which follows disturbance by fire. Cleared sites regenerate at an intermediate rate but the rate is closer to that for mined sites than for burned ones. Regeneration of cleared and sand mined sites, as measured by both understorey height and the amount of vegetation, had not resulted in a return to the pre-disturbed vegetation structure even after 17 years. However, it appeared that canopy cover had reached pre-disturbance levels. It is important that monitoring continues to see if contemporary, more sophisticated rehabilitation methods are promoting faster regrowth. The next stage of the study will be to build on the hierarchy and nature of the response to each single disturbances, so far established, to study the effects of multiple disturbances.

Language: Eng

Proceedings of Workshop on Indicators of Ecosystem Rehabilitation Success (Eds. C.J. Asher and L.C.Bell), pp. 139-140

Effects of Habitat Logs on the Succession of Small Mammal Species within RZM's Sand Mining Rehabilitation near Newcastle, NSW

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ABSTRACT

Sand mining near Tomago, north of Newcastle NSW, has been conducted by RZM Pty Ltd since 1967, where deposits of rutile, zircon, and ilmenite are mined, concentrated and explored. Rehabilitation of mined areas has progressively followed mining throughout the life of the operation with the main objective being to re-establish native ecosystems similar to those existing prior to mining. Two aspects of the rehabilitation at Tomago are unique. Firstly, this scheme included the reintroduction of habitat logs following mining. Habitat logs from the forest are stored along with the topsoil during mining and are redispersed throughout the area as a part of the rehabilitation process. The aim of this procedure is to provide habitat niches for various fauna species. Secondly, following improvements in processing technology, some of the rehabilitated areas at Tomago have been re-mined, meaning that these areas have been intensively disturbed twice in a relatively short time period. While a relatively large body of research exists on vegetational succession in rehabilitation, relatively little published research is available regarding the long-term successional patterns for fauna. The aim of this project was to determine the effect that the post-mining dispersal of habitat logs has on the re-introduction and successional sequence of small mammal species at Tomago.

Eight sites were selected, consisting of single mined and re-mined rehabilitation of three ages (3, 5 and 8 years) with two forest control sites. Four trapping sessions (three of which have been completed) were undertaken, one in each season of the year. Trapping was performed for four consecutive nights. Three transects of 100 m in length were placed approximately parallel to each other within each of the eight study sites. Transects were 50 m apart and at least 50 m from access roads. Eleven 'A' size Elliott traps were placed at approximately 10 m intervals along each transect within each study site, giving a total of 4224 trapping nights for the entire study. Mammals caught were identified, sexed, weighed, tagged and released at the point of capture. A vegetation survey was also conducted at each study site and included an estimate of volume of habitat logs, a measurement of leaf litter cover and depth, an estimate of crown foliage cover using a densiometer, an assessment of vegetation structure using a 3.9 m levy pole, and soil moisture analysis.

Preliminary results indicate the capture of 168 small mammals with a total of 118 recaptures (some individuals had been caught a number of times) throughout the first three trapping seasons. Capture success was highest on the third day of trapping. A common seasonal trend was evident among the mammal species, where animal captures declined from the Autumn to the Spring season of trapping. A successional trend was also displayed whereby recently rehabilitated areas have been invaded by early colonists such as Mus domesticus, *followed by* Pseudomys novaehollandiae. *Areas of older rehabilitation (>5 years) were more species rich and included* Antechinus stuartii, Smithopsis murina *and a number of rat species including* Rattus lutreolus. *Although vegetational patterns differed greatly between study*

sites, there was a tendency for a higher capture success within areas that contained greater densities of habitat logs.

Preliminary results indicate that the dispersal of the habitat logs as part of an intensive rehabilitation scheme may result in a faster re-introduction of small mammal species from the surrounding undisturbed ecosystems. Further discussion on the long term management of the rehabilitated areas in relation to the re-introduction of fauna is also provided.

Fauna Habitat Reconstruction after Mining Workshop (Eds. C.J. Asher and L.C. Bell), pp.73–79.

RECONSTRUCTION OF KOALA HABITAT AFTER TITANIUM MINERALS MINING IN THE TOMAGO SANDBEDS, NSW

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ABSTRACT

The Tomago Sandbeds is a coastal barrier sand dune system north of Newcastle on the mid north coast of New South Wales, Australia. The Sandbeds contain a significant regiona groundwater aquifer. Most of the area has, historically, been protected from developmen pressures to help ensure long term viability of this resource. The site contains a diverse rang of undisturbed native ecosystems, including several flora and fauna species listed a threatened and, in particular, a nationally significant koala population which has been the focu of some attention.

Titanium minerals have been mined from the area since 1967. These operations will cease late 1998. Approximately 10% disturbance of the total area of the Sandbeds will have occurre over the 31-year mine life. During this time, mine and rehabilitation planning objectives hav endeavoured to accommodate sound ecological principles, and rehabilitation of the site native ecosystems has been carried out progressively with mining throughout the life of th operation. The key elements of a successful site management program are resourcing th collection of detailed scientific information during premining site investigations to enable caref mine and rehabilitation planning, and paying attention to this detail in delivery of post mini field programs.

Subsequent fauna impact assessment and ecological monitoring at the Tomago Sandbeds s have demonstrated that, although a time lag of about ten years exists between establishm and initial use of regenerating areas by koalas, the species will browse regenerating preferr habitats to a degree similar to undisturbed areas. Although studies have not taken into acco social factors, the information suggests that, contrary to some local opinions, the local ko population will not be adversely affected in the long term as a result of the mining activities.

1.0 INTRODUCTION

The Tomago Sandbeds is a coastal barrier sand dune system north of Newcastle on the r north coast of New South Wales, Australia. The Sandbeds occupy an area of about 11, hectares, as a relatively narrow band averaging about 3 kilometres wide and stretcl approximately 30 kilometres from the Hunter River in the south-west to Port Stephens in north-east.

The site contains an unconfined groundwater aquifer of regional significance and has b managed since the 1930s as a water reserve by the now Hunter Water Corporation, primar supplement domestic supply to the Newcastle area. Protection of this groundwater reso has been a high priority. This has resulted in the Sandbeds experiencing relatively development over the years, and consequently, many of the site's intrinsic natural va remain. A mosaic of native ecosystems typical of coastal dune systems with a high rel

water table exist in the area, influenced by complex and dynamic soil–water relationships. These factors have led to the area attracting a *de facto* nature reserve status.

Apart from water harvesting infrastructure, the Tomago Sandbeds are also traversed by a variety of competing land uses. These include public utilities, urban infrastructure and defence facilities, and surrounded by other competing land uses including various forms of agriculture, extractive industry, a domestic airport and rural residential and urban areas. These developments have contributed to habitat fragmentation throughout the area, while peripheral developments, and in particular rural-residential subdivision in recent years, have played a significant role in isolating the Sandbeds from other core natural areas, although substantial links do still remain.

1.1 Mining Operations

Since 1967, economic grades of the titanium minerals group (rutile, zircon and ilmenite) have been mined by RZM from the relic beach strand sequences within this area. Whilst the total area of mining disturbance over the projected life of the mine will be less than 10% of the area of the Sandbeds, mining operations have been concentrated in the main orebodies which exist in the central south-west of the area, and some relatively large areas have been cleared, resulting in further fragmentation of some habitats.

During this time operations have been controlled both by Company policies and practices and by prescriptive licences and approvals imposed by regulators, in a climate of changing legislation. Despite this, mine and rehabilitation planning objectives have remained consistent and have endeavoured to accommodate sound ecological principles as far as possible, along with maintenance of existing land uses.

From 1986 to 1995 mining operations in the Sandbeds primarily entailed remining of areas originally disturbed between 1972 and 1985. To the projected end of the mine's life, mining will continue in two orebodies in this area with a significant proportion of the area (up to 50%) to be mined being remining.

Whilst rehabilitation of the site to native ecosystems has been carried out progressively with mining throughout the life of the operation, recently there has been strong opposition to mining by community interest groups. In this respect, clearing of undisturbed koala habitat has remained the main contention in protest action and associated legal challenges. Significant compromises have been volunteered by the Company in recent mine planning, to further facilitate fauna population movements and sustainability. Animal welfare management at the mines, in the form of modified work procedures, has been implemented.

1.2 Legislative Framework

In New South Wales, mining controls are enforced by the Department of Mineral Resources under the Mining Act (various revisions), and Mining Leases granted under this legislation contain prescribed conditions relating to wildlife management and site rehabilitation, amongst other issues. The main government agency responsible for the management of wildlife is the National Parks and Wildlife Service, and controls are enforced under the National Parks and Wildlife Act (1974) where almost all native fauna are protected by these laws, and the Threatened Species Conservation Act (1995) in respect of endangered or vulnerable species.

Generally, with respect to fauna management, mining operations have been conducted at a level beyond the regulatory standard, aiming at best practice management, and legislative controls have therefore not impacted on operational practices.

2.0 FAUNA IMPACT ASSESSMENT AND MANAGEMENT

The faunal assemblage in the Tomago Sandbeds is diverse, in direct relation to the wide diversity of habitats present across the site. This assemblage logically contains representation from all of the major vertebrate fauna groups — amphibians, reptiles, birds and mammals (large and small terrestrial, arboreal and microchiropteran bats). Some of these groups include species listed in state legislation as "threatened". Whilst a range of animal species exist in habitats on and near the mine path, and a total ecosystem management approach is adopted at the mines, the focus of attention has been on koalas, due to the significance of the local population and the vulnerability of the species, because of its biology and role as an ecological specialist. This emphasis does not imply that site management programs are biased towards koala management, and indeed some species are considered more recalcitrant than the koala in these circumstances.

Development consents for RZM's current mining operations predate much of the existing planning legislation, and despite the message commonly reported by antagonists of mining, there is a commonsense goal, and willingness, from the Company to help ensure long term regional sustainability of local ecosystems, which necessarily includes koala populations. With the advent of specific endangered species legislation, in recent years, the Company undertook a major fauna impact assessment program at the Tomago Sandbeds mine sites to investigate potential impacts of the remaining mining operations on native species. Together with existing pre-mining, operational, and post mining monitoring and assessment, this program also helped gauge the success of fauna return to existing rehabilitation areas.

The results, combined with existing scientific knowledge and data from local wildlife care organisations, such as the Native Animal Trust Fund and the Koala Preservation Society, enabled the Company to review mine planning and further refine operational controls and rehabilitation methods to accommodate fauna population management and welfare interests, thus mitigating against adverse long term effects as far as possible.

2.1 Habitat Assessment

Preferred fauna habitat management is based on the premise of application of sound ecological principles and a detailed understanding of the local site (habitats). Collection of comprehensive baseline information is fundamental to this process.

A relatively detailed habitat assessment process was undertaken to identify the likely fauna impact, as a result of mining operations in the area, and in particular, the magnitude of impacts on the local koala population. General fauna surveys entailed standard practices of diurnal and nocturnal trapping and searches, call playback for owls (and possibly their prey), pitfall trapping, hair tube placement, and searches for direct and indirect evidence of various species according to their respective habits.

Koala habitat quality is variable across the Sandbeds and, to specifically identify and assess koala habitat usage, a more detailed site assessment approach is required. The Port Stephens Council, in conjunction with the New South Wales National Parks and Wildlife Service, has been reviewing a draft Plan of Management for koalas in the shire (Callaghan et al. 1994). The

information gathered for this exercise was available as a base from which to work, and this was expanded for the project area (the forward mine path and surrounds).

Vegetation in the project area was mapped from interpretation of enlarged air photos at a scale of 1:10,000, and ground-truthed to determine major communities and specific dominant species associations to determine accurately preferred browse habitat areas and their extent. The aim was to categorise these into classes of feeding preference. Habitats were sampled for specific vegetation characteristics and faecal pellet density using plot survey methodology modified from that developed and used by the Australian Koala Foundation in their koala habitat atlas mapping program (Phillips 1995, Phillips and Callaghan 1995). The results of these surveys were combined with existing knowledge of the animals' habits, from previous fauna surveys, radio-tracking research work in the area, and knowledge of their physiology, to identify preferences in feeding habitats. This program did not take into account social interactions between individual animals in the population which, in some habitat classes, can be a determinant of population density and hence also of browsing.

It is recognised that, whilst the information collected from this program is reasonably detailed, it probably still falls within the realms of basic habitat assessment from a scientific viewpoint. There are further levels of investigation which could be explored to better determine habitat characteristics, and in this respect, the data may be considered limited. However they do provide a sound basis for critical management decisions. This is relevant from a commercial viewpoint, as was necessary, as well as making some contribution to the scientific knowledge base.

Results of the koala habitat assessment program (Fig. 1) are consistent with local knowledge of the range of habitat quality existing in the Tomago Sandbeds (Callaghan et al. 1994) and with studies at other locations (Braithwaite et al. 1983, Cork et al. 1990). They generally indicate a preference for swamp forests over dry forests, and utilisation of a range of intermediate species and communities whose presence is determined by specific soil-water relationships. Koalas utilise regenerating environments from about 10 years of age, and of interest to the study was the usage levels recorded in "advanced" (approximately 20 year old) regenerating areas. Regenerating swamp forest had similar usage potential to undisturbed wet forest, and regenerating dry forest was similar to its undisturbed forest. Both of these areas were remote from core undisturbed areas, indicating either opportunism or the koalas' willingness to roam regenerating environments to select preferential browse. This finding sits well with the objectives of the site rehabilitation program.

2.2 Mine Planning

The information gained in field investigations, along with that of studies elsewhere, was used in discussions with specialist consultants and regulating authorities to devise a mine plan which, in conjunction with current mine planning practices, went further to reduce long term impacts on the species in the mine area.

This involved the sacrifice of significant areas of mining reserves in the immediate forward mine plan and the forfeiting of some longer-term resource areas. A significant area of the immediate forward mine path was assessed as being prime koala habitat and as a result about 30% of this area was excluded from mining. Refuge islands of vegetation were retained in strategic locations where possible in the mine path and a habitat corridor system was planned to link undisturbed or older regenerating areas as best possible.

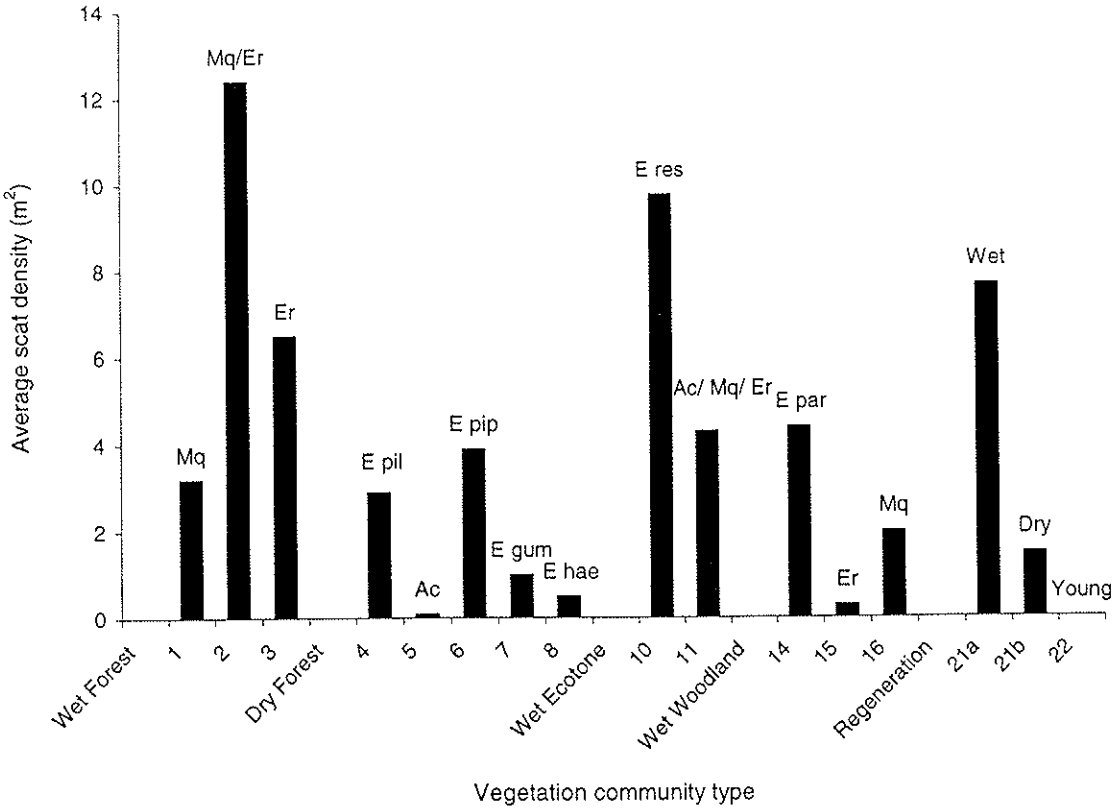

Fig. 1. Faecal pellet density

2.3 Site Rehabilitation

Rehabilitation of mined areas has progressively followed mining throughout the life of the operation with the main objective at this site being to re-establish native ecosystem environments similar to those existing prior to mining, consistent with the planned long-term management of the Sandbeds by Hunter Water Corporation for water harvesting.

A total ecosystem approach to site rehabilitation is adopted at the mine. This approach has generally been applied since commencement of mining operations, and has been significantly refined since the mid 1980's. It encompasses ideas typical of such programs such as:

- minimum disturbance philosophy;
- re-establishment of premining relief;
- management of site endemic ecological processes;
- habitat reconstruction to a similar diversity as existed prior to mining;
- respreading of cleared brush material; and
- use of local provenance seed stocks in all revegetation works.

In addition to these standard techniques, other site rehabilitation methods aimed at fauna management have included:

- strategic habitat corridor reestablishment. This is being carried out after research work in older regenerating areas had identified preferred use of this type of planting by arboreal marsupials.
- standing of stag trees. This has been undertaken, again in strategic locations to provide medium and long-term roosts and refuges for birds and arboreal mammals.

- installation of nest boxes in older rehabilitation areas. This has been tested with apparent limited success. However, it is planned to expand this program and to include their placement in stag trees also.

Field observations and anecdotal evidence suggests that the first two of these additional methods are of benefit to koalas (and other species) by providing movement protection and preferred browse within 10 years of establishment.

2.4 Site Management Protocols

In addition to minimising site disturbance other site management protocols operate as part of the company's ISO 14000 accredited environmental management system to help reduce direct impacts on resident fauna. These methods mainly comprise work instructions for site preparation where set procedures are followed to control clearing operations and ensure that observed or potential fauna are not injured and have time to escape to refuge areas prior to works being undertaken.

3.0 DISCUSSION

Like it or not, we all contribute to the problems of land and resource management by living in a society which demands an easy lifestyle, and this is particularly the case for so-called well developed countries such as Australia. We all use the products that are manufactured from mining, and some people forget that mining companies are in business to supply consumer demand.

Mining operations have the potential to affect dramatically natural environment processes such as ecological stability, hydrology and erosion at various scales, depending on the magnitude and influence of disturbance and the operating environment. However, these threats are seldom realised in Australia because careful planning and effective management controls are implemented to ensure that damage to the environment is minimised during mining activities, and that subsequent land use in the post mining landscape is sustainable.

Apart from the location of major titanium minerals deposits near population centres, the sensitivity of development pressures in the coastal environment is an issue of prime consideration in itself. It is the cumulative effect of all of these impacts which manifests itself in problems such as species decline (Reed and Lunney 1990). It is not necessarily the activities themselves, but the rate of the disturbance and habitat fragmentation (or degeneration) which is critical in this whole equation and must be managed. It should be realised that site rehabilitation is not a substitute for conservation of undisturbed habitat, and for this reason, successful management of fauna populations relies on responsible application of ecological principles in the planning stages of development as much as effective habitat reconstruction. Of course, environmental management and rehabilitation objectives are predicated on a scientific knowledge base which may or may not exist. In the case of commercial operations, the collection of baseline data may be limited by economic imperatives but this should be sufficient to back decisions with some degree of certainty.

Animal welfare interests in themselves are emotive and provide only short-term answers to fauna management problems, yet so many groups focus on this outcome. Welfare information that can be combined with scientific knowledge and incorporated into strategic ecological management programs can be used successfully to address important issues, such as the maintenance of viable koala populations, in an objective manner. Put simply, what is required is a population-based management approach, not one which focuses on individual animals. In this

regard, the implementation of a successful habitat rehabilitation process should employ system-based methods rather than ones aimed simply at returning the structural elements of the local environment.

Similarly, antagonists of mining often focus on the emotive and short-term issues when their resources may be better spent investigating the "bigger picture": confrontation does not usually lead to a positive outcome. Unless this is tactical, it demonstrates a naive understanding of the broader issues and how social, political and economic pressures all influence the decision making process. There is no substitute for detailed information when making technical decisions, such as on ecological issues, and in the case of the Tomago Sandbeds, RZM has sought to obtain an appropriate level of information on which to base long-term decisions that will help ensure the sustainability of the site's flora and fauna beyond the life of the mine.

4.0 CONCLUSIONS

Mining operations are a fact of life and active management is required to implement effective environmental controls and mitigate associated impacts. In the Tomago Sandbeds, some of the strategies employed in the mine planning process and techniques used in the site rehabilitation program have already proven to be effective in wildlife management. These strategies are based on a detailed knowledge of local environmental variables and the application of sound ecological concepts in their implementation. It is anticipated that the expansion of these ideas, whilst recognising that a certain time lag is inherent in the re-establishment of native ecosystems, will further reduce medium to long term fauna impacts and even enhance the rate of recolonisation into rehabilitation environments. The results of habitat assessment studies indicate that regenerating environments can provide satisfactory browse for koalas from about 10 years of age, and it is expected that social interactions will resume to ensure the long term viability of the species in the mining area. This information suggests that, contrary to some opinions, the local koala population will not be adversely affected in the long term as a result of mining activities.

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ECOSYSTEM RESTORATION FOLLOWING MINERAL SAND MINING - LEGACIES FROM FRASER ISLAND

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INTRODUCTION

Fraser Island, the world's largest sand island with an area of about 163 000 ha, is part of the Great Sandy Region - a series of large sand deposits comprising the Cooloola area north to Tin Can Bay and Fraser Island (Commission of Inquiry 1991). Fraser Island lies about 250 km north of Brisbane and is separated from the mainland by Great Sandy Strait, which broadens in the north to form the large shallow embayment of Hervey Bay (Figure 1).

Between late 1975 and December 1976 D.M. Minerals mined approximately 150 hectares on the southeast of Fraser Island within their lease, ML 102. At the end of 1976 sand mining on Fraser Island ceased as a consequence of the Fraser Island Environmental Inquiry 1975 (Commission of Inquiry 1976) when the overseas export of mineral sand derived from the Island was prohibited by the Commonwealth Government.

During the period of mining and until October 1977, rehabilitation of the lease was undertaken by the mining company. No further management intervention occurred at the site after October 1977, although in 1981 the sites were visually assessed (Cameron MacNamara 1981).

Because of continuing controversy about the management of Fraser Island, which has focused on commercial forestry, tourism and recreation, and proposals for extended conservation reserves and potential World Heritage Listing; the Queensland Government commissioned Mr Tony Fitzgerald to enquire into the future conservation, management and use of Fraser Island and the Great Sandy Region.

The enquiry reported to the Queensland Government in April 1991, and made 76 recommendations including the following (No. 65) which related directly to mining:

> "Mining be prohibited in the region and all mining leases and mining lease applications be surrendered or revoked, subject to the possible qualification provided for in the following recommendation."
>
> "The plan of management incorporate a mining management strategy"

This mining strategy envisages the possibility of mining areas on the two main remaining leases on the island located on the eastern side south of the village of Eurong. The possible mining areas are north and south of the areas previously mined by DM Minerals in 1975 and 1976.

The report concludes that:

> "Consideration has been given to a mining management strategy to provide environmental safeguards to ensure that any mining which proceeded was compatible with the inscription of the region on the World Heritage List.
>
> "An appropriate set of conditions could be formulated, based on the conditions recently imposed by the Commonwealth Government in connection with proposed sand mining in Shoalwater Bay."

The fact that the Commission contemplated mineral sand mining within a potential World Heritage Area is a significant change from the attitude of the 1975 Fraser Island Environment Enquiry. It suggests that with further demonstration of the technical capacity of the industry to achieve restoration objectives, further flexibility in land use allocation for mining may exist.

The research upon which this paper is based, was commissioned by the Pivot Group, who now hold the former D.M. Minerals leases, to provide an assessment of the present status of ecosystem restoration on the mined sites as part of their submission to the 1991 Fraser Island Commission of Inquiry.

Field work was conducted between mid June and mid July 1990. Details of the methods employed and the results of this work are available in Bowman and Little (1990) and Buzer et al. (1990).

In the short time available for field work, objectives had to be kept modest. Consequently we sought to identify merely the extent of reconstruction of the landforms of ML102, and landform processes as they effect the spatial pattern and dynamics of the reconstructed ecosystems.

- the extent to which post mining vegetation floristics and structure resemble the unmined condition; and
- the extent to which the macro fauna, in particular the fauna of the soil and litter of the reconstructed ecosystems reflect the unmined condition.

The study represented a rare opportunity to evaluate a restoration program after the comparatively long period of 13 - 15 years from the completion of mining. It is especially significant as it relates more directly to north Australia than any other mineral sand mining experience to date and we believe it has lessons for the future direction of post mining restoration practice. In this respect it addresses the issue raised in the 1976 Fraser Island Commission of Inquiry that:

> "The successful rehabilitation of vegetation affected by mining would be unlikely".

ECOLOGICAL RESTORATION AND MINERAL SANDS MINING

Revegetation of sites following mineral sand extraction has been an integral part of the mining process from the time mining operations moved beyond the beach and into the older vegetated dune systems (Brooks 1988). Public pressure has encouraged the rapid evolution of post-mining revegetation objectives. The initial objective of stabilising disturbed sand using an appropriate cover crop has been replaced by multiple objectives which range from the reconstruction of pre-mining landscapes to the provision of land suitable for a variety of other land uses including agriculture, urban development and recreation (Brooks 1989).

In the Queensland mineral sands industry the objective of post-mining treatment in recent years has been entirely directed towards the reconstruction of original landscapes. To distinguish this purpose we use the term **restoration** to describe the actual painstaking recovery of the plant and animal communities native to the area (Jordan et al 1987).

The major shift in thinking on behalf of the mining industry, must now be further impressed upon the Australian ecological community, from whom much of the expertise for implementation of restoration must come. It is clear from recent edited publications such as Jordan et al. (1987) and Buckley (1989) that restoration ecology and landscape ecology is becoming imbedded in mainstream ecological thinking and practice in Europe and North America. However, apart from some didactic papers, such as that of Westoby (1984) there is not a lot of evidence that ecosystem restoration is being grasped in the same way in Australia, where it remains largely the province of practitioners employed by the mining industry.

Restoration ecology has a long history in applied ecology but only in the past decade has there been a concerted effort to place what has been basically good horticultural practice into

n emerging theoretical framework. Jordan et al.(1987) provide number of examples where the practice of ecological estoration has contributed uniquely to deeper theoretical nsight into the functioning of ecosystems. Indeed Bradshaw 1987) refers to restoration as 'the acid test' against which ecological theory must be evaluated.

The theoretical formalisation of restoration practice in relation to the mining industry is important from a practical point of view since it is only from this theoretical 'high ground' that we are able to rigorously assess how good our restoration activities have been.

In 1990 the Pivot Group were able to propose the following widely accepted objectives for successful post-mining restoration (Pivot Group 1990).

- that the plant community (should reach) a state of being self sustaining, that is not requiring any further human assistance;
- a reasonable representation of the original structure, species and functions (should be) achieved; and
- that faunal habitats similar to those existing prior to mining (should be) re-established.

These criteria incorporate the five criteria defined by Ewel (1987) for assessment of restoration success - sustainability, invasibility, productivity, nutrient retention and biotic interactions.

The technology required for restoration of sand mined areas has been developed progressively over the last 30 - 40 years from a combination of agricultural and ecological insights, and appears to be successful in the most common situations (Brooks and Bell 1984; Rogers 1982).

Nevertheless there remain a number of concerning and difficult issues that have not yet been resolved. In particular the results of rehabilitation at similar stages after mining appear to vary unpredictably in terms of dominant plant species and structure within a region (e.g., Stradbroke Island - P. Foot pers comm. 1990) and between regions (e.g. central NSW and Stradbroke and Fraser Islands (J. Lewis pers comm. 1990). Furthermore the length of time required for reconstruction of high dune sites can still only be guessed at (Rogers and Mokrzecki 1984).

Three factors influence the resolution of these issues. The first is differences between the physical environments of mined sites and the fact that initial and post establishment treatment has changed, as technology and approaches have evolved. The second is the body of knowledge which is available for undisturbed high dune systems - a subject that has been profoundly addressed for the Cooloola Sand Dune Mass by CSIRO Division of Soils (Thompson and Moore 1984). The third is that there is very little detailed monitoring of long term change on

mined sites. The Bridgehill Ridge in the Myall Lakes is one exception (J. Lewis pers comm.1990) and when the data is published it will provide a most valuable benchmark.

THE PRESENT STATUS OF RESTORATION ON FRASER ISLAND

Topography

The topography of the lease area before and after mining was determined photogrammetrically from stereoscopic aerial photography captured in June 1958 and September 1986, respectively. As there were no accurate field datum points precise measurements were not possible; however, relative contour intervals of 10 metres were achieved for both situations with five metre contours in some cases.

This analysis revealed pre-mining, parabolic dune ridges up to 1 000 metres in length, rising from 10 metres to 65 metres above the coastal wetland area adjacent to the beach. The regular southeast to northwest alignment of the dune ridges is consistent with the pattern seen throughout the entire Fraser Island parabolic dune system. Aspect in relation to this dominating topography is a major ecological parameter.

At the time of mining, Queensland Government lease conditions restricted slope restoration to a maximum of 20 degrees. In consequence, the reconstructed topography is much reduced (Figure 2). Significantly, there is also a lack of directional ridges which has simplified the habitat diversity within the mined areas and this is reflected in the pattern of the plant communities now on the mined sites.

Soils

The most notable feature of the post-mining soils is the retention of the 'returned topsoil' layer in almost its original condition after 14 years. Little mixing or pedoturbation has occurred.

Evidence of active soil formation was not apparent throughout most of the mined area. The early stage of incipient podzolisation was indicated at one site where small bleached zones, iron staining of the topsoil, and an organic line precipitated at the topsoil-subsoil boundary was observed. This result contrasted with the very rapid incipient podzolisation observed after only 4.5 years under similar returned topsoil in mineral sand mining areas at Myall Lakes, NSW (Paton et al. 1974). The reason for the marked contrast in rates at these two east coast sites needs careful examination.

Levels of triacid extractable Na, K, Ca and Mg tended to be higher throughout the unmined soil profiles than in the mined profiles, especially on stable sites such as dune crests. Oxalate and pyrophosphate extractable aluminium and iron were higher in the mined sites compared to the unmined sites.

Accordingly, the ratios of triacid to oxalate extractable, and triacid to pyrophosphate extractable, aluminium and iron are markedly higher in the unmined soils. This result suggests that mobile forms of iron and aluminium are present in greater amounts in the mined soils as a result of the artificially emplaced topsoils.

In terms of nutrient status, calcium, phosphorus and carbon are consistently lower in the mined soils where regrowth is poor than in mined sites where regrowth is good or where there is original forest. Low calcium and phosphorus status may be limiting plant growth, although the values are not low compared to many of the older soils on the island (Little 1985).The status of trace elements is not known.

In general the major nutrient situation bears fair comparison to natural soils, especially as the original vegetation has a low nutrient requirement and the mined soils do contain minerals (notably delta Al and delta Fe) thought to be valuable sinks for nutrients in soils with very little clay.

There may be soil physical problems in the mined areas. Infiltration rates in the mined areas were observed to be substantially lower than in the unmined areas, an observation consistent with the apparently greater surface compaction and surface displacement of sand by hydraulic action. Mined areas throughout the site contain sheets of surface colonising moss presumably responding to moist surface conditions on the reconstructed slopes; but perhaps also indicating the long term affect of phosphorus additions in the fertiliser applied in the initial post mining planting (Specht 1963).

Mined sites which were underlain by rich ilminite stockpiles showed vigorous plant growth, perhaps related to the better water holding capacity of the fine-grained ilminite, although these sites were also fortuitously associated with high values for oxalate extractable aluminium, which would also help the nutritional status of the soils.

Vegetation

The mined areas of the lease have been successfully stabilised by plant species native to the area. However, the success in restoring original species composition and plant community dynamics has been more patchy and problematical.

A total of 107 vascular plant species were recorded with species richness varying from 13 to 19 per 0.1 ha. There was no consistent difference between the mined and unmined sites on the Lease, although the species richness of the control site just to the north of the lease area was higher at 21.5 per 0.1 ha. Only 10 species were found on the control site that were not present on the lease; but as we argue elsewhere (Buzer et al 1990) their absence is unlikely to be related to process change to the mined sites and it is expected that they will

arrive with the further passage of time.

Nevertheless, much of the mined area, especially in the northern two-thirds of the lease was still essentially monogeneric communities of **Acacia**, particularly **A. leiocalyx** but also **A. flavescens** (Figure 2). This association is very different from that of the original forest where an irregular canopy stratum is dominated by mixed eucalypt species with a complex sclerophyll shrub understorey. In this association mature **Acacia** is rare, although present in very high densities in the seedling layer (Table 1).

Stand structure was patchy, varying from dense and regular canopy cover where there was effective elimination of all other canopy species, to less regular and more open stands where other canopy species, notably **Eucalyptus signata**, were present. At these sites the eucalypts were typically stunted and most carried ringbarked branches due to heavy insect predation. In the south of the lease restoration appeared to have been more successful with species composition and stand structure more closely resembling adjacent undisturbed sites, although acacias still tended to dominate the canopy.

Restoration appeared to have been most successful along the western side of the lease where vigorous growth of **Eucalyptus signata** is associated with a complex, species-rich community with several strata and a number of age cohorts. Why this area is so different from that immediately to the east and north where **Acacia** spp dominate the canopy and there is essentially no understorey, is not clear.

Post mining treatment at all sites was reportedly identical. A suggestion that differences among seral communities may be related to the time of the year when initial planting occurred (c.f. Brooks and Bell 1984) conflicts with the account of the restoration programme provided by the former Site Supervisor (D.S. Bower pers comm. 1990). Mr Bower emphasized that intensive supplementary planting followed initial treatment throughout the area. Seedlings and seeds were reportedly planted throughout the mined areas of the lease following rain episodes until October 1977.

The foliage projective cover (FPC) measurements range between 42 and 61% as predicted for mature plant communities in this climatic region.

The data, summarised in Table 1, on stand basal area and density, seedling recruitment and mortality in mined and unmined areas of the Lease and at the unmined area to the north of the Lease reinforces the emerging picture of **Acacia** dominance over the mined portions of the Lease area. This picture contrasts starkly with the unmined sites and with the natural 'control' site to the north of the lease. The high stand basal areas in the unmined sites compared to the mined sites are a reflection of age of dominant trees in particular

large **Angophora costata**. This is further reflected in the total stem density figures.

Future dynamics of the **Acacia** dominated areas are more difficult to predict. There was a much higher density of dead stems in the mined sites due to die back in the canopy stratum of the wattles and far fewer seedlings. The absence of seedlings and the generally dense and compacted layer of litter on the ground raises questions about the future direction of change in the community. Will seeds of other species, dispersing into the **Acacia** dominated areas be able to germinate, and become established ? Or will the canopy progressively open, drying the litter and increasing the risk of fire, which may further encourage the germination of **Acacia** seeds stored in the litter, and releasing nutrients, notably Phosphorus (Heddle and Specht 1975).

Comparative observations of the mined areas at Inskip Point also dominated by **Acacia** communities are quite interesting. Here the communities are up to 12 years older than at the Fraser Island sites and the process of senescence and death is well advanced. At these sites there has been a significant invasion of coastal cypress **Callitris rhomboidea** suggesting that at least this species is able to germinate and establish. However, more information is required to examine the conditions under which species change occurs. There remains the possibility of continuing cycling of **Acacia** from generation to generation.

Invertebrate Soil Fauna

Preliminary surveys of the ground dwelling ant fauna, the litter fauna and the epigaic fauna were completed as part of the assessment of restoration. The invertebrate fauna are generally ignored during ecological assessments such as this because of the specialist expertise that is required to sample, identify and interpret the results of species distributions and abundances. An exception is the survey commissioned by Cliffs International Incorporated in the West Angelas area of the Pilbara (Majer 1982).

Ants, because of their numerical abundance, size and species richness are a prominent taxonomic group and appear to be closely related to specific ecosystem types (Greenslade and Thompson 1981). An investigation in nearby Cooloola National Park revealed over 300 species, which is on a par with observations from some rainforests in Papua New Guinea (Wilson 1971). Ants play a highly significant ecological role in processes of soil conditioning and organic decomposition. Majer (1984a,) reports that there is a successional wave of ants species through time in land revegetated after mining.

The results obtained in the present survey are still preliminary because of the restricted period of the study. The present work did not include those species which primarily use

trees and shrubs. Furthermore the comparisons of mined with unmined sites were within the mining lease only. Collections from areas outside the lease would have been valuable as a check on the probable peripheral disturbance through mining of the small patches of unmined dune sites on the lease.

A total of 27 major invertebrate taxa were extracted from the litter samples collected from the unmined and the mined sites. Fifteen ant types/species were recorded in the litter. Nine of these species were recorded in the unmined sites and ten in the mined sites

The major invertebrate taxa were represented in similar proportions in both mined and unmined sites. However there were considerably more individuals present in the unmined litter. The majority of these are the Acari and Collembola. These two groups are strongly implicated in the decomposition of organic matter and mineralisation. The more of these animals present the more rapid is the decomposition of organic matter and the release of nutrients in a form available to plants (Reichle, 1977). It is possible that the litter in the mined sites is less palatable to invertebrate fauna than that in the unmined sites because of the higher phenolic content of the dense acacia litter. Litter such as this, particularly if water-logged for any time, has a slower rate of decomposition and consequent nutrient release (Plowman, 1979).

The epigaic fauna of all the sites are similar, although in the two unmined sites sampled, individuals appeared larger and were moving actively about the soil surface. Collembolans were the dominant group and were mostly comprised of a few relatively large species and or genera. The numbers of individuals seems to be related to the depth of the litter cover.

Ten species of ants were recorded in the mined sites and nine species in the unmined. Evenness was also higher in the mined sites. Wallace (1989) reported that after fire the numbers of individuals increased on burnt sites while the number of species dropped. Majer (1982; 1984a; Majer et al, 1982) notes that successional waves of ant species can be tracked through time in regrowth vegetation, with changes in the numbers of species present at any one time.

While there was considerable overlap of species between mined and unmined sites there were also ecologically significant differences.

Two species of **Leptomyrmex**, regarded as dominant species in coastal sand dunes at Cooloola (Greenslade and Thompson, 1981), were recorded in the unmined sites. This genera was not recorded in the mined sites.

The soil surface of mined and unmined sites were also substantially different and appeared to be used by the fauna in a more patchy fashion.

There was little ground cover vegetation in the unmined sites and the litter appeared more compact and less disturbed. The litter in the mined sites was predominantly made up of organic matter from one plant genera, **Acacia**, and the litter probably contains high quantities of phenolic substances. This is presumably the reason that preliminary soil analysis suggests that unmined soil pH may be higher (i.e., less acidic) than those in mined sites.

The interface between the litter and the soil was clearly demarcated in the mined sites, whereas in the unmined sites there were fragments of organic matter in the surface sand. The surface in the unmined sites was generally mobile sand stabilised by vegetation, whereas in the mined sites the surface was generally compacted. **Aphaenogaster** sp nests were noted in the unmined sites. This genera is characteristic of the coastal dunes and nests in mobile sand (Greenslade and Thomson,1981). Species of this genus were not recorded in the mined sites. Bandicoot diggings in the mined sites indicated that these were the only obvious fauna affecting soil development. (It is interesting to note that Eneabba restoration studies by Majer et al, 1982 the soil was also difficult to penetrate.)

ECOSYSTEM DYNAMICS - EVALUATING RESTORATION

How successful has the restoration program been ? Although our work is still very preliminary the following responses can be made.

A plant community has been established on the Lease area which in several respects reflects the floral and faunal diversity of the site prior to mining.

However, it is also clear that the plant community, the litter and soil fauna and the soil environment itself are functionally very different from the unmined condition. Thus, while it is most likely that an ecosystem will continue to mature on the site, it is not at all clear in which direction, or at what rate, further change will occur.

The ecological status of the regenerating ecosystem is unexpected and complex. Nevertheless the result is intriguing and valuable. Once the present and future dynamics of the site are better understood, we believe they will offer important insights into the future practice of restoration on these coastal sand-heath environments of the sub-tropics and tropics.

The result obtained on Fraser Island conflicts with experience of restoration of sand mined areas on the central NSW coast (Dr J. Lewis pers comm.), Western Australia (Brooks 1989) and parts of Stradbroke Island (Brooks 1989; Rogers 1982), where regenerating sites are as old and older than at Fraser Island. However, it is consistent with some patterns emerging from high dune sites of similar age on Stradbroke Island (P. Foot pers

comm.)

We propose that part of the explanation of these important differences between southern and northern Australia lies in the longevity and life history of the colonising species of Acacia.

In the central NSW areas, the species of **Acacia** is **A. longifolia**, a shrub species which does not occur naturally in the coastal heath communities (R.L. Specht pers comm. 1990). **A. longifolia** grows to a height of about two metres only, and is relatively short lived at about four or five years (J. Lewis pers comm.). The species is planted and also recruits easily to a mined or otherwise disturbed site and serves a valuable role in stabilisation and in modifying the microclimate in favour of the longer lived, slower growing species of eucalypt that form the canopy of the mature community.

This pattern of seral replacement of an early colonising species within a short time frame of five years reinforces a widely held understanding of acacias as colonising species. It also conveniently suits a pattern of post-establishment monitoring and supplementary maintenance, which is common practice in sand mining restoration. Such an interpretation would appears to have underlain the assessment of progress in the restoration of ML102 on Fraser Island by the late Dr J.E. Coaldrake (Cameron and MacNamara 1981) about five years after operations were finalised on Fraser Island by D.M. Minerals.

Dr Coaldrake's conclusion from his examination of the site was that:

> 'The early dominance of acacias is now waning over much of the area as eucalypts and other types of plant become prominent. There is already an adequate variety of native plants present for development of distinct groupings (communities) of native vegetation in a mosaic suited to the rolling landform. These processes are slow and will need to continue for many years yet, but this is vigorous vegetation on stable land with seed set and natural reproduction now happening'.

This interpretation of the status of the restoration work is not consistent with the recollection of the status of the site in October 1977 by the Site Supervisor, Mr D.S. Bowers when D.M. minerals left the island (Bowers unpublished ms 1990).

By October 1977 it appears that a good establishment rate of seedling tree and shrub species had been achieved throughout the mined area. Initial plantings of at least ten major canopy species had been undertaken on a randomised pattern that reflected 'relatively small scale vegetation differences' across the site prior to mining. The initial planting of seedlings was followed by supplementary planting of further seedlings and seeds throughout the area during the period of mining and subsequent restoration work, in order to take advantage of periods of rain. As a result of this treatment, a

diverse assemblage of prospective canopy species had reached a height of approximately one metre. Seedling acacias, germinating from a seed store in the mulch, were prevalent, but still small.

Thus, between the latter half of 1977 and mid 1981, a stratum of established **Acacia** seedlings appears to have grown through the canopy of planted stock and to have succeeded in overtopping the earlier well-established eucalypts and other species.

It would appear therefore, that the situation observed by Dr Coaldrake was an intermediate stage in the competitive replacement of the planted stock by a vigorously growing cohort of 'naturally' recruited acacias. Whether a competitive advantage was gained by the acacias during the intervening dry years of 1977 to 1981 (Bureau of Meteorology records obtained for Eurong 1974 - 1977) is not clear, but is likely. Dr Coaldrake refers to the vegetation 'continuing to improve despite several years of low rainfall that lowered the water in the lakes to some of the lowest levels ever known' (Cameron MacNamara 1981). However, he also observed that the **Acacia leiocalyx** and **Acacia flavescens** also suffered severe dieback during this period.

Acacias may have additional competitive advantages. Jehne and Thompson (1981) argue that endomycorrhizae are important to plants colonizing coastal sand dunes in the subtropics. These fungi are able to utilize phosphorus bound in the thin sesquioxide coatings around sand grains. The application of mulch and leaf litter would have introduced soil fungi into the mined site and must have contributed significantly to the acacia seed store. Addition of phosphorus in the fertiliser applications, combined with the capacity of the acacias to fix nitrogen, may have also contributed to the long term success of the acacias in comparison to other species. This question bears more detailed examination.

This pattern of vegetation development appears to have culminated in a canopy cover of acacias over much of the mining lease, sufficiently dense to have excluded and removed the original eucalypt and other species from much of the Lease area. It resulted from an agressive 'natural' recruitment to the disturbed sites by **Acacia** species which were also comparatively long lived and had the life form of a small tree rather than a shrub. The dynamics of these species, in particular the dominant **Acacia leiocalyx**, which may live to around twenty years (R.W. Rogers pers comm.) may tend to a cyclic recolonisation of a site rather than seral replacement by other species.

Ecophysiological studies of these acacias would be valuable in understanding their competitive capacity in environments that may be seasonally and periodically very stressful. As an analogue, it appears that the life history of the species and

the dynamic pattern of the community that they dominate, may closely approach that of the brigalow **Acacia harpophylla**, and other related tree acacias of the semi-arid tropics.

In the southern part of the lease, where replanting was not preceded by mulching green brush, the acacias are less abundant, the eucalypts more common and vigorous. In the central part of the lease area, which was initially mulched with green brush, an open and senescent stand of acacias is separated from a vigorously growing stand of eucalypts, dominated by several cohorts of **Eucalyptus signata**. It is not clear whether these different responses reflect lower seedling densities of acacias or growing conditions which favoured seedling eucalypts (e.g., aspect, altitude, water conditions, nutrient availability, topsoil conditions or time of initial planting (cf., Brooks and Bell 1984; Majer 1985). Slope appears to have been an important factor.

Site dominance by **Acacia leiocalyx** appears to explain a number of other anomalous situations observed in the litter and the soil fauna, which, combined with greater soil compaction following mining, has contributed to the poor development of the surface soil profile within the A Horizon.

While there is considerable variation between quadrats, this initial survey indicates a deeper and more compacted layer of leaf litter, capable of retaining sufficiently high levels of moisture to impede oxygenation of the underlying soil. Combined with acid conditions and an apparent 'crustiness' in the surface soil, there has been little penetration and pedoturbation of the soil profile by interstitial fauna. The acid conditions may also have modified the soil microflora and fauna, favouring fungal activity over that of soil bacteria. In addition, colonization of the soil is influenced by the quality and diversity of the organic resource (i.e., leaves and wood). Litter such as this is slow to decompose as it is not an attractive resource for many soil organisms. The decomposition and mineralisation processes themselves act to maintain particular soil/litter environments and prevent colonization by other organisms, including plant species.

The direction of further change in those areas of the mining lease dominated by **Acacia** is unclear. Without further intervention, a likely hypothesis is for a continuing cycling of new generations of the present dominant canopy species. The few persisting seedlings of eucalypts, and other species, e.g., **Callitris**, may gain ground gradually during the cycling process, but such floral changes would probably be threatened by fire, which would favour the **Acacia spp.**

Fire may further exacerbate problems with restoration by liberating macronutrients. Specht has argued against the use of phosphate fertilisers, suggesting that many species of heath and dry sclerophyll plants respond adversely to imbalances in the levels of phosphorus, nitrogen and possibly potassium

(e.g., 1963, 1975, 1976, Specht et al 1977; Specht et al 1984; and see also Lewis 1976). This imbalance is set in train by the use of phosphate fertilisers.

Specht (1976:113) argues as follows:

> "Theoretically, a balanced fertiliser application can be devised for revegetating heath/dry sclerophyll forest vegetation on sand-mined areas. However the critical stage is many years later, when a bush fire, a natural part of these ecosystems, occurs. At this stage nitrogen escapes from the ecosystem; phosphorus stored in the foliage and stems is released as ash and some is released from formerly unavailable compounds in the soil. An imbalance between phosphorus and nitrogen results, leading to 'phosphorus toxicity' in the species which impart long term stability to the community" (cf., the 'structuring species' of Westoby 1984).

The result is the death of sensitive seedlings in regrowth after fire. The problem does not affect the short lived pioneer species apparently because they are capable of fixing nitrogen or have large nitrogen stores in their seeds. Specht and his associates further argue that the growth (particularly in the long term) of long lived heath species is also affected by the use of fertilisers.

CONCLUSIONS

There are major differences between the vegetation of the mined sites on ML102 and the undisturbed coastal dune ecosystems in the unmined portions of the lease and in undisturbed areas immediately to the north. Certain portions of the mined area do resemble more closely the undisturbed condition and this is presumably self-sustaining. However, these areas are small compared with those dominated by acacia. The reasons for this difference are not yet completely understood but are of considerable scientific interest.

On the basis of our observations on Fraser Island, restoration of the pre-existing dune ecosystems appears to depend upon the successful implementation of at least these three strategies:

- land form reconstruction which pays special attention to the orientation and shape of the major features.

- a revegetation program that is based upon an understanding of native ecosystem processes, including seral changes in the vegetation and in the fauna, decomposition, mineralisation and nutrient recycling, and soil profile development.

- continuing monitoring and management even beyond the point in time where the restored are is considered to be self-sustaining.

Notwithstanding contractual and engineering safety considerations, land form reconstruction needs to ensure the maintenance, as far as possible, of the original aspect and slope. The **Acacia** dominant areas of the mined areas on Fraser Island are generally flat, the more diverse vegetation occurs on the slopes.

Topsoil and mulch should be stored for the least possible time to minimise losses in seed viability and mycorrhizae, and changes in soil chemistry. The methods applied on Fraser Island should have achieved this, and the use of innovations such as venturi 'vacuum cleaners' to transfer leaf litter appear still to be still very appropriate.

Tongues of natural vegetation left within the mined areas provide recruitment sources for native seeds and the critical elements of the fauna. The retention of patches of native dunes within the mining lease was a significant benefit to the results observed in this study.

In retrospect the resources that were put into planting potential canopy species did not yield significant long term benefit. It may therefore be more appropriate to consider local species which colonise disturbed ground (such as **Acacia leiocalyx**) as the primary vegetation and invest restoration effort in supplementary interplanting of additional desired native canopy species several years later. The most appropriate way of implementing such an approach will require further consideration.

On Stradbroke Island Consolidated Rutile are overcoming the problem of domination of the canopy by mature acacias by selective physical removal.

The hypothesis that the use of phosphate fertilisers was a critical factor in tipping the balance between the desired heath/eucalypt community and the acacia dominated community appears to be strongly supported. A restoration procedure which does not use artificial fertilization should be the subject of further experimentation and development.

The acacias on the mined sites on Fraser Island are now reaching the end of their life. The extent to which the next generation represents a cycling or a succession of species is of considerable scientific interest.

Studies of the progress of sequential development of plant communities and other biological indicators would be of great benefit in predicting and planning restoration programmes for areas disturbed by mining and other activities in sub-tropical and tropical coastal sand environments.

Maintenance and monitoring will need to be undertaken for a number of years. In these sub-tropical/tropical environments time frames of 15 to 20 years may be more appropriate than the

more commonly adopted practice of five years, because of the longevity of the major competitive species.

There is clearly much more to be learnt and understood about the restoration of diverse, multi-species ecosystems. We suggest that more attention needs to be directed to understanding the relationships between microflora and soil faunal colonisation, and the natural processes which maintain soil fertility and plant growth within heath/sclerophyll habitats. Techniques need to be further modified to more closely align with the processes occurring in these ecosystems.

There is no doubt in our minds that the lessons that are contained in the experiences on Fraser Island,and in other sites where mine restoration work is in progress will assume a central place in the management of a variety of human ecosystem impacts, as land and other natural resources become limited and in demand from various sectors in the community. The management of tourism and recreation impacts will particularly benefit from the experiences which have been generated by the experience of the mining industry.

The Fitzgerald Inquiry concluded:

> "While land systems such as those which contain economic mineralisation on Fraser Island can be revegetated to a self-sustaining level of development, the information available does not indicate that pre-mining ecosystems could be re-established. It is not demonstrated that revegetation could fully satisfy land rehabilitation objectives."

Just as the passage of time has allowed the Commissioner to determine that the areas mined on Fraser Island can be rehabilitated to a self-sustaining level, it is likely that eventually it will be demonstrated that the plant community structure, species richness and botanical composition and wildlife habitats which are reasonably representative of the pre-existing situation can also be achieved.

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BRIDGE HILL RIDGE — A REVIEW

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ABSTRACT

The mining of Bridge Hill Ridge for mineral sands from 1974 to 1983 by BHP Titanium Minerals was preceded by up to 15 years of exploration, planning, and permitting, and has been followed by a similar period of rehabilitation works and management. Environmental excellence has been recognised at community, industry, and government levels.

This review returns to the 1973 EIS and re-examines the performance against the statements made in that document.

1.0 INTRODUCTION

In 1991 New South Wales Mineral Industry conferred upon BHP Titanium Minerals (formerly Mineral Deposits Limited) the Environmental Excellence Award for its mine rehabilitation at Bridge Hill Ridge. The company has now been responsible for almost 40 years of integrated planning, mining, and land management on this site.

Prior to the development of Bridge Hill, the Company had been active in the extraction and export of Rutile and Zircon won from diverse coastal locations from Brisbane, Queensland, to Tea Gardens, New South Wales. It had established integrated mining and processing facilities centred on Southport, Queensland, and Crescent Head and Hawks Nest in New South Wales. It had worked closely with various government departments to further develop and promote techniques in surface stabilisation and the regeneration of native species in a post-mining environment.

2.0 DISCOVERY

A substantial mineral-bearing sand ridge was located during exploration in the early 1960s, adjacent to the shores of Smiths Lake and Myall Lakes. During that decade, assessment of the deposit proved-up an orebody of 240 million tonnes of ore containing over one million tonnes of heavy minerals.

3.0 APPROVALS

A Development Application was outlined to the local Council in July 1968, formally submitted in March 1971, and received Council approval. As was provided in the planning instruments in force at the time, this approval required also the concurrence of the State Government agency, the State Planning Authority (SPA). This was not forthcoming due to concerns stated by the SPA relating to the stability of the ridge post mining, the probable inability to restore the dunes to their original height, the risk of failure in re-establishing a vegetative cover and the adverse impact on surrounding ground and surface water quality.

An appeal under the provisions of the Local Government Act was lodged by the Company and resulted in an inquiry being set-up, taking submissions from parties from March until August 1973. To support its case, the Company prepared an Environmental Impact Study

containing the scientific and engineering data relevant to the project. The EIS process was novel to Australia at that time, and was one of the first to be so presented. This document played a significant part in the process of appeal. The preparation of the EIS was supervised by the Company senior mining engineer assisted by the Company Ecologist, who had been engaged in 1971 and had been involved in formalising rehabilitation procedures on other high dune minesites. An approval was eventually issued for a reduced area, viz. 40% of the application.

3.0 OBJECTIVES AND ACHIEVEMENTS

3.1 Mining

3.1.1 EIS — "To recover the mineral resource by controlled mining within the area delineated ..."

Development commenced in the middle of 1974, commissioning took place late in that year, and the operation was expanded the following year to its designed capacity. The equipment consisted of a floating concentrator, capable of 12 million tonnes per annum, fed by two independent cutter-suction dredges. The mine life was estimated at 8.2 years.

Throughout the life of the mine, the format of the operations was essentially that originally proposed. However, to meet a sunset clause in the consent, plant capacity was increased by a notional 200 tonnes per hour towards the end of the mine life. The actual mining life was just over 8.5 years. Mineral recovery rates were as expected, there being no appreciable metallurgical difficulties throughout.

3.1.2 EIS — "To replace all the tailings in the excavation"

The approved plan of operations generally allowed a service buffer thirty metres wide around the mining path. Other than for the planned procedures associated with the initial mine development, all processed rejects were successfully placed within the mine excavation footprint. This target was met despite the need to alter the angle of repose of some dune slopes to facilitate rehabilitation procedures, and the relocation of features up to one kilometre (the length of the mine excavation) from their origin.

All of the tailings from the operation, about 99.7% of the dredge feed volume, were continuously pumped as a sand in water slurry to the rear of the concentrator. The placement system consisted of a low-level jetting unit, a mid-level jetting unit, a mid-level landline unit and a high-level landline unit. Both landline units had pulp de-watering facilities to assist in stabilising the dune under construction. All pumps had speed and density control. Stability of the mass was further enhanced by bankfoot de-watering wellpoints and bores to intercept the slurry water and recycle it back to the operation. Stability of the containment berm and access roads was ensured by the installation and daily monitoring of water table observation wells on the wetland side of the mine.

As placement was completed to the predetermined volumes in the tailings management plan, contouring and gradient reduction was effected by profiling with bulldozers. Formal placement plans were in active use throughout the operation, being reworked several times a year to account for minor variations in actual progress and altered conditions. Monitoring of operations for these purposes was carried out on a weekly schedule without exception.

3.1.3 EIS — "To reshape the tailings to ... slopes of at least 25 degrees where appropriate, to heights no less than existing and to blend ... into the undisturbed land forms ..."

One of the concerns aired by the SPA in their initial refusal of consent was that the dune system might not be re-constructed to its original spectacular heights, i.e. over 100 metres. The Company was able to achieve full height by designing and introducing a tailings placement system which had the required height ability and which was manageable in its daily operation.

Dune reconstruction yielded slopes of 25 degrees in most fold valleys and mid-slope falls. The process of dune steepening through micro erosion has been observed in action as the vegetative cover matures, indicating that this natural process, which was partly responsible for the pre-mining topography, continues. Experimental work off-site had previously determined the limit of practical slope reconstruction. The combination factors of fixed footprint and reduced slope meant that heights would not just be attained, but had to be exceeded on most occasions.

Sidebanks varied in elevation from 60 to 100 metres and were successfully blended into the reconstructed dune by moving the dune crest closer to, or further from the sidebank. In many places, the dune crest is considerably higher than the sidebank.

3.1.4 EIS — "To reshape the tailings into a sandridge which undulates in longitudinal section"

The need to meet parameters previously mentioned would generally not facilitate variation in longitudinal relief as the available volume is diminished each time a restriction is placed on the system. We find, however, that elevations ranging from 60 to 90 metres have been achieved and the greatest length of "flat" crest is not much more than 300 metres, all in a dune system almost four and a half kilometres long.

The mobility built into the tailings placement equipment permitted regular undulation and was augmented on three occasions by a instituting planned relocation or "step-up" which created a substantial feature. The monitoring of this process was a major agenda item of many of the annual committee inspections during operations.

3.1.5 EIS — "To establish a wind-erosion resistant cover of plants proceeding ... an ultimate forest cover ..."

A key element in successful rehabilitation is the conservation of the humus and nutrient-rich layer of soil commonly termed the "topsoil". In the relatively young soils of the Quaternary sand dunes such as existed at Bridge Hill, there is a clearly definable junction between the dark grey topsoils and the underlying leached-white or orange-stained subsoils. The capacity of the latter to support a complex surface vegetation matrix is low, therefore the conservation of the former is essential.

As the Bridge Hill mine advanced only ten metres per week, and the operating mine was generally over one kilometre long, it was recognised that excessive topsoil storage time would be experienced if the standard method of stockpiling soil at the mine side was deployed. A storage time of up to two years would have resulted, and both the soil nutrient levels and seed bank would have suffered unacceptably.

The method proposed and adopted was to remove the soil on a continuous basis at the mine face and transport it to the rear of the mine for almost immediate spreading over prepared tailings. Thus the critical time from "soil off" to "soil on" was shortened to between one month and three months, depending on seasons.

Following the transfer of topsoil from the mine face to the recontoured tailings, specially equipped low ground pressure bulldozers spread the topsoil down slope to both sides of the mine path. The light ground-pressure characteristics of the machines permitted tracking both down and up the slope without contamination of the topsoil by the underlying tailings.

To prevent wind and water damage to the surface, a cover crop was established soon after the spreading of topsoil. Usually a hybrid sorghum was used, occasionally a cereal rye, depending on soil temperatures at the time. Sowing rates were generally 10 kilograms per hectare and attended by an application of fertilizer (N:P:K ratio 10:4:4), applied at about 350 kilograms per hectare. This crop was always established from certified sterile seed to avoid any long term establishment of any exotic species. With only minimal regrowth in its second year, the crop could be relied upon to give the emerging native cover protection for up to two years, then disappear from that block forever. In areas having a high exposure, additional measures such as the erection of dune mesh fences and the laying of natural brush matting was also applied. The dune crest was usually treated this way.

3.1.6 EIS — "To re-establish as many as possible of the species now existing, with particular emphasis on the tree species"

During the production of the EIS, detailed vegetation surveys had been carried out which would provide a record of the type of vegetation existing before mining, and which would be established after mining. Most of the site was covered with a dry sclerophyll forest community, varying its dominant species (Blackbutt, Redgum, Bloodwood), from one sub-community to the next.

Mature trees in the mining path were felled by bulldozer as the dredges approached and, after harvesting of seed and brush heads where applicable, burnt in windrows on the mining path. Understorey cover which was not actually removed in this process was dealt with in topsoil removal.

At the time of sowing the cover crop, a mix of native plant seed was introduced, consisting typically of up to one kilogram per hectare of local mixed acacia, and about 100 grams per hectare each of Blackbutt, Bloodwood and Redgum, as well as half a kilogram of Sawtooth Banksia. Other species were included in specific areas.

In the second season after the spreading of the topsoil, tube stock from the Company's native plant nursery at Hawks Nest was planted out. A total of almost 200 hectares have been so treated with some 187,000 shrubs. Additional seeding was done on areas deemed deficient by internal monitoring procedures, until a total of over 200 kilograms of cleaned seed had been applied to the 200 hectare site. For some years additional fertilizer was applied by air to promote the build-up of phosphorus in the biomass.

3.1.7 EIS — "To hasten the return of a vegetative cover similar to that now existing which has a similar appearance in the landscape and which conforms to a land use ... for recreation or public enjoyment of open space."

In 1977, the first of a series of formal Environmental Reports was prepared and released by the Company Ecologist on the actions, progress, and planning related to the mine rehabilitation. Others followed at six monthly intervals. These reports have been reviewed by a committee made up of technical officers representing the Ministers for Mines and Planning, the National Parks and Wildlife Service (NPWS), and the Company. Recommendations have been, and are still being made by this committee.

Forty-six permanent transects were set up over the life of the mine monitoring the return of native vegetation to the site, and charting the progress of their growth. A monitoring program to check for the migration of nutrients, particularly phosphorus, was instituted in the early years and involved sampling waters in observation wells and the adjacent wetlands. The results obtained were deemed satisfactory, and the program was terminated after several years.

A procedure developed during the first year of operations at Bridge Hill was the transplanting of Burrawang Palms. It was recognised that, although the species grew well from seed, the time taken for a plant to mature and become self-regenerating was quite long, and that a delay of thirty to forty years might ensue if reproduction by seed alone was adopted.

Accordingly, the Company embarked on a campaign to transplant almost every specimen of the species from just ahead of the active mine to a permanent location in the advancing rehabilitation. Some 24,000 were so treated, and today, the majority have recovered from the trauma of disturbance, have flowered and produced second generation seedlings. In the later years of operations, 1300 kilograms of cleaned seed was collected from source areas and sown about the mine site. Woody Pear, another species of special interest, was treated similarly.

4.0 CONCLUSION

When benchmarked against the EIS, Bridge Hill Ridge has achieved its goals, but was a project from another time. The EIS has functioned as the master plan with a little modernising ("moving the goalposts") along the way. The focus of interest was primarily on vegetation and landform although the EIS was thorough in its range of subjects. Water quality impact was assessed as being low and the limited monitoring indicated this. However, in a present day scenario, water quality would be placed under a closer scrutiny. Likewise the monitoring of fauna which has not been part of the management requirements.

The area has been included in the Myall Lakes National Park for some years, although this was not the original intention of the community at the time of the EIS. Dual management, by NPWS and BHPTM, has existed for this period without conflict. Now that all requirements for revegetation have been met, hand-back is in progress.

The first rehabilitation processes took place at Bridge Hill Ridge 23 years ago and can only be said to be at an end now.

FAUNA IMPACT ASSESSMENT AND MINE PLANNING IN THE TOMAGO SANDBEDS, NSW, AUSTRALIA

3307

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ABSTRACT

The Tomago Sandbeds is a coastal barrier sand dune system north of Newcastle on the mid-North coast of New South Wales, Australia. The Sandbeds contain a major groundwater aquifer and have historically been protected from development for protection of this resource. Titanium minerals have also been mined from this area since 1967 and the operation is due for completion in 1998 with approximately 10% disturbance of the total area of the Sandbeds (11,000 ha). Whilst rehabilitation of the site to native vegetation has been carried out progressively with mining throughout the life of the operation, there has recently been strong opposition to mining by community interest groups. The site contains a diverse range of flora and fauna and, in particular, a significant koala population. The mining operation has been controlled by licences imposed by regulators in a climate of changing legislation, however, clearing of disturbed habitat has remained the main contention in protest action and legal challenges. Although a time lag of about ten years exists between establishment and use of regenerating areas by koalas, assessment of unmined and rehabilitated areas has indicated that the species will utilise regenerating preferred habitats to a similar degree as undisturbed areas. Although operating licence conditions mainly relate to koala management issues, other fauna species are also affected by habitat disturbance from mining and have been included in licensing. Mine and rehabilitation planning has endeavoured to accommodate sound ecological principles as far as possible whilst still maintaining flexibility in the mining operation and this is extended to animal welfare management at the mines in the form of work procedures.

INTRODUCTION

The Tomago Sandbeds is a coastal barrier sand dune system north of Newcastle on the mid-north coast of New South Wales, Australia. Whilst the Sandbeds occupy an area of about 11,000 hectares, the sands occur as a relatively narrow band averaging about 3 kilometres wide and stretching approximately 30 kilometres from the Hunter River in the south-west to Port Stephens in the north-east.

The site contains an unconfined groundwater aquifer of regional significance and has been managed historically by the Hunter Water Corporation as a water reserve to supplement domestic supply to the Newcastle area since the 1930's. Protection of this groundwater resource has resulted in the Sandbeds experiencing little development over the years and consequently many of the site's intrinsic natural values remain. A mosaic of native ecosystems typical of low relief coastal dune systems exist in the area and these are further influenced by fluctuations in the relatively high water table.

The Sandbeds are surrounded by a variety of other competing land uses including agriculture, extractive industry, air force base and domestic airport and residential areas. These peripheral developments, and in particular residential subdivision in recent years, have played a significant role in isolating the Sandbeds from other core natural areas although substantial links do still remain.

MINING OPERATIONS

Since 1967 economic grades of the mineral sands rutile, zircon and ilmenite have also been mined from the strand sequences within this area. Whilst the total area of mining disturbance over the projected life of the mine will be less than 10% of the area of the Sandbeds the mining operations have been concentrated in the main orebodies which exist in the central south-west of the area and some relatively large areas have been cleared.

Rehabilitation of mined areas has progressively followed mining throughout the life of the operation with the main objective at this site being to reestablish native ecosystem environments similar to those existing prior to mining, consistent with the planned long term continued management of the Sandbeds by Hunter Water Corporation for water harvesting.

From 1986 to 1995 mining operations in this area have primarily entailed remining of areas originally disturbed between 1972 and 1985. To the projected end of the mine's life in late 1998, mining will continue in this area and a significant proportion of the area to be mined will entail remining (up to 50%) with the remainder being previously undisturbed.

LEGISLATIVE FRAMEWORK

In the state of New South Wales the main government agency responsible for the management of wildlife is the National Parks and Wildlife Service. This is enforced by the National Parks and Wildlife Act (1974) where almost all native fauna are protected by these laws. Mining is controlled by the Mining Act and Mining Leases granted under this legislation typically contain conditions relating to wildlife management, as do Development Consents which are approvals issued in relation to major (designated) developments under the Environment Planning and Assessment Act (1979) where a project's environmental impact assessment is revised by government agencies and the general public prior to the granting of development consent.

FAUNA IMPACT ASSESSMENT AND MANAGEMENT

The faunal assemblage in the Tomago Sandbeds is diverse in direct relation to the wide diversity of vegetation communities present across the site. This assemblage contains representation from all of the major vertebrate fauna groups - mammals (large and small terrestrial, arboreal & bats), birds, reptiles and amphibians and some of these groups include species listed in state legislation as "endangered" or "threatened".

Whilst a range of animal species exist in the habitats on and in the vicinity of the mine path, that which has been perceived to be most vulnerable because of its biology and ecology is the koala; and in any case whilst this species is not necessarily a biological indicator it is quite often used as a means to an end in conservation arguments due its status as an Australian icon and the fact people can readily identify with it being a symbol of bushland areas.

Habitat Assessment

A relatively detailed assessment process was undertaken to identify the overall likely fauna impact as a result of mining operations in this area and, in particular, the magnitude of impacts on the local koala population. The general fauna surveys entailed standard practices of diurnal and nocturnal trapping and searches, call playback for owls (and possibly their prey), pit trapping, hair tube placement and searches for direct and indirect evidence of various species according to their respective habits.

In order to identify and assess koala habitat usage a different approach is required and in order to best assess this an intensive site assessment program is needed. The local Port Stephens Council in conjunction with the New South Wales National Parks and Wildlife Service is presently in the process of reviewing a draft Plan of Management for koalas in the entire shire. The information gathered for this exercise was available as a base to work from and this has been expanded for the project ares (the forward mine path and surrounds)

Vegetation in the project area was mapped from interpretation of enlarged air photos at a scale of 1:10,000 and ground truthed lo determine major communities and specific dominant species associations for the purposes of accurately determining preferred browsing habitat areas and hopefully categorising these into classes of feeding preference, as has been done in some other projects. Habitats were sampled for specific vegetation characteristics and faecal pellet density using a plot methodology modified from that developed and used by the Australian Koala Foundation in their koala habitat atlas mapping program. The results of these surveys were combined with existing knowledge of the animals habits, from previous fauna surveys and radio tracking research world, and physiology to predict preferences in feeding habitats.

Mine Planning

This information was then used in discussions with specialist consultants and regul authorities to devise a mine plan which, in conjunction with current mine planning prac went further to minimise long term impacts on the species m the mine area.

This involved the sacrifice of significant areas of mining reserves in the immediate for mine plan and the forfeiting of some longer term resource areas. A significant area o immediate forward mine path was assessed as being prime koala habitat and as a result a 30% of this area was excluded from mining. Islands of vegetation were also retained i mine path. Refuge islands were retained in some relatively large planned disturbance areas a habitat corridor system was planned to link undisturbed or older regenerating areas as possible.

Site Rehabilitation

A total ecosystem approach to site rehabilitation has always been adopted at the mine. approach encompasses ideas typical of such programs such as minimum disturb philosophy, management of site endemic ecological processes, habitat reconstruction similar diversity as existed prior, respreading of cleared brush material and use of provenance seed stocks in all revegetation works.

In addition to these normal techniques strategic habitat corridor reestablishment is being ca out after research work in older regenerating areas identified preferred use of this typ planting by arboreal marsupials and the standing of stag trees has been undertaken, aga strategic locations, in order to provide medium and longer term roosts for birds and refuge arboreal mammals. The installation of nest boxes in older rehabilitation areas has been tr with apparent limited success, however, it is planned to expand this program and to inc their placement in stag trees also .

CONCLUSIONS

Some of the strategies employed in the mine planning process and techniques used in the rehabilitation program have already proven to be effective in wildlife management. It is h that the expansion of these ideas, whilst recognising that a certain time lag is inherent in situations, will further reduce medium to long term fauna impacts and even enhance their ra recolonisation into rehabilitation environments.

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2. Community development on North Stradbroke Island

Mining for heavy mineral sands commenced on North Stradbroke Island in 1950, and since 1985 Consolidated Rutile Limited (CRL) has been the only heavy mineral sands mining company operating on the island. CRL is responsible for large areas of rehabilitation and has the overall objective of "re-establishing as close as is practicable within the limitations of current technology and BPEM, the indigenous floral and faunal communities which existed in the area prior to mining" (CRL 1998). On many of the reconstructed dunes, the target plant community is a *Eucalyptus pilularis* forest, characterised by an overstorey of *Eucalyptus pilularis*, a mid-storey of *Allocasuarina littoralis* and *Banksia aemula* and an understorey consisting of various shrub and herbaceous species.

Like Weipa, site stability and growth conditions (edaphic and climatic) are not major limitations to successful rehabilitation. Here, the development of an appropriate community structure in relation to the target plant community is an important consideration for rehabilitation success. Community development, however, has been found to be dependent on the life cycle of *Acacia concurrens*, a species that until recently had been included in the rehabilitation seed mix. In the establishment phase of the post-mining plant community, *A. concurrens* quickly dominates and excludes a large range of other shrub species. In general, other canopy species such as *Allocasuarina, Callitris* and *Eucalyptus* are able to establish, but these are in lower numbers and are less influential in the early system due to slower growth rates. It is not until the competitive dominance of *A. concurrens* has been weakened through senescence, and the canopy is replaced by a more open *Eucalyptus* overstorey, that a range of understorey shrubs can re-establish in the older rehabilitation areas (Figure 3).

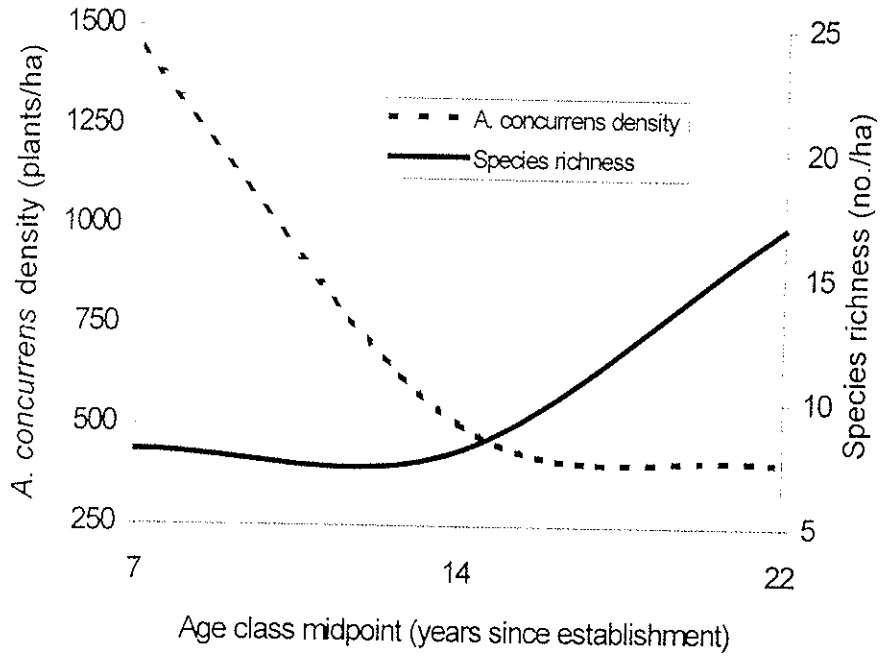

Figure 3. Relationship between the density of *Acacia concurrens* and the species richness of understorey shrubs in rehabilitated communities on North Stradbroke Island.

This pattern is predicated on the exclusion of disturbance, since the *Acacia* seed bank may be stimulated by fire (Purdie 1977, Auld 1986) and the dominance of the species may be perpetuated. Newer rehabilitation with *A. concurrens* excluded from the seed mix has been found to be closer in species composition and diversity to that of unmined communities, thereby potentially accelerating the process of community development (Harwood *et al.* 1998). In addition, the risk of rehabilitated communities remaining in an acacia-dominated 'sub-climax' is reduced. Thus, by understanding the ecological processes of succession, in particular the part played by *Acacia concurrens*, and the role of fire in the developing ecosystems, it is possible to begin defining success criteria for rehabilitation.

3. Seedling recruitment on tailings at Kidston gold mine

The decommissioning plan for the 310 ha tailings dam at Kidston Gold Mine includes the establishment of a sustainable native plant community directly on the final tailings surface. No topsoil is available for use as a capping material. While studies have shown that the oxide waste rock from the pit provides a suitable substrate upon which to establish vegetation, the use of this material for large-scale capping of tailings would be a very high cost.

In February 1994, as a part of a trial monitored by the CMLR, almost 1800 native trees and shrubs representing 21 species across five genera were planted directly into tailings as six-month old tubestock. The tailings in this 11 ha trial area had been deposited between 1985 and 1988. The plants were drip-irrigated on a regular basis until the commencement of the 1994/95 wet season. In May 1997, the average overall survival rate was 70%, and at a species level, survival ranged from 17% (*Acacia melanoxylon*) up to 95% (*Casuarina cunninghamiana*). After three years, two of the eucalypts (*Eucalyptus camaldulensis* and *Corymbia citriodora*) averaged 5.5 m in height. *Acacia holosericea*, *A. salicina* and *A. auriculiformis* were the best performing acacias, while *Casuarina cunninghamiana*, *C. glauca* and *Melaleuca leucadendron* were the tallest among the other genera. Across the trial site, pH ranged from 3.0 to 8.6 (median 7.2) and EC (1:5 aqueous extracts) from 0.1 to 6.7 dS m^{-1} (median 1.6 dS m^{-1}) (Roseby *et al.* 1998). However, attempts to correlate soil chemistry to plant performance were confounded by the heterogeneity of the material. The source of such variation arose from both the nature of the tailings deposition process and the oxidation processes that have occurred since.

Many of the planted species have produced seed, although only a few (e.g. *A. holosericea*, *A. leptoloba* and one or more of the *Melaleuca* and *Casuarina* species) have shown secondary recruitment on the site. As the initial establishment was from tubestock, ensuring that the community (including the locally dominant ironbarks) can successfully replace itself through establishment from seed is of paramount importance to ecosystem sustainability. As the original planting was irrigated, these plants were buffered against the harsher local environment that would otherwise confront germinating seed. Furthermore, over time, weathering processes and capillary rise may render some of the surface substrates more acid and/or saline, making successful establishment from seed an even greater risk. To address these issues, trials are currently underway to establish a greater range of species from seed sown directly on tailings. At the same time, analyses to predict the long-term chemical changes in the tailings will assist in assessing the likely characteristics of the seedbed in time. Such characterisation will identify the degree of tolerance that may be required by species, and indeed communities, for successful germination, growth, and longer-term survival.

Indicators of Ecosystem Rehabilitation Success.

Stage Two – Verification of EFA Indicators.

Final Report

For the Australian Centre for Mining Environmental Research

Produced by David Tongway and Norman Hindley

CSIRO Sustainable Ecosystems

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CSIRO

Australian Centre for Mining Environmental Research

Centre for Mined Land Rehabilitation

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EXECUTIVE SUMMARY

Field and laboratory investigations were undertaken at eight mines in Australia and one in East Kalimantan, Indonesia, covering bauxite, mineral sands, hard rock and coal. The principal objective was to verify that previously suggested indicators of minesite rehabilitation success produced by the Ecosystem Function Analysis (EFA) monitoring procedure properly represented soil properties measured by conventional field and laboratory procedures. Overall, the EFA indicators were shown to have a very high degree of correspondence with these measured soil properties.

Mined lands present unique and difficult places in which to do conventional soil measurements, most of which were developed for agricultural field application. Where verification was not fully met, the reasons behind this have been explained in the context of the specific locations.

An original design objective was for the field procedure to be generically applicable, this was reviewed at each mine and no changes were necessary. As the minesite locations varied from sandy deserts with 200 mm yr^{-1} rainfall to tropical rain forests with about 4000 mm yr^{-1} rainfall the method has shown very broad potential application. Many relationships are strong enough to use EFA indices in modelling soil stability, infiltration and nutrient cycling at hillslope and beyond scale. EFA has the potential to provide a spatially extensive data set over time at a fraction of the cost of biophysical measurements.

We developed a new way of representing the functional role of vegetation to overall ecosystem function at the hillslope scale by presenting vegetation cover on a layer-by-layer basis. Thus, we can show the value of the ground layer in ameliorating overland flow, soil erosion and the transport of organic matter as well as examining the role of the developing canopy in terms of wind interception, causing turbulence and thus reducing the capacity to retain resources in suspension. The sites where competent foliage is present may well become "sinks" for mobile airborne resources such as dust, organic matter and propagules.

The Stage 1 Final Report proposed that an analysis of the curve shape formed by the index values obtained by time series monitoring of rehabilitating lands should be the tool by which rehabilitation success is analysed. EFA has up to five landscape organisation indices and three soil surface indices, which can be used to delineate the ecosystem "trajectory". Between them, these indices cover the range of scales and the complexity of issues that make up ecosystem function. Only one study site in Stage 2 permitted the full demonstration of this proposition, the Alcan bauxite mine at Gove. This was the only site to have a history of very similar rehabilitation technique used over a long time period. This enabled us to trace the full shape of the trajectory curve that denoted functional rehabilitation success. The "trajectory" proposal was demonstrated well enough to recommend implementing, as evidence of ecosystem rehabilitation success.

We supplied each major Sponsor with Site Reports soon after our initial field visit, followed up at half term by a Sponsors meeting at which we presented interim results. Final Site Reports have recently been prepared and this overall Project Final Report completes the reporting process. A revised technical manual is also being prepared as a final communication. This manual will replace the existing field manual and training course notes and will include the Excel templates and a greatly expanded image library. We expect to make this available as a CD and on the CSIRO, Sustainable Ecosystems and ACMER websites.

Indicators of Ecosystem Rehabilitation Success.

Stage Two – Verification of EFA Indicators.

For the Australian Centre for Minesite Environmental Research

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Indicators of Ecosystem Rehabilitation Success.

Stage Two – Verification of EFA Indicators.

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Figure 4.7.d. The trajectory of the nutrient cycling index at Gove. The full sigmoidal shape is not evident. The ecosystem has not yet reached a true plateau, even after 26 years, due to the growth of *Eucalyptus tetradonta*, which will continue for some time yet, with increasing canopy biomass and litter fall. However, the utility of the early steep rise of the sigmoidal curve remains as a useful tool in assessing ecosystem development. 31

1. INTRODUCTION

This project is the second stage of a process to evaluate Indicators of Ecosystem Rehabilitation Success. The first stage evaluated the potential of a CSIRO developed approach called "Ecosystem Function Analysis" (EFA). The initial study was initiated by the ACMER (then ACMRR) and funded through the Australian Minerals Industry Association Ltd (AMIRA). That project commenced in March 1996 and concluded in September 1997.

The first study (Tongway *et al* 1997) showed that the EFA evaluation procedure could potentially be used for a wide range of mine types distributed across the continent without modification between sites. The EFA approach uses simple, rapidly assessed, visual indicators of surface processes mediated by both physical and biological components of the system, so repeated assessment is a cost-effective design feature. Overall, the indicators developed from the approach were shown to provide a wide range of information about the biogeochemical functioning of developing ecosystems on mined lands. The outputs appeared to agree with both measured attributes of ecosystems and also intuitive assessment of success. In addition, a procedure enabling the evaluation of "relative rehabilitation success" at different stages was proposed as a framework for future implementation (Tongway *et al* 1997). However, at the final Sponsors' meeting, it was decided that although considerable progress had been made towards project goals, the relationship between the EFA indicators and measured ecosystem variables had not been demonstrated in a sufficiently rigorous scientific fashion to create a strong case for Regulatory acceptance of the approach.

Accordingly CSIRO, through ACMER, was invited to submit a new proposal for a research project that would provide this level of evidence to the Industry. At length, the funding, the project and the personnel came together for a project commencement in April 2001 as ACMER Project No. 31.

2. PROJECT OBJECTIVES

There were four agreed objectives in the approved proposal for Indicators of Ecosystem Rehabilitation Success; Stage 2- Verification of Indicators and Transfer of Monitoring Technology project.

1. Conduct a program of scientific verification of the informing capacity of EFA indicators on a range of mine types (coal, bauxite, mineral sands, base metals and uranium) using conventional field and laboratory measurements.
2. Develop further links between existing methods of vegetation assessment and ecosystem function.
3. Report the results of testing the hypothesis by preparing papers for peer review and publication in the appropriate scientific literature.
4. Communicate the EFA techniques and the results of the Stage 2 study to minesite rehabilitators, regulatory agency personnel and community groups through the use of demonstration workshops, and prepare a new EFA manual for the Mining Industry.

These objectives were to be met by a coordinated study involving CSIRO and the Universities of Queensland and Western Australia. An MSc student from each University was included in the research team, with co-supervision by CSIRO and the relevant University. Towards the end of the first year of the project, the UWA student, Ms Vicki McAlister withdrew from the project and her work was taken over by the CSIRO team members, placing some strain on meeting time objectives and maintaining the budget outlays.

3. METHODOLOGY

3.2. What is EFA?

EFA is a monitoring procedure that establishes how well an ecosystem works as a biophysical system. It uses simple, visual, rapidly assessed indicators that focus on soil surface processes. As such it differs from conventional monitoring that typically records the presence and/or abundance of selected biota. EFA is comprised of four modules: a conceptual framework, a field methodology, a data reduction package and an interpretational framework. The conceptual framework was published in Ludwig *et al* (1997). The procedure uses a number of simple indicators easily assessable in the field. A full description of the EFA field methodology, including worked examples of real data, the data reduction software template and interpretational framework is available on the CSIRO Sustainable Ecosystems web page at:

www.cse.csiro.au/research/program3/efa/

Other relevant publications are Tongway and Smith (1989), Ludwig and Tongway (1992), Ludwig and Tongway (1995), Tongway (1995), Tongway and Ludwig (1996), Ludwig and Tongway (1996), Tongway and Murphy (1999).

The data are collected on line transects oriented in the direction of resource movement, which is usually down slope. There are two scales of data collection. At the coarser or hillslope scale, the line transect is differentiated into zones where resources tend to accumulate (patches) and zones where resources tend to be transported (interpatches). The width of the patches on the contour is also measured. These data produce a set of indicators of "landscape organisation" that can be used for time series monitoring, to see if the patches are growing larger or becoming more numerous, or the reverse, over time. At a finer scale, five replicates of each zone type are selected at random and the status of eleven indicators related to surface processes assessed. These individual indicators are assigned, by an Excel spreadsheet, to generate three emergent soil surface properties reflecting stability or resistance to erosion, water infiltration rate and nutrient cycling. The template is freely available on the web page above and contains some weighting factors so that the indices are well related to measured variables (from empirical evidence). With practice, each indicator takes about 5 seconds to assess. A manual provides illustrations of each indicator (Tongway and Hindley 1995). This process assesses the biophysical "quality" of the patches and interpatches, which is expected to improve over the course of rehabilitation.

EFA is designed for repeated use so that the development, or degradation, of a site can be assessed over time. The interpretation module proposes an analytical process that enables the user to examine the "trajectory" of the ecosystem being monitored and to use this information to decide if the site is converging on a "target" functional state, or needs further work to ensure ultimate success. Target values signifying rehabilitation success can be obtained by assessing "analogue" sites that represent the desired end point of rehabilitation. With EFA, a site may be deemed to be highly functional from a biogeochemical perspective, but have an inappropriate species composition, for example, weeds. This information might suggest that weed control is all that is necessary, and not major rehabilitation works.

An analogue site is one that is self-sustaining and has many of the attributes of the final landscape, particularly in respect to function. Note that the term "homologue" would be used to specify a landscape that would be replicated in every respect. Analogue sites would have similar slope, soil texture, resource regulation and many of the vegetation species required on the mature

rehabilitation. Data from analogue sites forms part of the monitoring procedure through time, so that varying seasonal conditions result in a "band" of values to act as targets for rehabilitation.

3.2. Field and Laboratory measurements

Published procedures for physical, chemical and biological soil measurements were selected from the soil science literature for the verification process. As many measurements as possible were done in the field, so that data analysis and repeat measurements could be done on site. The budget did not allow for multiple site visits.

1. **Soil Stability**. To measure soil stability, intact cores of soil 75 mm in diameter and 50 mm in depth were sampled in metal rings from sites where the EFA assessment was made. These were returned to the laboratory, the soil was expressed from the ring in layers of 0 to10 mm and 10 to 30 mm. Wet-sieving a weighed sample through a nest of sieves assessed the stability of the soil fragments. A stability value was calculated by the procedure of Chaney and Swift (1984). At the Gove site only, a field-going water drop applicator was used to wet the soil with simulated rain directly in the field before sampling and sieving.

2. **Infiltration**. We used a disk permeameter (Perroux and White 1988) to assess saturated flow infiltration directly in the field. We initially attempted to also measure unsaturated flow infiltration, but technical difficulties in setting up this type of infiltrometer on minesites lead to the abandoning of this procedure. We recorded until a steady state of infiltration had occurred for some minutes.

3. **Nutrient cycling**. We used two procedures; soil respiration measured as evolved CO_2 in the field over a 24-hr period, which reflects biological activity in the soil and the pool size of biologically acquired soil nutrients. The latter included total nitrogen and carbon, plant available phosphorus and mineraliseable nitrogen. The underlying assumption being that as an ecosystem develop all these nutrients would tend to accumulate in the surface layers of soil due to plant growth, litter fall and decomposition, as has been shown in the general soil fertility literature of natural ecosystems.

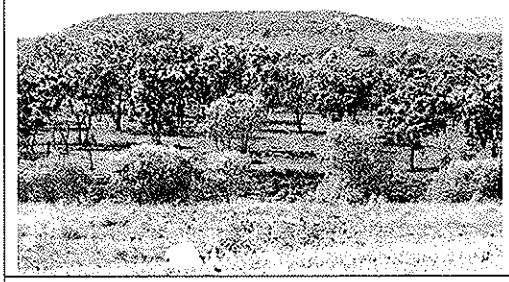
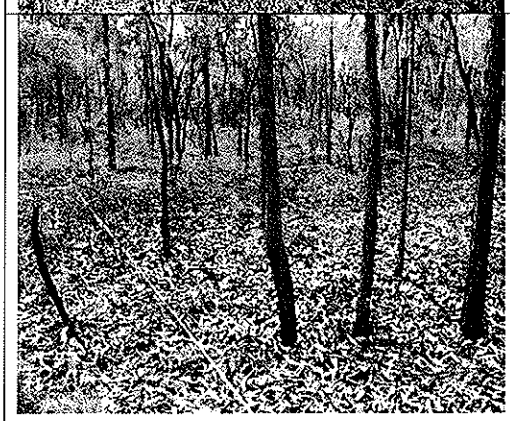
There is a huge potential range of possible measurements, but those selected optimised the informing capacity within the limitations of the budget.

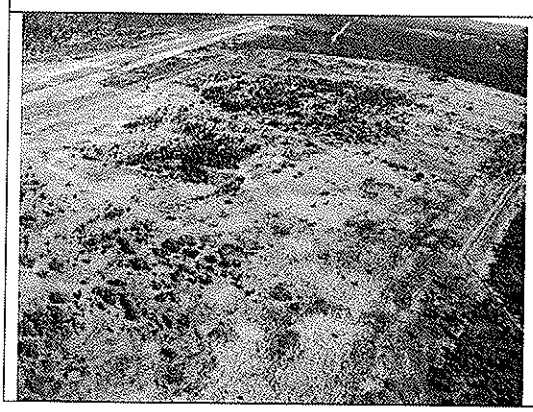
3.3. Verification Procedure

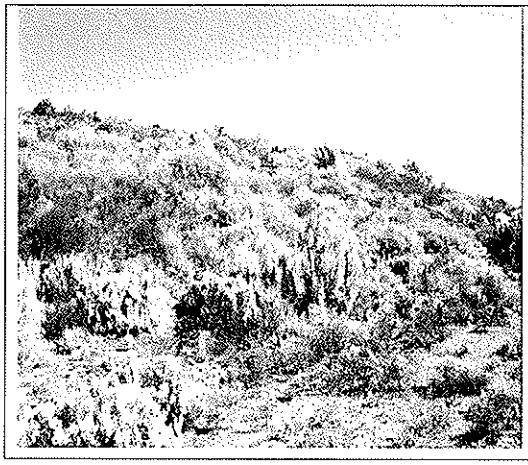
The EFA monitoring procedure generates three principal index values that represent respectively, soil stability or resistance to erosion, infiltration of rainfall and nutrient cycling. We tested those indices by regressing them against the field and laboratory measurements. The following criteria were proposed as comprising successful verification.

1. The regression relationship between the EFA index and relevant measured variables should have a high statistical relationship over a wide range of indicator values, with few outliers.
2. The relationship should preferably be linear, implying that the indicator sensitivity to environmental change was similar throughout the indicator range.
3. Data acquired from rehabilitation sites and relevant analogue sites should occupy the same data space (a sausage-shaped cloud of points), thus providing assurance that EFA index numerical values can be taken to have the same meaning throughout a given mine.

3.4. Sites Descriptions

	1. Brocks Creek gold mine (Anglo Gold) is located near Pine Creek, 150 km south of Darwin. Mining is now concluded and rehabilitation is well under way. The Faded Lily WRD in the background of the image is the location of our work. The rehabilitation methods for the lower, middle and top lift were all different, making the verification of EFA values difficult. *Eucalypt* savanna woodland (foreground) would be an appropriate vegetation structure, so couch grass established on lift 2 is incongruous.
	Carnilya Hill (WMC) is a former nickel mine located 30 km to the east of Kambalda in WA. Mining here is complete and rehabilitation has taken place in just two discrete time steps. There has been some erosion of the ripped WRD surface, but *Atriplex numularia* plants have established well. Decomposition of plant litter is stabilising the surface soil, which was initially dispersive. Feral goat grazing is having an affect on seedlings and needs to be controlled.
	Eneabba mineral sand mine (Iluka) is located 150 km south of Geraldton, WA, in a heath landscape. It remains an active mine but sections have been rehabilitated since the early 1970s, using a variety of techniques. Early rehabilitation has not progressed to the degree expected in 25 years, with bare soil areas remaining prominent, but more recent methods using an initial mulch comprised of slashed heath seem to have an improved the initial response rate.
	Gove bauxite mine (Alcan) is located on the eastern side of Arnhem Land, NT. It remains an active mine, but sections have been promptly and progressively rehabilitated since the early years with notable success. A successful technique was used right from the start and this is the only mine in the group where the trajectory of ecosystem development can be demonstrated. Topsoil management is the most benign of all mining procedures. Fire has been largely excluded from the rehabilitation, whereas fire is an annual occurrence outside the mine. This makes selection of an off-mine analogue site very difficult.

	Kelian is a gold mine (Rio Tinto) approaching closure, located near the equator in East Kalimantan, Indonesia. The reliable high rainfall, and availability and good management of high quality topsoil have resulted in spectacular growth. Full ground cover is achieved in just weeks, thus protecting the soil surface from rainsplash. The rehabilitation in the image is just 14 months old. "Horticultural" introduction of key species is a feature of the ongoing management of the rehabilitation. At about 4000 mm, this site has the highest rainfall of the study group.
	The Granites gold mine (Newmont) is about 600 km NW of Alice Springs, NT. There are a number of ore bodies. The image is of the Triumph WRD, looking out onto the Tanami Desert plain. This rehabilitation, comprised mainly of Spinifex is just a few years old, but the East Callie site also studied is dense *Acacia* shrubland. Erosion and deposition are active processes here despite having the lowest mean annual rainfall of the all the study sites at about 200 mm. Rocky surfaces prevented a full range of verifying data from being collected here.
	Gregory coal mine is located in the Bowen Basin, 200 km west of Rockhampton in Qld (BHP Billiton). It remains an active mine, but many areas mined in the past have been rehabilitated as rolling pasture grasslands intended for grazing by beef cattle. Dragline spoil is one of the more difficult media to rehabilitate because of the random mixture of soil materials from anywhere in the top 50 m of the regolith. The soils can be quite dispersive and rills may form early in the life of the rehabilitation. The available analogue site was a savanna woodland, and so not properly functionally comparable to the grassland in the rehabilitation.
	Nabarlek is a former uranium mine (*eriss*), located in west Arnhem Land, NT. There were two rehabilitation types examined here, the filled evaporation ponds (in image) and the former pit area. The EP area has subsided since being filled, creating a "perched depression" landform at the headwaters of 3 creeks. The area is substantially a grassland, but with areas of *Melaleucas*. The grass biomass creates a massive fire risk for young trees. The site is stable to erosion. The pit has mainly fire-prone *Acacias*, so also has some vulnerability to fire and erosion on its domed landform.



New Celebration gold mine (Newcrest) is 30 km SE of Kalgoorlie in WA. There are many ore bodies involved in the mining process, but we looked at the Pernatty WRD (image), which has been rehabilitated as a chenopod shrubland, though there are *Eucalypts* saplings present. Success has been patchy: good where water outflow from the top of the WRP is controlled and poor where runoff water concentrated from a large area is discharged down the batter slope. Soils are initially dispersive, but stabilize with plant litter decomposition. Second generation chenopods have established. A newly rehabilitated site at Mt Marion, about 30 km distant was also studied.

4. RESULTS

4.1. Summary of verification tasks

Table 4.1 summarises all the EFA soil surface index verification analyses from all the mines studied. The degree of verification is in table 4.2. The EFA indices were verified to a large degree: of the 9 mines x 4 indices matrix = 36 verification tests, 16 were strongly verified, 9 were moderately, 1 weakly verified, 9 not verified and for 2 it was not technically possible to be verify. Each of these circumstances is discussed in succeeding sections. "Strong verification" means that the points plotted in the correlation between the rehabilitation and relevant analogue site form a single "cloud", so that the EFA index numerical value has the same biophysical meaning across all sites on the mine. " Partial verification" means that whilst the rehabilitation sites and analogue sites each have EFA indices and measured properties that are internally well correlated, the respective regression equations are different, so that cross-calibration is needed to arrive at EFA index values to act as targets for rehabilitation. This cross-calibration would only need to be done once at each mine.

Table 4.1. Summary of the degree of verification for each mine, correlating the assessed indices against the measured variables. The meaning of the symbols is in Table 4.2.

Mine	Stability	Infiltration	Soil Respiration.	Nutrient Pool Size
Brocks Creek (Gold)	nv	nv	√√√	nv
Carnilya Hill (Nickel)	√√√	√√	√√√	√√√
Eneabba (Min. sands)	np	√√	√√√	√√√
Gove (Bauxite)	√√√	√√	√√√	√√√
The Granites (Gold)	√√	nv	√√√	√√√
Gregory (Coal)	√√√	nv	√√√	√√
Kelian (Gold)	√√	√√√	nv	√√√
Nabarlek (Uranium)	√√	nv	nv	nv
New Celebration (Gold)	np	√√	√√	√

Table 4.2. The verification classes used in table 4.1.

Class	Abbrev.	R^2	P
Strongly Verified	√√√	>0.60	< 0.01
Moderately Verified	√√	0.40 to <0.60	< 0.01
Weakly Verified	√	0.20 to <0.40	< 0.01
Not verified	nv	>0.20	< 0.01
Verification not Possible	np	-	-

4.2. Examples of successful verification of EFA indices.

Due to the large number of relationships deemed to have fully verified the EFA indices, only a sub-set will be discussed here. All the verification analyses are presented in the Appendix. The degree of verification between a measured variable and the EFA index is defined in Table 4.2. For a strong verification, the plotted data from both rehabilitation and analogue sites form a single "cloud" or "sausage". Typically, this means that there is a linear regression, but sometimes in tropical ecosystems, there is an exponential relationship with an upper asymptote. EFA was deliberately designed to be more sensitive at the less functional end of the spectrum so that early rehabilitation could be assessed accurately in case a problem needing early attention was present.

4.2.1. Stability.

Data from the Carnilya Hill nickel mine site near Kambalda (Fig. 4.2.1a..) in the semi-arid lands of Western Australia and the Gove Bauxite mine (Fig. 4.2.1b.) in the seasonally humid tropics in the Northern Territory represent successful verification of the stability index. These mines have contrasting soil types, very different climates and differing rehabilitation techniques. The data represent a wide dynamic range of values of both the index and the measured variable.

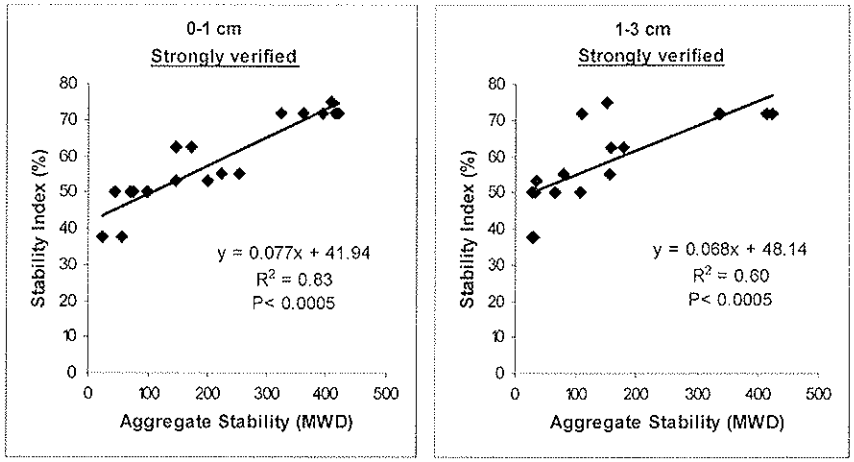


Figure 4.2.1.a.. Carnilya Hill site. The relationship between the laboratory measurement of stability (mean weight diameter) of the 0-1 and 1-3 cm layers of soil and the EFA stability index. The data show a good linear fit with high statistical significance across all rehabilitation and analogue sites. This verifies that the EFA stability index can be used to assess ecological development in the Kambalda district landscapes.

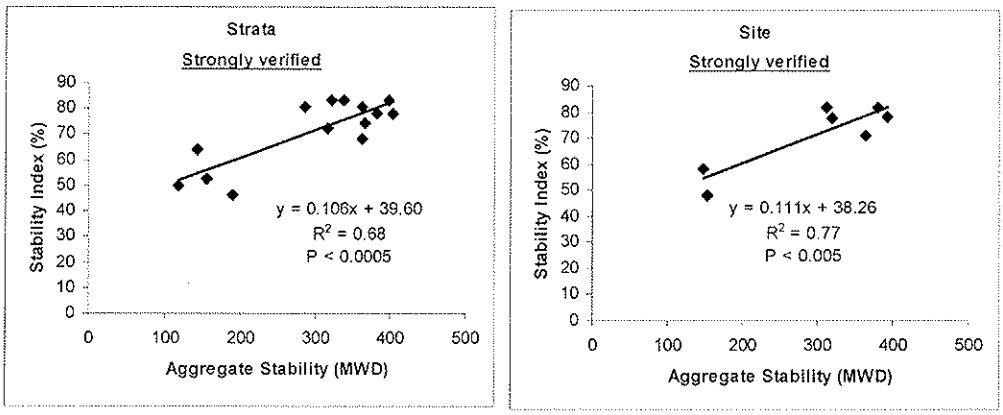


Figure 4.2.1.b. Gove site. The relationship between the EFA stability index and water stable aggregation measured in the field. The left hand graph represents the mean values of individual landscape zones and the right hand graph the mean site values.

4.2.2. Infiltration

Data from the Gove bauxite mine (Fig. 4.2.2.a.), a tropical woodland and from New Celebration (Fig. 4.2.2.b.) a gold mine near Kalgoorlie are presented as representing successful verification of the EFA infiltration index. These sites also differ in mining techniques, rehabilitation procedures, soil types and climate.

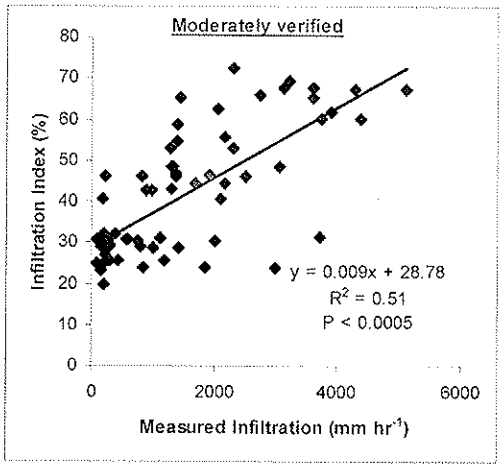


Figure 4.2.2.a. Gove. The relationship between the EFA infiltration index and the field measurement of saturated flow infiltration. All data for all sites are included and although there is a scattered in the data the large number of observations make the relationship significant.

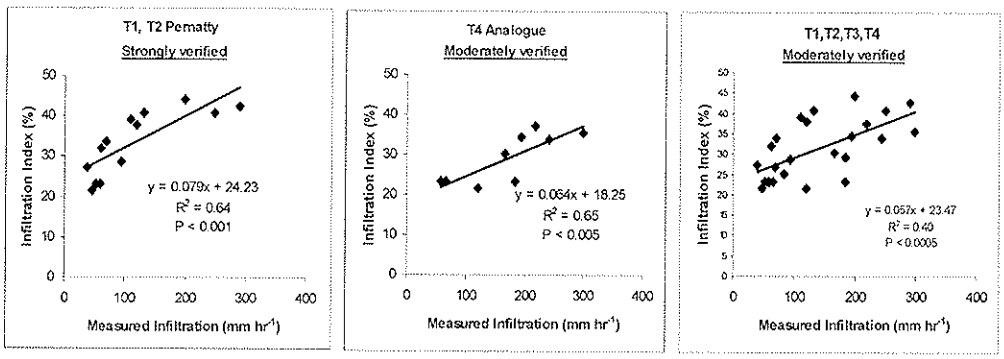


Figure 4.2.2.b. New Celebration. Relationships between the EFA infiltration index and the field measurement of saturated flow infiltration presented as linear regressions. The left hand plot contains the 2 rehabilitation sites on Pernatty, the middle plot the analogue site alone and the right hand plot the above 3 plus Mt Marion combined. Although there is more scatter in the combined graph, the relationship is statistically significant.

4.2.3. Soil Respiration

Data from the Gove bauxite mine, (Fig. 4.2.3.a.) and the Granites gold mine (Fig 4.2.3.b.) are presented as representing strong verification of the EFA nutrient cycling index using soil respiration as the measured property. Soil respiration represents the biological activity due to soil microbes and roots. It is an integrating measure, depending on the size of the microbial population and the quality of the organic substrate for them to process. In both cases the data from all rehabilitation and analogue sites studied at the mine are included in the analysis.

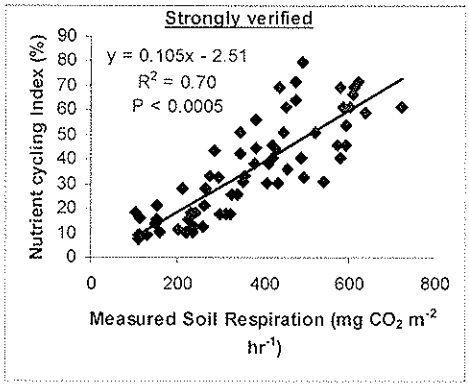


Figure 4.2.3.a. Gove Mine. The relationship between the EFA nutrient cycling index and soil respiration, using data from all studied rehabilitation, and the analogue sites. The good relationship verifies that the EFA nutrient cycling index can be used as a surrogate for costly and time-consuming measurements in monitoring rehabilitation progress.

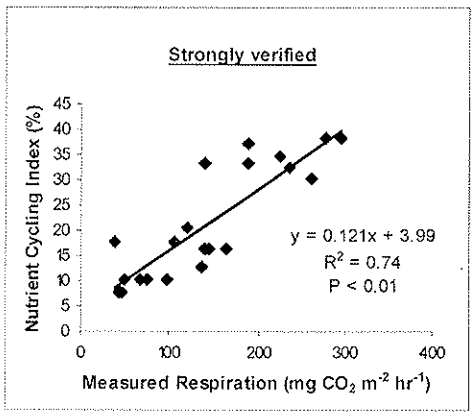


Figure 4.2.3.b. The Granites Mine. The relationship between soil respiration rate and the EFA nutrient cycling index. Data from all sites are represented.

4.2.4. Nutrient pool size.

Data from the Granites gold mine (Fig. 4.2.4.a.), the Nabarlek uranium mine (Fig. 4.2.4.b.) and the Gove bauxite mine are presented as representing successful verification of the EFA nutrient cycling index by examination of the soil nutrient pool size. This is not as simple a verification as the soil respiration procedure as there can be "old, recalcitrant" carbon and nitrogen present in the soil, especially that which comes from deep within the regolith and not "recent" nitrogen and carbon fixed by contemporary biological processes. In some mines, there was clearly a complex and heterogeneous mix of soil materials that militates against the use of C and N measurements as representing cycling nutrients.

Soil phosphorus estimated by dissolution in 0.5M sodium bicarbonate generally gave results not related to the nutrient cycling index. This is due to applications of superphosphate often provided at the outset of rehabilitation. That is, the P measured was a mixture cycling P and applied P.

Total soil sulphur was measured with C & N in an analyser but S chemistry on minesites is not well-enough understood to make use of the data in this study.

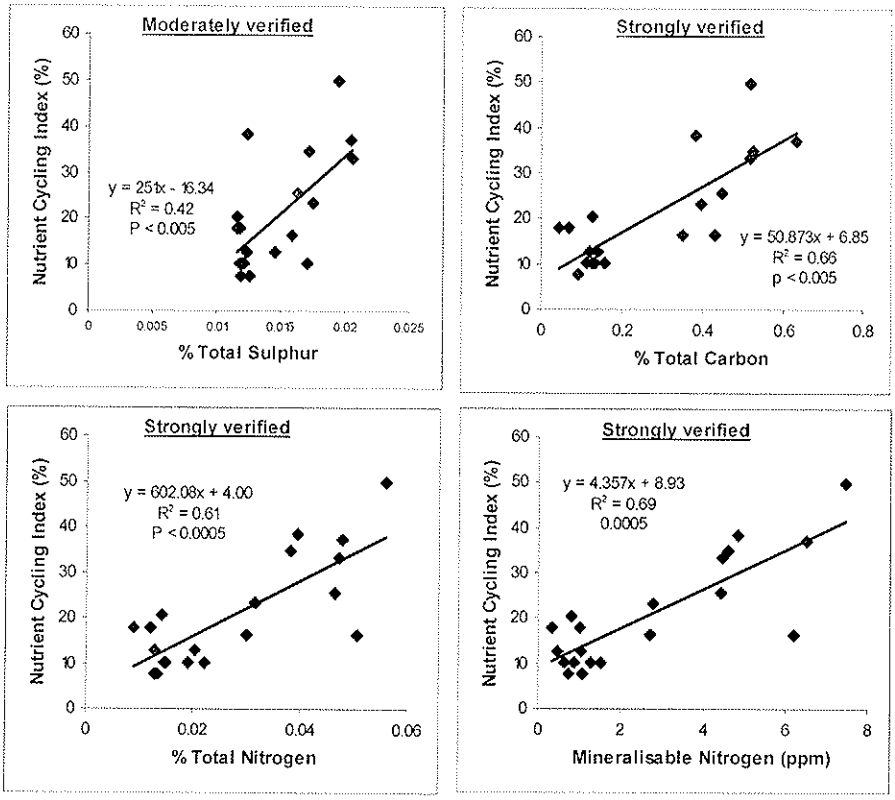


Figure 4.2.4.a. The Granites Mine. The relationship between the EFA nutrient cycling index and the concentrations of soil organic nitrogen and carbon, total sulphur and mineraliseable nitrogen from all sampling sites on the mine.

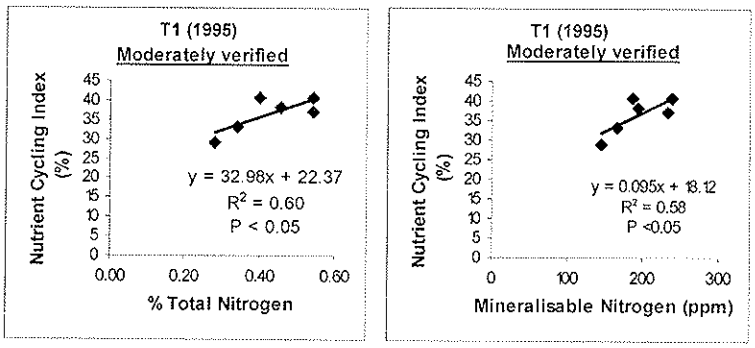


Figure 4.2.4.b. Nabarlek Mine. The relationship between EFA nutrient cycling indices with organic nitrogen concentration and mineralisable nitrogen pool size in the surface soil.

4.3 Examples where verification was partially achieved.

The extent to which verification was partially achieved is shown in the ensuing notes and graphs. Not all graphical relationships for all relevant sites are shown, but are available in the respective site reports and in the Appendix to this report.

a. Soil respiration at Brock's Creek. The nutrient cycling index was well related to soil respiration on the analogue site and also to the two rehabilitation sites where it was technically possible to make measurements. However, the two regression equations were quite different (Fig 4.3a). Combining all available data produced a non-significant and noisy relationship. We concluded that the soil differences between the analogue and the rehabilitation sites were such that the EFA values could not be reliably used across all the sites as representing the same measured property value. It is possible to calibrate the analogue site against the rehabilitation sites to rescale the Index appropriately.

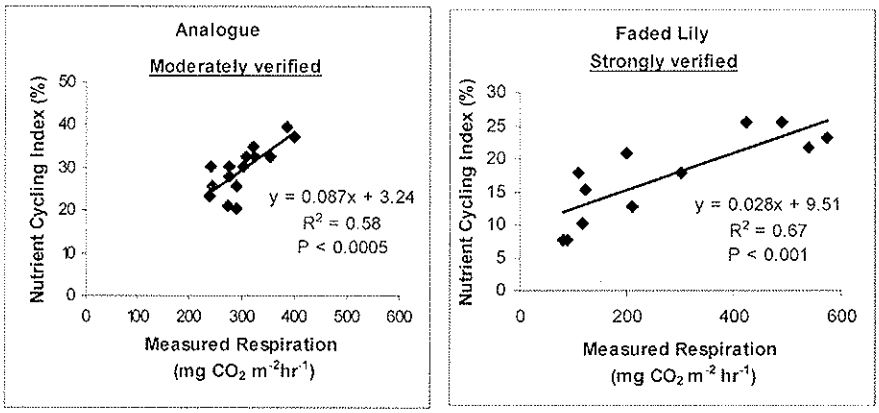


Figure 4.3.a. The regressions between measured soil respiration and the nutrient cycling index at Brocks Creek, showing that the analogue site and the rehabilitation sites have different response shapes, though both have acceptable correlations, taken singly. There is higher biological activity in the rehabilitation sites than in the analogue for a given EFA index value. Neither graph covers the full index range by itself.

b. Nutrient pool size at Brock's Creek. As with the soil respiration, there were some good within-site relationships (Fig.4.3.b.). Combining the data from all sites produced a very noisy graph. Cross-calibration of the analogue with the rehabilitation sites was not fully possible in this case, suggesting that soil materials used in different stages of rehabilitation were quite diverse to begin with.

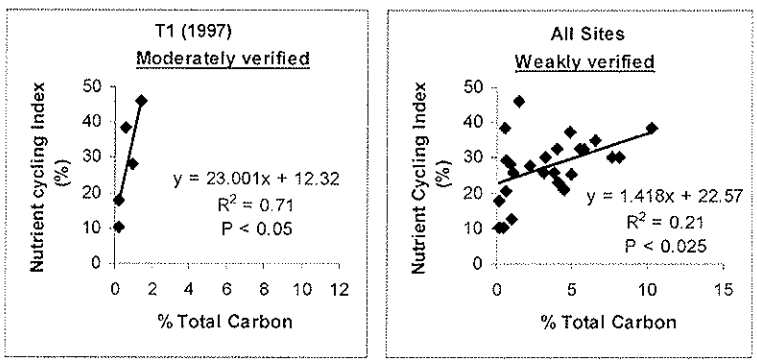


Figure 4.3.b. Illustrates that the correlation between measured % Total Carbon (0-1 cm) and EFA Nutrient Cycling Index. The lower lift on Faded Lily had a different slope compared with sites combined.

c. Soil infiltration at Eneabba. Within-site correlations (Fig. 4.3.c.) were quite acceptable but each site had a different regression equation (slope and intercept) (Fig 8.2), so that EFA index values could not be directly used to assess the degree of rehabilitation success or set target values. Cross-calibration would provide an appropriate link. When the infiltration rate rises above about 600 mm hr^{-1} on the rehabilitation sites, the EFA index is a poor predictor of actual infiltration. The individual indicators making up the infiltration index have been selected to discriminate soils with low to moderate infiltration. This capacity can be seen on the graph for infiltration rates below 600 mm hr^{-1}. At Eneabba, deep non-cohesive sands with no trace of a physical surface crust can have very high infiltration rates, as shown. A similar situation occurs at Nabarlek with highly structured clay soils when infiltration was measured in the dry season.

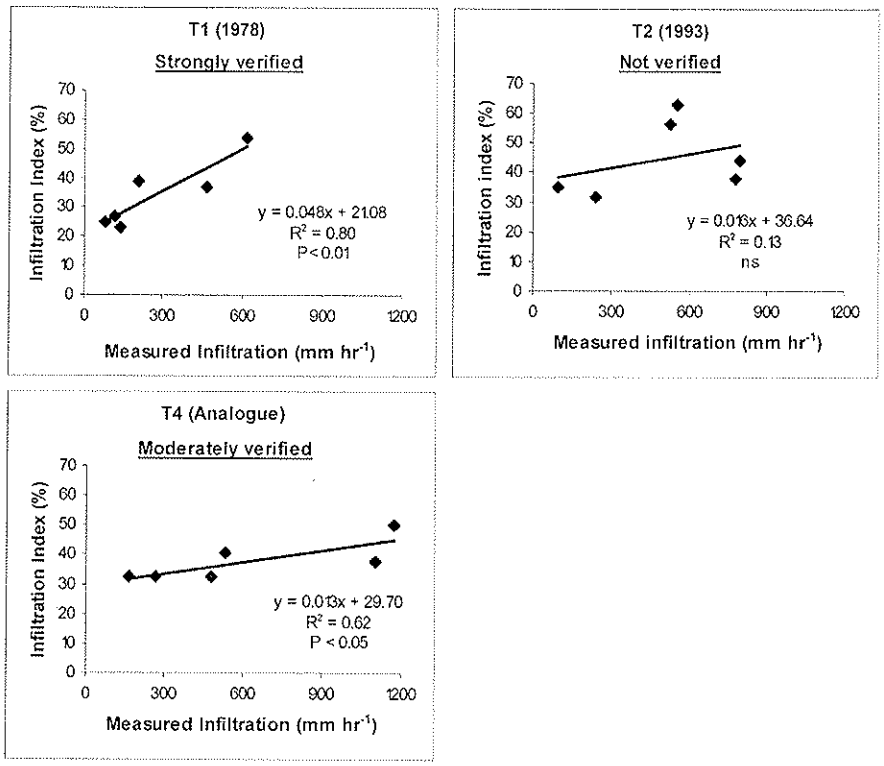


Figure 4.3.c. The relationship between the EFA infiltration index and measured infiltration on rehabilitated and the analogue sites at Eneabba. The within-site relationships are good, but the gradients of the relationships differ significantly.

d. Soil respiration at Eneabba. Within-site correlations showed good relationships but combining sites together produced a noisy plot (Fig. 4.3d). There were differences in the rehabilitation procedure at each of the rehabilitation sites, so the differences are not surprising. A very recently established site had high respiration, implying that current procedures, using a substantial application of shrub mulch give a biological kick-start to the system whereas former procedures did not. Cross-calibration is possible, however.

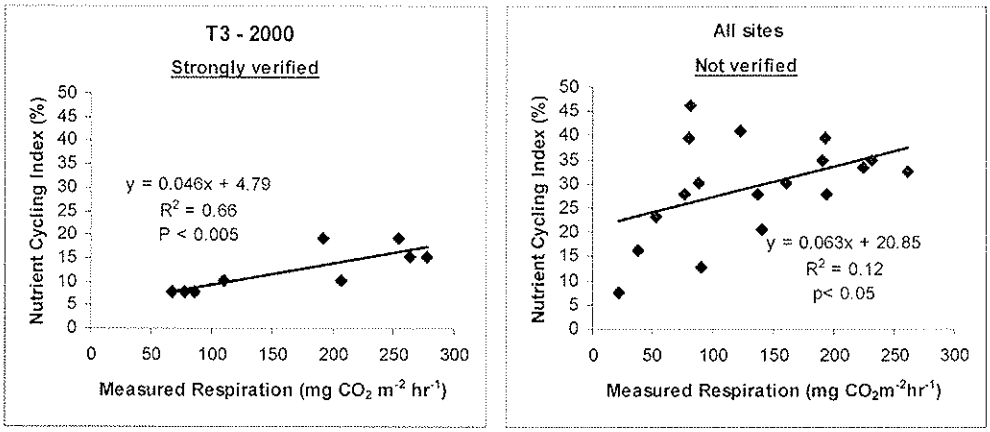


Figure 4.3.d. The within site correlations for T3 (2000 rehabilitation) were whereas combining all sites produced a very noisy and un-useable relationship.

e. Soil respiration at Nabarlek. Again, there were good within-site correlations, but between-site comparisons were incompatible to the extent, with existing data, that cross-correlation is probably not possible (Fig. 4.3.e.). A new grassy analogue site should be selected, that will be well away from the mine area, probably a small flood plain.

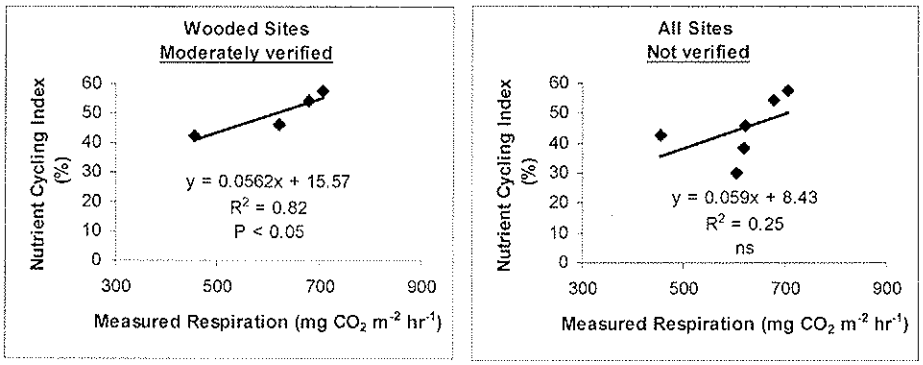


Figure 4.3.e.The relationships between measured soil respiration for the wooded sites (left) and all sites (right), and the nutrient cycling index.

The wooded sites were consistent across the range of analogue and rehabilitation but the grassy rehabilitation sites had a different response (lower points added in fig 4.3.e. right). This implies that grasslands and woodlands are not likely to be comparable. Calibration with the nearby analogue site would not be appropriate in this case and a new analogue site (off the mine area) should be selected.

f. Soil respiration at New Celebration. The rehabilitation sites had a combined data set that showed a good correlation, but which had a different regression equation to the analogue, which was also internally consistent (Fig. 4.3.f). Cross-calibration to reconcile this difference is possible.

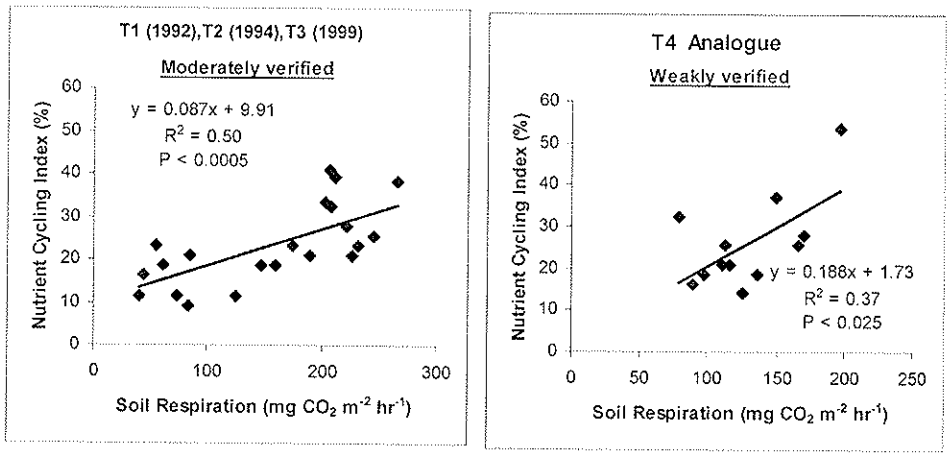


Figure 4.3.f. Combined Pernatty sites and Mt Marion rehabilitation sites (T1, T2 and T3), and the Analogue site (T4) showing the relationship between the nutrient cycling index and soil respiration rate. The slope of the relationship between the rehabilitated sites and the analogue were different, although individually significant, so could not be combined into a single graph.

4.4. Examples where verification was not possible.

At some sites, the soil surface conditions were such that some verification measurements were not technically possible within the financial resources and instruments at hand. Each of these circumstances will now be discussed in turn. Obviously, there are no data available for these sites

a. Stability at Eneabba. The soil here comprised non-cohesive sands that had no natural soil aggregation or "clumping" as such, and in our laboratory test no data was generated that properly reflected soil stability on wetting. Using a rainfall simulator and measuring sediment yield would better assess stability.

b. Infiltration at The Granites. Soil rockiness at the analogue site and one of the rehabilitation sites prevented a correlation being developed (Plate 4.4.c). Portable rainfall simulation equipment might be able to generate useful data.

Plate 4.4. This stony surface at The Granites prevented infiltration measurements with the instrument available.

c. Microbial respiration at Kelian. A minimum 24 hr. respiration period is necessary to obtain reliable values and we were not able to do this at Kelian as the respiration rates were very high, saturating the KOH solution too quickly. A return to the site with new field equipment is not now possible.

d. Stability at New Celebration. These samples were taken with the expectation that they would be processed at the UWA laboratory. With the withdrawal of Ms. Vicki McAlister from the project, the samples were ultimately sent on to Canberra for analysis, but arrived in such a disturbed state (powder!) that a soil stability test was not possible. However, a re-sampling, now not possible within the budget would most likely produce useful data as the soil has an aggregate structure suitable for water stable aggregation analysis.

4.5. Where verification failed.

Nabarlek infiltration. The soils on the evaporation ponds at Nabarlek were well-structured clays and we were present in the field in the mid dry season by which time the soil was dry to 50 cm (Plate 4.5). We measured extremely high infiltration rates on these soils, of the order of 2000 mm h^{-1}, with no relation whatever to the EFA index. Gillman and Bristow (1990) have also recorded similar values in well-structured clay soils on the Atherton Table-land when they were initially dry. The EFA index does not take account of the soil fabric within the profile, so no cross-correlation is possible. Advice from *eriss* personnel who visit Nabarlek during the wet season tell of water logging on the evaporation ponds. That might be a time to measure field infiltration, though it would be technically difficult to do so at those times. It is important to note that the index did not predict high infiltration when low infiltration was the reality. That is, no problem was overlooked. We found a similar problem at a newly developed site at Kelian that was comprised of paddock-dumped well-structured clay soil materials.

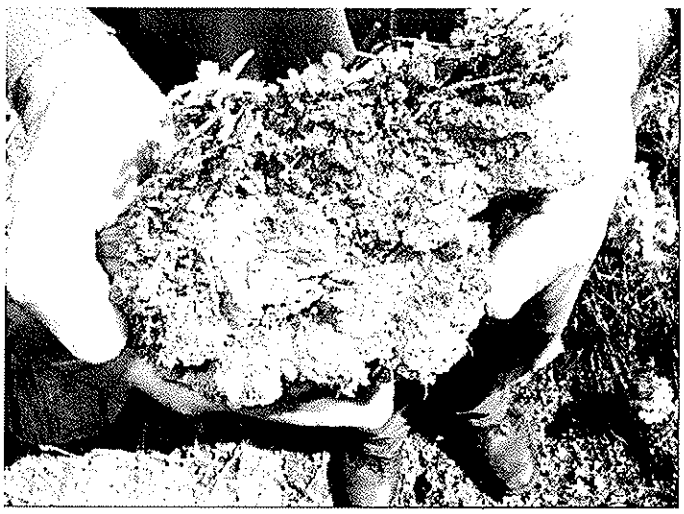


Plate 4.5. The clay soil from the grassy evaporation pond rehabilitation at Nabarlek, showing copious amounts of macro organic matter in the surface undergoing decomposition. Note the soil "darkening" and the open friable/fractured structure of the sub-surface clay when dry. These properties confer very high measured infiltration when as dry as this. In the wet season, the clay would swell and air voids would be eliminated: the infiltration would more closely resemble saturated hydraulic conductivity.

4.6. The functional role of vegetation: a new graphical index.

Typically, vegetation development assessment on mines looks at species composition and growth rates. These assessments use well-developed and time-honoured procedures that have been in the literature without substantive modification for decades, and with a history extending back well over a century (Bonham 1989). These methods remain important, but like many other monitoring procedures, represent past behaviour without a strongly developed predictive capacity. That is, if species X is missing from the rehabilitation, there is nothing in the basic data that can assist in deciding why this is so or what to do about it. Individual observers may use their experience and intuition to make practical suggestions, but these are not intrinsic to the data.

In working towards addressing Objective 2, we have devised a vegetative index that has value at "site" or "hillslope" scale and refers specifically to the role of vegetation in ameliorating resource transfer by wind and water. This adds to the information collected in the "landscape organisation" (L/O) stage of EFA data collection. In L/O, both biotic and abiotic features that regulate the flow of resources are assessed on a line transect. This provides an assessment of the "mix" of resource regulation processes, but probably under-samples the role of the variety of vascular plant life forms.

The new vegetation index involves collecting data on plant cover and structure and differentiating the data in 0.5 m height classes, so that the role of vegetation in affecting surface flow can be distinguished from the role of canopy in affecting wind flow and turbulence. Typically, we collect data according to life form: grasses, low shrubs, tall shrubs and trees in separate assessments. The data are collected by "distance measuring, plot-less" procedures such as point-centred quarter (PCQ) for dense vegetation and wandering quarter (WQ) for sparse vegetation (Bonham 1989). At each plant, we measure the major and minor canopy dimensions, total height, height from the ground to the canopy base, and canopy density (McDonald et al 1990). These data are displayed as cover in square metres per hectare (x-axis) in 0.5 metre height classes (y-axis). This presentation provides a pictorial view of vegetation canopy distribution as a cross-section (Fig. 4.6.a). This presents data representing 3-dimensions on a 2-dimensional diagram. These data can be analysed by the "ecosystem trajectory" procedure described in section 4.7, below. An expectation for successful rehabilitation would be a high initial ground cover spread, followed by a vertical extension of canopy as shrubs or trees developed, heading towards values exhibited by the analogue site. Reaching analogue site values would not be expected in the time frame of deciding on rehabilitation success, as some life forms take many decades to fully develop. It is the shape of the trajectory as shown by regular monitoring that would comprise the evidence for success in rehabilitation.

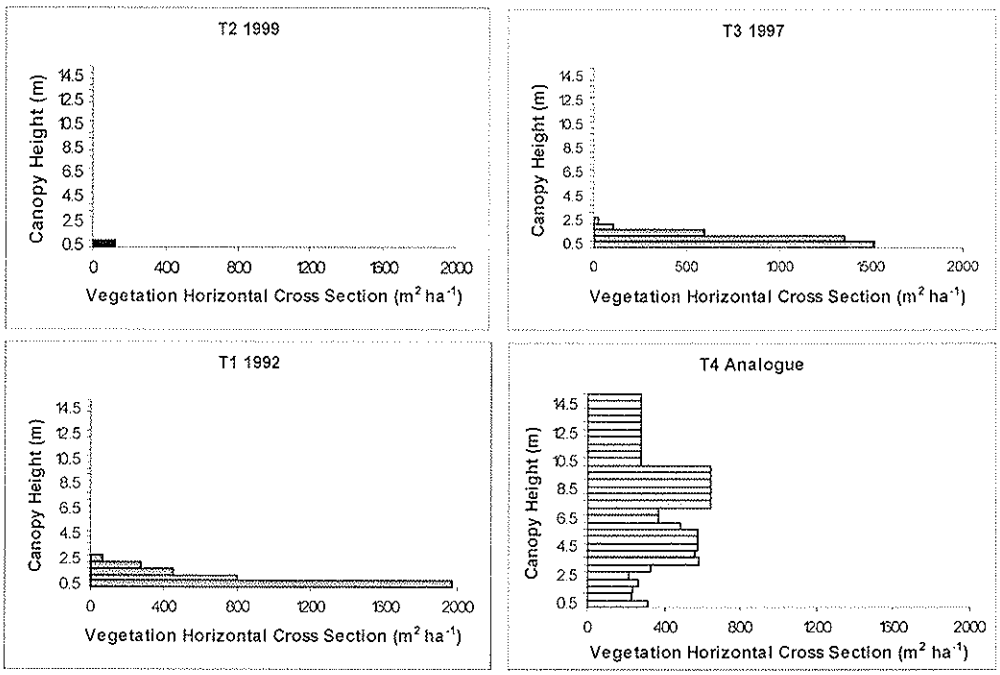


Figure 4.6.a. The vegetation horizontal cross sectional area for three rehabilitation sites and the analogue at Carnilya Hill, resolved into 0.5 metre vertical height intervals. There is still a considerable difference in the functional vegetation structure between the rehabilitation and the analogue. However, Eucalypt saplings growing on the rehabilitation sites are now 3 m high, so appropriate progress is being made.

Studies of non-mined lands subject to degradation by management have shown that good ground cover of vegetation and plant litter is a common feature in all highly functional landscapes. Conversely, on dysfunctional landscapes, lack of vegetation at ground level or lack of plant litter causes resources to be transported by wind or water too rapidly across the surface, resulting in too little water infiltrating or excessive loss of soil or plant litter by erosion. This is a concept that applies to mined lands as well. Initially of course, it is the geotechnical set-up of the mined lands that provides the initial site stability, but over time, biological development should dominate the ecosystem function. In Stage 1, this concept was presented as in Fig 4.6b. EFA data can be discriminated to show this transition from physically dominated to biologically dominated processes.

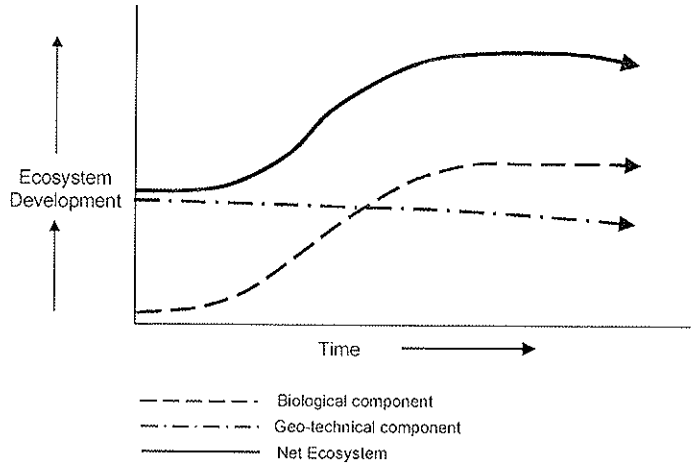


Figure 4.6.b. The change from Physical to Biological control on Rehabilitation over time and overall system response.

4.7. Assessment of rehabilitation success

4.7.1. The conceptual framework and practical application.

Most monitoring systems or procedures are focused solely upon showing that the proposed indicators suitably reflect the status of the system under examination. Many have been developed to assess degradation scenarios. EFA explicitly sets out to not only track the system status over time but also to specify a target region for EFA index values for self-sustaining ecosystems and assess the rate at which the system is approaching that target region. These design features are vital for full practical implementation. EFA data is well placed to achieve these design criteria because the emphasis is on ecosystem processes, rather than biota as such. Deficient or slowly changing process rates quickly indicate to rehabilitators the nature of the problem they face, whereas if, say, ants are the indictor, the nature of needed restitution is not clear and requires further investigation. It is the focus on processes that makes EFA a tool that can be used across a wide range of landscape/climate combinations and land use variation.

In Stage 1, the concept of "ecosystem trajectory" was proposed: the comparison of rehabilitation sites over time with appropriately selected analogue sites as the means of assessing rehabilitation success (Fig. 4.7.a). The curves would be generated by time series monitoring, plotting successive values of each index over time and examining the emerging curve shape. In the early stages of rehabilitation, assessment should be relatively frequent in order to observe the initial response rate. These data would give rise to a true time-series graph in normal practice. This concept arose from empirical observations of rehabilitation progress on a large number of mines: successful rehabilitation was characterised by a steep initial response, slowly flattening off over time, whereas less successful rehabilitation was characterised by either very slow response rate or a "saw-tooth" trajectory with no perceptible functional improvement. The data collected to develop this concept were from pseudo time series; sites assessed at a single time at different time intervals from the commencement of rehabilitation. This would never be as accurate as a true series where the same site is monitored over time. Nevertheless, a potentially useful concept emerged.

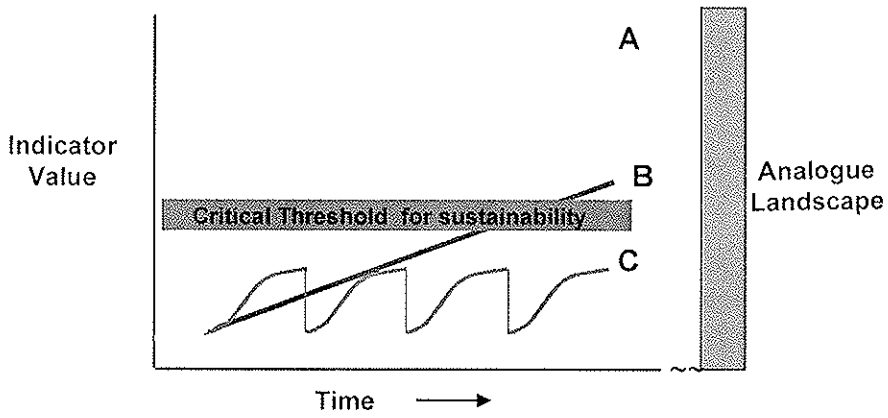


Figure 4.7.a. Three contrasting ecosystem function trajectories. Trajectory A shows a satisfactory response over time, passing rapidly through a critical functional threshold and continuing to improve. It is likely to be self-sustaining. Curve B represents a system that develops slowly and hence subject to stochastic events and possible failure. Curve C represents a system that frequently succumbs to external threats and fails to develop into a self-sustainable system.

In this project, we were able to provide a demonstration of this concept, but only at the Alcan Gove Bauxite Mine site. This site was the only one where a very similar rehabilitation technique had been regularly used for a continuous 26-year period. All other mines in the Stage 2 study had either changing techniques over time or had been operating for too short a period, or both for a proper demonstration of the concept. We had independently looked at the concept in a rangeland context for the National Land and Water Audit (Tongway & Hindley 2000), and found that a sigmoidal curve was particularly useful in describing the behaviour of the data. Previous practical and theoretical studies had already strongly suggested that this form of the curve was particularly appropriate and useful (Graetz & Ludwig 1978, Krebs 1972, Noy-Meir 1981). This curve shape indicates that, in the minesite context, after the initial rehabilitation landscape set-up, there was a brief plateau period followed by a rise in landscape function as the biota established and became active in ecosystem processes that gently flattened off after some years, eventually forming an upper plateau representing the biogeochemical potential of the site. This latter region is the "target" mentioned earlier and depends on the parent soil material and the climate. In practice, this value in mined lands, would come from a set of analogue sites similar to the final rehabilitation landscape phase in terms of slope, soil surface properties, vegetation composition and land use. Assessing the analogue sites would be an integral part of monitoring rehabilitation and in practice would generate a "band" of values depending on seasonal effects as well as stochastic events like storms, droughts and fire. Fig. 4.7.b shows the sigmoidal curve derived from a pseudo time series analysis of the Gove sites over a 26-year time span for the stability index. The sigmoidal shape conforms to the conceptual shape very neatly, with the exception of year 8.

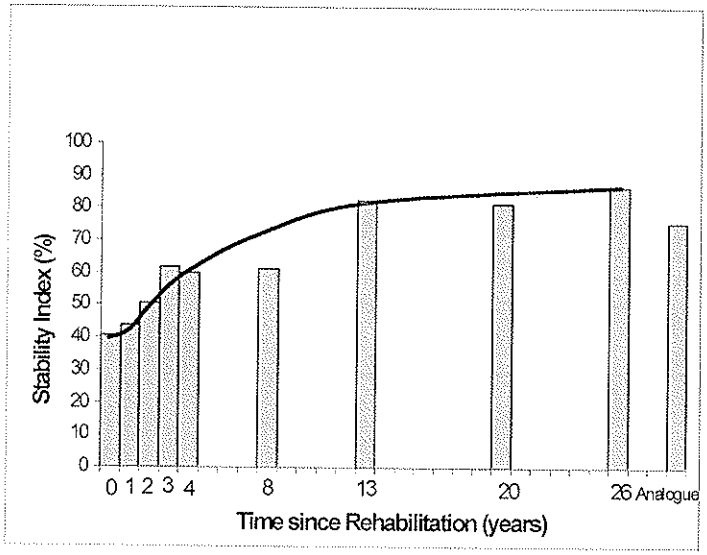


Figure 4.7.b. The sigmoidal curve reflecting the trajectory of stability development at Gove. Note that the analogue site value is lower than the older rehabilitation sites, due to effect of frequent burning of the analogue site.

This set of curves shows a considerable difference in the time to achieve plateau values. Both infiltration (Fig 4.7.c) and nutrient cycling (Fig. 4.7.d) rely on the long-term development of the vegetation and the increasing activity of soil "ecosystem engineer" fauna (Lavelle 1997). However, looking at the rate of increase of the indices at early times remains a powerful tool. At Gove, the utility of the analogue concept is affected by the unremitting fire regime in lands outside the mine.

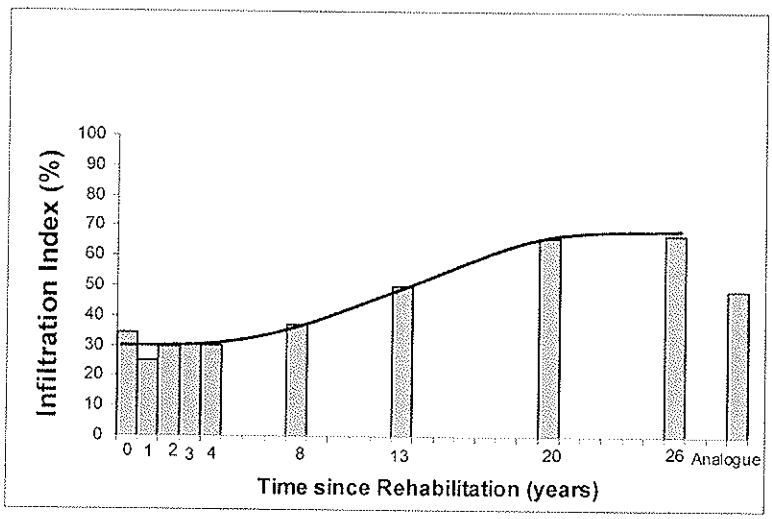


Figure 4.7.c. The sigmoidal curve reflecting the trajectory of infiltration development at Gove. Note that the development "lags" the stability curve (Fig 4.7b).

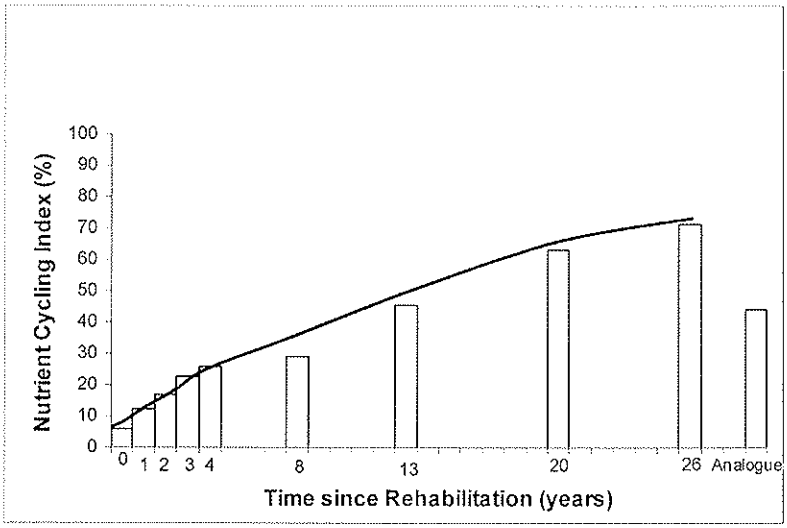


Figure 4.7.d. The trajectory of the nutrient cycling index at Gove. The full sigmoidal shape is not evident. The ecosystem has not yet reached a true plateau, even after 26 years, due to the growth of *Eucalyptus tetradonta*, which will continue for some time yet, with increasing canopy biomass and litter fall. However, the utility of the early steep rise of the sigmoidal curve remains as a useful tool in assessing ecosystem development.

5. CONCLUSIONS

5.1. EFA index verification.

The EFA indicators were shown to have a very high degree of verification with the measured properties in the surface soil. Mined lands present unique and difficult places in which to do conventional soil measurements, most of which were developed for agricultural field application. Where verification was not fully met, the reasons behind this have been explained in the context of the specific locations.

The EFA procedure was the same for all the minesites, which was an original design factor: generic procedure. As the sites varied from sandy deserts with 200 mm a^{-1} rain fall to tropical rain forests with about 4000 mm a^{-1} rainfall the method has shown very broad potential application. It may well be possible to refine the procedure for use at specific locations to give results with higher precision, or greater sensitivity, but this is not necessary for monitoring, especially if the ecosystem trajectory process is followed.

The quality of many of the relationships were sufficient to use to model soil stability, infiltration and nutrient cycling if hillslope and above scale studies were required. EFA could provide an extensive data set of adequate quality at a fraction of the cost of direct measurements.

5.2. Functional vegetation data.

We have proposed a new way of representing the contribution of vegetation to ecosystem function at the hillslope scale by representing the role of vegetation layer by layer. By this means, we can show the value of the ground layer in ameliorating overland flow and erosion of soil and transport of macro-organic matter at the same time as the role of developing canopy spread that will intercept wind, causing it to become turbulent and losing the capacity to hold resources in suspension. The sites where competent foliage is present may well become "sinks" for airborne resources such as dust organic matter and propagules.

5.3. Ecosystem trajectory: a tool to judge rehabilitation success.

The Stage 1 final report proposed that an analysis of the curve shape formed by the index values obtained by regular monitoring of rehabilitating lands should be the tool by which rehabilitation success should be judged. EFA has up to five landscape organisation indices and three soil surface indices with which to look at the ecosystem "trajectory". Between them, these indices cover the range of scales and the complexity of issues that make up ecosystem function. Only one study site in Stage 2 permitted the full demonstration of this proposition. This was because a very similar rehabilitation technique had been used over a long time period, which enabled us to trace the full shape of the trajectory curve. This was a useful demonstration in showing functional rehabilitation success. The Eneabba site had an example of 22-year-old rehabilitation that had made slow functional progress, but due to the "once-off" assessment, the shape of the trajectory was not possible to delineate. The "trajectory" proposal appears to be well enough demonstrated to recommend implementing as an overall indicator of ecosystem rehabilitation success.

5.4. Communication of the Project outcomes.

We supplied each major Sponsor with Site Reports soon after our initial field visit, followed up by a Sponsors meeting at about half term at which we presented data already processed. Draft Final Site Reports have recently been prepared and sent to the relevant Sponsor with this overall Project Final Report complete the reporting process. A revised technical manual is also being prepared as a final communication. This manual will replace the existing field manual, the training course notes and

will include the Excel templates. We expect to make this available as a CD or on the CSIRO Sustainable Ecosystems and ACMER websites.

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7. APPENDICES

These appendices contain the processed data from the project. Each mine's data is presented in the same order.

7.1 Brocks Creek

7.1.A. Stability

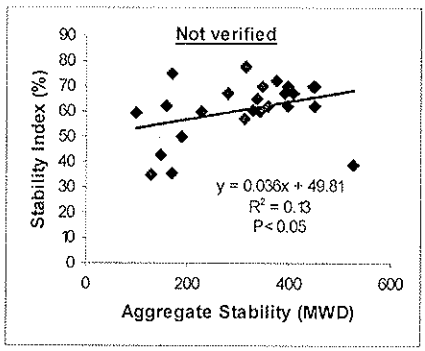

Figure 7.1.a. The relationship between measured aggregate stability and the EFA stability index. The relationships for individual sites were poor or non-existent and all sites lumped give a noisy plot.

7.1.B Infiltration

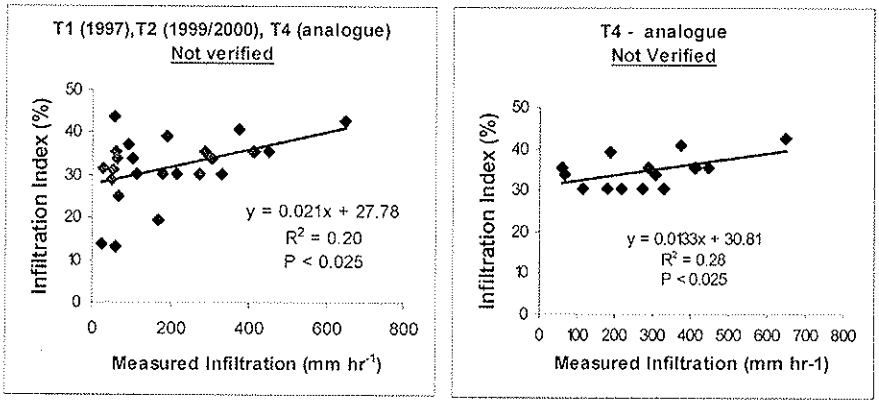

Figure 7.1.b. Due to the rocky nature of the surface, it was difficult to set up the infiltrometer without disturbing the soil and thus the relationships obtained were poor.

7.1.C. Soil Respiration

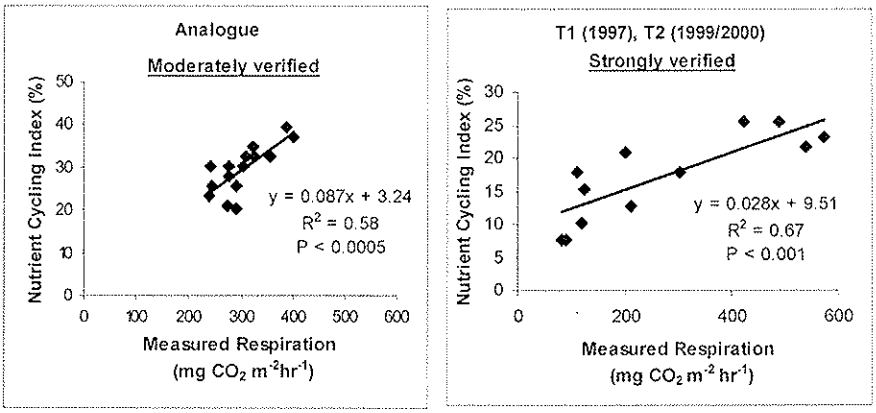

Figure 7.1.c. Soil respiration as a function of the EFA nutrient cycling index. The analogue site and the rehabilitation sites had good independent relationships, but different response functions.

7.1.D. Nutrient Pool size.

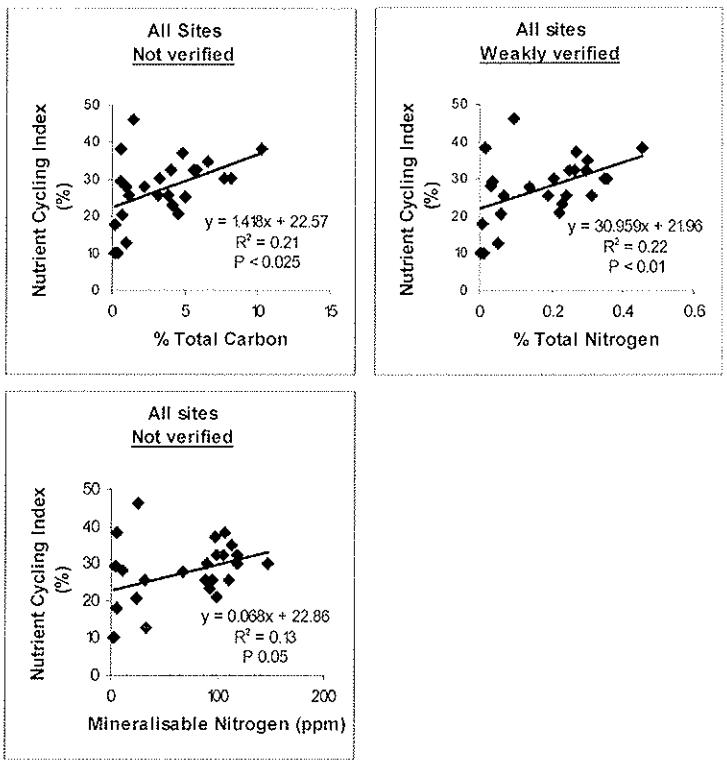

Figure 7.1.d. The relationships between total N, total C and mineralisable N verus the EFA nutrient cycling index.

7.1.E. Functional Vegetation

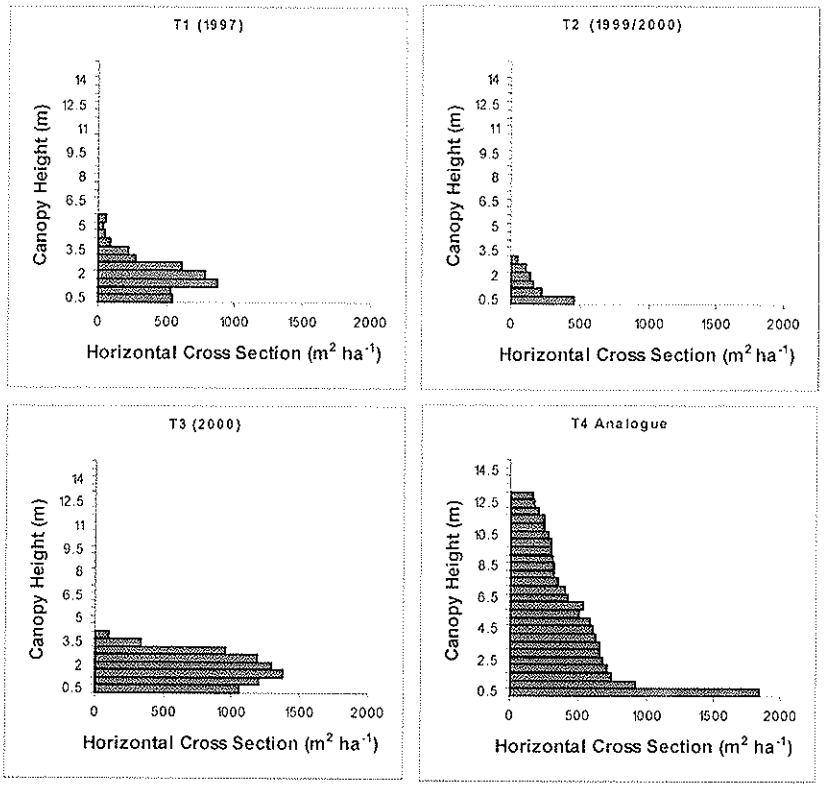

Figure 7.1.e. The horizontal cross sectional area of the foliage is plotted against the canopy height in 0.5 metre intervals to show the vegetation development over time. It would be anticipated that successful rehabilitation would reach the values of the analogue in a reasonable time.

7.2 Carnilya Hill

7.2A. Stability

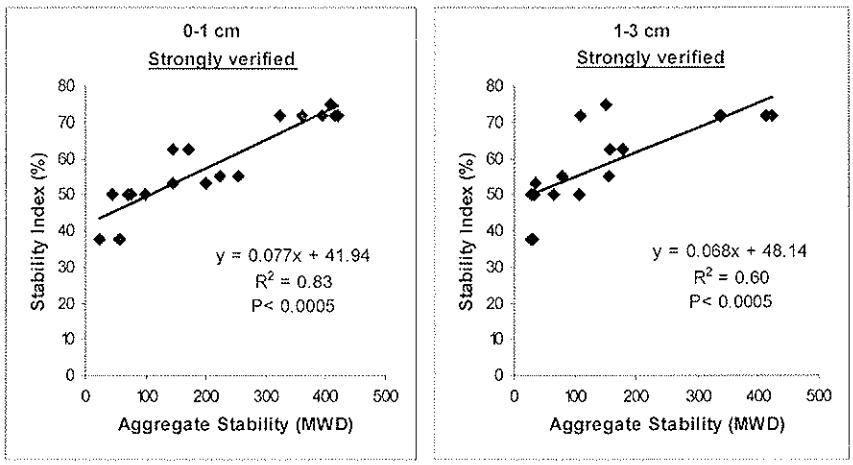

Figure 7.2.a. show the relationship between the laboratory measurement (MWD) of stability of the 0-1 and 1-3 cm layers of soil and the EFA stability index. These data show that a good linear fit with high statistical significance for the data across all rehabilitation and analogue sites. This verifies that the EFA stability index can be used to assess ecological development across all the Kambalda landscapes.

7.2.B. Infiltration

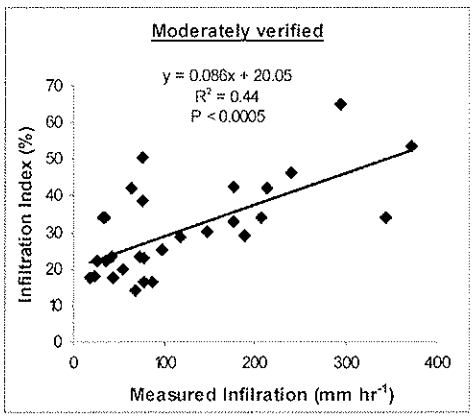

Figure 7.2.b. The relationship between saturated flow infiltration rate and the EFA infiltration index.

7.2.C. Soil Respiration

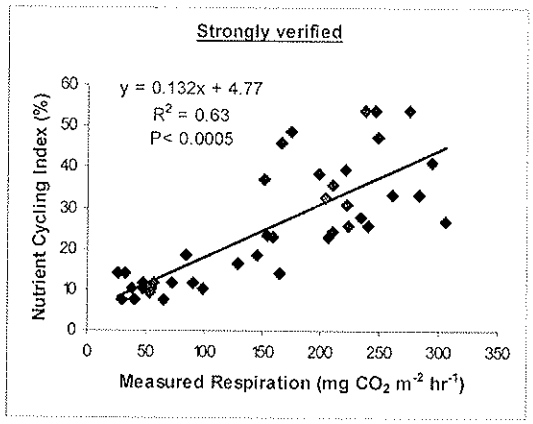

Figure 7.2.c. The relationship between soil respiration rate and the EFA nutrient cycling index for all sites combined at Carnilya Hill.

7.2.D. Nutrient pool sizes

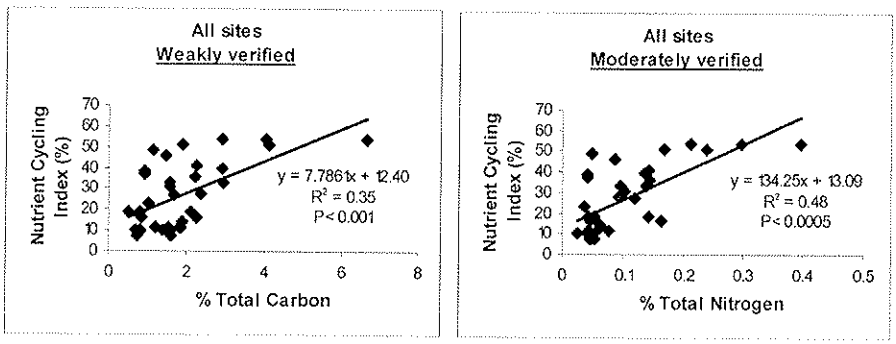

Figure 7.2.d.(i) The linear relationship between total organic carbon and total nitrogen and the EFA Nutrient Cycling Index across all sites at Carnilya Hill.

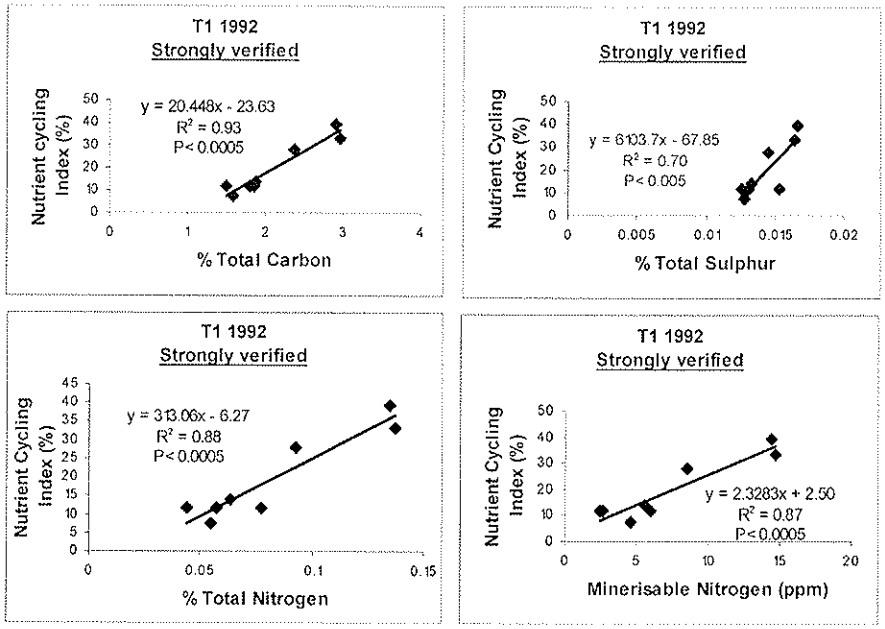

Figure 7.2.d.(ii). The relationship between laboratory measured values for total carbon, total sulphur, total nitrogen and mineralisable nitrogen and the EFA nutrient cycling indices for transect 1.

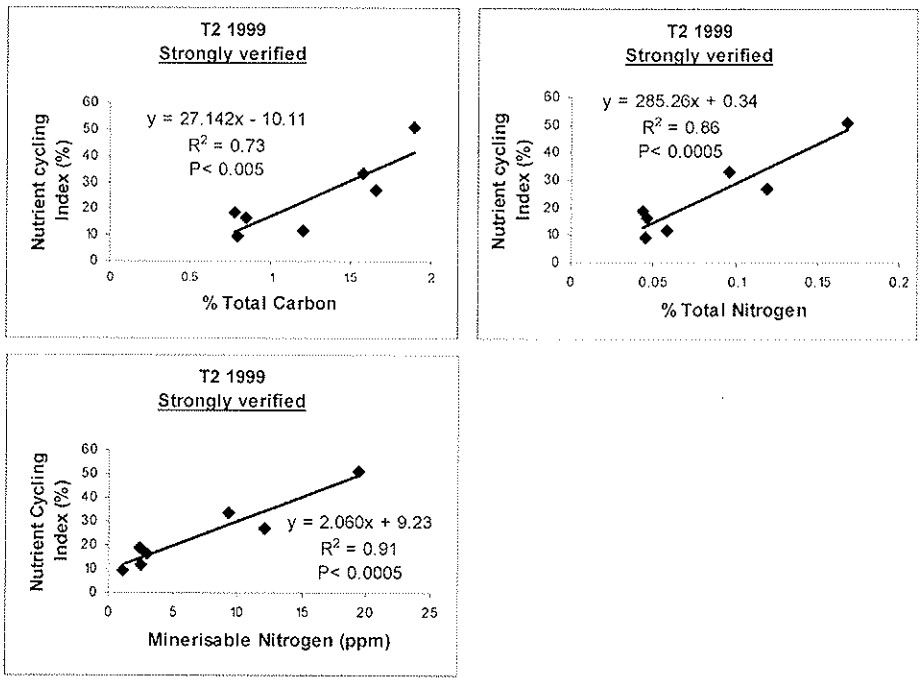

Figure 7.2.d.(iii). The relationship between laboratory measured values for total carbon, total nitrogen and mineralisable nitrogen and EFA nutrient cycling indices for transect 2, Carnilya Hill.

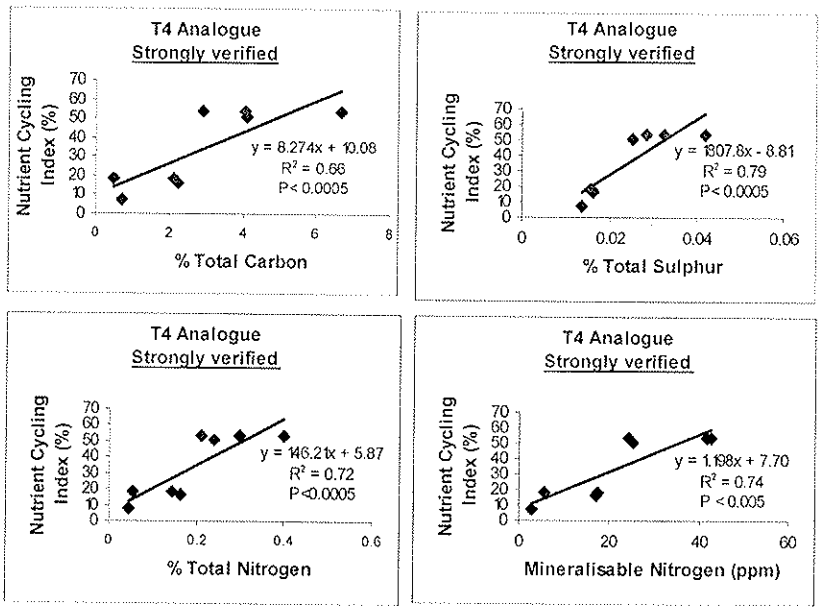

Figure 7.2.d.(iv). The relationship between laboratory measured values for total carbon, total sulphur, total nitrogen and mineralisable nitrogen and EFA nutrient cycling indices for the analogue site .

7.2.E. Functional Vegetation Index

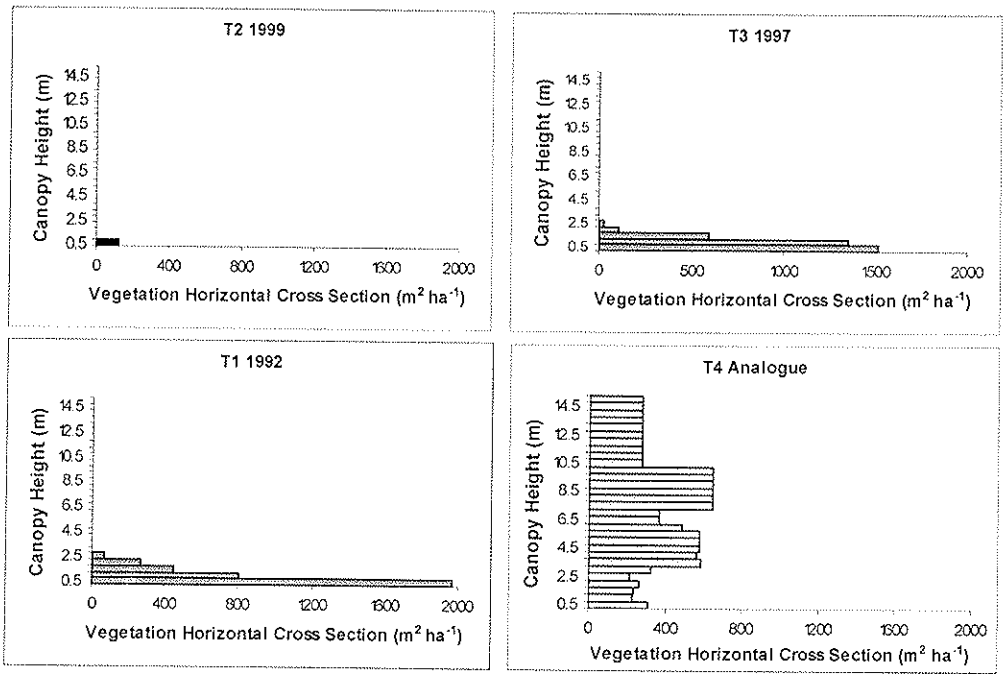

Figure 7.2.e. The vegetation horizontal cross sectional area for all four sites measured resolved into 0.5 metre vertical height intervals.

7.3. Eneabba

7.3.A. Stability

It was not possible to verify this index at Eneabba because the sandy soil at the site was non-coherent and the water stable aggregation procedure did not yield any interpretable results.

7.3.B. Infiltration

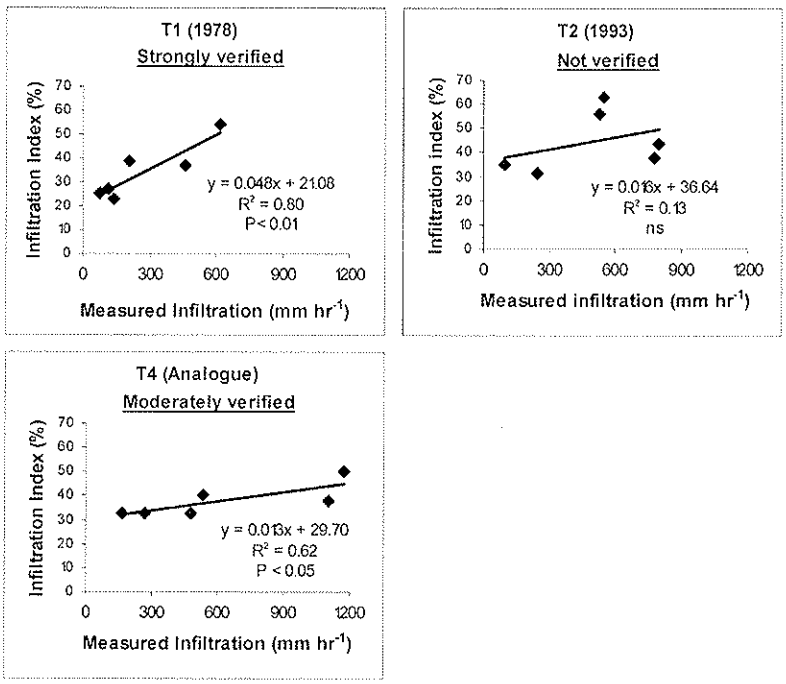

Figure 7.3.b. There were variable and only partially reconcilable relationships at Eneabba for infiltration. The analogue site and T1, the oldest site, both possessing surface crusts had useful, within-site relationships, but T2, with a mobile, non-cohesive sandy soil did not.

7.3.C. Soil Respiration

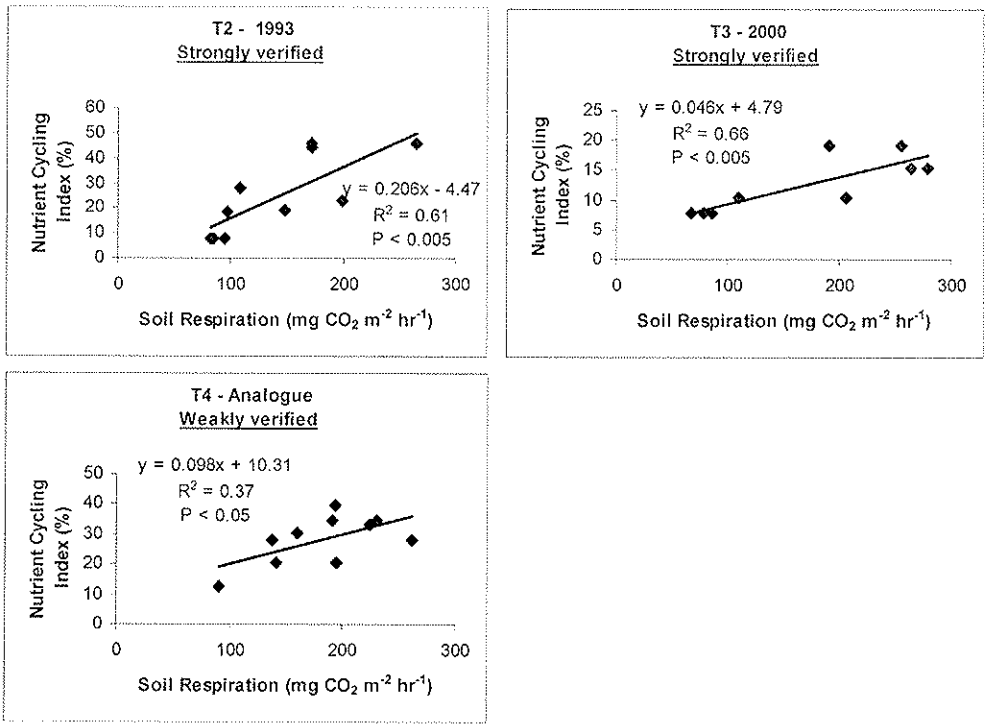

Figure 7.3.c. Soil respiration rate verus the EFA nutrient cycling index. These graphs reveal significant correlations between the index and measured respiration for the two 'younger' sites T2 (1993) and T3 (2000) and the analogue site, but the differing slope values militate against a single practical regression.

7.3.D. Nutrient Pool size

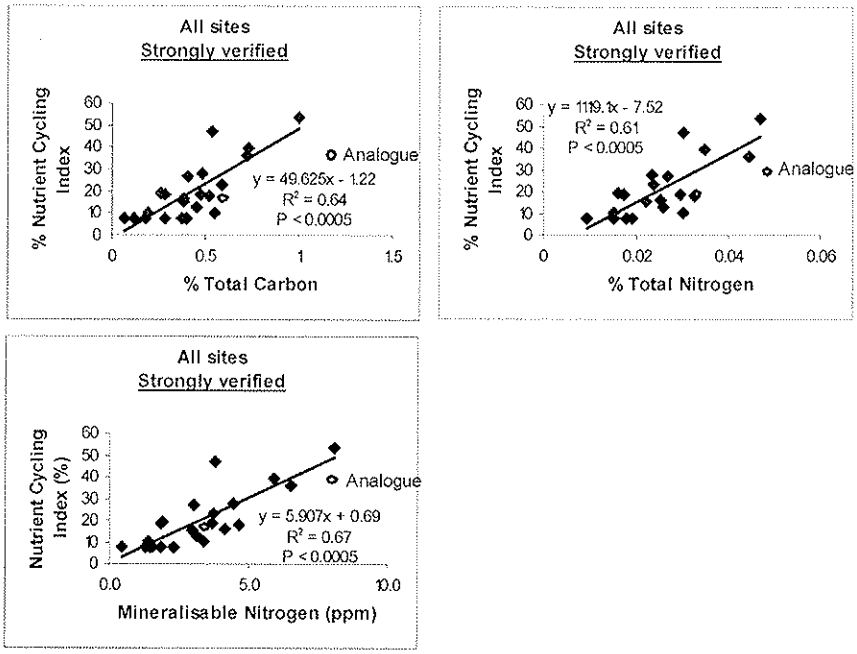

Figure7.3.d. The relationship between the EFA nutrient cycling index and soil total N, total C and mineraliseable N.

7.3.E. Functional Vegetation Index.

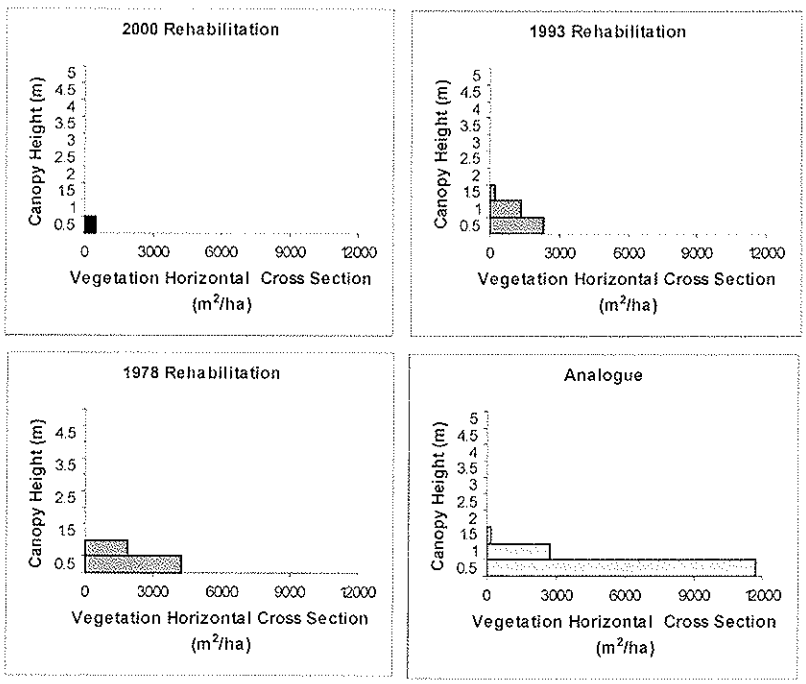

Figure 7.3.e. The vegetation horizontal cross sectional area for all four sites measured, resolved into 0.5 metre vertical height intervals.

7.4. Gove Bauxite Mine

7.4.A. Stability.

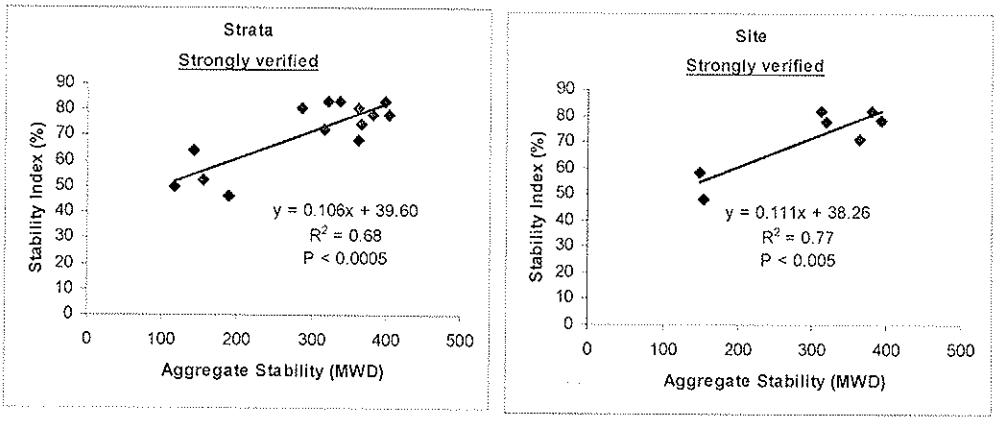

Figure 7.4.a. Shows the significant relationships between the EFA stability index and measured aggregate stability for all individual landscape strata (left hand) and the site means (right hand).

7.4.B. Infiltration.

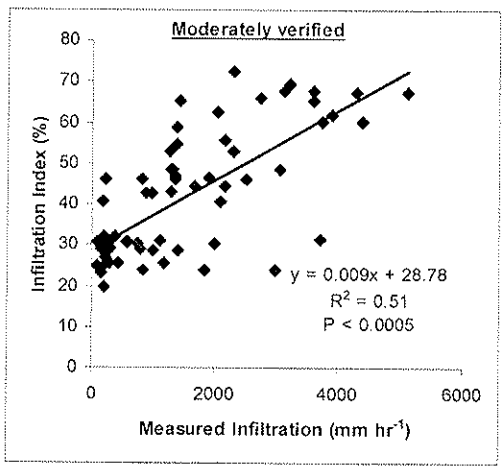

Figure7.4.b. Shows the relationship between the EFA infiltration index and measured infiltration. Data from all studied rehabilitation sites and the analogue site are included in this plot. Note that measured infiltration rates are very high at all locations and do not present a problem in rehabilitation.

7.4.C. Soil Respiration.

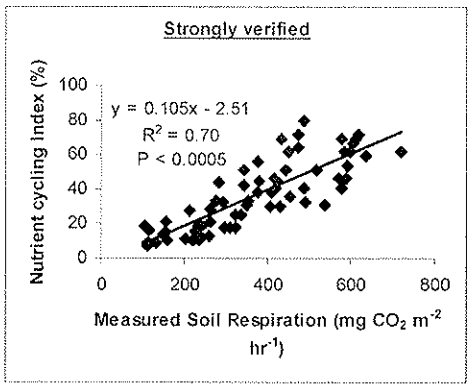

Figure 7.4.c. Shows the relationship between the EFA nutrient cycling index and measured soil respiration, using data from all the studied sites. The tight relationship verifies that the EFA nutrient cycling index can be used as a surrogate for costly and time-consuming measurements in monitoring rehabilitation progress.

7.4.D. Soil Nutrient Pool Size

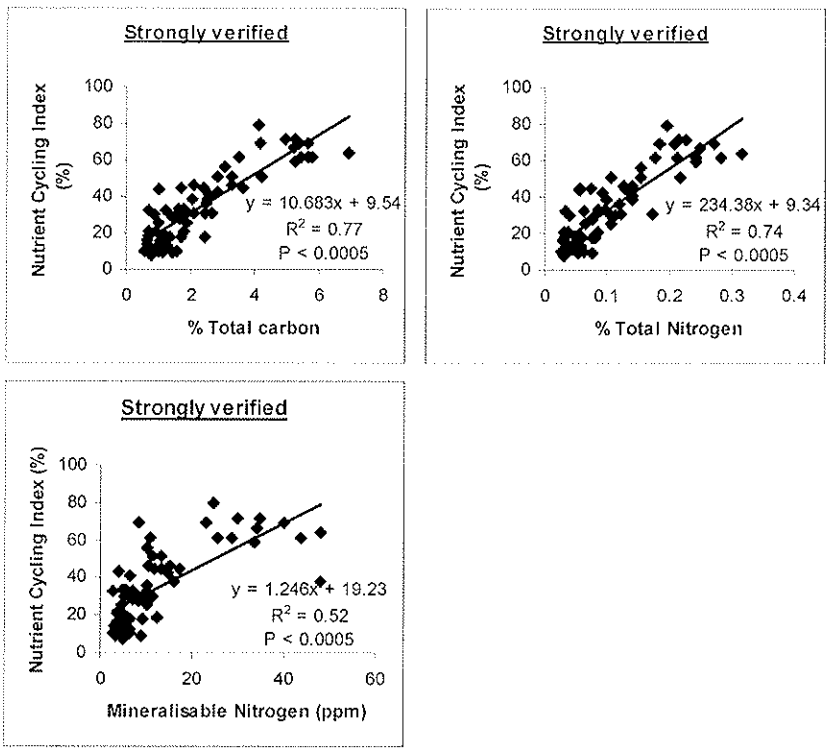

Figure7.4.d. The relationships between the EFA nutrient cycling index and measured levels of total carbon, total nitrogen and mineralisable nitrogen at Gove.

7.4.E. Functional Vegetation Index.

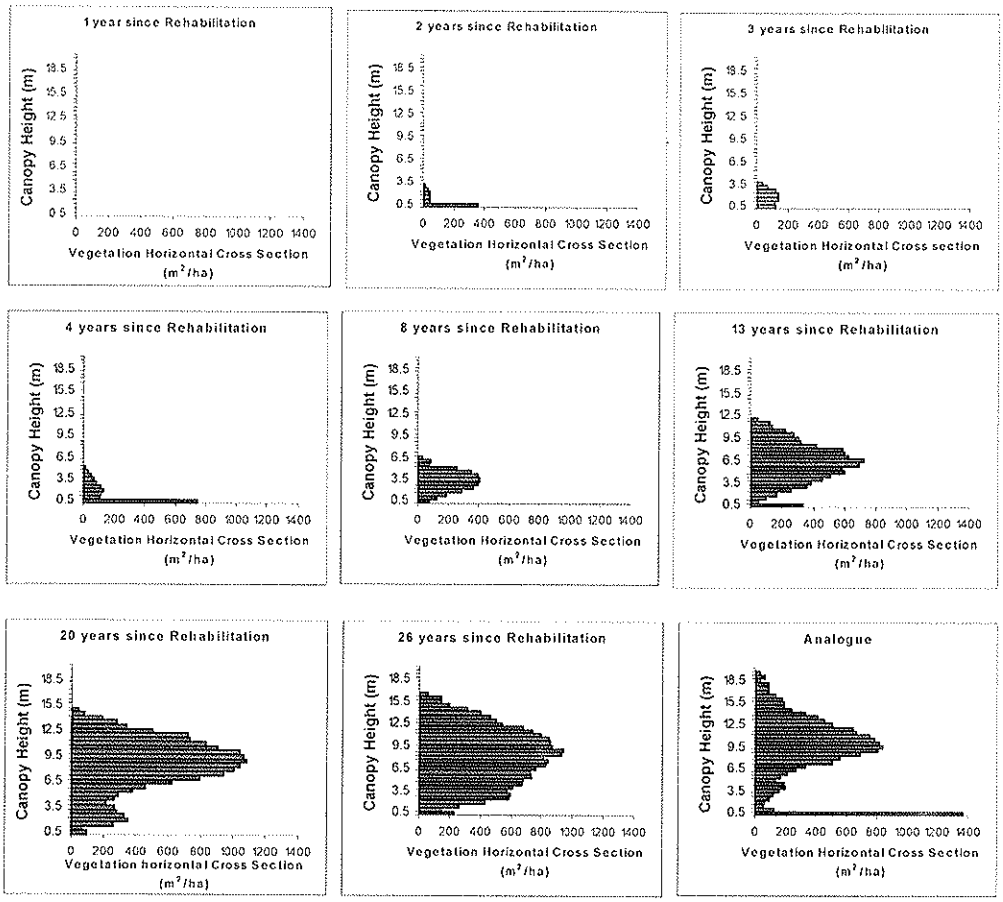

Figure 7.4.e. The vegetation horizontal cross section area for all nine sites measured, resolved into 0.5 metre vertical height intervals. Note that the rehabilitated lands develop a good canopy structure, but do not emulate the ground cover of the analogue site.

7.5 The Granites

7.5.A. Stability.

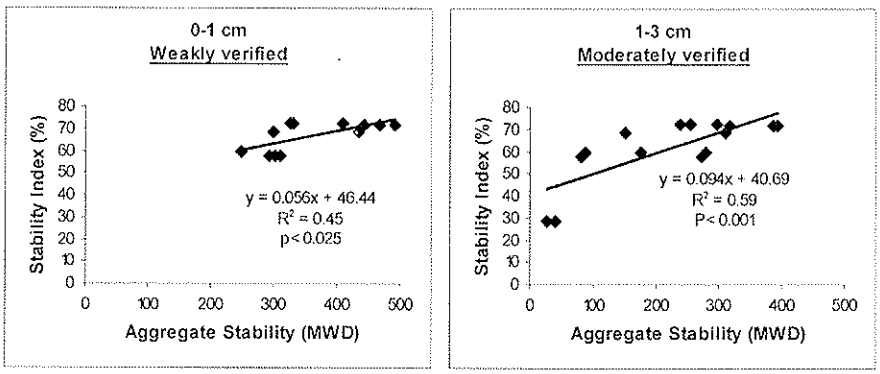

Figure 7.5.a. Two relationships are shown. The left hand graph represents the 0-1 cm layer, where data were available: a reasonable relationship but restricted in dynamic range, due to the loose, non-coherent nature of some samples. The right-hand side graph represents the 1-3 cm layer, where a full set of data was available.

7.5.B. Infiltration.

At The Granites the relationship between EFA infiltration index and measure infiltration was not verified, due to the rocky nature of many of the soils. Too few data were collected and over a very limited dynamic range.

7.5.C. Soil Respiration.

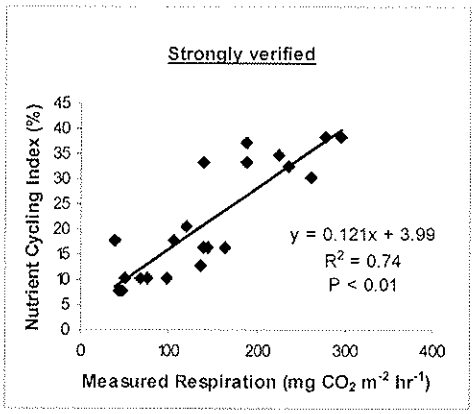

Figure 7.5.c. The relationship between soil respiration rate and the EFA nutrient cycling index. Data are combined from all sites.

7.5.D Soil Nutrient Pool size.

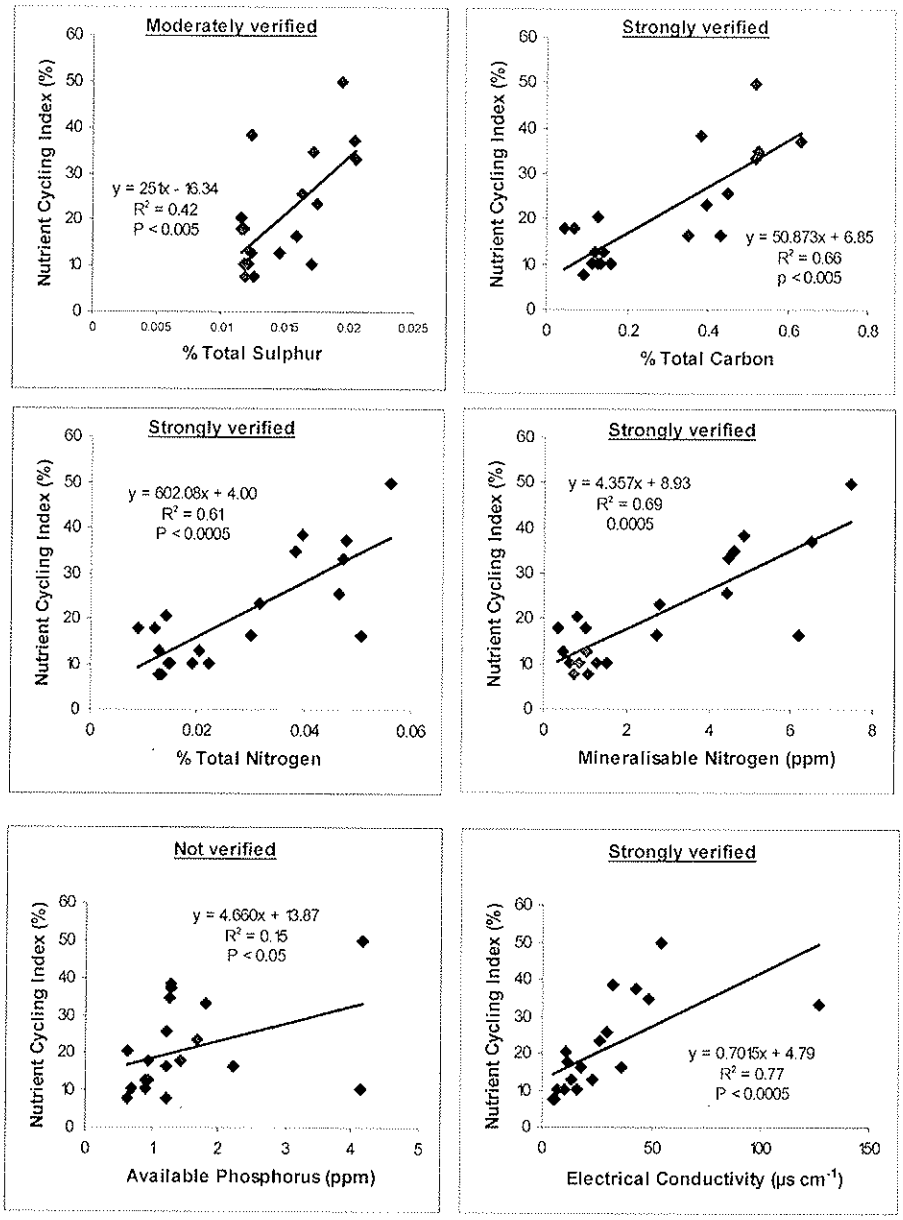

Figure 7.5.d. The relationships between the EFA nutrient cycling index and measured levels of total carbon, total sulphur, total nitrogen, available phosphorus, mineralisable nitrogen and electrical conductivity in the top 1 cm of the surface.

7.5.E. Functional Vegetation Index.

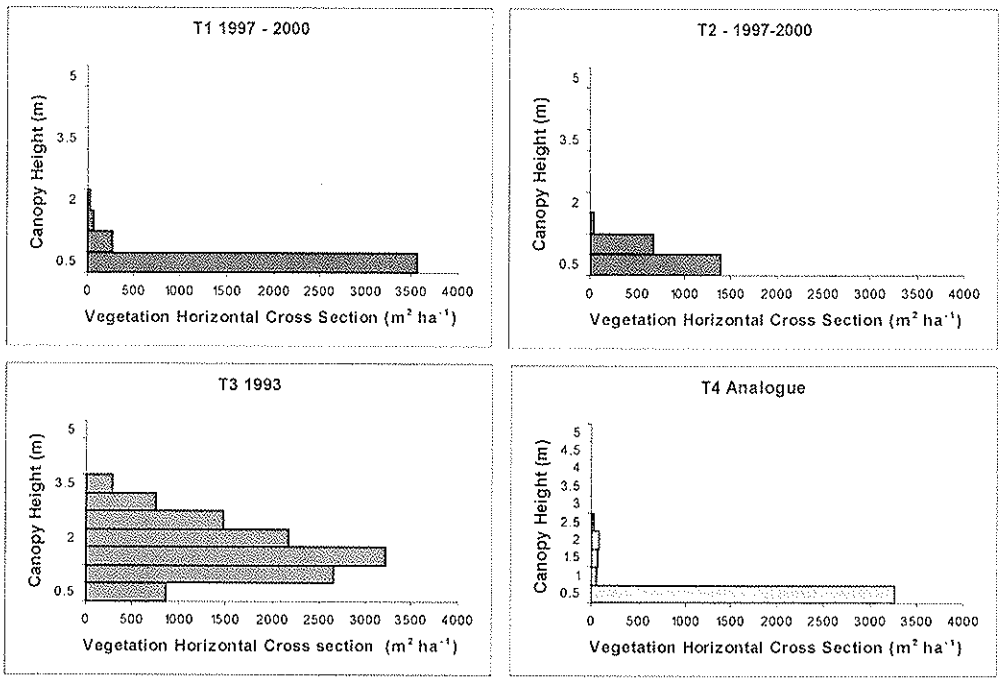

Figure 7.5.e. The vegetation horizontal cross-section area (m^2 ha^{-1}) expressed at 0.5 m height increments at each site.

7.6 Gregory Coal Mine

7.6.A. Stability

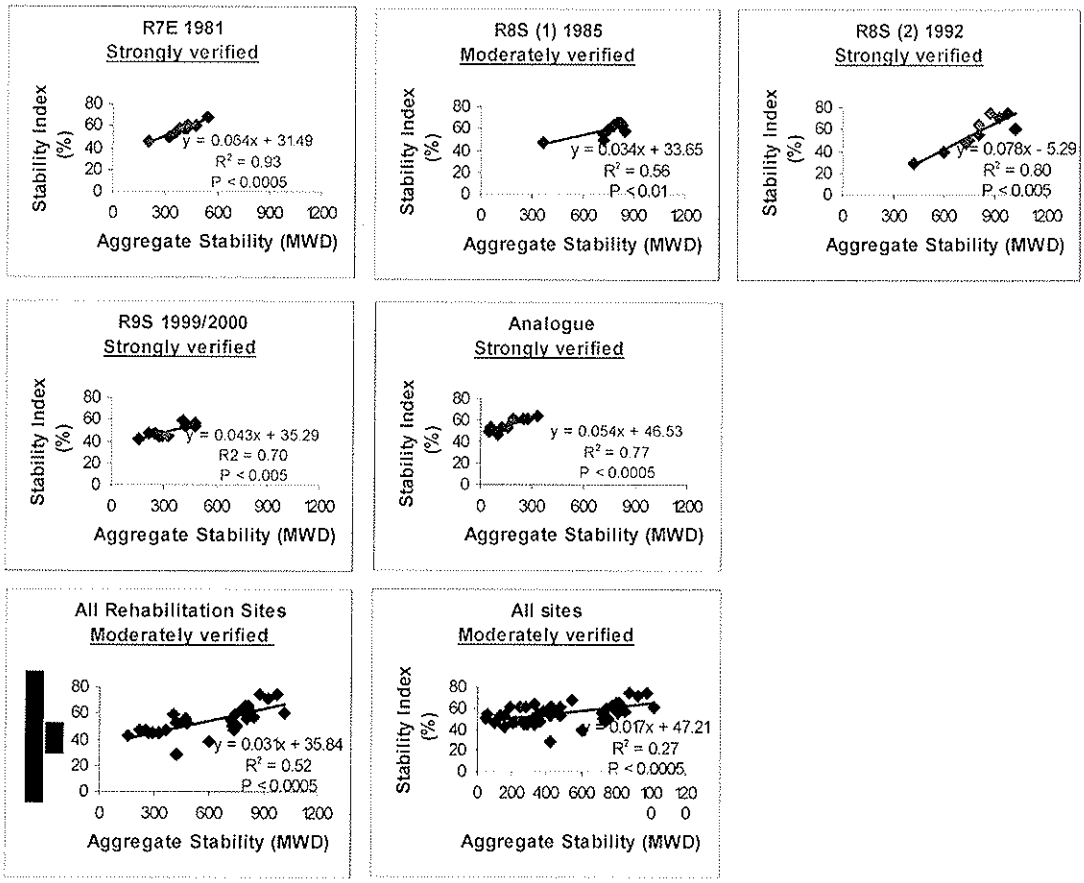

Figure 7.6.a. Showing the significance of relationships between the EFA stability index and measured aggregate stability for each of the individual sites, all rehabilitation sites and all sites. It be seen from the regression equations that the slope of the relationship is different for each site, and that the combined data set is noisy.

7.6.B. Infiltration

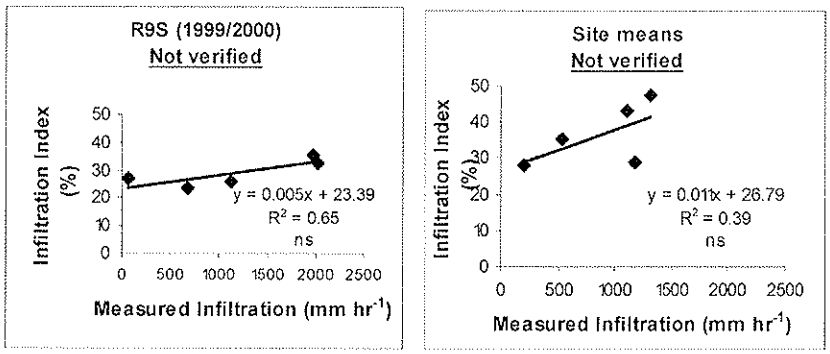

Figure 7.6.b. Showing the relationship between the infiltration index and measured saturated for R9S (left) and site means (right). It was technically difficult to set the infiltrometer up due the nature of the soil surface and the prevalence of root masses under the grasses present on some sites.

7.6.C. Soil Respiration

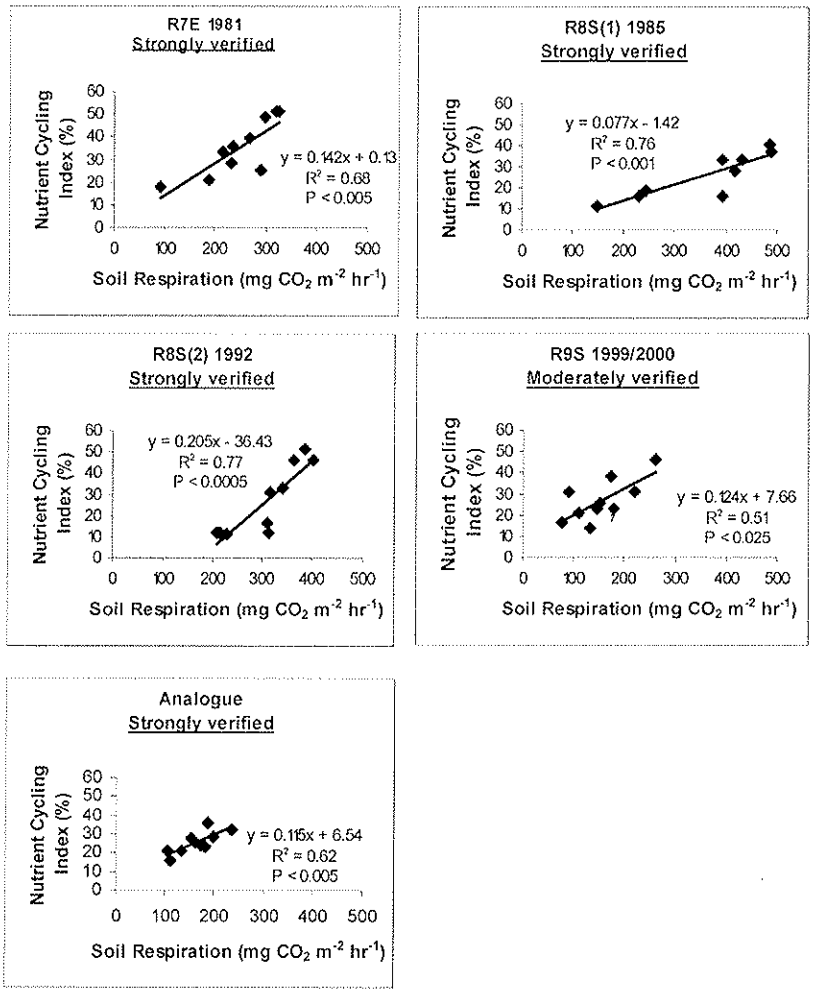

Figure 7.6.c. The relationships between the EFA nutrient cycling index and the measured soil respiration.

7.6.D. Soil Nutrient Pool Size.

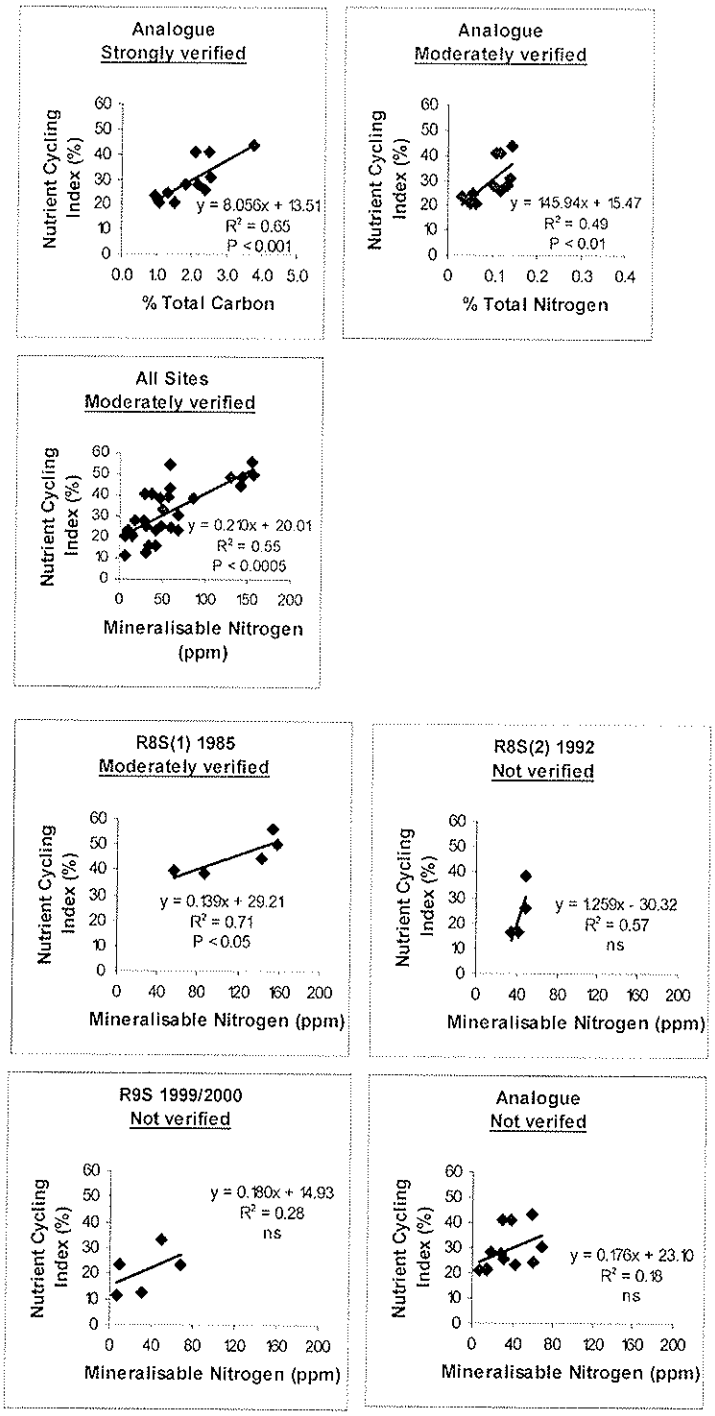

Figure 7.6.d. Showing the relationship between the EFA nutrient cycling index and total nitrogen, total carbon for the analogue and mineralisable nitrogen.

7.6.E Functional Vegetation index

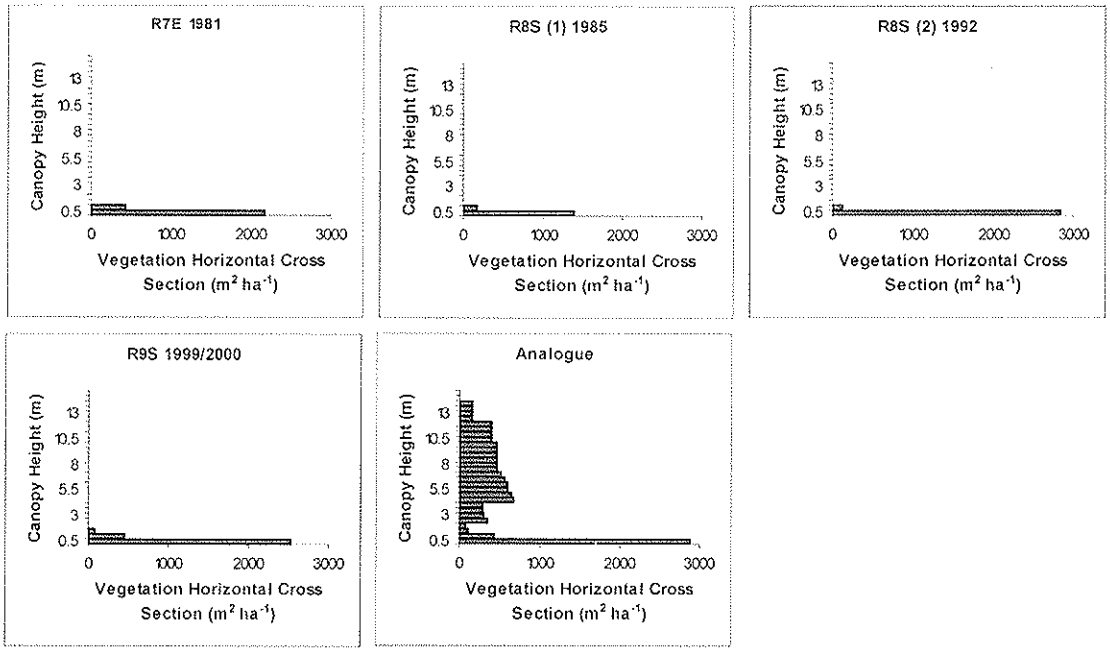

Fig 7.6.e. The vegetation horizontal cross section area for all sites at Gregory. The rehabilitation sites after about 20 years are similar to the analogue site for the cover to 0.5 m., but there is no tree canopy developing (by design), so the rehabilitation will always lag the surrounding woodlands.

7.7 Kelian Gold mine

7.7.A. Stability

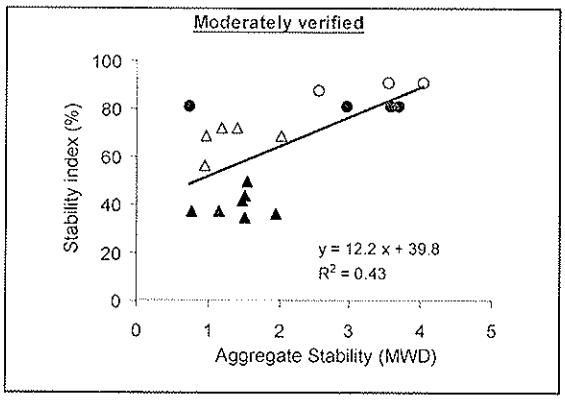

Figure 7.7.a. The relationship between the EFA stability index and aggregate stability.

7.7B. Infiltration

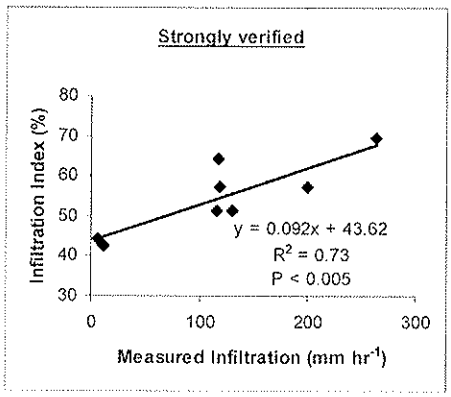

Figure 7.7.b. Relationship between the EFA infiltration index and measured infiltration. Note that data where measured infiltration was more than 300 mm hr^{-1} have been excluded from this graph. Those data were collected in paddock-dumped well-structured clay soils and some measurements indicated infiltration rates in excess of 3000 mm hr^{-1}. EFA was designed to identify soils with low infiltration, high runoff and erosion issues, and this figure shows a good relationship in those circumstances.

7.7.C. Soil Respiration .

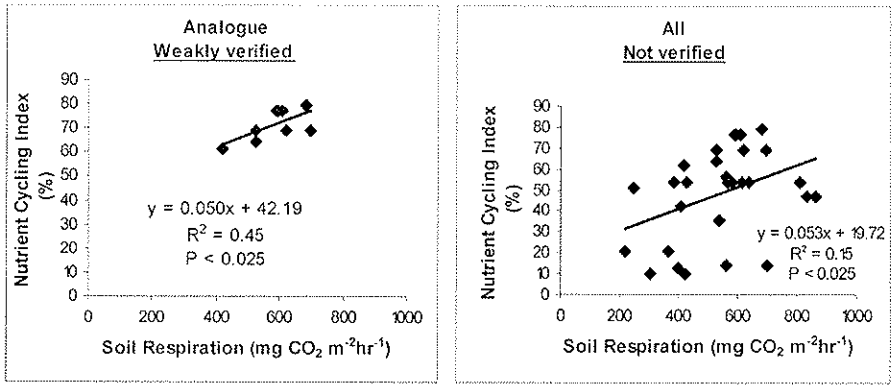

Figure 7.7.c. Shows the relationship between the EFA nutrient cycling index and measured soil respiration. This relationship, although having some significance is highly compromised in terms of representing verification. We did not have the appropriate instrumentation to run a full 24-hr respiration run for respiration rates this high. A further complicating factor was that the recently paddock-dumped soil piles retained their highly biological activity from their original location (a valley floor) because they were removed and dumped in a single operation. The EFA indicators (perennial plants and litter) for the nutrient cycling index had been destroyed in transport, giving them low values.

7.7D. Soil Nutrient Pool Size.

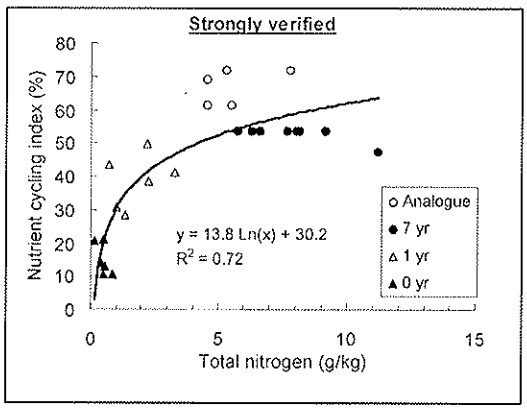

Figure 7.7.d.(i). Shows the relationship between the size of the organic nitrogen pool and the EFA nutrient cycling index for the 0-1 cm depth.

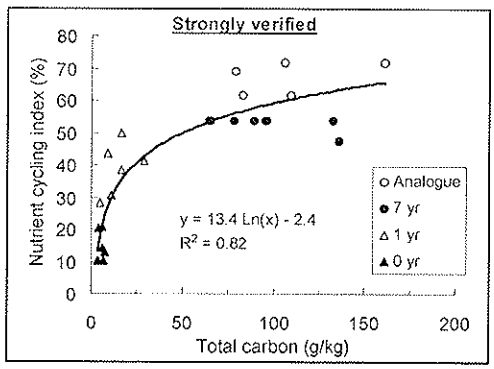

Figure 7.7.d. (ii). The relationship between the size of the organic carbon pool and the nutrient cycling index for the 0-1 cm depth .

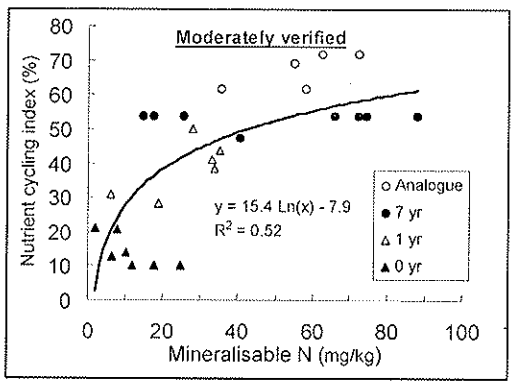

Figure 7.7.d. (iii). Shows the relationship between the size of the mineralisable nitrogen pool and the EFA nutrient cycling index for the 0-1 cm depth.

7.7.E. Functional Vegetation Index.

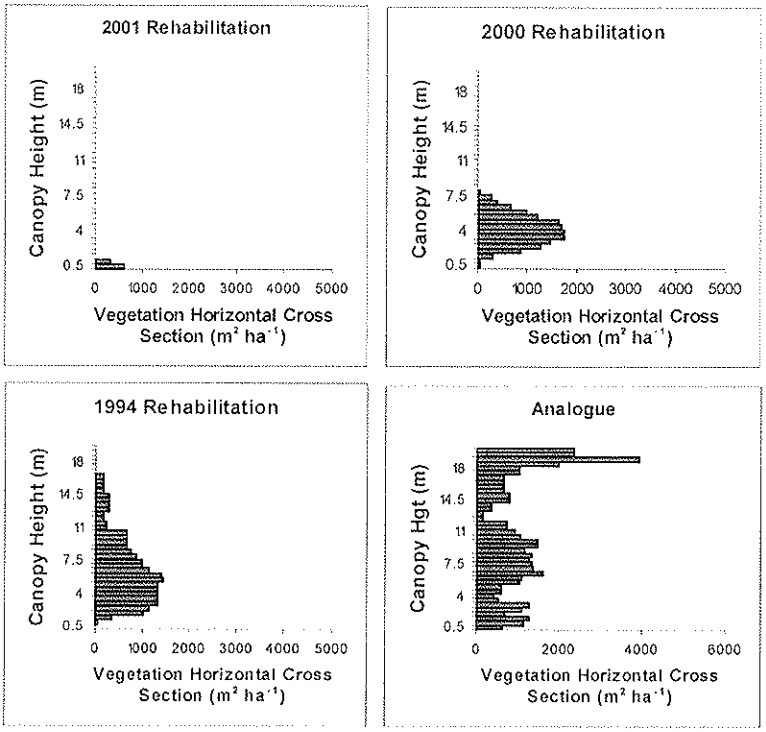

Figure 7.7.e. The horizontal cross sectional area of the canopy at the 4 sites studied at Kelian, showing a rapid mid canopy growth. Tree seedlings are being planted in the rehabilitation in a horticultural/forestry context, so vegetation is more highly managed through time here than at most mines.

7.8 Nabarlek Uranium Mine

7.8.A. Stability

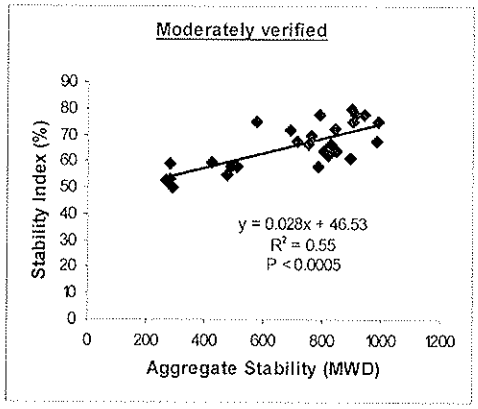

Figure 7.8.a. The relationship between aggregate stability as measured by mean weight diameter and the EFA stability index .

7.8.B Infiltration

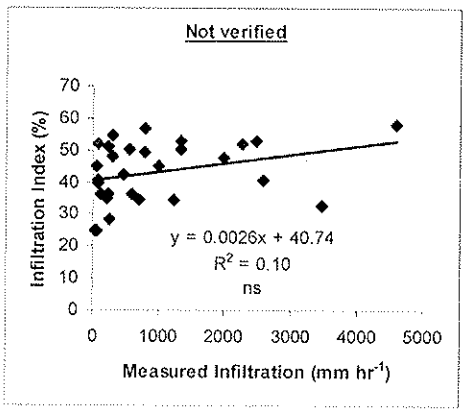

Figure 7.8.b. The infiltration index was not verified at the Nabarlek site. This was mainly due to extremely high infiltration rates (in excess of 2000 mm hr^{-1}) being measured on the dry, well-structured clay soils. Anecdotal and informal evidence suggests that these soils do become saturated in the wet season and measurement of saturated hydraulic conductivity at this time would be an appropriate way to verify the index in these soils. Other sites had too high a proportion of rock to install out infiltrometer ring without major disturbance.

7.8.C. Soil Respiration.

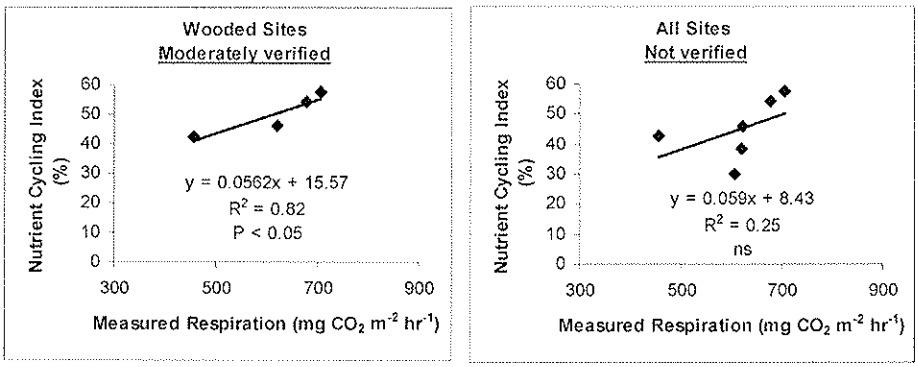

Figure 7.8.c. Mean respiration rate vs. mean EFA nutrient cycling index for wooded sites alone (left) and with grassy sites added (right). There was more biological activity under the grassland than the indicators suggested. It is possible that algal slimes were activated by the measurement process.

7.8.D. Soil Nutrient Pool Size

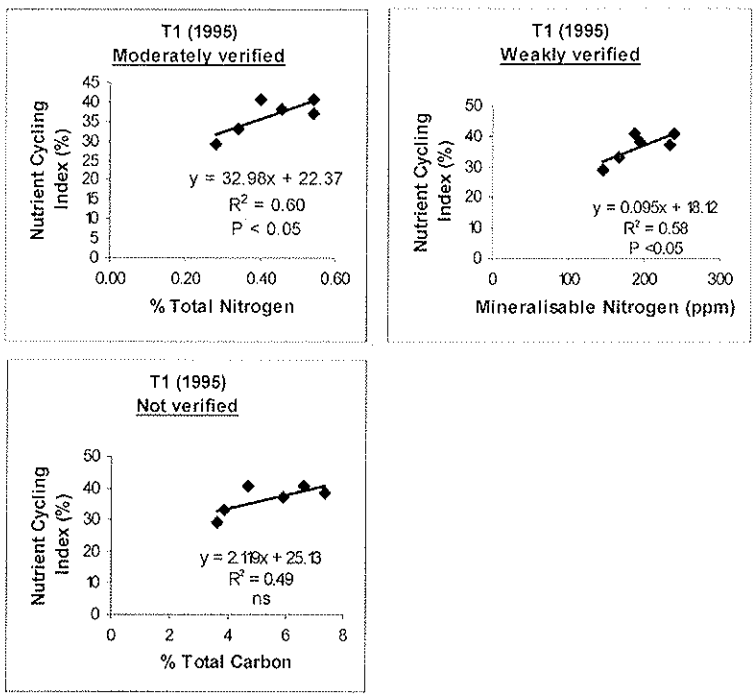

Figure 7.8.d. Shows the relationship between the measured soil nutrient pool and the EFA nutrient cycling index for the dense grass site. All the rehabilitation at Nabarlek was done at a similar time, using a mix of soil materials. As the time for the establishment of a new nutrient profile is only about 6 years, a strong relationship was not expected. There was a low dynamic range in the EFA index, militating against more useful relationships.

7.8.E. Functional Vegetation Index

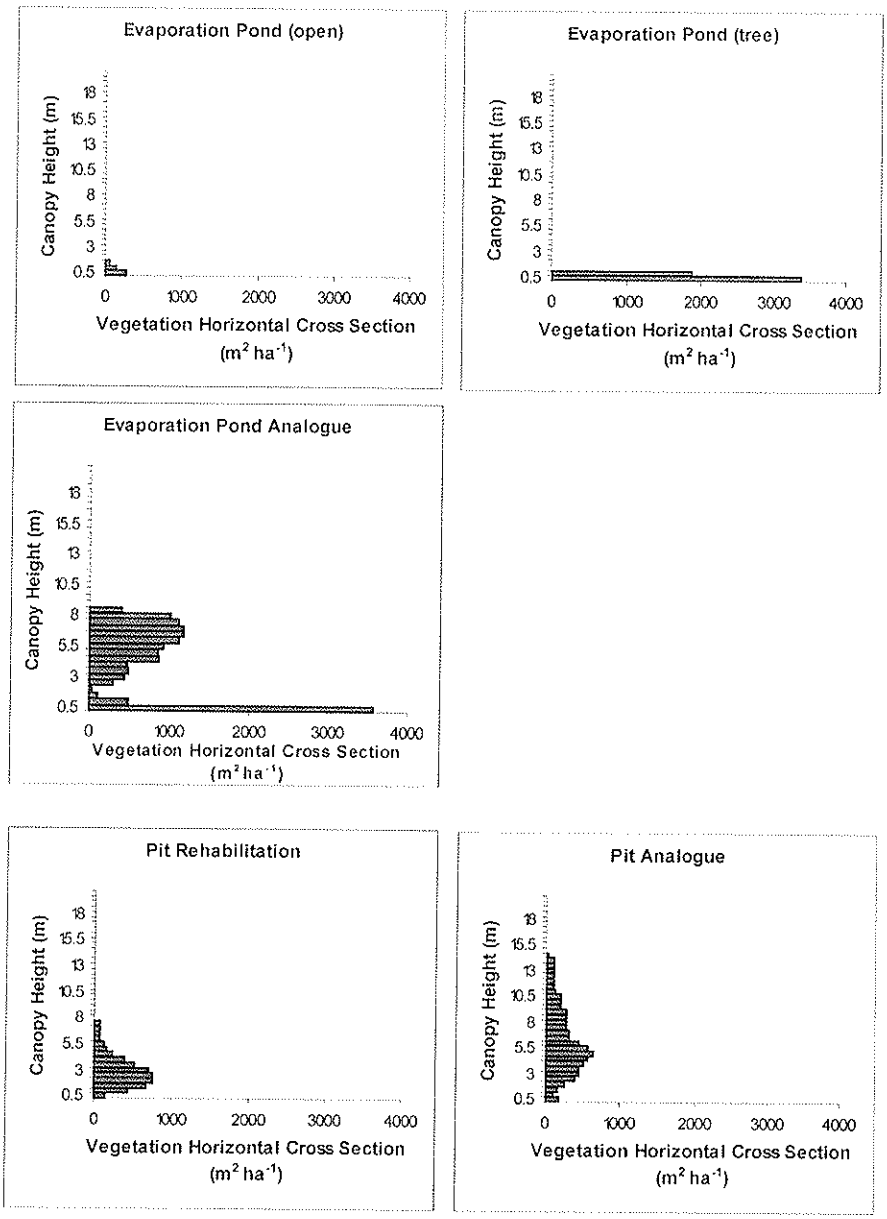

Figure 7.8.e.(i) Vegetation horizontal cross section area (m^2 ha^{-1}) differentiated into 0.5 m height classes for all study sites at Nabarlek. Note that the grassy *Melaleuca* analogue site for the evaporation pond is still very different to the rehabilitated sites and any convergence is still a long time off.

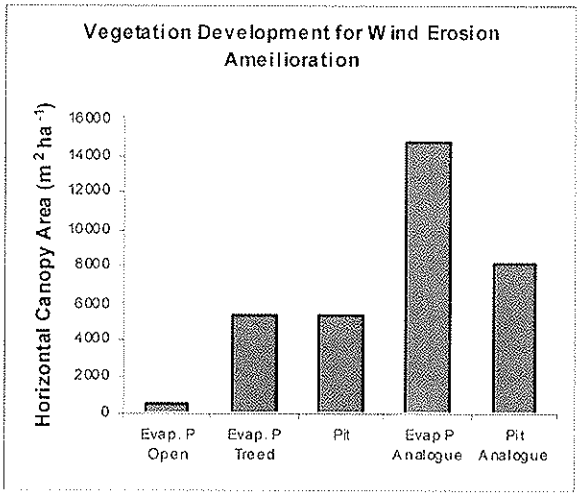

Figure 7.8.e (ii) The vegetation cross section area cover for all sites at Nabarlek. This clearly shows the contribution of foliar cover of woody species compared to grasses.

7.9 New Celebration Gold Mine

7.9.A. Stability

This index was not verified at New Celebration because the samples were destroyed during transport and re-visiting the site for this purpose was not financially possible.

7.9.B. Infiltration

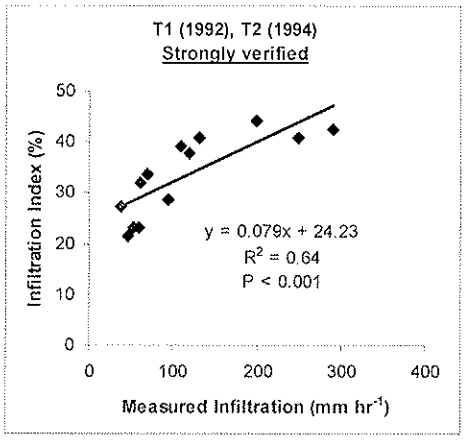

Figure 7.9.b.(i). Verification of the EFA infiltration index with measured infiltration. Pernatty WRD: upper and lower lifts combined. Too few measurements were possible at the Mt Marion site for verification purposes, due to the low mechanical strength of the soil in many places, which was unable to carry the weight of the infiltrometer. The form of this graph suggests that an exponential fit would improve the relationship.

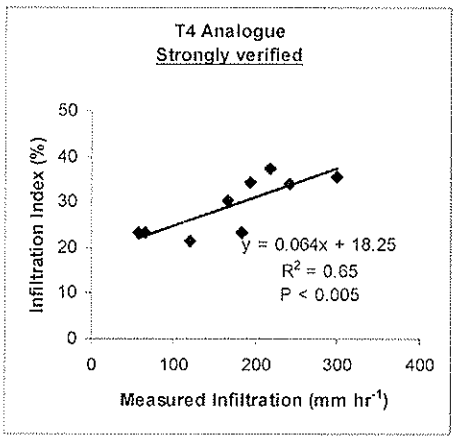

Figure 7.9.b. (ii). Shows the relationship between the EFA infiltration index and measured infiltration for the analogue site alone. Note that the rehabilitated sites and the analogue have different slopes and intercept values, so that the use of the infiltration index would need to be cross-calibrated to be of the highest precision.

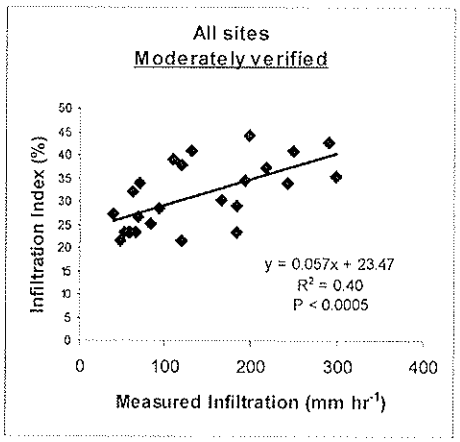

Figure 7.8.b. (iii). The relationship between the EFA infiltration index and measured infiltration for all 4 sites at New Celebration.

7.9.C. Soil Respiration.

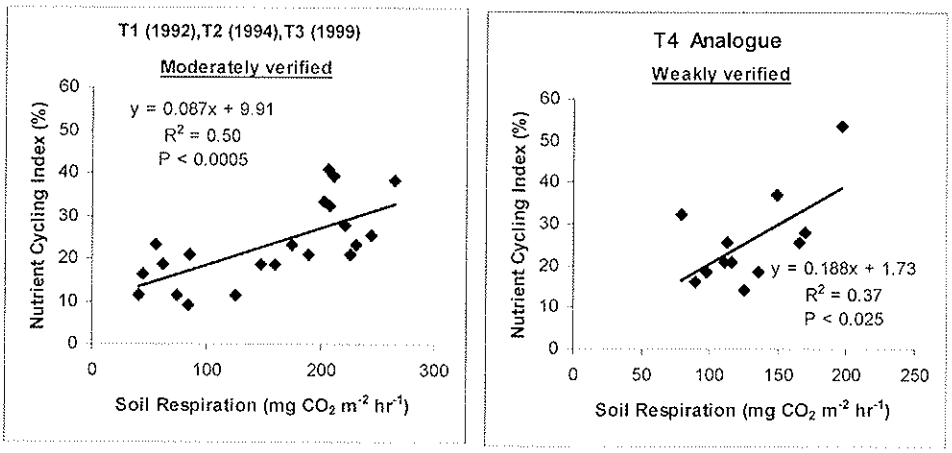

Figure 7.9.c. The rehabilitation sites (left hand graph) have an internally consistent relationship with the LFA nutrient cycling index and soil respiration rate but different to the analogue site (right hand graph), that is also internally consistent. This prevents a simple overall verification relationship, but cross correlation is possible. The explanation for this difference lies in the maturity and species composition of the vegetation at the analogue site.

7.9.D. Nutrient Pool Size

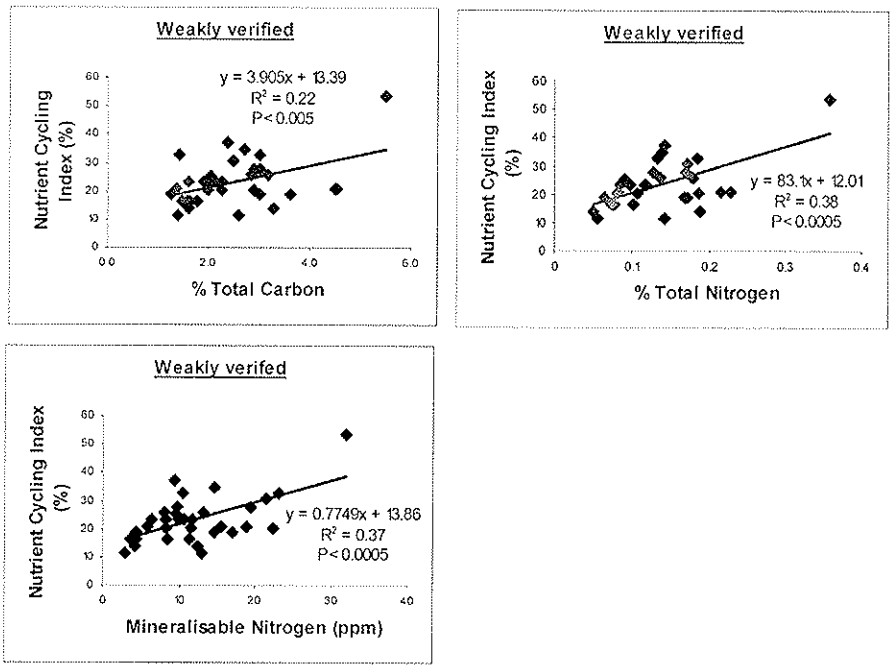

Figure 7.9.d. The relationships between the EFA nutrient cycling index and measured concentrations of soil total carbon, soil total nitrogen and mineralisable nitrogen.

7.9.E. Functional vegetation Index

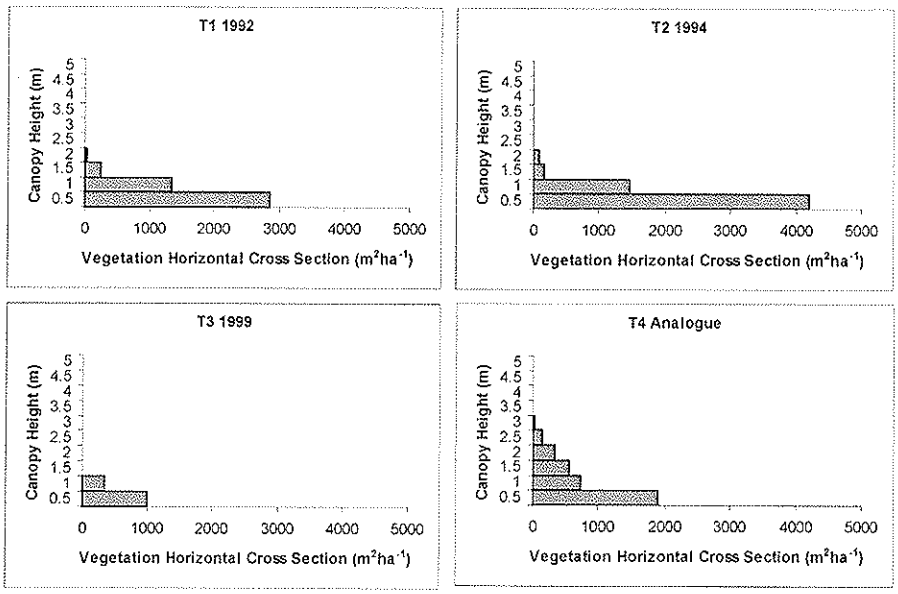

Figure 7.9.e. Horizontal cross section vegetation cover (m^2 ha^{-1}) resolved into 0.5 m height increments for each site shows how the vegetation has developed to regulate ground surface flows (0-0.5 m) and create wind turbulence at other heights. The analogue site clearly has taller vegetation and lower ground cover than the rehabilitation sites due partly to age of rehabilitation and partly to the analogue site being a woodland.

Revegetation and Wildlife

What do we know about revegetation and wildlife conservation in Australia?

A report to Environment Australia

December 1999

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Summary

There is widespread recognition in Australia of the need for revegetation to counter the detrimental effects of excessive clearing of natural vegetation. These effects include degradation of soils and water resources, decline and loss of biodiversity, loss of agricultural productivity, and increased production of greenhouse gases. Protection and management of existing natural vegetation is a critical first step, but will not be enough, on its own, to arrest ecological degradation and ensure long-term sustainability of natural resources. It is essential that vegetation cover be increased in many landscapes in Australia, either through natural regeneration or strategic replanting programs.

Revegetation activities are widespread in rural environments throughout Australia, particularly in Victoria and Western Australia, with 35% of farmers surveyed nationally in 1994 having undertaken some revegetation in the 3-year period 1991-94. The principal motivation for revegetation on farms is to provide shade and shelter for livestock and crops, and to rehabilitate or protect degraded land. However, although only 2% of recent plantings on farms were undertaken to conserve native vegetation or wildlife, more than a quarter of farmers reported that this was an important function of revegetated areas. Most revegetation is in the form of linear strips of trees (e.g. shelterbelts), and the size of revegetated areas is usually small (rarely more than 5 ha). Revegetation is predominantly carried out by using a mix of native, but not necessarily indigenous, species.

Revegetation has the potential to enhance conservation of wildlife in rural environments, but few studies have tested whether the proposed benefits are realized. Published information is sparse and largely restricted to the use of revegetation by birds. While a range of species have been recorded in revegetation, many of those resident in revegetated habitats are 'generalists' or 'open country' species that occur widely. This lack of basic monitoring and research is a critical gap in our knowledge, particularly given that revegetation activities are widespread in Australia, are supported by national environmental programs, and are promoted as having substantial benefits for biodiversity conservation.

Surveys of the occurrence of wildlife in revegetated mining sites (e.g. bauxite mines, Western Australia) show that most species typically present in adjacent unmined habitats utilise revegetated habitats. This use is related to the extent to which the restored habitat provides resources for foraging and shelter, such as leaf litter, logs, tree hollows, a diverse understorey and vegetative cover. However, there are fundamental differences between mine-site rehabilitation and revegetation in agricultural landscapes. The primary objective of revegetation on mine sites is to restore the indigenous flora and fauna, whereas most revegetation in rural landscapes has conservation as a secondary or incidental objective. Further, the spatial context of mine-site revegetation, a small area of restored vegetation surrounded by relatively undisturbed vegetation, differs markedly from the planting of trees within cleared open farmland.

The use of revegetated habitats by wildlife has been studied in a range of countries around the world, but mainly relating to linear habitats such as hedgerows, shelterbelts and plantations. A wide range of species, especially birds, have been reported using these habitats, but there has been little critical analysis of their conservation value as a habitat compared with larger blocks of vegetation or nature reserves. Clearly, they provide shelter and foraging resources for many species, and increase faunal diversity in heavily cleared landscapes. However, many species that use linear planted habitats are 'edge' species, 'generalists' or 'open country' species rather than those of greatest conservation concern. It is likely that their greatest value lies in supplementing the area of existing wooded habitats and increasing connectivity in fragmented landscapes.

Where the primary goal of revegetation is to enhance biodiversity, insights can be gained from ecological studies of remnants of natural vegetation. Characteristic features of the composition and structure of remnant vegetation, and its vulnerability to disturbance from land use, can be used for planning revegetation at the site level. Planning for revegetation at the patch or block level can be assisted by studies of the way in which wildlife communities use remnant patches of vegetation. Such studies indicate the importance of patch size and shape, the maintenance of habitat resources, and the patch location in relation to sources of disturbance. At a landscape level an important issue is the location of habitat patches in relation to other habitats, and thus the degree of isolation or habitat connectivity for populations of animals living within

revegetated blocks. These insights can be used to plan revegetation activities to increase their value as habitat, either for single species or for faunal communities.

Revegetation in rural environments offers a number of benefits for sustainable land management and agricultural productivity. Revegetation can be used to reduce land and water degradation by strategic establishment of vegetation in order to minimise the rise of saline groundwater, reduce waterlogging of soils, reduce erosion and protect water quality. These measures, together with planting of vegetation as shade and shelter for stock and crops, enhance agricultural productivity on farms. Revegetation in the form of agroforestry has the potential to provide economic benefits through a variety of timber products. By storing carbon in plant biomass, revegetation may also provide an opportunity to slow the rate of increase of greenhouse gases in Australia.

Planning revegetation projects requires careful consideration of issues such as the location of planting, selection of plant species, design of planted areas, timing of planting and the most suitable methods for establishing plants. Techniques and approaches for establishing plants in the ground appear to be the most thoroughly studied aspect of revegetation, and there is now a large and growing body of knowledge on species selection and optimum planting procedures for many areas in Australia.

There are four main areas in which future directions in revegetation can learn from past experience. First, greater emphasis needs to be given to nature conservation in the way revegetation activities are planned and implemented in rural environments. Second, there is a critical lack of knowledge of the value of revegetated habitats for flora and fauna and for maintaining ecological processes. This needs to be addressed urgently by carefully designed studies. Third, there is great potential for integrating ecological monitoring and research with on-ground activities as a way of simultaneously gaining knowledge and improving management practices. Last, strategic planning at the landscape and regional scale is required to provide a framework within which revegetation actions can most effectively contribute to local as well as regional environmental issues.

Preface

Revegetation is an important issue in natural resource management in Australia. There is widespread recognition in the community that extensive revegetation is needed to counter environmental problems arising from the loss of native vegetation. One of these problems is a decline in biodiversity in heavily developed landscapes – an ongoing loss of plant and animal species and disruption to ecological processes in which they have essential roles. Revegetation has been promoted as having substantial benefits for the conservation of biodiversity, but the reality is that this is not the main objective of most revegetation activities. In most cases, the primary motivation for revegetation is to address urgent problems in land degradation, to provide shelter for stock or crops, or to enhance the aesthetic environment.

This project was instigated by Environment Australia to identify ways in which revegetation activities can be undertaken to maximize their value as habitat for wildlife. We have addressed a number of objectives, as follows:

- to review existing information relevant to understanding the habitat requirements of native fauna and techniques for maximising habitat quality within plantations (ranging from narrow windbreaks to larger plantings);
- to provide a resource list where further information can be obtained;
- to develop technical guidelines, based on sound ecological principles, for maximising the habitat value of planted systems;
- to use case studies to illustrate how these guidelines might be applied at a range of scales;
- to develop a product for disseminating these principles, guidelines and case studies to relevant stakeholders such as landowners, Landcare and Bushcare facilitators, agroforestry officers and conservation planners.

This report fulfils the first part of the task by presenting a review of current knowledge on revegetation and wildlife conservation, together with an annotated list of references where further information can be obtained. Our focus is on the role of revegetation in the protection and maintenance of biodiversity, particularly in rural environments in southern Australia. We are primarily concerned with the way in which revegetation can be used to provide habitat for the native fauna, and thereby contribute to nature conservation in developed landscapes. Because relevant published information is scattered in the literature, the annotated list of references provides an introduction to this material for those wishing to read more widely. The second part of the task is achieved in a companion publication (Bennett *et al.* 1999) that sets out a series of principles for revegetation activities that will enhance their habitat value for wildlife. It is based upon the material reviewed in this report, but because it is intended for a wider audience less attention is given to citing scientific literature.

The report has been produced using funding from the Natural Heritage Trust, under the Bushcare Program. We thank Environment Australia for initiating the project and in particular Kathy Tracy for her support. The project was guided by a Steering Committee, comprising Kathy Tracy and Peter Lyon (Environment Australia), Geoff Barrett (Birds Australia), Stephen Platt (Flora and Fauna, NRE Victoria), Dale Tonkinson (Greening Australia Victoria) and Des Stackpole (Centre for Forest Tree Technology, NRE). We gratefully acknowledge their advice and constructive contributions. We also warmly thank those people who attended a one-day workshop in February 1999, and whose deliberations and discussion on a range of issues related to revegetation provided valuable input. They were Geoff Barrett, Karen Barton, Hugh Bramwells, Neil Collins, Ian Davidson, Louise Gilfedder, David Goldney, Ian Hislop, Richard Hobbs, Richard Loyn, Peter Lyon, David Parkes, Stephen Platt, Doug Robinson, Jim Robinson, Des Stackpole, Dale Tonkinson, Kathy Tracy and Brenan Wotherspoon. Members of the Landscape Ecology Research Group at Deakin University, Jim Radford, Rodney van der Ree, Grant Palmer and Jenny Wilson, also provided valuable comments and discussion.

1. Introduction

There is widespread recognition in Australia of the need for revegetation to counter the detrimental effects of excessive clearing of natural vegetation. The call for revegetation is coming from many sectors of the community – rural landholders, scientists, conservation organisations, state and local government agencies – and has been reflected in national policy commitments by the Commonwealth Government. In recent years, major government initiatives that have promoted revegetation activities have included the One Billion Trees Program, the National Corridors of Green Program and the Decade of Landcare (1989-1999). These, and other initiatives that promote revegetation, are now incorporated within the Natural Heritage Trust, a national program that coordinates a range of activities aimed at protecting and restoring the natural environment in Australia (Natural Heritage Trust 1999).

Revegetation activities may be undertaken for a number of reasons: to counter the degradation of land or water resources, to maintain or enhance agricultural productivity, to enhance the aesthetics of human-dominated environments, to protect and restore biodiversity and, more recently, to create carbon sinks to counter potential 'greenhouse' climate change. The focus of this report is the role of revegetation in the protection and maintenance of biodiversity, particularly in rural environments in Australia. It is primarily concerned with the way in which revegetation activities can be used to enhance habitats for the native fauna, and thus contribute to nature conservation in agricultural landscapes dominated by intensive human land use.

This report has three main objectives:

1. to present a review and synthesis of what is currently known about revegetation and its capacity to enhance habitats for wildlife in Australia;
2. to provide an introduction to relevant literature for those wishing to read more widely or gain access to primary resource material; and,
3. to form a scientific basis for developing principles for revegetation activities that will enhance wildlife habitat values. These principles are presented in a companion publication to this report (Bennett *et al.* 1999).

Introductory chapters in this report outline the need for revegetation in Australia, and review what is known about the extent and pattern of recent revegetation activities in rural environments across the country. This is followed by a summary of current knowledge of the use of revegetation by wildlife and its value as habitat. There is a scarcity of such published information in Australia, and consequently we have also included a brief review of relevant scientific literature from other countries. An alternative approach to understanding the potential value of revegetated habitats is to gain insights from the way in which wildlife species and communities use remnant stands of natural vegetation in rural environments. Consequently, studies of the status of wildlife in remnant vegetation in Australia have also been reviewed.

To provide a broader context for understanding the role of revegetation in rural environments, we have reviewed literature relating to the benefits of revegetation for sustainable land use in agricultural landscapes. A short summary is also given of sources of information concerning the planning and implementation of revegetation projects, although this topic is covered in much greater depth in other guides and books. The report concludes by discussing the effectiveness of revegetation to date, and raises issues concerning future directions for revegetation activities.

The scope of this report is largely restricted to revegetation in rural environments in Australia. It is mainly concerned with revegetation activities at a relatively small scale, such as those that typically may occur on farms or be carried out by community groups. We do not attempt to review information relating to major commercial revegetation activities, such as plantations of pines and other introduced tree species for timber, or the growing range of commercial agroforestry ventures using native species, such as the recent expansion in plantings of fast-growing eucalypts in southern Australia. Some attention is given to published results on the use by wildlife of revegetated mine sites, to compare this with revegetation activities in rural landscapes.

Selected references are cited in the text, and a list of key references and references for further reading are presented for each chapter. An annotated list of all references, including full publication details and a brief summary of the content of each, is given in the final section of the report.

Key references:

Bennett, A.F., Kimber, S.L. and Ryan, P.A. (1999). Revegetation and Wildlife: Principles for Enhancing Habitat Values for Wildlife Conservation. Environment Australia: Canberra.

Natural Heritage Trust (1999). Natural Heritage Trust Home Page. Available URL: http://www.nht.gov.au.

2. The need for revegetation in Australia

The primary reason for undertaking revegetation activities in Australia has been to counter the detrimental effects of the excessive loss of natural vegetation. Here, we briefly review the extent of vegetation loss in Australia, and the main consequences of such loss that have led to calls for restoration of vegetation.

Loss of natural vegetation

Clearing of native vegetation for agriculture, mining, forestry and human settlement is widespread throughout the world. In many regions it has been accompanied by widespread land degradation and substantial declines in biodiversity. In Australia, clearing of vegetation has occurred since the earliest days of European settlement, although the rate of clearing has been greatest in the past 50 years (Department of Environment, Sport and Territories 1995).

A snapshot of changes to terrestrial ecosystems in Australia over the last two centuries has been described in the recent State of the Environment Report (State of the Environment Advisory Council 1996):

- about 43% of forests in Australia have been cleared;
- nearly 90% of temperate woodlands and mallee have been cleared;
- more than 99% of temperate lowland grasslands in south-eastern Australia have been lost;
- about 75% of rainforests have been cleared; and
- more than 60% of coastal wetlands in southern and eastern Australia have been lost.

The most extensive loss of native vegetation has occurred in the temperate regions of the continent, particularly the fertile inland slopes and plains of south-eastern and south-western Australia that once supported diverse woodlands. These areas are now dominated by broadscale agricultural production – primarily grazing and cropping – and remnants of the former woodlands now exist as fragments of various size, shape and isolation, scattered throughout the landscape (Goldney and Bowie 1990; Hobbs and Hopkins 1990; Saunders *et al.* 1991; Hobbs *et al.* 1993; Robinson and Traill 1996; Bennett and Ford 1997).

Robinson and Traill (1996) have summarised the extent of change in a number of regions:

- in the Western Australian wheatbelt (14 million ha), only 7% is now covered by native vegetation;
- in South Australia, native vegetation remains on only 2% of the 1.5 million ha Narracoorte Plains and 9% of the 3.4 million ha Murray-Darling region;
- in the Victorian Murray-Darling region, only 9% of 3.3 million ha retains native vegetation;
- in Victoria's Box-Ironbark region, 19% of 3.7 million ha has native vegetation remaining; and
- in the Riverina region of New South Wales, 9% of 3.4 million ha still has native vegetation cover.

The extent of clearing of native vegetation has been increasingly recognised as a primary factor in land degradation in rural regions, reduced agricultural productivity and declines in biodiversity (DEST 1995). The need to restore vegetation to the landscape has, therefore, become widely accepted.

Implications of the loss of natural vegetation

Clearing of natural vegetation to establish farmland has been essential to the development of Australia's agricultural industries. However, the excessive and near-complete loss of natural vegetation in many districts has led to major environmental problems including degradation of land and water resources, increased production of 'greenhouse gases' (e.g. carbon dioxide), and the decline and loss of biodiversity in rural environments.

Degradation of land and water resources

There is abundant evidence that, in many parts of Australia, substantial degradation of the land and waterways has occurred and continues to occur. Many forms of degradation can be identified, most of which are directly associated with the excessive loss of vegetation. These include:

- *surface waterlogging* caused by rising water tables as groundwater uptake has been reduced by loss of deep-rooted vegetation;
- *salinisation of surface soils* resulting from rising water tables depositing salts in the subsoil layers;
- *loss of soil fertility* due to direct loss of topsoil by erosion, removal of nutrients by cropping, and loss of closed nutrient cycling processes;
- *soil erosion* resulting from increased wind velocity and water runoff following loss of vegetation cover;
- *erosion of stream banks* where stabilising vegetation has been removed; and
- *deteriorating water quality* of surface and groundwater supplies as a result of:
  - salinisation, where rising water tables leach salt and other minerals into water supplies;
  - siltation, where increased runoff, and erosion of slopes and stream banks deposit silt in water supplies;
  - pollution, where pesticides, herbicides, and fertilisers enter the aquatic system; and
  - eutrophication, where increased water temperatures result from removal of shading vegetation, and loss of fringing vegetation reduces the capacity to buffer the aquatic environment from inputs of nutrients and silt.

Other types of land degradation also of concern, but not directly related to vegetation clearing, include pest animal infestations, weed infestation, soil acidity, soil compaction and loss of soil structure.

Dryland salinity is a particularly widespread and increasing problem. Gretton and Salma (1996) report:

- in Victoria, 100,000 ha are affected by dryland salinity and a further 198,600 ha are considered at risk;
- in Western Australia, 900,000 ha are affected and 2.4 million ha are considered at risk; and
- in Queensland, 30,000 ha are affected and 70,000 ha are considered at risk.

Predictions are that the areas affected by salinity will continue to increase.

The extent and type of land degradation problems vary between agricultural zones – pastoral, wheat-sheep and high rainfall. Many farmers are faced with multiple land degradation problems. Figure 2-1 summarises the results of a survey of land degradation problems reported by 1800 farmers from across Australia in 1995-96 (Mues *et al.* 1998). This is likely to be an *underestimate* of the extent of these problems because degradation may not be noted until it is well advanced, and some landholders may not be aware of certain issues.

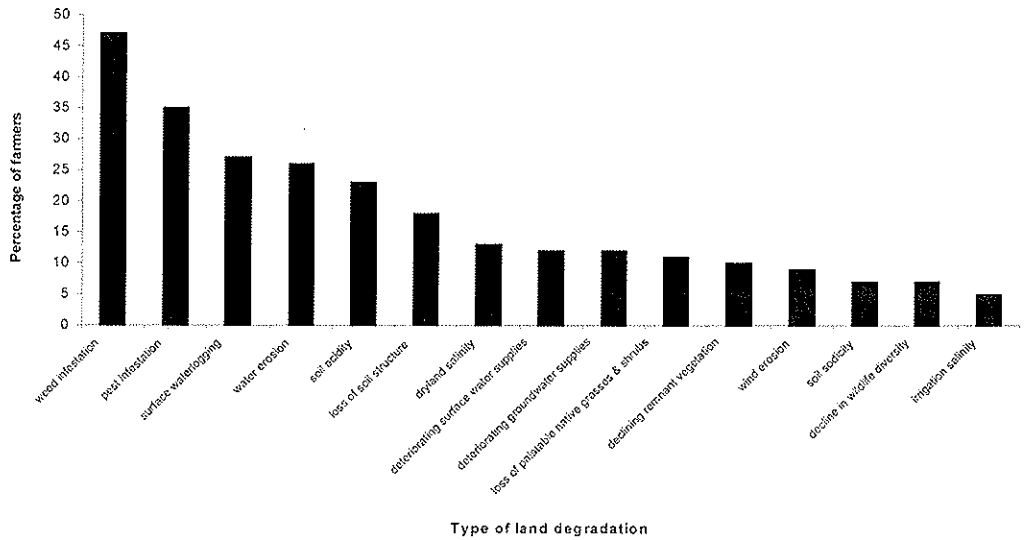

Figure 2-1. Land degradation problems reported by farmers in Australia. Values are the percentage of farmers from all broadacre and dairy industries reporting significant problems with different types of land degradation in 1995-96 (data from Mues et al. 1998).

Land degradation in some agricultural regions has advanced to a point where the future viability of production is seriously threatened and urgent measures must be taken to prevent further environmental decline.

Decline in agricultural productivity

Land degradation results in loss of productivity. Estimates of the cost of such losses, while difficult to measure, indicated that in 1995-96 the national impact was around $1 billion a year, or around 4% of the gross value of agricultural production (Mues *et al.* 1998). A more recent assessment (1997) has placed these figures at $1.2 billion per annum or 30% of total annual production costs (Lefroy and Hobbs 1998). Examples of the impact of land degradation on production in Australia include:

- at the national level, erosion has been estimated to be responsible for an annual loss of production of $80m, waterlogging $180m, soil acidity $300m, and both loss of soil structure and salinity have been attributed with a $200m loss of productivity (DEST 1995; Lefroy and Hobbs 1998);
- at the state level, in Western Australia, estimates of the annual cost of land degradation are $62m for dryland salinity, $90m for waterlogging, $70m for soil structure decline, and $21m each for water and wind erosion (Gretton and Salma 1996); and
- at the regional level, in the Lachlan Valley, NSW, the estimated production equivalent of soil loss due to sheet and rill erosion in 1989-90 was $4.27m, the equivalent of $21.1m for the Lachlan region (Gretton and Salma 1996).

Throughout the temperate woodlands, more than $400m in lost agricultural production annually has been attributed to soil degradation. It is expected that this figure will rise markedly as more than 60% of some districts become salt-affected in the next 20 years, and the area of croplands and irrigated pastures increase (Robinson and Traill 1996).

Effects on crop production

Crop yields can be affected by the extent of shelter from, or exposure to, wind. A number of examples of impacts have been reported by Reid and Bird (1990):

- *loss of soil fertility.* Exposure to strong winds has been shown to result in removal of fertile top soil. For example, about 27 t of soil was removed per ha during a five hour period in the Western Australian wheatbelt, 10% of which was the fertile top 4 mm of soil. In economic terms, removing the dust fraction from the top 8 mm of soil can reduce the yield of the subsequent crop by 25%. The rate of replacement of lost soil is only 2.5 mm in 100 years and is unlikely to occur because of continued loss.
- *physical damage to crops.* Physical damage to horticultural and vegetable crops caused by wind has been well documented. Blossom can be destroyed prior to setting, fruit discoloured or damaged, fruit fall can be increased and insect damage increased. A 37% increase in yield, and improved size and appearance of fruit, has been recorded in citrus crops in New South Wales in response to artificial shelter.

Effects on livestock production

Loss of shade and shelter provided by vegetation also contributes to declines in livestock productivity. Research in Australia has indicated that heat and cold stress can markedly reduce milk production, weight gain and stock fertility, and increase lamb, calf and off-shear mortality. Examples reported by Reid and Bird (1990) include:

- a reduction in milk production of almost 26% expected from stock exposed to high levels of solar radiation (e.g. on a 27°C day); and
- a study of fine-wool merinos in the Armidale area of NSW found mortality over 30% for single lambs without shelter where windspeed exceeded 15 km/h in combination with rain or low temperatures (less than 5°C) during the critical period.

Climate change and the Greenhouse effect

Greenhouse gases – carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) – form a blanket around Earth. Light energy from the sun penetrates this blanket and is converted to heat energy at the surface. Most

of this heat is re-radiated, but some is trapped by the gases in the atmosphere. This is known as the 'greenhouse effect'. It is a natural process that maintains Earth's temperature at a level necessary to support life (Australian Greenhouse Office 1998).

However, human actions such as land clearing and the burning of fossil fuels are substantially increasing the atmospheric concentrations of natural greenhouse gases. Ice core air samples have indicated that, since the Industrial Revolution (over 200 years ago), the atmospheric concentrations of carbon dioxide have increased by more than 30%, methane by more than 145% and nitrous oxide by about 15% (SEAC 1996). Humans have also created additional greenhouse gases, known as halocarbons (e.g. CFC's) (AGO 1998). Increases in concentrations of greenhouse gases have raised concerns about potential change to the global climate. This is a complex process that has been termed the 'enhanced greenhouse effect'.

Clearing of natural vegetation has been a major contributor to the enhanced greenhouse effect. It was estimated that in 1990 about 25% of the total greenhouse gas emissions in Australia resulted from land clearing. By 1995, this proportion had declined to 17%, a result explained in part by a rise in energy-related emissions, but also by a reduction in the extent of vegetation clearing (Krockenberger 1998).

Forests and woodlands play an important role in the global carbon cycle. Through photosynthetic activity, carbon dioxide from the atmosphere is transformed into inorganic material, including wood. Trees are therefore often regarded as a carbon 'sink'. Estimates suggest that forests store approximately 600 billion tonnes of carbon with a further 1,500-1,600 billion tonnes stored in soils and forest detritus (Intergovernmental Panel on Climate Change 1994 cited in Alexandra 1995). When forests are cleared, much of this carbon is converted back to carbon dioxide by burning or decomposition of organic material (AGO 1998), thereby contributing to increasing levels of greenhouse gas.

Decline in biodiversity

"Biodiversity is the variety of all life forms – the different plants, animals and micro-organisms, the genes they contain, and the ecosystems of which they form a part" (DEST 1996).

Australia contains a diverse range of biogeographic regions – the arid interior (70% of the continent), tropical monsoon areas to the north and regions experiencing mediterranean and temperate climates to the south. There are also several snow clad mountain regions, and external territories extend to subantarctic and antarctic regions. Australia's biological diversity has great scientific value and many of its elements are unique (DEST 1996).

> *"Six families of mammals, four of birds, and 14 of flowering plants are endemic – far more families than in any other country. Further, at the species level about 82 per cent of our mammals, about 45 per cent of our land birds, about 89 per cent of our reptiles, and 93 per cent of our frogs occur nowhere else.....Of the estimated 600 inshore fish species in the southern temperate zone, about 85 per cent are found only in Australia. Australia contains eight endemic families of fish, and more than half the shark and ray species are confined to Australian waters." (DEST 1996)*

However, over the last 200 years there have been profound changes to the flora and fauna in most parts of Australia. A number of species have become extinct, many species are now listed as endangered or vulnerable (Table 2-1), and a much greater number have declined, and continue to decline, in status although not officially listed as threatened species.

A number of factors have been implicated in the decline and loss of species, including the impacts of introduced species, direct destruction by humans, disease, and utilisation of natural resources, but it is widely recognised that clearing, fragmentation and degradation of natural vegetation has been the primary cause of decline. Clearing and fragmentation of natural vegetation affects the flora and fauna:

- by direct loss of habitat and reduction in population size;
- by fragmentation of habitat, and subsequent decline in small, isolated populations; and
- by disruption to, and modification of, ecological processes in remaining vegetation.

Table 2-1. The number of species listed as endangered, vulnerable or extinct in Australia under Schedule 1 of the *Endangered Species Protection Act 1992* as at 22 July 1999. (ns – not stated in report)

Taxon	Endangered	Vulnerable	Presumed extinct
Mammals	35	20	19
Birds	36	60	23
Amphibians	13	3	0
Reptiles	13	40	0
Fish	11	12	0
Invertebrates	(ns)	4	(ns)
Non-vascular plants	1	1	(ns)
Plants	370	700	68
Total	**479**	**840**	**110**

(Data from: Environment Australia 1999).

There are a growing number of reports documenting changes to the fauna in agricultural regions of southern Australia, including the wheatbelt region of Western Australia (e.g. Kitchener *et al.* 1980; Saunders 1989; Saunders and Curry 1990; Hobbs *et al.* 1993), central and northern New South Wales (e.g. Goldney and Bowie 1990; Barrett *et al.* 1994) and western and northern Victoria (Bennett 1990a; Robinson and Traill 1996; Bennett *et al.* 1998). Although it is seldom possible to show a direct cause and effect relationship between clearing of native vegetation and regional decline of fauna, it is clear that the greatest faunal changes at local, district and regional levels have occurred where there has been greatest loss of vegetation. For example, in the Northern Plains of Victoria there is a significant relationship at the landscape level between the number of woodland-dependant bird species and the extent of tree cover. There is a dramatic decline in bird species in those landscapes where less than 10% tree cover remains (Bennett and Ford 1997). Rural landscapes with little natural vegetation have a depleted fauna compared with that prior to clearing.

In many cases, the decline of species continues to occur long after the initial clearing of habitats. Populations of native animals are likely to experience further decline and extinction as fragmented populations succumb to the effects of isolation. These issues are compounded in the agricultural zone, where the conservation reserve system is inadequate and not representative of the former vegetation mosaic (Ryan in press). Presently, more than two-thirds of Australia (some 500 million ha) are managed by private landholders, compared to about 40 million ha within the terrestrial reserve system (DEST 1996). Increasingly, wildlife managers are looking towards habitats outside conservation reserves to complement the reserve system.

Summary

Clearing, fragmentation and degradation of native vegetation is a major issue in Australia, particularly in agricultural environments. The consequences of excessive land clearing include degradation of soils and water resources, decline and loss of biodiversity, increased production of greenhouse gases, and loss of agricultural productivity. Achieving ecological sustainability in land use and conservation of biodiversity in agricultural landscapes is now a major challenge facing land-use planners and land managers in Australia. Recognition of the role of vegetation cover in addressing these issues has stimulated a shift in the attitude of many planners and land managers toward the value of retaining, protecting and enhancing native vegetation in agricultural landscapes. This is of critical importance for the conservation of biodiversity. However, management of existing vegetation will not be enough on its own to arrest ecological degradation, to protect the flora and fauna, and to restore the land to a sustainable condition. It is essential that vegetation cover be increased in many agricultural landscapes through natural regeneration or strategic revegetation programs.

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Further reading:

Alexandra (1995), Bennett (1990a), Bennett and Ford (1997), Bird *et al.* (1992), Goldney and Bowie (1990), Hobbs and Saunders (1991), Hobbs *et al.* (1993), Johnston and Williams (1993), Kitchener *et al.* (1980), Krockenberger (1998), Lefroy and Hobbs (1998), Oates (1995), Ryan (in press), Saunders (1989, 1996), Saunders and Curry (1990), Saunders and Hobbs (1995), Saunders *et al.* (1991).

3. Revegetation activities in Australia

In the broadest sense, revegetation is the deliberate planting or re-establishment of vegetation by humans. However, the term revegetation encompasses a wide range of activities that restore vegetation to land, including various forms of forestry, agroforestry, land protection and site rehabilitation (e.g. mining, dune stabilisation). Revegetation methods include allowing areas to naturally regenerate, re-establishing indigenous native vegetation by planting or direct seeding, and planting native or exotic species for a particular purpose.

A principal motivation for much revegetation in Australia has been to maintain or increase the productivity of agricultural land through protection and enhancement of soil and water quality, and by providing shelter for crops and livestock. Hence, a large proportion of revegetation in Australia is undertaken by rural landholders. However, revegetation is also undertaken by road management authorities (e.g. planting along highways), government departments (e.g. parks and land management – Parks Australia, Environment Australia), conservation groups (e.g. Australian Trust for Conservation Volunteers), forest plantation companies, mining companies, forest managers, and local government. In this chapter we review the way in which revegetation is occurring in rural Australia.

Revegetation on farms in Australia

Surveys of approximately 2000 farmers in Australia were carried out in 1994 by the Australian Bureau of Agricultural and Resources Economics (Wilson *et al.* 1995). The following summary of revegetation activities by this sector of the community is based largely on their report. Similar results were also reported in a survey conducted by CSIRO in 1995 (Nicoll and Dobbie 1996). Other than these reports, it is difficult to obtain published data summarising revegetation activities in rural environments in Australia. Given the extensive activity that is currently underway, there is a need to collate statistics at catchment and state levels on the amount, location and pattern of revegetation (including size, shape, and species composition).

Revegetation on farms is occurring throughout Australia, although the extent of involvement varies between states. Landholders in southern states have been more involved in revegetation activities over previous years than those in the Northern Territory and Queensland (Figure 3-1).

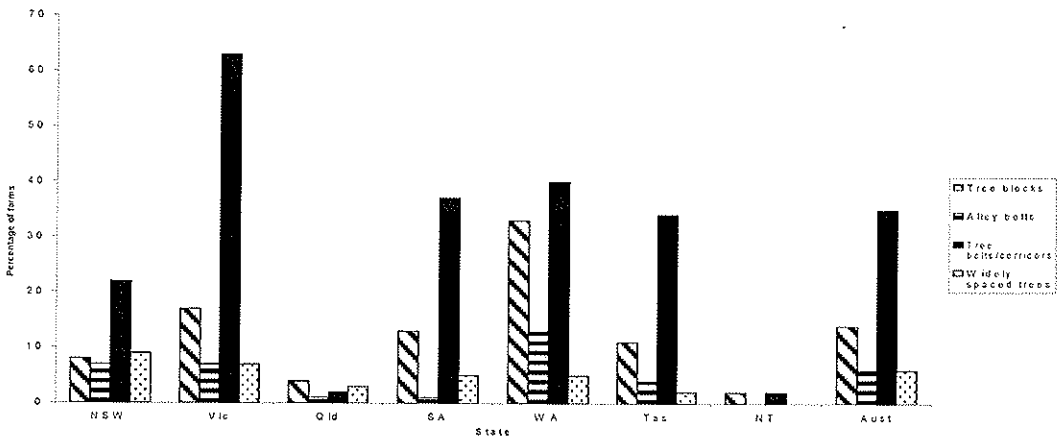

Figure 3-1. Percentage of farms with planted trees by state and planting regime (data from Wilson et al. 1995).

A similar trend is evident when the most recent (1991-94) tree planting activities are examined (Figure 3-2). Overall, 35% of farmers reported planting trees in this three year period, including 34% in the wheat-sheep

zone (approx. 250-600 mm mean annual rainfall), 39% in the high rainfall zone (approx. >600 mm rainfall), and 4% in the pastoral zone (< 250 mm rainfall) (Wilson *et al.*1995).

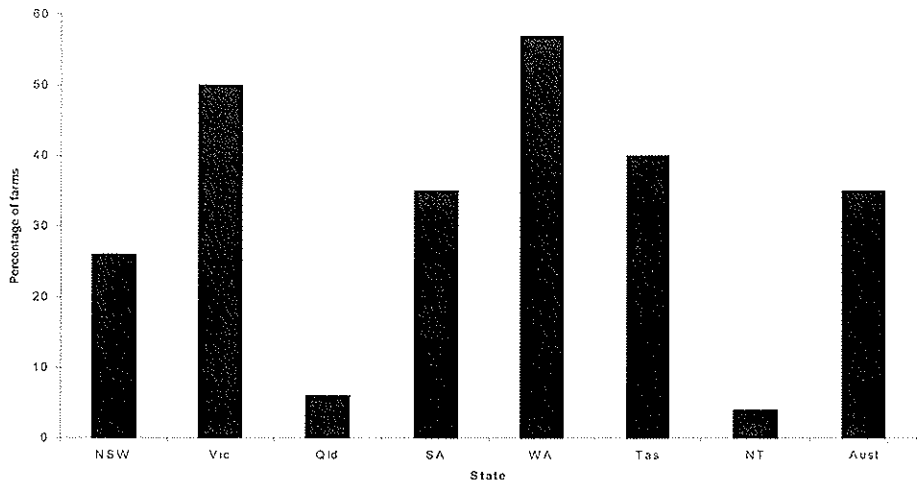

Figure 3-2. Percentage of farmers who planted trees 1991-1994 (data from Wilson et al. 1995).

The most common forms of revegetation in agricultural regions are linear strips (which may or may not connect to other patches of vegetation) and block-style plantings. Linear strips include shelterbelts, farm forestry plantations, wildlife corridors and alley farming belts; block-style plantings include agroforestry woodlots and livestock havens.

Nationally, of the farmers surveyed (Figure 3-3):

- 35% responded that they had *tree belts and corridors*: trees planted in a strip with a maximum width of approximately 30 m;
- 14% had *tree blocks* (includes woodlots): trees grown in high density clumps;
- 6% had *alley belts*: two or more strips of trees with grazing or cropping between the strips; and
- 6% had *widely-spaced plantings*: separated and scattered trees on farms, with grazing or cropping carried out between the trees.

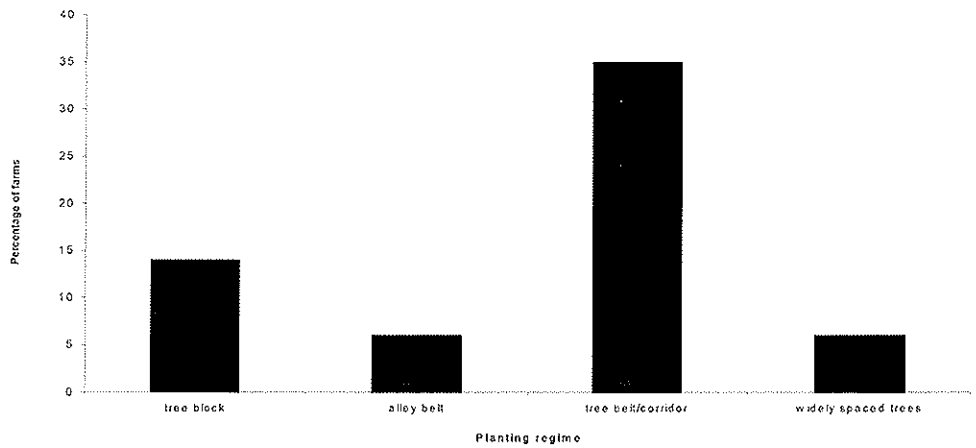

Figure 3-3. Types of revegetation plantings on farms in Australia (data from Wilson et al. 1995).

Reports from farmers with planted trees showed that the average area per farm for each planting regime was 5.2 ha of tree belts and corridors, 11.7 ha of tree blocks, 9.9 ha of alley belts (includes the area of pasture or crop between tree belts), and 406 widely-spaced trees (no measure of area reported).

Farmers were asked to list up to three main functions of planted trees on their property (Figure 3-4). In the wheat-sheep and high rainfall zones, the most common functions listed were:

- to provide shelter and shade (81% wheat-sheep; 93% high rainfall);
- to rehabilitate degraded land and/or protect land from degradation (58% wheat-sheep; 38% high rainfall); and
- to conserve native vegetation and wildlife (29% wheat-sheep; 26% high rainfall).

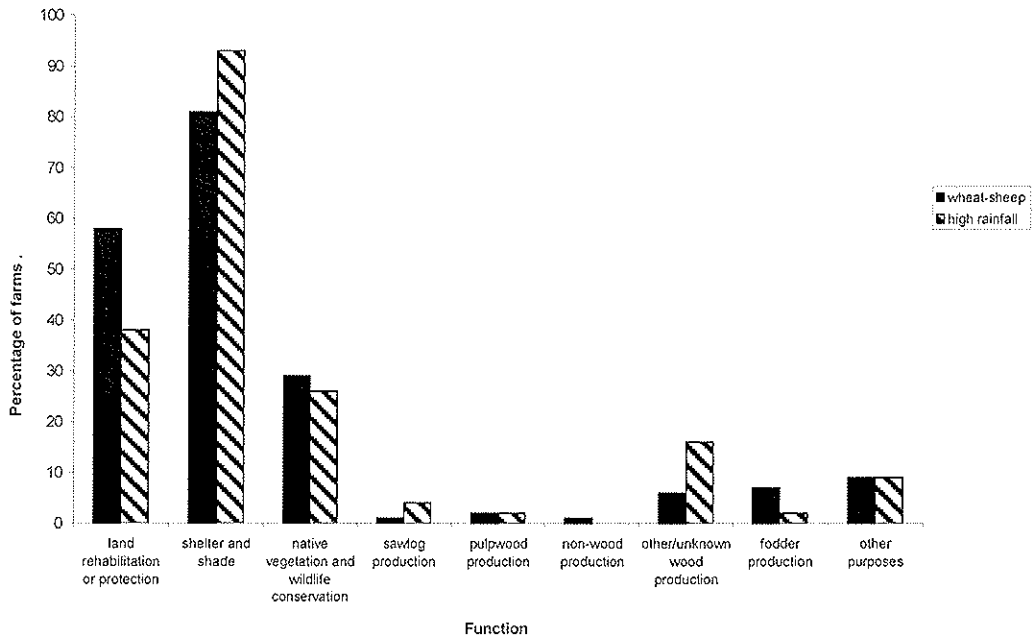

Figure 3-4. Main functions of planted trees on farms, as reported by Australian farmers (data from Wilson et al. 1995).

At the national level, planting trees to produce timber or non-timber products for sale was of relatively minor importance at the time of the ABARE survey (1994). However, in the last five years, there has been a large increase in agroforesty activities, with many plantings of blocks of fast-growing tree species on farms in southern Australia (e.g. Tasmanian Blue Gum *Eucalyptus globulus*).

Characteristics of recent plantings on farms (1991-92 to 1993-94)

Farmers with planted trees on their property were asked for detailed information about the most recent stand established during the three years 1991-92 to 1993-94 (Figure 3-5). Nationally, 63% of farmers had planted tree belts/corridors, 20% planted tree blocks, 8% planted alley belts, and 9% planted widely spaced trees.

The average area of the most recent stand of trees planted on each farm between 1991-94 was 5.1 ha for tree blocks, 3.8 ha for alley belts (including the area between tree belts), 657 trees per planting for tree belts and corridors (no measure of area reported), and 5.7 ha for widely-spaced trees.

The majority of recently established trees were local native species. For example, at the national level, 67% of farmers had established predominantly local native species, while 21% and 12% had established other native and exotic species, respectively.

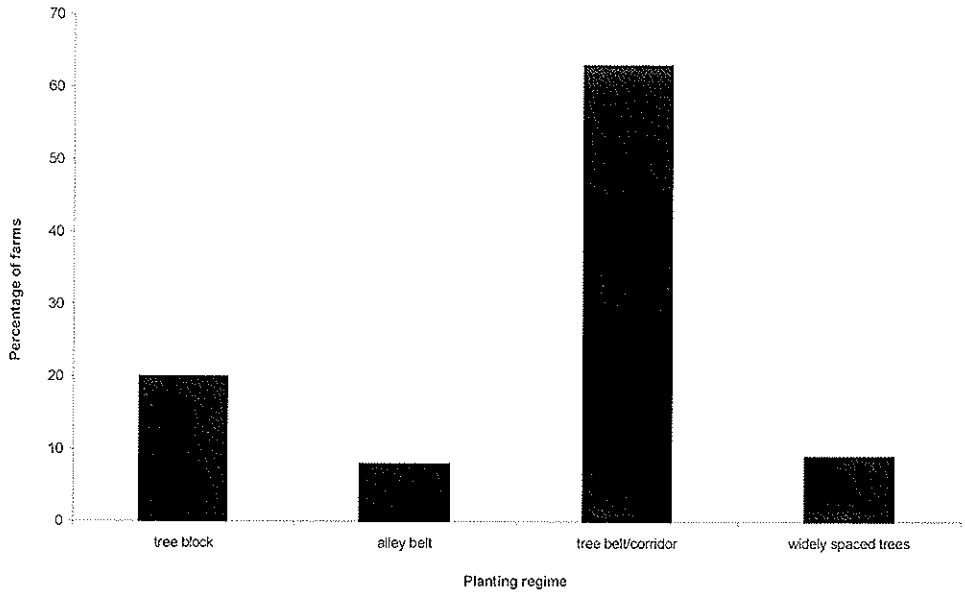

Figure 3-5. Form of the most recent (1991-1994) tree planting activities on Australian farms (data from Wilson et al. 1995).

The most commonly stated purpose for a farmer's most recent planting was (Figure 3-6):

- the provision of shade and shelter (68% of respondents);
- to rehabilitate or protect degraded land (21% of respondents); and
- to conserve native vegetation and wildlife (2% of respondents).

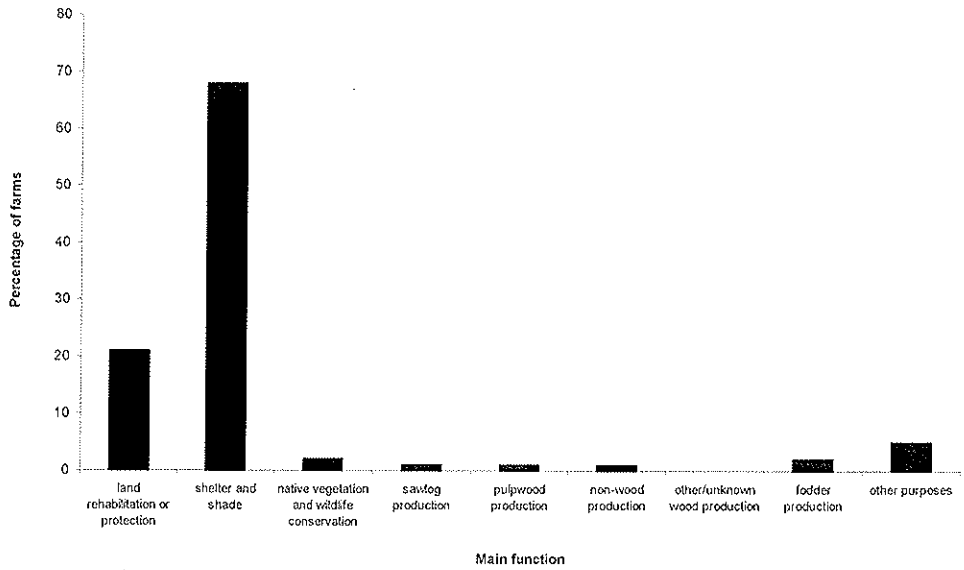

Figure 3-6. Main purpose of the most recent stand planted on farms 1991-1994 (data from Wilson et al. 1995).

Nationally, in both the wheat-sheep and high rainfall zones, less than 2% of farmers listed production of timber and non-timber products as the main function of the most recent tree plantings.

Case studies of revegetation in rural districts

Revegetation activities vary throughout Australia (Figure 3-2), and so several case studies are useful to illustrate revegetation activities at a catchment level.

1 – South Tammin Catchment, Western Australia

The South Tammin Catchment covers an area of 25,000 km^2 within the wheatbelt region of Western Australia. This region experiences a Mediterranean climate of hot dry summers and cool wet winters. Europeans progressively settled south-west Australia from 1827 and the wheatbelt now consists of an almost entirely human-modified environment. A mosaic of pastures, arable land and salt pans, with thousands of small remnants of native vegetation scattered across the landscape, now characterises most areas of the Western Australian wheatbelt (Saunders *et al.* 1993).

Revegetation in the South Tammin Catchment

Revegetation activities in the South Tammin Catchment commenced in 1982. For the first seven years a maximum of half of the 15 catchment members were involved in revegetation activities, with the area planted on an annual basis less than 50 ha. Only after dedicated funding for the catchment in 1989 did all catchment members become involved, with a subsequent boost in the area planted annually to between 100 and 300 ha (Brandenburg and Majer 1995).

Size and shape of revegetation

Most revegetated areas within the catchment are small (Figure 3-7) and linear in shape. Of the 123 revegetated areas within the catchment:

- 70% are less than five ha in size and 81% are less than 10 ha in size. Of the remaining 22 larger revegetated areas, 13 have been established primarily to enhance farm productivity (e.g. areas comprising fodder crop plantations, boudain and alley farming).
- 77% of plantations are linear in shape and comprise 63% of the total revegetated area within the catchment, while 23% of plantings are blocks and comprise 37% of the total revegetated area.

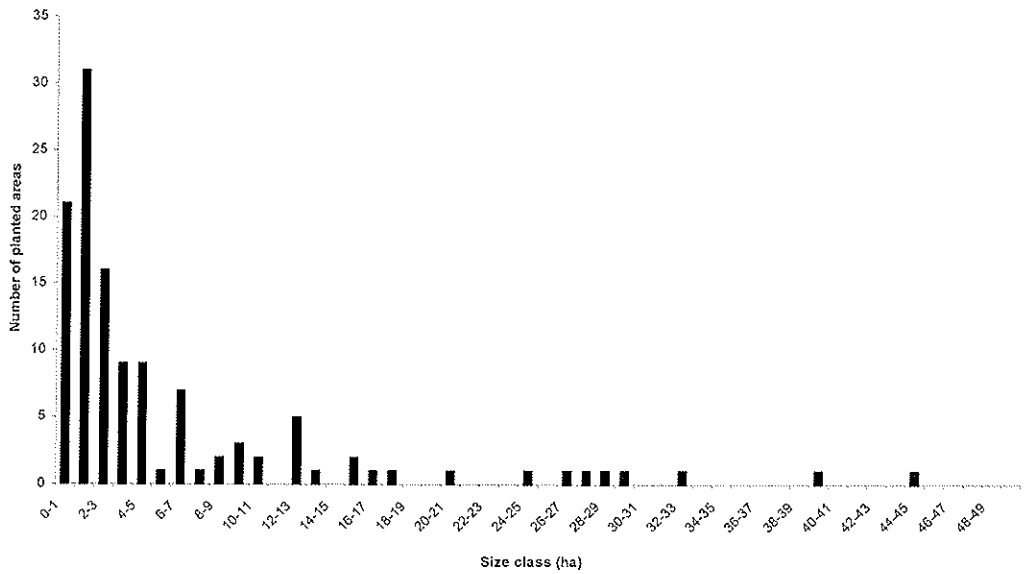

Figure 3-7. Size class distribution of revegetated areas in the South Tammin Catchment, Western Australia (redrawn from Brandenburg and Majer 1995). Note: there are two additional blocks of around 90 ha not included here.

Most revegetated areas in the South Tammin Catchment are extensions of existing remnants (68%), while 24% have no connection to remnant vegetation or other revegetated areas. Only 8% of the entire revegetated area connects at least two areas of uncleared native vegetation (Brandenburg and Majer 1995).

Species planted
A mix of native plants are present in about 70% of the total revegetated area. More than two-thirds of the areas revegetated with native plant species are characterised by native species alone, with the remaining third being characterised by the addition of tagasaste and saltbush species to the planting mix. Less than 1% is comprised entirely of local, Tammin or Western Australian natives.

Purpose and management
Revegetation within the catchment has been primarily aimed at achieving soil and water conservation rather than conservation of flora and fauna. To date, no information has been collected on the value of revegetated areas for fauna conservation. Brandenburg and Majer (1995) concluded that, in agricultural regions of Western Australia, revegetation is rarely undertaken for the primary purpose of biodiversity conservation.

2 – Goulburn Broken Catchment, Victoria

The Goulburn Broken Catchment in Victoria covers 2.4 million ha, stretching from the forested mountain ranges of the Great Dividing Range in the south, to the flat riverine plains adjacent to the Murray River in the north. The variations in geomorphology, climate and soils have produced a diversity of vegetation ranging from sub-alpine communities and tall wet forests to sparse woodlands and grasslands. European settlement began in the catchment during the late 1830's with the fertile riverine plains of the lower catchment providing suitable land for intensive agriculture. Irrigation systems were introduced in the late 1880's and now support one of the most important agricultural and food processing regions in Australia. Economic output from the irrigation region alone (approximately 20% of the catchment area) is estimated at more than $4.5 billion annually (GBCLPB 1997).

However, development of intensive agriculture has come at the expense of the natural environment. Salinity, waterlogging, soil erosion, declining water quality and loss of biodiversity are widespread problems throughout the catchment. Such widespread degradation has had a substantial impact on production and is now threatening the social, environmental and economic future of the catchment. Substantial investment is being made by government and community, working closely in partnership to address these serious issues.

Revegetation in the Goulburn Broken Catchment

The Draft Native Vegetation Management Strategy for the Catchment establishes targets for vegetation cover and outlines the need to restore depleted vegetation types to address land degradation, loss of production and biodiversity declines. Revegetation work is stimulated through a range of government funded incentive programs, as well as a substantial input from the community. Innovative cost-sharing arrangements recognise the benefits to both land owners and to the wider community and are designed to maximise the on-ground benefit. In the order of $2-$3 million is invested in revegetation and remnant habitat protection throughout the catchment each year.

Two salinity management plans for the catchment set targets for revegetation in priority areas to address hydrological cycles, productivity and water quality issues. Between 1990-1994, approximately 5000 ha of revegetation was established. At an average rate of around 1200 ha per year, the target is to establish over 40,000 ha of native vegetation to control rising groundwater and other land degradation issues. In addition, approximately 400 ha of farm forestry sites are being established per year, with predictions that this will rise rapidly in the near future as industry investment in farm forestry increases.

Design of revegetation
While the rates of revegetation for the Goulburn Broken Catchment as a whole are high, with many landowners participating, the amount of revegetation carried out on individual properties is low. For example, in the intensive irrigation region the average size of revegetated areas is 1.2 ha. Most sites are narrow linear strips to provide shade and shelter, with 2-3 rows of mainly native trees and shrubs.

Landowners suggest that loss of production, high land value, cost of establishment and the need to incorporate revegetation into the farm layout are major reasons for limiting the size of revegetated areas.

In the more extensive dryland region of the catchment, individual plantings are generally larger, averaging approximately 2.8 ha. In this region, plantings are more variable in design and include shelterbelts, 'break of slope' plantings to intercept groundwater, woodlots, and planting to bolster the width of roadsides for threatened fauna. Farm forestry is prevalent in the higher rainfall belt of this region, with some 1100 ha of plantations established since 1996 under one government-assisted scheme. Again, however, the average area of revegetation on individual properties is relatively small (approximately 12 ha), and consists mainly of monocultures of native and introduced species.

The potential benefits of revegetation for wildlife are minor considerations for the majority of revegetation activities in the catchment. There are, however, a small number of revegetation projects where biodiversity conservation is the primary aim. Species-specific revegetation programs in particular, have attracted widespread community support, an example of which is the Superb Parrot Project.

Revegetating habitat for the Superb Parrot

The Superb Parrot Project is a revegetation and remnant fencing program initiated to address the decline of the endangered Superb Parrot *Polytelis swainsonii*. Since 1992, the project has been responsible for planting 62,000 trees and shrubs on 74 sites and erecting 54 km of fencing. Landowners and volunteers have contributed over 12,000 voluntary hours and 100 ha of land for revegetation and the protection of remnants. Most of the work has occurred along the important 'fly ways' on private land that link the parrot's feeding areas with nesting sites in nearby forested areas. By linking remnant patches and providing *Acacia* species for food trees (the Superb Parrot feeds on a range of *Acacia* seeds) private landowners and agencies are attempting to provide the elements for the species survival in the area. On-going monitoring has revealed that Superb Parrots are now utilising the revegetated areas for feeding and are frequently recorded flying along revegetated strips between remnants.

Summary

Revegetation activities are widespread on farms throughout Australia, particularly in Victoria and Western Australia, with 35% of farmers sampled having undertaken some revegetation in the 3 year period 1991-94. The principal motivation for revegetation on farms is to provide shade and shelter for livestock and crops, and to rehabilitate or protect degraded land. Although only 2% of recent plantings on farms were undertaken to conserve native vegetation or wildlife, more than a quarter of farmers (29% in the wheat-sheep zone and 26% in the high rainfall zone) reported that this was an important function of revegetated areas. Most revegetation on farms is in the form of linear strips of trees (e.g. shelterbelts), and revegetated areas (either linear strips or blocks) are usually small in size (rarely more than 5 ha). Revegetation is predominantly carried out by using a mix of native species, but not necessarily species indigenous to that area. Case studies of revegetation from catchments in WA and Victoria confirm the results from a national survey – that revegetation activities on farms are overwhelmingly carried out for reasons related to agricultural production, are mainly linear strips, and are predominantly small areas.

Key references:

Brandenburg, S.A. and Majer, J.D. (1995). A database for revegetated areas in the Tammin region of Western Australia: implications for land owners, managers and researchers. Pp 258-270 *in* Nature Conservation 4: The Role of Networks. (Eds. D.A. Saunders, J.L. Craig and E.M. Mattiske). Surrey Beatty and Sons: Chipping Norton, New South Wales.

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Saunders, D.A., Hobbs, R.J. and Arnold, G.W. (1993). The Kellerberrin project on fragmented landscapes: a review of current information. *Biological Conservation* **64**: 185-192.

Wilson, S.M., Whitham, J.A.H., Bhati, U.N., Horvath, D. and Tran, Y.D. (1995). Trees on farms – survey of trees on farms: 1993-94. ABARE Research Report 95.7. Australian Bureau of Agricultural and Resources Economics: Canberra.

4. Revegetation as a habitat for wildlife in Australia

There are frequent references to the value of revegetation for biodiversity in rural environments (e.g. Hobbs 1993a; Oates 1995; Bennett and Platt 1996), but there is little empirical data clearly documenting the use of revegetated areas by fauna in Australia. This is a critical deficiency considering the intensive effort now being directed towards revegetation in Australia, and the frequent assumption that revegetation will have great benefits for biodiversity. Greater attention has been given to the use of revegetated and restored areas where mining has occurred, and this literature is also briefly reviewed. However, there are major differences between revegetated habitats in rural and mining landscapes and these are also discussed below.

Potential benefits of revegetation for wildlife

There are several ways in which revegetation can be undertaken to provide conservation benefits for wildlife (Hobbs 1993a; Lambeck 1999). *Direct* measures are those that provide direct benefits to wildlife by providing habitats in which species may live or from which they obtain resources such as food and shelter. *Indirect* measures are those that do not necessarily provide an immediate benefit to wildlife species, but by preventing the decline and degradation of wildlife habitats they indirectly benefit the species that use these habitats. Direct and indirect benefits from revegetation are not mutually exclusive – both may be achieved depending upon the way in which revegetation is planned and undertaken.

Direct measures

There are three ways in which revegetation can directly benefit wildlife species (Figure 4-1):

- by increasing the *amount* of suitable habitat in the landscape, either by adding to existing vegetation or by establishing new blocks of habitat;
- by improving the *quality* of existing habitats by selective replanting or seeding of additional plant species, to enhance the composition or structure of existing vegetation (e.g. adding understorey shrubs where only a tree layer is present); and
- by promoting the *connectivity* of existing habitats in the landscape by filling in 'gaps' to complete the continuity of an existing link (e.g. roadside or streamside vegetation), establishing a new habitat corridor to link several habitats, or providing 'stepping stones' between two habitats.

Figure 4-1. Means by which revegetation can directly benefit wildlife species: 1) increasing the amount of habitat, 2) improving quality of existing habitat, or 3) connecting existing habitats.

Indirect measures

Revegetation can be used to protect or enhance *existing* habitats in several ways:

- by acting as a buffer for existing vegetation to counter micro-climatic changes at the edge of the stand (such as wind speed, exposure to sun), and to minimise impacts from surrounding land uses (such as fertiliser drift, invasion by weeds or pest animals); and
- by stabilising disturbance processes in surrounding lands to prevent the entire landscape (including wildlife habitats) from being degraded. Revegetation to counter soil erosion and drift, waterlogging and rising saline water tables will increase the sustainability of the landscape and reduce the likelihood that natural habitats will be further degraded or lost.

Key references:

Hobbs, R.J. (1993a). Can revegetation assist in the conservation of biodiversity in agricultural areas? *Pacific Conservation Biology* **1**: 29-38.

Lambeck, R.J. (1999). Landscape Planning for Biodiversity Conservation in Agricultural Regions: A Case Study from the Wheatbelt of Western Australia. Biodiversity Technical Paper No. 2. Department of the Environment and Heritage: Canberra.

Faunal use of revegetation in agricultural landscapes

While it is recognised that revegetation has the potential to benefit wildlife, little is known about the use of revegetated habitats in agricultural landscapes by either the vertebrate or invertebrate fauna.

Vertebrates

Birds

Several reports provide information on the use of revegetated habitats by birds in Australia (Biddiscombe 1985; Ryan 1993; Crome *et al.* 1994; Leary 1995). Biddiscombe (1985) recorded the avifauna of four revegetated sites in south-west Western Australia over a seven year period from the time of planting. Ryan (1993) censused the avifauna of 24 revegetated strips and five stands of remnant roadside vegetation over a single seasonal cycle in northern Victoria. The study by Crome *et al.* (1994) included four windbreak plantings in north Queensland as part of a larger study of the fauna of remnant habitat in farmland. Leary (1995) censused and compared the avifauna of planted vegetation, remnant vegetation and cleared agricultural land in the Monarto region of South Australia.

Several further studies are in progress, for example:

- an investigation of birds in windbreaks/corridors in the Northern Tablelands of NSW being conducted by Cilla Kinross (University of Sydney);
- a study of the avifauna of revegetated sites in northern Victoria being undertaken by Paul Ryan (Deakin University); and
- a community-based study involving surveys of birds in different types of vegetation on farms, including revegetated habitats (Birds on Farms Scheme, Birds Australia).

Evidence indicates that revegetated areas provide habitat for a wide range of bird species; from the studies reviewed, at least 112 species have been recorded using planted vegetation, and anecdotal evidence suggests that many other species will utilise revegetated areas (Ryan in press). However, most of those species recorded from revegetated areas are regarded as 'generalist' species that occur in a wide range of habitats, or as 'open country' species that are able to live in modified habitats amongst farmland. Seasonal fluctuations in species diversity and abundance within revegetated habitats are evident. This seasonal change and the occurrence of many species represented by only a few sightings suggests that many species of birds are able to locate revegetated plots during movements through the landscape. Whether they are *resident* within revegetation depends upon the size and suitability of the habitat.

The pattern of occurrence of birds in revegetated areas has been found to be positively associated with a number of inter-related factors:

- *age of revegetated habitats:* Bird species diversity increases with habitat age as the structure of planted vegetation develops. For example, preliminary results of the Birds on Farms survey indicate that about six species are present per half hectare in areas with trees about three years old, while more than 13 species are present per half hectare in seven year old plantings (Francis 1997).
- *structural complexity:* Bird species diversity and abundance increases with structural complexity and vegetation density, because more layers provide a greater range of potential nesting sites, shelter and food sources. Structural complexity develops with age and also depends on the plant species present (floristic diversity).
- *floristic diversity:* Bird species diversity generally increases with the number of plant species present within revegetated habitats, because a variety of plant species may provide a range of food and shelter sources year-round or at specific times of the year.
- *season:* Changes in bird communities in revegetation have been associated with seasonal events such as migration periods and the flowering of eucalypts.

At a broader scale, increases in bird species diversity have been reported on a number of properties where revegetation has increased the area of habitat available to birds. Two examples have been noted from properties in Victoria:

(1) As part of a 'Wildlife on Farms' demonstration project (forerunner to the Land for Wildlife scheme), Alan and Mary Burgess invited several government agencies to create wildlife habitat on their property, a 35 ha dairy farm in irrigated farmland near Shepparton. The project began in the late 1970s and regular bird counts recorded a subsequent increase in the total bird species known from the property from 24 species in 1979 to 110 species in 1992 (Clugston 1992).

(2) John and Cicely Fenton on 'Lanark' at Branxholme, western Victoria, have been committed to a long-term plan of tree planting and restoration of wetlands since the early 1960s (Campbell 1991). Over the years, they have transformed 18% of their 800 ha property to timber or wetland with over 80,000 trees planted in belts, clumps, agroforestry plots and along waterways and swamp verges (O'Neill 1999). The bird population has burgeoned, with the number of native species recorded from the property increasing from some 39 species in 1956 to 155 resident or itinerant species by 1996 (O'Neill 1999).

Key references:

Biddiscombe, E.F. (1985). Bird populations and farm plantations in the Hotham River Valley, W.A. *Western Australian Naturalist* **16**: 32-39.

Crome, F., Isaacs, J. and Moore, L. (1994). The utility to birds and mammals of remnant riparian vegetation and associated windbreaks in the tropical Queensland uplands. *Pacific Conservation Biology* **1**: 328-343.

Francis, J. (1997). Revegetation brings back the birds: survey backs benefits. *Australian Farm Journal* **June**: 66-67.

Leary, D.E. (1995). An ecological assessment of the Monarto Revegetation Program. B.App.Sc. (Hons) Thesis. Department of Zoology, University of Adelaide: Adelaide.

O'Neill, G. (1999). Renaissance on Lanark. *Wingspan* (Supplement) **9**: 1-16.

Ryan, P.A. (1993). The utilisation of natural and planted linear habitats by birds in northern Victoria. B.Sc. (Hons) Thesis. School of Ecology and Environment, Deakin University, Melbourne.

Ryan, P.A. (in press). The use of revegetated areas by vertebrate fauna in Australia: a review. Chapter 19 *in* Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration. (Eds. R.J. Hobbs and C.J. Yates). Surrey Beatty and Sons: Chipping Norton, New South Wales.

Further reading:

Campbell (1991), Clugston (1992), Oates (1995).

Mammals

Published information on use of planted vegetation by native mammals is currently restricted to a survey of a single windbreak in northern Queensland in which no ground-dwelling or arboreal mammals were found.

Key reference:

Crome, F., Isaacs, J. and Moore, L. (1994). The utility to birds and mammals of remnant riparian vegetation and associated windbreaks in the tropical Queensland uplands. *Pacific Conservation Biology* **1**: 328-343.

Reptiles and amphibians

No studies have yet been published on the use of revegetation by Australia's herpetofauna in agricultural landscapes.

Invertebrates

Limited information is available on the use of revegetation by invertebrates. Trees were found to facilitate an increase in richness of soil fauna in agricultural landscapes on the Northern Tablelands, New South Wales (Chilcott *et al.* 1997). Revegetated sites in the Monarto region of South Australia were found to support 23 orders of surface-active, flying and tree-dwelling invertebrates compared with 24 orders in remnant vegetation and 21 in cleared agricultural land. Relative abundance of invertebrates was highest in cleared paddocks and lowest in the remnant vegetation (Leary 1995).

Key references:

Chilcott, C., Reid, N.C.H., and King, K. (1997). Impact of trees on the diversity of pasture species and soil biota in grazed landscapes on the Northern Tablelands, NSW. Pp 378-386 *in* Conservation Outside Nature Reserves. (Eds. P. Hale and D. Lamb). The University of Queensland: Brisbane.

Leary, D.E. (1995). An ecological assessment of the Monarto Revegetation Program. B.App.Sc. (Hons) Thesis. Department of Zoology, University of Adelaide: Adelaide.

Summary

Revegetation has the potential to enhance the conservation of wildlife in rural environments in a number of ways, but there have been few studies to test whether the proposed benefits are realized. Published information is restricted to several studies of the use of revegetation by birds. These show that, while a wide range of species have been recorded in revegetation, many of those resident in revegetated habitats are 'generalists' or 'open country' species that occur widely. This lack of research and published information is a critical gap in our knowledge, particularly given that revegetation activities are becoming increasingly widespread in Australia, are being supported by millions of dollars of funding by national environmental programs, and are being promoted as having great benefits for biodiversity conservation.

Wildlife and revegetation in mining landscapes

Use of rehabilitated mine sites by vertebrates and invertebrates has been studied at several locations in Australia, particularly following bauxite mining in the Jarrah *Eucalyptus marginata* forests of Western Australia, which is used as a primary example in this review (Kabay and Nichols 1980; Majer *et al.* 1984; Nichols and Watkins 1984; Majer and Bamford 1985; Nichols and Burrows 1985; Majer 1989; Nichols *et al.* 1989; Majer 1990). Other studies have included the occurrence of selected faunal groups following mining in the Northern Territory (Majer 1983, 1985), sand mining in coastal New South Wales (Fox and Fox 1984), and following coal mining in heathland at Anglesea, Victoria (Kentish 1983).

Vertebrates

Birds

The avifauna of revegetated plots on rehabilitated bauxite mines in the Jarrah forests of Western Australia have been the subject of several studies (Kabay and Nichols 1980; Nichols and Watkins 1984; Collins *et al.* 1985). These studies found that sites revegetated as recently as 4-5 years with a wide range of understorey species can support similar numbers (over 80%) of bird species, in similar densities and diversities, to unmined sites in upland Jarrah forest:

- 12 species were found nesting in rehabilitated sites (primarily those that nest in shrubs and lower canopy foliage); and
- 21 species used rehabilitated sites for foraging or hunting.

Species richness, density and diversity of birds were positively associated with a dense understorey and midstorey, and plant species diversity.

Bird species not found in rehabilitated areas included those that are migratory, dependent on forests with mature trees, rare or present only in low densities in the surrounding Jarrah forest. For example, Nichols and Watkins (1984) noted several species absent from revegetation:

- the Rufous Treecreeper *Climacteris rufa* which would be expected to utilise rehabilitated areas when bark and leaf litter have sufficiently developed to support suitable insect populations and when tree hollows for nest sites are available;
- species such as Willie Wagtail *Rhipidura leucophrys* and Singing Honeyeater *Lichenostomus virescens* that are characteristic of open, cleared or inhabited areas and are not common in the surrounding forest block;
- Varied Sitellas *Daphoenositta chrysoptera* which would not be expected in rehabilitated areas until more mature trees are present; and
- migratory or nomadic species, such as the Purple-crowned Lorikeet *Glossopsitta porphyrocephala* and Sacred Kingfisher *Todiramphus sanctus*, that were not common in surrounding Jarrah forest and may utilise rehabilitated areas when flowering eucalypts and insects become more abundant.

In all surveys, certain species (e.g. Grey Fantail *Rhipidura fuliginosa*, Striated Pardalote *Pardalotus striatus*, Western Rosella *Platycercus icterotis*) were found in rehabilitated areas in densities lower than those in unmined forest. As most bird species in the northern Jarrah forest were found to use rehabilitated areas it was concluded that such areas will not prove an ecological barrier to species movement (Nichols and Watkins 1984).

Key references:

Collins, B.G., Wykes, B.J. and Nichols, O. (1986). Recolonisation of restored bauxite minelands by birds in southwestern Australian forests. Pp 341-354 *in* Birds of Eucalypt Forests and Woodlands: Ecology, Conservation, Management. (Eds. A. Keast, H.F. Recher, H. Ford and D. Saunders). Surrey Beatty and Sons: Chipping Norton, New South Wales.

Kabay, E.D. and Nichols, O.G. (1980). Use of Rehabilitated Bauxite Mined Areas in the Jarrah Forest by Vertebrate Fauna. Environmental Research Bulletin No. 8. Alcoa of Australia Limited: Perth.

Nichols, O.G. and Watkins, D. (1984). Bird utilisation of rehabilitated bauxite minesites in Western Australia. *Biological Conservation* **30**: 109-131.

Mammals

Surveys in rehabilitated bauxite mine sites recorded most of the four introduced and eight native mammal species (excluding bats) known to use upland Jarrah forests (Kabay and Nichols 1980; Nichols *et al.* 1989). The Southern Brown Bandicoot *Isoodon obesulus* was the only species not found using the rehabilitated area, but individuals were trapped within 20 m of the site.

Key references:

Kabay, E.D. and Nichols, O.G. (1980). Use of Rehabilitated Bauxite Mined Areas in the Jarrah Forest by Vertebrate Fauna. Environmental Research Bulletin No. 8. Alcoa of Australia Limited: Perth.

Nichols, O.G., Wykes, B.J. and Majer, J.D. (1989). The return of vertebrate and invertebrate fauna to bauxite mined areas in south-western Australia. Pp 397-422 *in* Animals in Primary Succession: The Role of Fauna in Reclaimed Lands. (Ed. J.D. Majer). Cambridge University Press: Cambridge.

Reptiles and amphibians

Over 75% of reptiles (17 of 23 species) and 75% of frogs (six of eight species) found in upland Jarrah forest in Western Australia were recorded in rehabilitated bauxite mine sites (Kabay and Nichols 1980; Nichols and Bamford 1985). Utilisation of revegetated mine sites by reptiles and amphibians was found to depend largely on food availability and the microhabitat preferences of particular species. Generally, the return of reptiles and amphibians was positively associated with leaf litter accumulation, log abundance, and availability of open foraging space.

Those reptiles not found in rehabilitated areas were species that favour thick understorey, abundant leaf litter or dense ground cover (Nichols and Bamford 1985). It is expected that as these resources become available, reptiles not currently present will return to rehabilitated mine sites. For example:

- arboreal foragers, such as the Marbled Gecko *Phyllodactylus marmoratus*, should utilise rehabilitated areas as trees mature and bark develops; and
- suitable habitat for species of legless lizard and blind snakes (e.g. Burton's Legless Lizard *Lialis burtonis* and Mitchell's Short-tailed Snake *Unechis nigriceps*) should become available as leaf litter increases, logs build up, the abundance of other small reptiles and termite mounds increase, and Grass Trees *Xanthorrhoea* spp. are introduced.

Those species found at lower densities in rehabilitated areas than in healthy unmined Jarrah forest are also expected to become more abundant as resources develop. For example, reptiles such as:

- *Ctenotus labillardieri* and *Egernia napoleonis* should increase in number as logs become more abundant; and
- *Morethia obscura* and *Lerista distinguenda* should become more abundant as trees mature and leaf litter accumulates.

Key references:

Kabay, E.D. and Nichols, O.G. (1980). Use of Rehabilitated Bauxite Mined Areas in the Jarrah Forest by Vertebrate Fauna. Environmental Research Bulletin No. 8. Alcoa of Australia Limited: Perth.

Nichols, O.G. and Bamford, M.J. (1985). Reptile and frog utilisation of rehabilitated bauxite minesites and dieback-affected sites in Western Australia's jarrah (*Eucalyptus marginata*) forest. *Biological Conservation* **34**: 227-249.

Invertebrates

Invertebrate utilisation of revegetated bauxite mine sites has also been investigated (Nichols and Bunn 1980; Majer *et al.* 1984; Nichols and Burrows 1985; Nichols *et al.* 1989; Greenslade and Majer 1993). In general, the return of ground invertebrates is enhanced by provision of dense litter cover, larger logs, and dense and diverse plant cover.

Predatory invertebrates

Utilisation of rehabilitated mine sites by predatory invertebrates (arachnids, centipedes, earwigs, cockroaches and ants) was found to vary depending on the ecological requirements of the groups concerned (Nichols and Burrows 1985). Presence of predatory invertebrates was generally positively associated with the presence of logs and dense leaf litter, shrubs, small amounts of cover, and open spaces.

Arachnids

A study of spiders, harvestmen and scorpions revealed that it may take a minimum of 8 years for populations to reach a richness comparable to forests (Nichols *et al.* 1989). Return of arachnids was positively related to vegetation structure (percentage ground cover and number of shrubs over 1.5 m), and leaf litter depth.

Termites

Return of termite species in revegetated habitats was found to be a successional process with species which feed on sound wood first appearing. With increasing age of revegetation, species which feed on decaying wood, and litter and grass-eaters also return (Nichols and Bunn 1980). Most species were present by 12 years after disturbance. Age of revegetated habitat seems to be the most important determinant of termite species composition. The abundance of termites in rehabilitated mine sites was positively influenced by the amount of litter, and small logs and other wood materials.

Ants

A study of 30 rehabilitated bauxite mine sites and three control sites in unmined forest recorded 42 species of ants in rehabilitated areas, although many of the original forest species were not present (Majer *et al.* 1984). Species richness and diversity was positively associated with plant species richness and diversity, time since rehabilitation, percentage plant cover, and percentage litter cover. Species composition of the ant community in rehabilitated areas was influenced by these same factors (Majer *et al.* 1984).

Seventy-two species of ants were collected from unmined Jarrah forest sites that were not recorded within rehabilitated areas (Majer *et al.* 1984). These included:

- a number which nest in dead standing timber or logs, and others which often nest under logs. The paucity of dead wood in the rehabilitated areas may explain the absence of certain species;
- specialised feeders, for example, *Cerapachys* spp. are specialised predators of other ants, *Anisopheidole* sp. may feed on termites and *Acropyga* sp. probably tends root aphids;
- litter dwelling species. While litter has been sufficient to facilitate the return of some species, its low depth and cover may still limit the presence of other species;
- species not common in the Jarrah forest, whose absence from rehabilitated areas may reflect the low chance of finding them within an immediate locality.

Collembola

A study of 30 rehabilitated bauxite mine sites and three forest plots in 1978-79 recorded 60 species of Collembola in rehabilitated areas and 28 species in forest plots; nine species found in the forest plots were not present on mined sites (Greenslade and Majer 1980). Species richness of the collembolan community in the rehabilitated areas was positively correlated with plot age (time since rehabilitation), plant species richness and diversity, and percentage plant cover. Litter depth and cover was also associated with a diverse collembolan fauna. Presence of suitable logs and other soil attributes may also be important factors determining the return of some species. Those species not collected on rehabilitated plots have habitat requirements, such as rotted logs, which may take up to 100 years to develop. While many rehabilitated plots were older than three years, results indicate that the collembolan fauna on rehabilitated areas only begins to approach that of forest plots after 10-13 years when trees attain canopy cover similar to a forest.

Key references:

Greenslade, P. and Majer, J.D. (1993). Recolonisation by Collembola of rehabilitated bauxite mines in Western Australia. *Australian Journal of Ecology* **18**: 385-394.

Majer, J.D., Day, J.E., Kabay, E.D. and Perriman, W.S. (1984). Recolonisation by ants in bauxite mines rehabilitated by a number of different methods. *Journal of Applied Ecology* **21**: 355-375.

Nichols, O.G. and Bunn, S. (1980). Termite Utilisation of Rehabilitated Bauxite Minesites. Alcoa of Australia Ltd Environmental Research Bulletin No. 9.

Nichols, O.G. and Burrows, R. (1985). Recolonisation of revegetated bauxite mine sites by predatory invertebrates. *Forest Ecology and Management* **10**: 49-64.

Nichols, O.G., Wykes, B.J. and Majer, J.D. (1989). The return of vertebrate and invertebrate fauna to bauxite mined areas in south-western Australia. Pp 397-422 *in* Animals in Primary Succession: The Role of Fauna in Reclaimed Lands. (Ed. J.D. Majer). Cambridge University Press: Cambridge.

Recommended management practices for facilitating wildlife return to revegetated sites in mining landscapes.

Recommendations for change to rehabilitation practices in bauxite mine sites to facilitate faunal recolonisation have been summarised by Nichols *et al.* (1989). They include the need to: return larger logs for shelter, increase litter cover, increase the return of woody material, increase diversity of the understorey, select trees with bark similar to Jarrah, manage the understorey by thinning, deep rip the soil to facilitate penetration of tree roots, use fire to make the structure of sites similar to unmined, upland forest, reduce the number of perimeter roads around pits, introduce nest boxes for hollow-nesting birds and mammals, and control feral animals – mainly cats.

It is notable that all of these recommendations relate to management of vegetation at the site level so that particular resources required by species are available. It will be necessary for long-term monitoring to be undertaken to confirm that all species *do* return to the rehabilitated sites as the habitat becomes more suitable with time. Because such mine sites are surrounded by unmined forest, which provides a source of colonists for the revegetation, the issue of landscape context and spatial pattern of the revegetation is of lesser importance than in agricultural landscapes (see below).

Differences between mine-site rehabilitation and agricultural revegetation practices

There are several fundamental differences between revegetation in agricultural landscapes and mine-site rehabilitation (Hobbs 1993a). First, the objectives of revegetation in areas disturbed by mining are different from that of revegetation in agricultural systems. The underlying motivation for revegetation on a mining rehabilitation site, in response to a legislative requirement or 'social' responsibility, is to restore the original biota to the site following mining. Thus, the primary goal is restoration of flora and fauna. In contrast, the aim of most revegetation projects in agricultural areas is to protect land and water resources, or provide shelter for livestock, which largely dictates the design, placement and plant species selection for revegetation at the property and catchment scale. The potential benefits of revegetation for biodiversity conservation have generally been a secondary or 'incidental' objective in most situations in rural environments.

Second, the major difference between mine-site rehabilitation and agricultural revegetation is the landscape context in which they occur. In the Jarrah forest of Western Australia, for example, sites rehabilitated following bauxite mining are surrounded by, or in close proximity to, extensive areas of native vegetation which provide a suitable source for recolonisation and maintain natural ecological processes. In contrast, agricultural revegetation is frequently surrounded by a production landscape, hostile to most faunal species, and containing only small fragments of the original vegetation dispersed across the landscape.

Successful recolonisation of revegetated sites by animals in agricultural landscapes relies on species being able to traverse the intervening matrix, with the chance of successful recolonisation decreasing as isolation of revegetated sites increases. While the careful placement of revegetated patches within networks of remnant vegetation, or the establishment of corridors may reduce these difficulties, rarely will the situation in agricultural landscapes approximate that of mine sites. Consequently, the apparently successful restoration of mine sites with functioning ecosystems that resemble the original habitat must be treated with caution when setting goals for revegetation in agricultural landscapes (Ryan in press).

Key references:

Hobbs, R.J. (1993a). Can revegetation assist in the conservation of biodiversity in agricultural areas? *Pacific Conservation Biology* **1**: 29-38.

Ryan, P.A. (in press). The use of revegetated areas by vertebrate fauna in Australia: a review. Chapter 19 *in* Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration. (Eds. R.J. Hobbs and C.J. Yates). Surrey Beatty and Sons: Chipping Norton, New South Wales.

Summary

Studies of the use of revegetated areas by wildlife on former mining sites in Jarrah forests in Western Australia, show that a majority of species typically present in unmined forest have been recorded from revegetated sites, although their relative abundance may vary. The use of revegetated sites by most faunal groups is related to the extent to which the restored habitat provides resources for foraging and shelter, such as leaf litter, logs and woody material, diverse understorey, tree hollows, and vegetative cover. It is expected that the fauna of rehabilitated sites will increasingly match that of undisturbed sites as the vegetation ages. Two fundamental differences between mine-site rehabilitation and revegetation in agricultural landscapes mean that results from mine revegetation can not be directly extrapolated to the rural situation. The primary objective of revegetation on mine sites is to restore the indigenous flora and fauna, whereas most revegetation in rural landscapes is undertaken primarily to increase agricultural productivity with conservation as a secondary or incidental objective. Second, the spatial context of mine-site revegetation, a small area of restored vegetation surrounded by relatively undisturbed vegetation, differs markedly from revegetation within cleared open farmland.

5. Revegetation and its value for wildlife in other parts of the world

Although there are few studies of faunal use of revegetated habitats in Australia, there is a larger body of research in other countries that has examined faunal use of planted habitats in agricultural landscapes.

Types of revegetation

Planted habitats in agricultural landscapes in other parts of the world are predominantly linear features, including shelterbelts and windbreaks, hedgerows, fencerows and tree rows.

Shelterbelts and windbreaks are widespread in the Great Plains of the United States, the Soviet Union, China and Japan. Their structure and composition vary greatly but they generally consist of rows of woody plants that have been deliberately planted by humans for a variety of purposes, including shelter from wind, to reduce soil erosion, as a source of timber, for wildlife habitat and for their aesthetic qualities. These plantings may be a monoculture of non-indigenous tree species or a mixture of indigenous trees and shrubs that mimic the natural vegetation.

The use of plantings as shelterbelts and windbreaks to protect houses and farm buildings, livestock and crops from damaging winds, windchill and snow drift, and to reduce environmental degradation is common practice in agricultural landscapes throughout the world. For example:

- in the former Soviet Union, more than two million ha of shelterbelts were planted to reduce land degradation in agricultural areas (Schroeder and Kort 1989).
- in the United States, tens of thousands of kilometres of shelterbelts were established on the Great Plains earlier this century to reduce soil erosion from cropping areas (Baer 1989).
- in China, trees were planted around cropping land to prevent wind erosion as early as 550 BC. In the 1950's, 4000 km of forest belts were planted to control sand movement. In 1978, the Chinese government initiated a shelterbelt development program to form a 'Great Green Wall' of 35 million ha of trees stretching 700 km across northern China. In the last 40 years, 30 million ha of shelter-forest has been established in China with further plans being approved to control 6.7 million ha of desertified land in the next 10 years by planting trees, shrubs and grasses (Bird *et al.* 1992; Mitchell *et al.* 1998).

These shelterbelts now provide not only landscape protection benefits, but also extensive areas of potential wildlife habitat, in some cases forming important additions to the area of vegetation available to the fauna in these landscapes.

Hedgerows are widespread in Great Britain and Europe. These narrow bands of woody vegetation have been planted to form boundaries to farmland and barriers to the movement of stock between fields. Many are dominated by a single or a few woody species, although floristic diversity is enhanced by grasses and herbaceous species of the ground layer, and structural diversity by associated ditches and raised banks. Hedgerows form extensive networks linking remnant woods and forests retained within the rural landscape.

Fencerows are characteristic of farmland in eastern United States and Canada. They can either be planted or develop by regeneration and dispersal of plants in a neglected strip of land between fields. The vegetation ranges from grass and herb dominated rows, to narrow lines of shrubs, to broad strips with mature woodland trees. Woody plant species in fencerows are primarily those with wind or animal dispersed seeds, but wider fencerows may support woodland-interior species. Fencerows may form extensive corridor networks amongst farmland and between woodland patches.

Grassy field margins are characteristic of farmland in numerous countries, such as France, the Netherlands and United Kingdom. Narrow grassy strips are established either by natural regeneration or by sowing grass, and are usually cut or sprayed annually. There is a growing interest in the ecological role of such field margins, including their value as habitat for invertebrates (Mineau and McLaughlin 1996).

Plantations are usually block-style plantings (can also be linear strips) established primarily as a source of timber, although they may also provide a variety of land conservation benefits. Plantations can form extensive areas of vegetation. For example:

- an urban reafforestation plan in the Randstad region of western Netherlands, which encompasses the four largest cities (Amsterdam, The Hague, Rotterdam, Utrecht), involves the development of about 10,000 ha of forests and recreation areas over a 15 year period (Harms and Knaapen 1988).
- in Britain, an initiative to establish a National Forest is underway, with the aim being to develop a multi-purpose forest across 500 km^2 of land over three Midlands counties. It will involve planting about 30 million trees, and will have an ecological, commercial and recreational focus (Pearce 1994).

The value of these revegetated areas for wildlife or nature conservation has been investigated in a number of studies. Other values of revegetation in agricultural landscapes that have been reported include economic, recreational and aesthetic values (Baer 1989; Cable and Cook 1990; Cable 1991; Cook and Cable 1995).

Key references:

Baer, N. W. (1989). Shelterbelts and windbreaks of the Great Plains. *Journal of Forestry* **87**: 32-36.

Bennett, A.F. (1999). Linkages in the Landscape: the Role of Corridors and Connectivity in Wildlife Conservation. IUCN: Gland, Switzerland.

Bird, P.R., Bicknell, D., Bulman, P.A., Burke, S.J.A., Leys, J.F., Parker, J.N., van der Sommen, F.J. and Voller, P. (1992). The role of shelter in Australia for protecting soils, plants and livestock. *Agroforestry Systems* **20**: 59-86.

Burel, F. (1996). Hedgerows and their role in agricultural landscapes. *Critical Reviews in Plant Sciences* **15**: 169-190.

Cable, T.T. (1991). Windbreaks, wildlife and hunters. Pp 35-55 *in* Wildlife and Habitats in Managed Landscapes. (Eds. J.E. Rodiek and E.G. Bolen). Island Press: Washington DC.

Capel, S. W. (1988). Design of windbreaks for wildlife in the Great Plains of North America. *Agriculture, Ecosystems and Environment* **22/23**: 337-347.

Cook, P. S. and Cable, T.T. (1995). The scenic beauty of shelterbelts on the Great Plains. *Landscape and Urban Planning* **32**: 63-69.

Forman, R.T.T. and Baudry, J. (1984). Hedges and hedgerow networks in landscape ecology. *Environmental Management* **8**: 495-510.

Mineau, P. and McLaughlin, A. (1996). Conservation of biodiversity within Canadian agricultural landscapes – integrating habitat for wildlife. *Journal of Agricultural and Environmental Ethics* **9**(2): 93-113.

Schroeder, W. R. and Kort, J. (1989). Shelterbelts in the Soviet Union. *Journal of Soil and Water Conservation* **44**: 130-134.

Further reading:

Burel and Baudry (1990), Cable and Cook (1990), Forman (1995), Harms and Knaapen (1988), Mitchell *et al.* (1998), Pearce (1994).

Faunal use of revegetation in agricultural landscapes

In many countries, planted areas are recognised as an integral part of the landscapes in which they occur, providing important habitat for a variety of species. Indeed, concern has been raised about the potential biological impacts associated with the loss and deterioration of hedgerow and shelterbelt habitats in areas of North America, Britain, France and elsewhere (Sorenson and Marotz 1977; Forman and Baudry 1984; Conyers 1986; Baltensperger 1987; Schaefer *et al.* 1987; Burel and Baudry 1990; Cable 1991; Burel and Baudry 1995; Burel 1996).

A number of studies have shown that revegetated habitats are used by a wide range of faunal groups and contribute to the persistence of species in agricultural landscapes. Different species respond to the presence of hedgerows, fencerows, shelterbelts and windbreaks in different ways; they may be used for breeding, shelter, foraging, and movement or dispersal. Seasonal changes have also been recorded in the structure of bird, mammal, reptile, amphibian and insect communities of hedgerows (Forman and Baudry 1984).

Vertebrates

Birds

Much attention has been given to use of hedgerows, shelterbelts and fencerows as habitat for birds (Pollard *et al.* 1974; Yahner 1982a, 1982c, 1983b; Arnold 1983; Best 1983; Osborne 1984; Shalaway 1985; Lack 1988; Johnson and Beck 1988; Green *et al.* 1994; Parish *et al.* 1994, 1995; Haas 1995, 1997; Macdonald and Johnson 1995). These habitats have been found to provide access to food in adjacent fields, cover from predators, song perches, roosting sites, nesting sites, foraging habitats and movement corridors for a number of bird species around the world (see Table 5-1).

While large numbers of species have been reported using these linear habitats, primarily for shelter, temporary refuge or foraging, only a smaller group of bird species (mainly farmland, forest edge or generalist species) use these linear strips as nesting habitat. Hedgerows and shelterbelts are not a replacement for large blocks of habitat, but their presence does appear to significantly increase the diversity of birds, especially woodland species, in open farmland. Sites with hedges, ditches or linear woods have more species than comparable areas of arable land (Arnold 1983). Hedge intersections are favoured by many breeding birds in comparison with straight sections, probably because a greater area of habitat is available within a specified distance (Lack 1988).

The occurrence and richness of bird species is related to the:

- *dimension and structure of the hedgerow or fencerow*: Species respond to different aspects of hedge structure, but in general 'bird rich' hedgerows are those that are tall and wide, have many trees and species of trees, few gaps, contain dead timber, and a diverse structure.
- *associated habitats*: Habitats associated with hedgerows along field boundaries, such as ditches and wide grassy margins, also enhance the overall boundary strip for birds.
- *surrounding land use*: Land use surrounding hedges is an influential factor on their suitability as habitat. There is greater bird species richness in hedgerows where permanent pasture occurs on either side, in comparison with those with adjacent arable lands. The presence of rivers and ponds and garden habitat nearby also encourages use by birds.

Key references:

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Table 5-1. Examples of the use of hedgerows, shelterbelts and fencerows by faunal groups around the world.

Taxon	No. of species	Habitat	Location	Source
Birds				
Birds	93	fencerows	New York State, USA	Petrides 1942
Birds	25	shelterbelts	Hokkaido, Japan	Hino 1985
Birds	73	shelterbelts, fencerows	Iowa, USA	Best et al. 1995
Birds	37	hedgerows	eastern England	Arnold 1983
Birds	65	shelterbelts	South Dakota, USA	Martin 1980
Birds	66	hedges	Dithmarschen, Germany	Busche 1997
Birds	62	copses	Germany	Jetzsch 1992
Breeding birds	62	shelterbelts	Great Plains, Kansas, USA	Schroeder et al. 1992
Breeding birds	21	shelterbelts	Ceanu Mare, Romania	Munteanu 1990
Mammals				
Small mammals	10	hedgerows	England	Eldridge 1971; Pollard and Relton 1970; Kotzageorgis and Mason 1997
Small mammals	12	hedgerows	New York, USA	Petrides 1942
Small mammals	11	shelterbelts	Minnesota, USA	Yahner 1982b, 1983a
Small mammals	9	fencerows	Ontario, Canada	Asher and Thomas 1985
Bats	2	hedgerows, tree lines, tree lanes	Netherlands	Limpens and Kapteyn 1991; Verboom and Huitema 1997
Invertebrates				
Invertebrates	77	hedgerows, fields	England	Lewis 1969
Carabids	57	hedges (9 years old)	Bonn, Germany	Gruttke and Kornacker 1995
Carabid beetles	54	treed lane	France	Burel 1989
Carabid beetles	42	hedgerow network	France	Burel 1989
Geometrides (moths)	52	hedges	Switzerland	Lavorel 1988
Aphid eating insects	1000/m^2	field boundaries	United Kingdom	Chiverton 1991 cited in Mineau and McLaughlin 1996
Spiders	95	hedges (9 years old)	Bonn, Germany	Gruttke and Kornacker 1995
Harvestmen	9	hedges (9 years old)	Bonn, Germany	Gruttke and Kornacker 1995

Mammals

Habitats such as shelterbelts and hedgerows have been shown to facilitate movement and to provide refuge, foraging and breeding habitat for populations of some mammals (see Table 5-1). Hedgerows in Great Britain and Europe provide cover for woodland rodents such as Wood Mice *Apodemus sylvaticus* and Bank Voles *Clethrionomys glareolus* (Pollard and Relton 1970, Eldridge 1971). Other mammals using hedgerow cover include rabbits, squirrels, weasels, hedgehogs, moles, Common Shrews *Sorex araneus*, Pygmy Shrews *S.minutus*, Water Shrews *Neomys fodiens*, Field Voles *Microtus agrestis*, Brown Rats *Rattus norvegicus*, Yellow-necked Mice *Apodemus flavicollis*, Harvest Mice *Micromys minutus* and House Mice *Mus musculus* (Eldridge 1971; Pollard *et al.* 1974; Forman and Baudry 1984; Kotzageorgis and Mason 1997).

Fencerow and shelterbelt networks in North America are also used by a range of animal species (Petrides 1942; Dambach 1945; Ogilvie and Furman 1959; Wegner and Merriam 1979; Swihart and Yahner 1982; Best 1983; Asher and Thomas 1985; Shalaway 1985; Henderson *et al.* 1985; Johnson and Beck 1988; Bennett *et al.*1994). Small terrestrial mammals such as shrews, voles, rabbits and chipmunks live within suitable fencerow vegetation, while medium-sized mammals such as the Red Fox *Vulpes vulpes*, Striped

Skunk *Mephites mephitis*, Woodchuck *Marmota moxa rufescens* and Raccoon *Procyon lotor* use fencerows for foraging, refuge and movement (Bennett 1999).

Mammalian use of revegetation has been mainly associated with the structure and dimensions of the habitat. Species respond to different aspects of hedge structure, but 'mammal rich' hedgerows are generally those that are tall and wide, have a diverse and complex vegetation structure and composition, few gaps, and have logs and other dead timber. Fencerows, hedgerows and shelterbelts, on their own, are unlikely to maintain viable populations of many species. However, in conjunction with other larger areas of habitat, they have a role in maintaining wildlife, albeit mostly common species, within the farmland mosaic (Bennett 1999).

Key references:

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Limpens, H. J. G. A. and Kapteyn, K. (1989). Bats, their behaviour and linear landscape elements. *Myotis* **29**: 63-71.

Petrides, G. (1942). Relation of hedgerows in winter to wildlife in central New York. *Journal of Wildlife Management* **6**: 261-280.

Pollard, E. and Relton, J. (1970). Hedges. V. A study of small mammals in hedges and cultivated fields. *Journal of Applied Ecology* **7**: 549-557.

Verboom, B. and Huitema, H. (1997). The importance of linear landscape elements for the Pipistrelle *Pipistrellus Pipistrellus* and the Serotine Bat *Eptesicus Serotinus*. *Landscape Ecology* **12**: 117-125.

Wegner, J.F. and Merriam, G. (1979). Movements by birds and small mammals between a wood and adjoining farmland habitats. *Journal of Applied Ecology* **16**: 349-358.

Yahner, R. (1982b). Microhabitat use by small mammals in farmstead shelterbelts. *Journal of Mammalogy* **63**: 440-445.

Yahner, R.H. (1983a). Small mammals in farmstead shelterbelts: habitat correlates of seasonal abundance and community structure. *Journal of Wildlife Management* **47**: 74-84.

Further reading:

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Reptiles and Amphibians

There has been little research on the ecology of reptiles and amphibians in revegetated habitats in agricultural landscapes. Pollard *et al.* (1974) provide an overview of knowledge of amphibians and reptiles in hedges in Britain. In comparison to other parts of the world, such as Australia, Britain has a species-poor herpetofauna. Nearly all of the common species have been recorded in hedges. In the United Kingdom, hedges are considered to provide food, shelter, breeding grounds and possibly movement pathways between habitats for vipers (Prestt 1971).

Factors identified as influencing herpetofaunal use of hedges include provision of suitable basking sites and hibernation burrows. Presence of leaf litter and provision of suitable habitat for prey items are also considered important.

Key references:

Pollard, E., Hooper, M.D. and Moore, N.W. (1974). Hedges. Collins: London.

Presst, I. (1971). An ecological study of the viper in southern Britain. *Journal of Zoology, London* **164**: 373-418.

Invertebrates

Hedgerows facilitate movements and provide habitat for many invertebrates (Pollard *et al.* 1974; Cameron *et al.* 1980; Lyngby and Nielsen 1981; Burel 1992; Thomas *et al.* 1992; Sparks and Parish 1995; Mineau and McLaughlin 1996; Paoletti *et al.* 1997) (see Table 5-1). Carabid beetles, for example, use hedgerows in France as resident habitat and may breed there, and some species disperse along them. Dispersion of carabids along hedgerows was found to vary with the ecology of species (e.g. 'core', 'peninsula' or 'corridor' species). Up to 100 m from forest, all carabid species were present in hedgerows but core woodland species did not extend further. Lanes, depending on their width and vegetation density may serve as stepping stones for forest carabid species. Abundance of forest carabid species in hedgerows was related to a dense herbaceous layer and presence of a tree layer (Burel 1989).

Invertebrate abundance and diversity is generally greater in hedgerows than in the surrounding fields. Factors influencing the use of hedgerows by invertebrates include the age of hedgerow, proximity to woodland remnants, vegetation cover and structure, floral species richness (butterflies), presence of grassy areas in the form of verges or ditch banks (butterflies), litter cover, and hedge management.

Key references:

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Burel, F. (1992). Effect of landscape structure and dynamics on species diversity in hedgerow networks. *Landscape Ecology* **6**: 161-174.

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Lewis, T. (1969). The diversity of the insect fauna in a hedgerow and neighbouring fields. *Journal of Applied Ecology* **6**: 453-458.

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Further reading:

Cameron *et al.* (1980), Lavorel (1988), Paoletti *et al.* (1997), Pollard *et al.* (1974), Sotherton *et al.* (1981), Thomas *et al.* (1992).

Summary

Studies of the use of planted or revegetated habitats by wildlife have been made in a range of countries around the world, but mainly relating to linear habitats such as hedgerows, shelterbelts and plantations. A wide range of species, especially birds, have been reported using these habitats, but there has been little critical analysis of their conservation value as a habitat compared with larger blocks of vegetation or nature reserves. Clearly, they provide shelter and foraging resources for many species, and have a role in increasing faunal diversity in cleared farm landscapes. However, many of the species that use them are 'edge' species, 'generalists' or 'open country' species rather than those of greatest conservation concern. It is likely that their greatest value lies in supplementing the area of existing habitats and increasing connectivity in fragmented rural landscapes.

6. Remnant vegetation and wildlife – lessons for revegetated habitats

In many parts of southern Australia, remnants of natural vegetation occur throughout cleared and developed environments as patches of various sizes and shapes or as linear strips along roads and streams. They share a number of similarities with revegetated habitats in that they are generally small in size, surrounded by other land uses, frequently isolated from other similar vegetation, and vary in the composition and management of their vegetation. Consequently, studies of the ecology of remnant vegetation and its use by different faunal groups may provide insights into the potential design and management of revegetated habitats and the values of revegetated blocks of different size, shape, composition and location.

Features of remnant vegetation

In situations where an important goal of revegetation is to re-establish natural plant communities or natural habitats for animal species, remnant vegetation can serve as a model. A number of features of remnant vegetation that have particular relevance to planning for revegetation are summarised below.

Variation in vegetation composition in relation to site characteristics
The composition of natural vegetation varies in relation to local features such as geological substrate, soil structure and nutrient levels, availability of moisture, and site aspect. This results in subtle variation in the relative composition of the plant community at a local scale, and changes in dominant plant species in relation to local environmental gradients (such as elevation or soil moisture).

Floristic diversity
Natural plant communities are generally composed of species representing a range of families and growth forms. Forest communities, for example, are likely to include trees, tall shrubs, low shrubs, ferns, grasses, sedges, herbs, creepers and mistletoes, as well as mosses, lichens, and fungi. This diversity of plant species in natural communities is integral to maintaining natural ecological processes and to the provision of resources used by animal species, especially for food.

Structural diversity
The structure of a natural plant community is influenced by the types of plant species present, the physical conditions of the site, and the relative growth stage of the community (e.g. regenerating, ecologically mature, recovering from recent disturbance). Aspects of structural diversity include the relative height of the vegetation, the number of vertical 'layers'of vegetation, the vegetative cover at different height intervals (or layers), the spatial heterogeneity (or patchiness) of dominant plants and vegetation cover, the size of individual plants (especially trees), and the amount of standing or fallen dead material. These structural characteristics have a critical influence on the availability of habitat components that animals use for shelter, refuge, movement, and breeding (e.g. Recher 1991).

Diversity in the soil and litter layer
The ground layer, including the soil and leaf litter, has an important but largely unrecognized role in natural communities. The enormous diversity of life in these layers, including fungi, bacteria, microarthropods and macroarthropods (many insects), has a key role in two essential processes. First, through their role in decomposition of dead organic material they are critical to the cycling of nutrients in ecosystems. Second, they form the foundation of detritus-based food chains responsible for much of the energy flow through terrestrial ecosystems. Soil and litter invertebrates are a food source for many animals in the food web.

Vulnerability to changes at edges
Remnant vegetation is particularly vulnerable to change at the interface between the vegetation and surrounding altered land (such as cleared farmland, urban development), and consequently vegetation edges differ from the 'interior' in a number of ways. Smaller remnants (such as those less than 10 ha in size) and narrow linear strips such as roadside and streamside vegetation are particularly vulnerable. In many

instances, the entire remnant may be influenced by 'edge effects'. Changes that frequently occur at the edges of remnant vegetation include (Saunders *et al.* 1991):

- changes in microclimate, such as increased solar radiation, increased exposure to wind, altered air and ground temperatures and changed humidity;
- modified hydrological regimes resulting from changes to soil moisture, changes in evapotranspiration and altered flows of water in surrounding land; and
- influx of nutrients, and plant and animal species from surrounding land uses.

All of these changes affect the composition and structure of the vegetation at, and near, edges and consequently the habitats for animal species. This, in turn, alters the composition of animal communities and may have far-reaching effects on ecological processes in the remnant vegetation.

Impacts of land use

Land use *within* remnant vegetation as well as *surrounding* the remnant has important effects on the quality of the habitat that it provides to plants and animals. Land uses such as sustained grazing by domestic stock can bring about marked changes to the soil structure, plant species composition, vegetative cover, and capacity for tree and shrub regeneration in remnant vegetation (Scougall *et al.* 1993). Changes to nutrient levels and the invasion of grasses and weeds can also result in profound change to the ground layer in remnants, and hence to habitats used by many invertebrates, reptiles, small mammals and ground-foraging birds.

Key references:

Hobbs, R. J. (1993b). Effects of landscape fragmentation on ecosystem processes in the Western Australian wheatbelt. *Biological Conservation* **64**: 193-201.

Hobbs, R. J. and Atkins, L.A. (1988). Effect of disturbance and nutrient addition on native and introduced annuals in plant communities in the Western Australian wheatbelt. *Australian Journal of Ecology* **13**: 171-179.

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Recher, H.F. (1991). The conservation and management of eucalypt forest birds: resource requirements for nesting and foraging. Pp 25-34 *in* Conservation of Australia's Forest Fauna. (Ed. D. Lunney). Royal Zoological Society of New South Wales.

Saunders, D.A., Hobbs, R.J. and Margules, C.R. (1991). Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* **5**: 18-32.

Scougall, S. A., Majer, J.D. and Hobbs, R.J. (1993). Edge effects in grazed and ungrazed Western Australian wheatbelt remnants in relation to ecosystem reconstruction. Pp 163-178 *in* Nature Conservation 3: The Reconstruction of Fragmented Ecosystems. (Eds. D. A. Saunders, R. J. Hobbs and P. R. Ehrlich). Surrey Beatty & Sons: Chipping Norton, New South Wales.

Faunal use of remnant vegetation

There have been many studies of the use of remnant vegetation by wildlife in Australia. These range from studies of single species (e.g. Suckling 1984; Saunders and Ingram 1987; Margules and Milkovits 1994) to studies of whole communities of different groups of animals (e.g. Bennett 1990a; Barrett *et al.* 1994; Smith *et al.* 1996). Some studies have included large areas such as national parks and other conservation reserves (Kitchener *et al.* 1980, 1982; Caughley and Gall 1985) while others have been concerned with small remnants amongst farmland (Howe *et al.* 1981; Fortin and Arnold 1997). There has also been a number of studies of the pattern of occurrence of various faunal groups in linear remnants, such as remnant roadside vegetation (e.g. Bennett 1988, 1991; Cale 1990; Arnold and Weeldenburg 1990; Keals and Majer 1991) and streamside vegetation (Hill 1995; Bentley and Catterall 1997).

These diverse studies have produced a wealth of information on the types of species that use remnant vegetation, the value of different types of remnants, and the effects of remnant size, shape, isolation and position. The following summaries highlight some of the main results derived from these studies. References for a range of source studies undertaken on birds, mammals, reptiles and invertebrates are listed below.

Species-area relationship

A highly significant relationship between the number of species present and the size of remnant vegetation has been consistently found in those studies in which sampling effort was proportional to remnant size (Figure 6-1). Larger remnants support a greater number of species of birds (Kitchener *et al.* 1982; Howe 1984; Loyn 1987), mammals (Kitchener *et al.* 1980; Suckling 1984; Bennett 1987) and reptiles (Kitchener and How 1982; Caughley and Gall 1985) than do smaller remnants. One of the reasons for this relationship is that larger remnants generally have a greater diversity of habitats than smaller remnants. Habitat diversity appears to be a particularly important influence on species richness of reptiles (e.g. Kitchener *et al.* 1980).

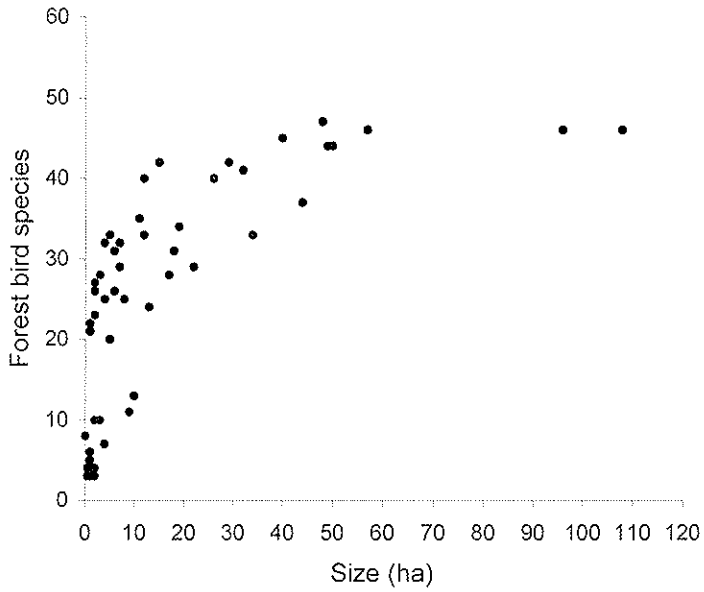

Figure 6-1. Species-area relationship for the number of forest bird species in forest remnants of varying sizes in Gippsland, Victoria (data from Loyn 1987).

Changes in species composition with remnant size

There are usually marked differences in the composition of the animal species present in remnants of different size classes. Larger remnants are more likely to support a faunal assemblage that is typical of the original unfragmented habitat, whereas faunal assemblages in small remnants are likely to be dominated by species that are able to persist in relatively disturbed habitats (Humphreys and Kitchener 1982). For some groups, such as mammals, there appears to be a relatively predictable pattern of occurrence of species in relation to remnant size with species 'dropping out' of the assemblage in successively smaller remnants (Bennett 1987; Deacon and Mac Nally 1998). It has proved difficult to reliably predict the kinds of species that are most vulnerable to habitat fragmentation, but a number of studies have provided insights for various groups (e.g. Laurance 1991; Saunders 1989; Sarré *et al.* 1995). Factors influencing vulnerability of species to fragmentation include:

- *body size*: larger species generally require larger areas of habitat in which to live (and in which to maintain a self-sustaining population) and so are less likely to occur in small fragments;
- *mobility*: species that are highly mobile and able to move between remnants are more likely to persist than those with limited capacity for movement; and
- *habitat specialisation*: species that require specialized habitats are likely to be more vulnerable to fragmentation than are species that have generalized habitat requirements.

Effect of spatial isolation

Remnants are isolated from similar vegetation by new forms of land use, such as cleared agricultural land or urban development. Those remnants in which animals are highly isolated from other populations, either by distance or by inhospitable surrounding habitats, are likely to be prone to species loss or change in the long

term (e.g. Arnold *et al.* 1993; Sarré *et al* 1995). Isolation reduces the ability of animals to move between remnants, and consequently restrict the opportunity for new individuals to supplement declining populations, for animals to recolonise should local extinctions occur, or to maintain ecological processes that involve animal movements.

Effect of time since isolation from similar vegetation
The numbers and types of species in remnant vegetation do not remain the same through time. Changes occur as some species disappear and new species colonize the habitat, often in relation to changes in surrounding land uses or disturbance processes in the vegetation (e.g. Saunders 1989; Recher and Serventy 1991). The trend is for a gradual long-term loss of species that are dependent on 'undisturbed habitat' and greater representation of species that are 'generalists' or favour disturbed habitats (including introduced species).

Quality of habitats in remnant vegetation
The way in which remnant vegetation is managed has a strong influence on the types of habitat resources available to animals. In turn, the availability and abundance of these resources has a significant impact on the occurrence and abundance of animal species, particularly for smaller and less-mobile species such as reptiles and invertebrates (e.g. Abensperg-Traun *et al.* 1996; Hadden and Westbrooke 1996; Smith *et al.* 1996; Bromham *et al.* 1999). Consequently, land uses such as grazing by stock and timber removal that result in changes to habitats (for example, loss of shrub and ground cover, soil compaction and disturbance, weed invasion, loss of large trees and associated tree hollows) may have greater effects on the occurrence of some species than factors such as remnant size or isolation (Abensperg-Traun *et al.* 1996; Deacon and Mac Nally 1998).

Values of linear remnants
In many regions of southern Australia, strips of remnant vegetation along roads and streams make up a substantial part of the remaining natural vegetation. A growing number of studies of these linear habitats (particularly roadside vegetation) show that they have value as remnant habitat for many species and that they enhance the connectivity of natural habitats in developed landscapes. Linear strips have been documented as a habitat for mammals (Bennett 1988, 1990a; Downes *et al.* 1997), birds (Arnold and Weeldenburg 1990; Cale 1990; Leach and Recher 1993; Bentley and Catterall 1996), reptiles (Brown and Bennett 1995) and some invertebrates (Keals and Majer 1991; Hill 1995).

In general, strips that are narrow and have experienced substantial degradation (such as grazing by stock or heavy weed infestation) are likely to be dominated by species that are common, habitat generalists, or that also occur in surrounding altered land. In contrast, wider and less-disturbed strips of remnant vegetation that retain key microhabitats, such as large trees, shrubs and ground cover and fallen timber, may be used by a large proportion of the fauna. They can have great value in their own right as a remnant habitat and support species that otherwise would not persist in the landscape (Bennett 1999).

Linear strips that connect two or more remnants have the potential to function as habitat corridors that assist the movement and interchange of species between otherwise-isolated populations. Such corridors may function by assisting the movement of single animals or by maintaining a resident population of animals along the corridor. Examples of the way in which remnant linear habitats enhance landscape connectivity and wildlife conservation are provided by Bennett (1990, 1999) and Saunders and Hobbs (1991).

Changes to ecological processes in remnants
Changes to ecological processes in remnant vegetation arise from the loss of native species, the introduction of new species (such as introduced predators or competitors), and changes to soil and vegetation from land use in the surrounding environment. To date our understanding of changes in natural ecological processes involving animals is limited, but it is likely that such changes have profound implications for the long-term conservation value of remnant vegetation (Saunders *et al.* 1991; Hobbs 1993b). Several examples of reported changes include:

- increased levels of predation on the native fauna by introduced predators (e.g. Red Fox *Vulpes vulpes*, Cat *Felis catus*);
- changes in levels of insect herbivory on plants (van Schagen *et al.* 1992);

- dominance of small remnants by Noisy Miners *Manorina melanocephala* with consequent aggressive exclusion of small insectivorous birds (Loyn 1987; Grey *et al.* 1997); and
- changes in relationships between mistletoes and host trees in woodland remnants (Norton *et al.* 1995).

Key references:

Birds

Arnold, G. W. and Weeldenburg, J.R. (1990). Factors determining the number and species of birds in road verges in the wheatbelt of Western Australia. *Biological Conservation* **53**: 295-315.

Barrett, G. W., Ford, H.A. and Recher, H.F. (1994). Conservation of woodland birds in a fragmented rural landscape. *Pacific Conservation Biology* 1: 245-256.

Cale, P. (1990). The value of road reserves to the avifauna of the central wheatbelt of Western Australia. *Proceedings of the Ecological Society of Australia* **16**: 359-367.

Howe, R. W. (1984). Local dynamics of bird assemblages in small forest habitat islands in Australia and North America. *Ecology* **65**: 1585-1601.

Howe, R. W., Howe, T.D., and Ford, H.A. (1981). Bird distributions on small rainforest remnants in New South Wales. *Australian Wildlife Research* **8**: 637-651.

Kitchener, D. J., Dell, J. Muir, B.G. and Palmer, M. (1982). Birds in Western Australian wheatbelt reserves - implications for conservation. *Biological Conservation* **22**: 127-163.

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Saunders, D. A. and de Rebeira, P. (1991). Values of corridors to avian populations in a fragmented landscape. Pp 221-240 *in* Nature Conservation 2: The Role of Corridors. (Eds. D. A. Saunders and R. J. Hobbs). Surrey Beatty and Sons: Chipping Norton, New South Wales.

Mammals

Arnold, G. W., Steven, D.E., Weeldenburg, J.R. and Smith, E.A. (1993). Influences of remnant size, spacing pattern and connectivity on population boundaries and demography in Euros *Macropus robustus* living in a fragmented landscape. *Biological Conservation* **64**: 219-230.

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Implications for the use of revegetated habitats by wildlife

Our present understanding of the ecology of remnant vegetation and its use by a range of faunal groups suggests that where the goal of revegetation is to mimic natural habitats for wildlife, planning and design need to be considered at several levels.

Site level

Important issues at the level of the individual 'site' include the need to select appropriate plant species to match the site characteristics, to plan for floristic and structural diversity of the revegetation, to consider the importance of developing ground-layer habitats, and to ensure that management is in place to deal with processes that might degrade the vegetation and habitat components. Time is a critical element in the quality of vegetation at the site level. It will take many years for newly-revegetated habitats to develop the characteristics of remnant vegetation, such as older trees with large canopies, large dead trees and large logs on the ground.

Block level

At the level of individual blocks of revegetation, important issues to consider include the size of the block, its shape and the length of edge, and its location in relation to surrounding land uses. Size is particularly important because habitat area is correlated with many other attributes such as population size, number of species, diversity of habitats, vulnerability to edge disturbance and so on. Increasing the area of revegetation is one of the most effective ways to increase its conservation value in the long term. Clearly, where the purpose of revegetation is to provide habitat for particular species, it is necessary to establish blocks of a size sufficient to meet that species' requirements.

Landscape level

A major issue at the landscape scale is the location of an area of revegetation in relation to other habitats, and how this will affect the capacity for animals to move to and from the new habitat. This is particularly important when the revegetation takes the form of linear strips (e.g. as shelterbelts or wildlife corridors). Additional issues at the landscape scale, when considering a broader conservation strategy, include the overall extent or amount of habitat and the representation of different vegetation types.

Summary

Where the primary goal of revegetation is to enhance biodiversity through the re-establishment of plant communities or animal habitats, useful insights can be gained from current knowledge of the ecology of remnant natural vegetation in rural landscapes. Characteristic features of the composition and structure of

remnant vegetation, and its vulnerability to disturbance from land use, can be used for planning for revegetation at the site level. Insights into important issues for revegetation at the block level come from studies of the use of remnant vegetation by wildlife species. These studies indicate the importance of size and shape, the maintenance of habitat components within blocks, and the location in relation to sources of disturbance. At a landscape level, studies of wildlife within remnant vegetation points to the importance of the location of the remnant in relation to other habitats, and thus the degree of isolation or habitat connectivity for populations of animals living within the remnant vegetation. These insights can be used in designing and planning revegetation to maximise its value as a habitat, either for single species or for broad faunal groups.

7. Benefits of revegetation and its role in sustainable land use

While revegetation has the potential to enhance the conservation of Australia's native flora and fauna, it also has significant benefits for sustainable land use including ameliorating land and waterway degradation, improving agricultural productivity, and enhancing human environments. This section outlines current knowledge of the benefits of revegetation and of how to design revegetation activities to achieve these benefits. An overview of the literature relating to revegetation techniques is presented in Section 8.

Protection of land and water resources

Revegetation has a role in overcoming land and waterway degradation, including minimising salinity and waterlogging, reducing erosion, landslips and mass movement of soil, and protecting water quality.

Salinity and waterlogging

Trees and other deep-rooted plants have the potential to minimise salinity problems resulting from rising saline groundwater. Through uptake of soil moisture and transpiration to the atmosphere, deep-rooted perennial vegetation distributed broadly across the landscape can reduce the amount of water added to the underlying water table and hence minimise or prevent its rise toward the surface. Measurements have been made of the rate at which trees take up and transpire water (Greenwood 1986; Schofield 1991; Williamson 1998), but the number of trees or percentage of a catchment that must be revegetated to stabilise or reduce the level of saline groundwater is a key area of research.

Research conducted in Western Australia provides an example of the extent to which revegetation (eucalypts in this instance) may assist with lowering water tables. Schofield *et al.* (1991) present a summary of the influence of revegetation on water table depth, particularly for revegetation of differing size, shape and location in the landscape. For example:

- *lower slope or discharge zone planting* was investigated at a farmland site where 44% of the natural vegetation had been cleared for agriculture. In 1979, 35% of the farmland was replanted with three species of eucalypts. Between 1979-1988 the water table beneath the revegetation fell by 2.0 m while the water table under pasture rose by 0.4 m.
- *strips or block style plantings*, which cover a smaller area of farmland (<15%), were also investigated at three sites. Tree planting had little or no effect on water tables, and it was suggested that in areas with annual rainfall greater than 700 mm, planting of less than 15% of farmland is unlikely to have a significant effect on the water table.
- *extensive high density plantings* covering more than 50% of farmland were investigated at two sites. At one site, 54% of an area cleared for agriculture was replanted with two eucalypt species. Between 1979-1988, the water table fell by 2.8 m, while under pasture there was a slight lowering (0.5 m) of the water table. At the second site about 70% of farmland was replanted, resulting in a lowering of the water table by 5.6 m under revegetation but a rise of 0.4 m under pasture, over the 1979-1988 period.

However, doubts have been raised about the effectiveness of water uptake by trees as a means of controlling problems of salinity and waterlogging at landscape and regional scales. For example, in a review of studies investigating groundwater uptake by plants, Thorburn (1996) found plants rarely take up more than 500 mm of saline groundwater (EC>5dS/m) per year, an amount similar to groundwater discharge rates from denuded soils. Furthermore, uptake of groundwater continually increases root zone salinity, further limiting water uptake by plants. Trials and simulations to test the role of different patterns of revegetation in controlling the water table are underway (see Land and Water Resources Research and Development Corporation 1999; Rural Industry Research & Development Corporation 1999 and CSIRO Australia 1999). Current knowledge suggests that changes in groundwater recharge over most of the landscape (e.g. tree planting across 50-70% of a catchment) are required to achieve significant reductions in the ultimate extent of salinity (T. Hatton, CSIRO, National Science Briefing June 1999, pers. comm. P. Lyons).

Nevertheless, selection of appropriate plant species and design of revegetation activities for ameliorating salinity and waterlogging have been widely addressed (see Riemer 1986; Oates and Clarke 1987; Morris and Jenken 1990; van der Moezel and Bell 1990 and other work cited within; Taylor 1994; Marcar *et al.* 1995; Oates 1995), but are not discussed further here.

Erosion

Vegetation can reduce water erosion of soils by:

- *intercepting rainfall:* Leaves intercept rainfall, absorbing much of the raindrop energy, thereby reducing the impact of rainfall splash which breaks down soil structure.
- *slowing water movement along the ground:* Extensive ground cover slows the downhill movement of water, regulates infiltration rates into the soil, and acts as a trap for silt and debris.
- *binding the soil:* The extensive, deep and spreading root systems of some plants bind the soil and improve its water absorption capacity. Movement of water deeper into the soil can be influenced by the presence of soil pores created by roots. Vegetation along stream banks reduces bank collapse by binding soils and slowing the flow of floodwater.

Wind erosion is reduced when vegetation slows the wind speed and reduces turbulence. Plantations of trees and shrubs can reduce wind speed over substantial distances , thus limiting the ability of wind to cause soil erosion. For example, shelterbelts can reduce wind speed by 50% in the zone up to 10 times their height (10H) and provide some shelter up to 30H (Haines and Burke 1993).

Vegetation can also assist with *preventing or controlling landslips and mass movement of soil.* The stability of the land surface is influenced by soil type, geology, soil moisture, slope and vegetation cover. Steep slopes in particular can be prone to mass movement of soil. Vegetation helps stabilise the soil by reducing soil water movement and maintaining groundwater levels. Roots and litter also bind soil particles together.

Design of revegetation and selection of plant species appropriate for controlling wind and water erosion have been addressed by Riemer (1986), Oates and Clarke (1987), Marshall (1990) and Harrison (1993), and are not discussed here.

Protecting and enhancing water quality and controlling stream bank erosion

Research into the role of vegetation in protecting water quality in Australia has been reviewed by Woodfull *et al.* (1993). Vegetation can protect aquatic environments in a number of ways:

- by acting as a filter for sediment and attached nutrients and contaminants (including bacteria);
- by providing shade and reducing fluctuations in water temperature and oxygen levels;
- by strengthening and stabilising banks, thereby helping to prevent erosion and subsequent silting of stream beds; and
- by providing terrestrial and stream habitat. Inputs of litter and other organic debris provide food for stream biota, and inputs of logs and other coarse debris provide habitat for invertebrates and fish.

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Agricultural productivity

Agricultural production can benefit directly from revegetation through shade and shelter for livestock and crops, protection of buildings, and through harvesting of tree products such as timber and firewood. It has been estimated that up to 20% of a farm may be planted with trees without loss of economic return from the property (Taylor 1994; Bird and Cumming 1996).

Shelter and shade for livestock and crops

Environmental stress reduces livestock production and plant growth. Shelter from wind, sun or extreme temperatures improve livestock performance and increase plant growth. The benefit of planted shelterbelts has been extensively studied in Australia (see Reid and Bird 1990; Bird *et al.* 1992; Haines and Burke 1993; Burke and Wilson 1995), and in other countries (see Forman 1995).

Shelter provided by trees can directly benefit livestock production. Higher survival rates of newborn lambs, improved weaning rates, better weight gain by meat-producing animals, increased milk production and greater wool production have been reported as benefits of reducing the stress of harsh weather on livestock (King 1993). For example, Reid and Bird (1990) reported that at Armidale, NSW, shelter was found to almost halve the death rate of single lambs from 17% to 9%, increase wool production by 31%, and increase liveweight gain by over 10% at the highest stocking rate assessed over a 5 year period.

Shelter can also directly benefit pasture and crop production by reducing wind speed and turbulence, reducing wind erosion, lowering the rate of evaporation and thereby conserving soil moisture, moderating air and soil temperatures, and reducing the extent of physical damage to plants by windblown particles. Vegetation can also trap diseases attached to airborne particles and other windborne pests (Oates 1995; Abel *et al.* 1997). Crops grown near shelterbelts can have higher yields than crops grown in exposed paddocks (King 1993). For example, Reid and Bird (1990) reported that pasture or crop yield may average 20-30% more in a sheltered zone extending to a distance equal to 10 times the height of the shelterbelts than on unsheltered land. Research into citrus crops in NSW recorded a 37% increase in yield and improved size and appearance of fruit in response to artificial shelter.

NATIONAL NATURAL RESOURCE MANAGEMENT MONITORING AND EVALUATION FRAMEWORK

Monitoring and Evaluation Framework endorsed by the Natural Resource Management Ministerial Council, 3 May 2002 Meeting

Attachment C under review

Last Revised: 8 April 2003

PURPOSE AND SCOPE

1. The Natural Resource Management Ministerial Council has established the National Natural Resource Management Monitoring and Evaluation Framework (National Framework) to assess progress towards improved natural resource condition through the development of accurate, cost-effective and timely information on the:
   - health of the nation's land, water, vegetation and biological resources; and
   - performance of programs, strategies and policies which provide national approaches to the conservation, sustainable use and management of these resources.

   The National Framework is based on a set of principles (see below) for the monitoring, evaluation and reporting on natural resource condition. It also provides a set of indicators for assessing change in resource condition and program performance. The roles and responsibilities for meeting national, state/territory and regional level reporting requirements are outlined.

PRINCIPLES

2. The National Framework is based on principles, which ensure that processes within the framework are useable, cost-effective, accurate, comprehensive and transparent. The framework is structured so that it:

(a) is **useful for all partners** in natural resource management – Commonwealth, States and Territories, and regions, communities and industries.

(b) is **simple, cost-effective, affordable and practical**. To suit these ends the data infrastructure required to support the framework:
   - *avoids duplication of effort*, and maximises the benefits of earlier investment in data collection, by building on existing State, Territory and Commonwealth initiatives for developing and sharing of data such as the NLWRA and State of the Environment reporting;
   - *uses data for multiple purposes*, where ever possible. In particular, data are collected so that they can be used for both monitoring resource condition and assessing program outcomes. This requires data to be collected in such a way as to permit their use at a range of scales and levels (national, state, regional and local);
   - *ensures that users can obtain the data*. Data are easily accessible to all sectors of the community in format, location, cost and under conditions that do not inhibit their use; and
   - *ensures that users can easily find out whether suitable data already exist*. All data are documented in the Australian Spatial Data Directory with sufficient information for users to determine whether the data are suitable for their intended purpose.

(c) recognises that **NRM interventions encompass a range of time-scales**. While interventions undertaken by Government are usually structured as programs conducted over periods of between five and seven years, significant changes in the resource condition will not be measurable for periods ranging from five to fifty years. Processes to measure and report resource condition change and the performance of NRM interventions must encompass this range of time-scales. This relationship is represented in a diagram included as **Attachment A**.

(d) supports **meaningful interpretation of data over time** by establishing standard national indicators, protocols for their sampling, measurement and interpretation , and data quality and management requirements. Protocols specify the quality of the data to be produced and ensure the data can be used for their intended purposes. To maximise their use and comparability, where required, data are developed and maintained to meet agreed international or national guidelines or standards for the management of spatial information as endorsed by ANZLIC or national coordination arrangements. Quality assurance and control requirements will ensure the consistency of the monitoring process over time and across jurisdictions and may necessitate the accreditation of complying monitoring programs.

(e) **specifies the assumptions** on which monitoring and evaluation activities are undertaken in a consistent manner which is open to all stakeholders.

STRUCTURE

3. The National Framework structures monitoring and evaluation processes at the national level and also provides a blueprint for monitoring and evaluation frameworks for programs, strategies and policies within the scope of the Council. The framework is represented diagrammatically in the following table:

	Natural Resource Condition	**Program, Strategy and Policy Performance**
Monitoring	• Natural resource condition monitoring at local, regional, State/Territory and national levels	• Monitoring of resource condition against Standards and Targets Framework) • Management action monitoring
Evaluation	• Evaluating progress towards improved natural resource condition at the national level	Performance evaluation of programs and strategies

Evaluating models & assumptions

The following discussion describes the elements of the above diagram.

NATURAL RESOURCE CONDITION

4. The health of the nation's natural resource is being assessed to provide a continuing reference point against which the appropriateness and effectiveness of national policies, strategies and programs may be judged. This assessment assists the Ministerial Council to identify areas of concern and to better target the use of resources.

Monitoring the Natural Resource Condition

5. The National Framework utilises nationally agreed outcomes and measures to report on the conservation, sustainable use and management of Australia's land, water, vegetation and biological resources. A core set of resource condition indicators measure progress towards the agreed national outcomes on a medium and long-term basis.

Reports on resource condition trends and associated measures are provided at least every five years.

6. In addition to the primary set of resource condition indicators providing information on resource condition trends, this monitoring will encompass measures of community and social processes relevant to or affected by NRM programs as well as measures of the adoption of sustainable development and production techniques.

7. Since changes in resource condition can result from a range of interventions, as well as from extrinsic factors, interpretation of this assessment will be supplemented with additional contextual information pertinent to the indicator being considered. These contextual data will also be required to permit examination of the scientific and program assumptions that underlie specific interventions.

Evaluating Progress towards improved natural resource condition

8. While most evaluation will focus on specific programs, there will be a need from time to time to commission evaluations of policies, strategies or activities pertinent to one or more programs. The appropriate evaluation focus will be defined on a case-by-case basis.

PROGRAM, STRATEGY AND POLICY PERFORMANCE

9. The performance of programs, strategies and policies is assessed to determine the achievements of these interventions towards improved natural resource condition. Monitoring and evaluation is used for both program improvement and accountability purposes and supports better targeting of resources.

10. It is recognised that factors beyond the scope of government and community interventions may affect natural resource condition. Participants will be accountable for the implementation and delivery of programs, strategies and policies for which funding has been provided, in accordance with contemporary best practice. It is not appropriate for participants to be held directly accountable for changes in resource condition where there is a lack of major scientific certainty or there are significant external factors over which the participants have little or no control.

11. Prior to the commencement of an activity (such as a major new program like the National Action Plan on Salinity and Water Quality), all stakeholders will agree to a monitoring and evaluation framework which will include:
    - Performance information, including all performance indicators and associated data protocols, data collection processes to meet the reporting, performance management and review requirements at local, regional, State/Territory and national levels;
    - Roles and responsibilities for collection, storage, analysis and reporting of monitoring data. This may include cooperative data-sharing arrangements where possible to minimise duplication and maximise efficient use of data;
    - A schedule of planned evaluations at all levels including roles and responsibilities for the identification, direction and conduct of

evaluations, processes for the implementation of recommendations, and arrangements for review of the schedule;

- Timelines, linked to the program implementation schedule, for reporting achievement of milestones, monitoring information, and production of planned evaluation reports; and
- An appropriate allocation of resources to implement the strategy, including cost-sharing arrangements for major stakeholders.

This Framework will be available for the Council to use in assessing progress against expected outcomes for that particular activity.

12. The design of each NRM program, strategy or policy is based on assumptions that particular interventions will bring about change in resource condition. These linkages and assumptions will be documented to provide an explanation of the logical structure of each program, strategy or policy and its intended impact on the resource condition and could include a high level goal, and one or more achievable and measurable, outcomes. The model also provides the basis for testing the assumptions and models adopted as part of the program design and assists in determining appropriate performance information to monitor and evaluate progress towards achieving the desired outcomes. A draft example of such a structure is included at **Attachment B.**

Monitoring of Resource Condition against Standards and Targets Framework

13. The National Framework utilises the national natural resource outcomes identified by the National Natural Resource Management Standards and Targets Framework. The national outcomes provide direction for catchment/regional communities to identify specific timebound and measurable targets for each region, which will move natural resource condition towards the achievement of the national outcomes. Relevant national outcomes and associated measures from the Standards and Targets Framework will be used to monitor changes in resource condition associated with each program, strategy or policy. Further detail on key aspects of the development and characteristics of these indicators are included at Attachment C.

14. Additional national outcomes and associated measures of interest to the Ministerial Council will be included in the National Framework as they are identified and developed.

Management Action Monitoring

15. Each national program, strategy and policy will require specific performance information to meet its accountability and management needs. This performance information will refer to the particular management actions established as part of the rationale. Performance of these management actions will be reported at investment, output and intermediate outcomes levels.

Performance Evaluation of Programs and Strategies

16. Performance evaluation will assess the appropriateness, effectiveness and efficiency of the program, strategy or policy in the achievement of its objectives and intended outcomes at all levels. The assumptions, logical

structure and performance indicators will be used to focus specific performance evaluations.

17. To improve program design and delivery, reorient investment during the life of the program, and fulfil accountability requirements, evaluations should be planned to inform key decision points throughout the life of the activity. Planning for evaluations must remain responsive to management needs with the flexibility to undertake evaluations as the program unfolds and as evaluation questions emerge.

18. Planned evaluations will be detailed in the Monitoring and Evaluation Framework for each program, strategy or policy. A register of evaluations will be maintained and reports from these evaluations will be available for the Ministerial Council and other stakeholders to use in considering achievement of expected outcomes for that particular activity.

CAPACITY BUILDING

19. Because the Commonwealth, States/Territories and Regions all benefit from effective monitoring and evaluation, all three levels share responsibility to provide the support required to ensure the effectiveness of these processes, including training/accrediting people to use the national protocols.

20. Where any element of monitoring and evaluation capacity needs to be developed to support the implementation of the National Framework, linkages to the capacity building strategy of the relevant program, strategy or policy will be established to ensure appropriate support.

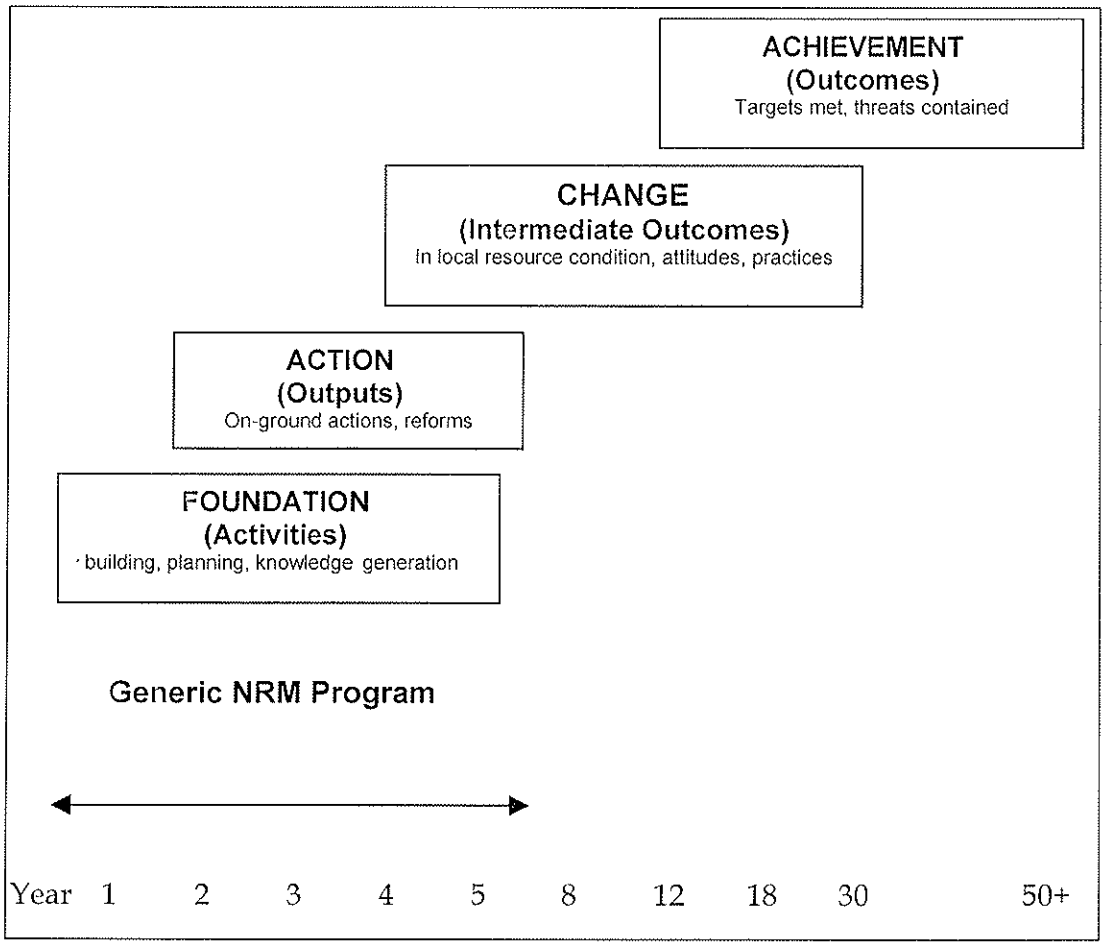
21. It is intended that activities will be fully costed prior to the commencement of each program, strategy or policy. Costs will be shared on an agreed basis established prior to the commencement of activities.

REPORTING

22. All stakeholders' reporting requirements are incorporated within the framework. Reporting products will be specified to meet the needs of stakeholders at all levels.

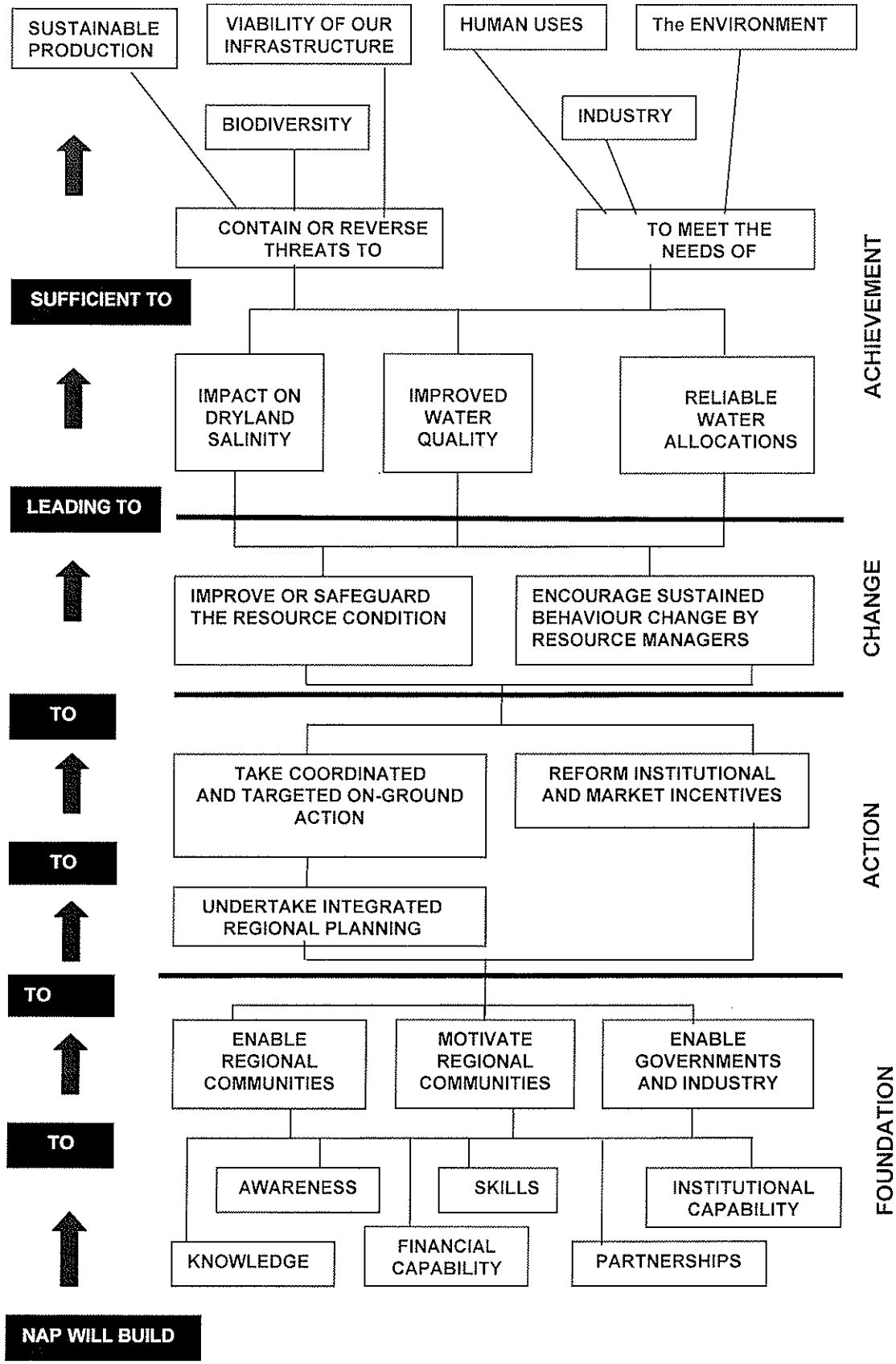
23. Each program, strategy or policy will establish a comprehensive reporting structure for monitoring and evaluation activities at all levels tailored to the needs of the stakeholder groups. The structure should support the transmission of complete and accurate information in the agreed format to the required stakeholders in time for it to be used in the processes for which it was collected. The reporting structure should include all statutory and accountability requirements including:
    - Agency annual reporting;
    - Budget document reporting;
    - Performance reporting within financial agreements at predetermined intervals;
    - Evaluation reports

ATTACHMENT A



ATTACHMENT B

CORE NAP LOGIC DIAGRAM



ATTACHMENT C

DEVELOPING A SET OF INDICATORS FOR THE NATIONAL FRAMEWORK FOR MONITORING AND EVALUATION

Introduction

The National Natural Resource Management Monitoring and Evaluation Framework (National Framework) will support the NRM Ministerial Council in assessing progress towards improved natural resource condition through the development of accurate, cost-effective and timely information on the:

- health of the nation's natural resources, and
- performance of programs, strategies and policies which provide national approaches to the conservation, sustainable use and management of Australia's land, water, vegetation and biological resources.

The National Framework provides information on the health of the nation's resources and on the conservation, sustainable use and management of Australia's land, water, vegetation and biological resources. The health of the nation's natural resource will be monitored on an intermediate and long-term basis through a coordinated, comprehensive and independent resource condition assessment process. This periodic assessment of resource condition will be used to judge the appropriateness and effectiveness of national policies, strategies and programs may be judged. This will assist in better targeting of natural resource management strategies.

The national framework establishes nationally agreed outcomes and measures to report on resource condition changes and associated institutional, social and economic matters. This process requires a primary set of resource condition indicators to provide information on resource condition trends, measures of community and social processes relevant to or affected by NRM programs and measures of the adoption of sustainable development and production techniques.

A core set of these indicators is required to measure progress towards the agreed outcomes on a medium and long-term basis.

Issues for which indicators are required

Identifying indicators will be assisted by the work already undertaken in developing the National Standards and Targets Framework. The framework identifies natural resource management outcomes and the minimum set of matters for which targets are set in integrated natural resource regional plans. The natural resource issues covered in the framework are of interest to the NRM Ministerial Council and reflect the current program, strategy and policy needs. The immediate priority is to define the indicators and measurement and reporting protocols for the salinity and water quality matters so far identified for the National Action Plan. The next priority will be to define the indicators and measurement protocols for the in-stream and terrestrial biodiversity matters, as required by the National Action Plan, when the national outcomes and minimum set of matters have been identified. Adopting the outcomes and the minimum set of matters developed through the National Standards and Targets Framework as core elements of the National Monitoring Framework should be the basis of a practical and cost effective process for the development of the monitoring framework. Other processes will need to be developed for matters

of interest to the Ministerial Council which fall outside of the Standards and Targets framework.

At the highest level the national framework will support the achievement of the NRMMC's objective: "to promote the conservation and sustainable use of Australia's natural resources". In addition, information at smaller scales, possibly down to the local catchment, will also be required to support on-ground management actions.

At the broad scale the main requirement is that indicators be able to monitor changes in resource condition and management practice. This will require indicators of resource condition, management and relevant socio-economic conditions. These indicators are selected according to the principles of cost, simplicity, consistency and practicality along with their ability to deliver information from the local through to the national scale.

A set of indicators that can operate through a range of spatial and temporal scales is needed. The ideal set of indicators would encompass long term outcomes, such as changes to the natural resource base, intermediate outcomes, such as changes in land management practices, and short term outcomes, such as increased awareness of NRM related programs.

Significant changes in the resource condition will not be measurable for periods ranging from five to fifty years. However, NRM interventions undertaken by Government are usually structured as programs conducted for periods of between five and seven years. Processes to measure and report resource condition change and the performance of NRM interventions therefore need to encompass a range of time-scales.

Under Review

Characteristics of indicators

Indicators in the national framework should be:

- simple (easily interpreted and monitored),
- measurable (statistically verifiable, reproducible and show trends);
- accessible (regularly monitored, cost effective and consistent),
- relevant (directly address the objectives of the NRMMC), and
- timely (provide early warning of potential problems).

Relevant indices for integrating individual indicators

To assist in understanding the information provided by indicators it is possible to aggregate sets of indicators to support indices such as Index of Catchment Condition developed by the NLWRA or the Index of Stream Condition developed in Victoria. While indices provide a summative profile of a particular condition, specific measures are also needed to provide key statements of individual critical elements. Indices are of limited value in accurately reflecting the condition of some aspects of the resource.

Need for consistent approach to monitoring, measuring and reporting

To be cost effective and practical, information needs to be used at a number of levels. This will require data to be aggregated from regional to national levels. This can be done in two ways by:

1. aggregating data;

2. determining performance at a local scale and aggregating performance measures.

Both methods of aggregation will need to be used. The choice will depend on the nature of the objective being considered.

The first approach requires the use of consistent methodologies. The NLWRA reports on dryland salinity and water resources provide evidence of the inconsistencies in methodologies used and data collected between jurisdictions. Where possible, the adoption of agreed national standards, such as the National Water Quality Management Strategy Water Quality and Monitoring Guidelines, would greatly facilitate the use of indicators. The MWG could review the existence of agreed national standards and provide support for their use. For example, the recently released Evaluation Framework for Dryland Salinity from the NLWRA could be adopted nationally.

The second approach is less exacting in its requirements for consistency and different types of monitoring may be used to suit different local conditions. For example, different catchments might use different methods to monitor performance with respect to their own water quality objectives and the high level indicator might be the number of catchments meeting their water quality objectives.

Under Review

Data sources

Relevant data for indicators are available from a range of State and Commonwealth sources. For example, the National Land and Water Resources Audit (NLWRA) and SoE have recently collated large amounts of information on resource condition.

Need for consistent approach to data handling and management

To ensure that data can be aggregated to regional or national scales requires the use of consistent methodologies and protocols for data exchange and management. Considerable effort has already been put into developing consistent methodologies for data management by the Australian and New Zealand Land Information Council (ANZLIC) and the National Land and Water Resources Audit (NLWRA). Adoption of these methodologies would be a major step forward. The development of agreed national custodianship arrangements for core data sets will be very important for developing an information base for natural resource management, which will assist in the implementation of core indicators. Work on achieving this is underway through initiatives such as the development of the proposed Executive Steering Committee on Land Use that will identify the fundamental land use datasets for natural resource management.

Data quality

The data quality requirements of the framework must be established to support meaningful interpretation over time. Where comparison or aggregation of data is required, common data specifications should be established to ensure that the data collection processes support the purposes of each aspect of the framework. To maximise their use and comparability, data should be developed and maintained to meet agreed international or national guidelines or standards for the management of spatial information as endorsed by ANZLIC or through national coordination arrangements.

Cost and efficiency

Establishing and maintaining monitoring programs are costly processes. To minimise the costs of developing the data infrastructure required to support the framework:

- Data should be collected once with the aim of using it to support many activities. For example, use a common set of data to support the many regional, national and international reporting obligations.
- Existing State, Territory and Commonwealth initiatives for developing and sharing of data. Should be built on to avoid duplication of effort, and to maximise the benefits from earlier investment in data collection.
- To ensure that users can obtain the data, data must be easily accessible to all sectors of the community in format, location, cost and under conditions that do not inhibit their use.
- To ensure that users can easily find out whether suitable data already exist, all data must be documented in the Australian Spatial Data Directory. The documentation must be current and provide enough information for users to determine whether the data are suitable for their purpose.

National S&T Framework

The National Framework for Natural Resource Management Standards and Targets (National S&T Framework) sets out consistent national directions and approaches to natural resource planning, target setting, monitoring and reporting. The statements of desired national natural resource outcomes are aspirational statements about desired national natural resource outcomes. They are expressed in a manner that allows an assessment of progress towards these outcomes to be made. While these outcomes are largely focussed on resource condition, they also include objectives related to changing land and water management systems and practices.

The Framework also identifies a minimum set of matters for which catchment and regional communities will need to identify specific timebound and measurable targets for each region, which will move the natural resource condition towards the achievement of the national outcomes. These targets are characterised as achievable resource condition targets (10-20 years) and are required for each matter in the minimum set, and management action targets (1-5 years) which would be implemented to achieve the resource condition outcomes and would vary across regions.

National protocols for monitoring and reporting of progress against targets will be identified to:

- promote consistency in setting and measuring progress towards targets within and across regions;
- allow aggregation and reporting on progress nationally;
- allow comparison of program achievements with national assessments of condition or trends in resource condition (eg NLWRA, SoE reporting); and
- enable feedback to regions on how they are contributing to achieving national outcomes.

Under Review

For the National Action Plan on Salinity and Water Quality (NAPSWQ), the National S&T Framework has initially identified national outcomes and a minimum set of matters for salinity and water quality/flows, and is currently developing national outcomes and matters for targets for in stream and terrestrial biodiversity. Additional work is also being undertaken to identify national outcomes and related measures for the extension of the Natural Heritage Trust.

Benefits and Issues

Adoption of the Standards and Targets Framework approach will most likely utilise indicators that are common to both the national resource condition monitoring and related performance monitoring of programs, strategies and policies. These common measures may support higher level analysis of the impact made by particular programs, strategies and policies on specific aspects of the national resource.

Further consideration will need to be given to the process for identifying outcomes and measures related to issues not related to the National S&T Framework, particularly the community and social processes relevant to or affected by NRM programs, and the adoption of sustainable development and production techniques.

Under Review

Process for identifying indicators

The development of these indicators has built on an existing process by using the National S&T Framework to identify natural resource outcomes and associated matters suitable for monitoring the trend on the condition of the nation's natural resource and the performance of programs, strategies and policies. A draft preliminary list of indicators is shown overleaf.

RECOMMENDED INDICATORS

Table 1. List of Indicators for Resource Condition Matters for Target developed to date

Matter for Target	Indicator Heading	Recommended indicators
Land Salinity	Area of land threatened by shallow or rising water tables	• Depth to groundwater • Groundwater salinity • Location and size of salt affected areas
Soil Condition	Soil condition	**For regionally significant soil condition issues that are the subject of targets in regional plans:** • Soil acidification • Soil erosion – water • Soil erosion – wind • Soil carbon content • Soil physical condition • Surface hydrophobicity • Soil biota activity
Native Vegetation	Native vegetation extent and distribution	**For regionally significant native vegetation that is the subject of targets in regional plans:** • The extent of native vegetation by IBRA subregion measured in hectares. • The extent of each present native vegetation type by IBRA subregion measured in hectares. • The proportion remaining of each native vegetation type by IBRA subregion measured as a percentage of the pre-European extent.
	Native vegetation condition	**For regionally significant native vegetation types that are the subject of targets in regional plans:** • The proportion of each native vegetation type in each IBRA subregion that is estimated to be in specified condition classes based on a selected set of attributes.

Under review

Matter for Target	Indicator Heading	Recommended indicators
Inland Aquatic Ecosystems Integrity	River condition	**For regionally significant reach based issues that is the subject of targets in regional plans, the indicators are:** • Benthic macroinvertebrate community assemblages • Fish community Assemblages • Benthic diatom community assemblages • Riparian vegetation community assemblages • Riverine physical structure and in-stream habitat • Water quality • Hydrology If all or most of these indicators are measured, it may be possible to use monitoring data to develop an index of river condition.
	Wetland ecosystem extent and distribution	• Extent of regionally significant wetlands
	Wetland ecosystem condition	• Condition of regionally significant wetlands
Estuarine, coastal and marine habitat integrity	Estuarine, coastal and marine habitat extent and distribution	• Area of each estuarine, coastal and marine habitat type measured in hectares
	Estuarine, coastal and marine habitat condition	• Condition of habitat at significant sites of selected estuarine, coastal and marine habitats
Nutrients in Aquatic Environments	Nitrogen in aquatic environments	• Total Nitrogen + flow leaving sub-catchment or whole catchment
	Phosphorus in aquatic environments	• Total Phosphorus + flow leaving sub-catchment or whole catchment
Turbidity / suspended particulate matter in aquatic environments	Turbidity / suspended solids	• Turbidity OR • Total Suspended Solids (TSS) + Flow
Surface Water Salinity in freshwater aquatic environments	In-stream salinity	• Total dissolved solids (TDS) + Flow OR • Electrical conductivity (EC) + Flow

Under Review

Matter for Target	Indicator Heading	Recommended indicators
Significant native species and ecological communities	Selected significant native species and ecological communities extent and conservation status	For significant species that are the subject of targets in regional plans: • Range area and location of each species: area • Area, location and condition of key habitat of each species: • Relative abundance of each species. For significant ecological communities that are the subject of targets in regional plans • extent of each ecological community: estimated area (in hectares) • condition of each ecological community.
Ecologically significant invasive species	Selected ecologically significant marine invasive species extent and impact	• Presence/absence of known or new: - exotic marine species and - native pest marine species in commercial ports and boat harbours.
	Selected ecologically significant vertebrate invasive species extent and impact	• Reduction in impact of regionally significant invasive vertebrate pests (excluding fish)
	Selected ecologically significant invasive vegetation species extent and impact	• The areal extent and density of weeds under selected regulatory control that are being addressed by regional bodies or community projects

Under Review

Indicators for Management Action Matters for Target

Management Action Matter for Target	Recommended indicators
Critical assets identified and protected	• The extent to which assets are identified during the planning process • Steps taken to protect assets identified in the plan
Water Allocation Plans developed and implemented	• Water allocation plans developed and implemented in accordance with relevant State or Territory legislation
Improved land and water management practices adopted	• Adoption of codes of practices or recommended practices that are identified in the regional plan

Indicators for Social and Economic Issues

- Effectiveness of information networks
- Youth net migration
- Index of Economic Diversity
- Median income and income distribution
- Farm cash income
- Farm family off-farm income
- Farm Debt-equity ratio
- Education
- Age and experience
- Population Growth

Under Review

To ensure the quality of the data produced and ensure that data can be used for their intended purposes, data protocols for the sampling, measurement and interpretation of these indicators will be developed. The proposed structure of these protocols is as follows:

Proposed Structure of Data Protocol

Indicator name eg Depth to Groundwater

1. **Definition**
   depth to groundwater in metres from the ground surface.

2. **Rationale**
   2.1.1 Monitoring location selection (scale)
   2.1.2 Why do we want to know it? ie. rationale for measuring it
   2.1.3 Context in which its been measured with regard to national, state and regional resource management programs.

3. **Monitoring Methodology**
   3.1 Monitoring location selection (scale)
   3.2 Monitoring frequency required
   3.3 Data measurement method
   3.4 Data collation/calculation method
   3.5 Data analysis and interpretation
   3.6 Robustness or quality assurance

4. **Reporting Products**

5. **Current Monitoring and Reporting Products**
   5.1 Monitoring location selection (scale)
   5.2 Monitoring frequency required
   5.3 Data measurement method
   5.4 Data collation/calculation method
   5.5 Data analysis and interpretation
   5.6 Robustness or quality assurance

6. **Proposed Responsibilities**
   5.1 Data collection (ie 3.1-3.3)
   5.2 Data collation (ie 3.4)
   5.3 Data analysis and interpretation (ie 3.5)
   5.4 Generation of reporting products (ie 4)
   5.5 Data storage and management
   5.5.1 sub-regional/sub-catchment data collection and collation
   5.5.2 data analysis and interpretation information
   5.5.3 reporting products

7. **Future development**

8. **Links to other indicators**

9. **Further information**

Selection of further indicators

This will occur in conjunction with development of the National Standards and Targets Framework as further matters for standards and targets are identified.

Under Review

GLOSSARY

Monitoring and Evaluation Framework
A monitoring and evaluation framework structures monitoring and evaluation activities to provide accurate, cost-effective and timely performance information for management decision making. Monitoring and evaluation activities use many of the same data sources, and complement one another in covering the full range of issues affecting the performance of an activity. Because of the differences between the two activities, each requires different management structures and processes.

Performance Information can be quantitative or qualitative. Quantitative performance information provides measures in respect of specific indicators of performance towards achieving a target or goal. Qualitative performance information is descriptive, and can include stories and case studies. Qualitative and quantitative performance information should generally be employed together, as quantitative measures usually require some interpretation and explanation to ensure they are properly understood.

Monitoring activities provide *indicative* information to track and review the performance of policies, strategies and programs at regular intervals to inform management decision-making. This indicative information must directly relate to the expected outcomes and outputs of the policy, strategy or program.

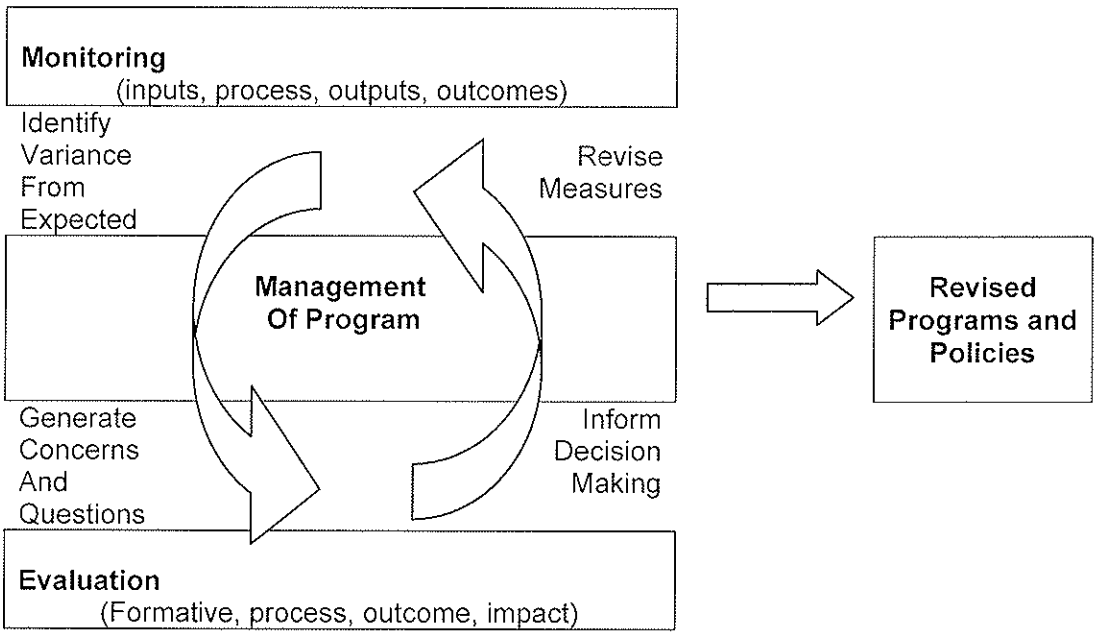
Monitoring is defined as the regular gathering and preliminary analysis of information needed for day-to-day management or evaluation. Monitoring activities provide indicative information to track and review the performance of policies, strategies and programs at regular intervals to inform management decision-making. This indicative information must directly relate to the expected outcomes and outputs of the policy, strategy or program. Monitoring provides information on what is occurring and what the program, policy or strategy is achieving.

Evaluation activities provide comprehensive *analysis* to determine how well, and in what ways the policies, strategies and programs are working. They include an assessment of efficiency (determining whether existing delivery arrangements can be improved) and effectiveness (determining whether or not the program strategies are having the desired effect) and are used to support management decision making at appropriate points in the life of the policy, strategy or program. The information used in these activities may be both directly and indirectly related to the expected outcomes and outputs of the policy, strategy or program. Evaluations may need to consider:

- unforeseen consequences, particularly social and economic impacts;
- external factors impacting on program delivery and likely achievement of outcomes (including social and economic factors)
- attribution - the extent that program strategies contribute to, or are responsible for, achievement of desired outcomes.

Accountability - the Parliament and the public need to be confident that program funds are being spent on actions that will make positive changes to Australia's environment.

Adaptive management enables lessons learned to be realised during the life of the program and to make necessary adjustments in response. Adaptive management utilises monitoring and evaluation activities to form a feedback loop in order to make necessary adjustments to the program. Monitoring and evaluation activities will use many of the same data sources, and complement one another in covering the full range of issues affecting program performance for consideration in management decision making. Regular collection of performance information provides a basis for identifying needs and opportunities for evaluation and policy review. For instance, information from monitoring data could be a trigger for further action or investigation, and agreements on the trigger mechanisms should be included within the program monitoring strategy. Evaluation serves as a powerful tool for reviewing the appropriateness of performance indicators and identifying gaps in the performance information structure. Evaluations will rely on, and therefore should be responsive to, monitoring information. For example, unexpected or undesired results from monitoring data might allow triggering of unscheduled evaluations. These linkages are represented in the following diagram.



Adapted from De Boer M. *Monitoring: the slow cousin of evaluation or an equal partner* Proceedings from the Australasian Evaluation Society Conference 2001

MMSD

Mining, Minerals and
Sustainable Development Project

Ameef.

MANAGING THE IMPACTS OF THE AUSTRALIAN MINERALS INDUSTRY ON BIODIVERSITY

Final Report

M.V. Lloyd, G. Barnett, M.D. Doherty, R.A. Jeffree, J. John, J.D. Majer, J.M. Osborne, O.G. Nichols

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EXECUTIVE SUMMARY

Biodiversity can be described as 'the variety of all life forms – the different plants, animals and microorganisms, the genes they contain and the ecosystems of which they form a part'. This report, which has been prepared by the Australian Centre for Mining Environmental Research (ACMER), as part of the Australian regional Mining, Minerals and Sustainable Development (MMSD) project, addresses (1) the impact of the minerals industry on biodiversity, (2) how the modern minerals industry manages its impact on biodiversity, (3) the potential for enhancement of industry practices to protect biodiversity, and (4) the opportunities for indigenous communities to assist the minerals industry in managing biodiversity.

Australia has 80 biogeographic regions containing a large number of endemic species, and is one of the most mega-diverse countries in the world. A number of threats to biodiversity by all forms of land use have been identified, namely habitat modification and fragmentation, overexploitation of species, impact of introduced species and genes, and the pollution of soil, water and the atmosphere.

Surveys have shown that the minerals industry affects less than 0.05% of the Australian terrestrial land surface, and it has not been responsible for the extinction of any known species (with possibly one exception). Rather, it is the agricultural and grazing industries that occupy the greatest amount of developed land area (greater than one-half), and these industries have caused the most adverse effects on biodiversity, with a total of 44 attributed extinctions.

Whereas the national impact on biodiversity by the industry has been low, past practices have been responsible for some severe local impacts on both terrestrial and aquatic biodiversity. These impacts, which have arisen from exploration, mining and mineral processing activities, have involved direct clearance of vegetation (and faunal habitat), on- and off-site impacts of acid mine drainage from waste rock dumps, tailings storage facilities and open pits, and poor rehabilitation practices.

The industry now recognises the potential for impact on both terrestrial and aquatic biodiversity and has developed measures for managing this impact through both in-house and commissioned research undertaken for environmental impact assessments for mining proposals and for rehabilitation of post-mine environments. As a result, the

industry may be credited with contributing a substantial amount of knowledge on Australia's biodiversity. Most of Australia's mineral reserves are located in remote locations, away from major population centres. Mineral reserves are concentrated on the basis of geology, rather than ecology or archaeology, and therefore are not necessarily in areas that would have received thorough biological studies without the presence of mining.

In certain fields of rehabilitation expertise, such as native ecosystem reconstruction, the minerals industry in Australia is seen as a leader in development of new technologies. With the continuing globalisation of the industry, these technologies may become useful in the other countries with similar climatic ranges.

Means by which the industry has the potential to enhance its protection of diversity include (1) developing a greater understanding of the ecology of reconstructed landscapes, (2) identifying, and obtaining agreement on, indicators of rehabilitation success, (3) improving the utilisation of native seed reserves, and (4) improving fire and weed management techniques.

The minerals industry is increasingly recognising the role that indigenous communities can play in the management of biodiversity. The knowledge of these indigenous communities of native flora is being utilised in (1) identifying the species important to these communities and thus the post-mine vegetation mix, and (2) developing contracts with communities to collect native seed for mine rehabilitation programs.

Through the development of environmentally sensitive practices and ongoing research and monitoring, the minerals industry has, for the most, demonstrated it to be a responsible land manager, which is aware of the temporary nature of its activities and its commitment to future land users. A number of case studies, both positive and negative, have been presented in this report to illustrate the improved awareness of biodiversity issues by the minerals industry.

1.0 INTRODUCTION

1.1 What is Biodiversity?

The term 'biodiversity' has come to encapsulate the vast varieties of life forms and their physical, chemical and biotic environments. There are numerous definitions that are associated with the term, of which one of the most accepted within Australia is the definition that was adopted in the 1996 Australia State of the Environment Report:

'*The variety of all life forms – the different plants, animals and microorganisms, the genes they contain and the ecosystems of which they form a part*'.

The broad nature of this and other definitions of biodiversity are the source of considerable scientific debate, as they have essentially become a synonym of 'all life', hence rendering the concept extremely difficult to operationalise (Doherty *et al.* 2000). Nevertheless, for the purpose of this report, the definition provided above will be used.

1.2 Biodiversity and Conservation in Australia

The Australian continent extends approximately 3,700 km from the south latitudes of 10° to 43° and contains a number of climatic zones from the tropical north, through the dry, arid interior to the southern temperate areas and alpine regions in the Australian Alps. A geologically stable land mass has resulted in highly weathered soils and few mountain ranges. The relatively flat landscape presents few opportunities for orographic rainfall, hence there is a scarcity of fresh water in inland Australia. Australia comprises slightly over 5% of the world's land area, but its proportion of surface runoff is less that 1% (Smith 1998) with the average annual runoff for Australia being 52 mm. On this basis, Australia may be considered the driest inhabited continent in the world.

The majority of Australia is characterised by unpredictable rainfall and vast seasonal variations in evaporation rates, especially in the extensive arid and semi-arid regions. The arid landscapes are dotted with fresh, brackish and hypersaline lakes encrusted with salt crystals with many inundated for only a few weeks annually. The area of saline wetlands in Australia far exceeds that occupied by freshwater (Mummery and Hardy, 1994). In addition, high variability of flow has caused a large number of

intermittent streams and lakes. The ephemeral nature of a large number of inland waterbodies (including streams and rivers), in turn, has helped to shape the unique survival strategies of Australia's flora and fauna.

Australia has 80 biogeographic regions containing a large number of endemic species (species that can be found only in this country). As a result, the International Union for the Conservation of Nature (IUCN) lists Australia among the mega diverse countries in the world. The degree of knowledge about biodiversity varies with geographical regions, ecosystems and taxa (Lovejoy 1997). The following are all currently investigated under the broad domain of biodiversity: inventory (cataloguing) at the level of taxonomic groups, genetic origin, evolution and dispersion of species; economic, ecological and indicator values of taxa; bio-invasion of exotic species; stress, fragmentation, destruction, and rehabilitation of habitats and ecosystems.

A number of threats to biodiversity in Australia have been identified, namely, habitat modification and fragmentation, overexploitation of species, the impact of introduced species or genes, and pollution of soil, water and the atmosphere (BDAC 1992).

The 1996 State of the Environment Report highlights Australia's loss of biodiversity and lists a number of key threats to sustainability. The single largest threat to biodiversity is reported to be land clearing and related activities, the extent of which, both previous and current, varies greatly between states. Currently, Australia experiences an annual net loss of native vegetation with an estimated average to be 300,000 to 340,000 hectares per year (NHT Midterm Review 1999). Many of our native ecosystems are now fragmented and in danger of becoming irreversibly degraded as clearing is compounded by more insidious, widespread, incremental losses of native vegetation. The majority of land clearing activities has been undertaken for agricultural purposes. In the State of Queensland over the period 1991-1999, 95.3% of clearing was undertaken for agriculture with only 0.6% of land clearing attributable to mining over the same period (SLATS 2000).

Since the 1996 State of the Environment Report was released, a number of Commonwealth initiatives have been established to address the increasing areas of degraded lands and establish a system of protected areas to conserve environmentally and culturally sensitive environments.

Australia is a participant in a number of international agreements and has developed national and State legislation that require an attempt to conserve biodiversity and minimise the effects of activities on 'sensitive' environments. The definition of sensitive environments in this context may include: proposed and current national parks, nature reserves, conservation parks, State forests, timber reserves, rainforest areas, areas of declared rare flora and fauna, mangrove communities and wetlands (Department of Minerals and Energy WA 1998), areas with slow rates of recovery due to climatic variables (alpine and arid zones) and culturally significant areas.

As signatories to the National Strategy for the Conservation of Australia's Biological Diversity (NSCABD), Commonwealth and State governments are committed to the establishment of a comprehensive and representative system of ecologically viable conservation areas. For conservation of biodiversity, the IUCN recommends protecting a minimum of 10 percent of natural areas worldwide. Currently, Australia has 60,273,030 ha of terrestrial lands on the mainland (including Tasmania) that are managed for nature conservation. This represents 7.84% of the Australian continent (including Tasmania).

1.3 Overview of the Minerals Industry in Australia

Australia has the world's largest reserves of several key mineral commodities (Australia's Identified Mineral Resources 2001) with the minerals industry contributing approximately $40 billion annually to the Australian economy (Commonwealth Minister for Environment and Heritage 2000). The minerals industry in Australia is taken to include the process of removing the targeted resource from the ground (mining) plus the associated minerals processing activities. The industry directly employs more than 80,000 people in the minerals, oil and gas industries and contributes more than 300,000 indirect jobs in the manufacture of downstream products (Commonwealth Minister for Environment and Heritage 2000).

Historically, environmental protection was relegated behind economic development on the list of national priorities. With scant knowledge of environmental protection measures and the consequences of incorrect disposal of wastes, the majority of the early development projects focussed on production with little regard for biodiversity and ecology. The activities of the minerals industry reflected the views of the time and left a legacy of unrehabilitated waste dumps, polluted waterways and deforested

hillsides. The Australian government has spent significant sums of money in correcting the environmental damage that has resulted from historical activities, such as Rum Jungle and Captains Flat, which remain a burden to the mineral industry's attempts to improve its environmental image (Brooks 1997).

Today, Australia is at the leading edge of rehabilitation technologies, especially mine-site rehabilitation (Beattie 1995). In 1996-97, the minerals industry spent $369m on environmental protection, primarily in waste water management and water protection as well as the protection of biodiversity and landscape ecosystems (Astolfi *et al.* 2000) yet the industry continues to have a negative environmental image in the community (Commonwealth Minister for Environment and Heritage 2000; Hancock and O'Neill 1996). The reality is that mining affects less than 0.05% of the terrestrial land surface (Bell 2001) and has significantly contributed to the knowledge and understanding of Australia's biodiversity.

With the exception of mining occurring on agricultural lands where the community wants the post-mined land use as improved pastures, the majority of mines place an emphasis on establishing native ecosystems after mining. As a result, there has been considerable effort placed on the development of technologies to ensure successful establishment and stability of these ecosystems often under adverse environmental conditions (Bell 2001). With many of our native ecosystems now fragmented and in danger of becoming severely degraded, unmined lands managed within mineral leases are frequently in a more pristine condition than the surrounding communities adjacent to the lease due to management of degradative factors.

Severe environmental pollution caused by mining operations within Australia, as evident by the denuded hillsides of Queenstown in Tasmania, and Australian offshore operations such as the Ok Tedi mine in Papua New Guinea and the Esmeralda Exploration cyanide spill in Romania, impact on the public image of the Australian minerals industry as a whole. With access to viable mineral deposits requiring approval of stakeholders and being dependent on the minerals industry indicating the potential for multiple land use, the continued activities of the Australian minerals industry will be increasingly determined by its environmental performance.

In 1996, the Minerals Council of Australia (MCA) launched the Australian Minerals Industry Code for Environmental Management (the Code). The primary purpose of

the Code was to provide a framework for continual improvement in environmental management and communication in each phase of mineral development, from initial exploration to closure and final rehabilitation (MCA 1999).

According to Hamblin (1998), the highly organised and commercial nature of the industry means that voluntary compliance by large mining operations is a viable option for gaining adoption of the Code by most sections of the industry. Participation in the Code is intended to facilitate the signatory's improvement of environmental performance and to communicate the performance to stakeholders and the community. The Code requires each signatory to produce annual environmental reports to verify implementation of the Code and to describe their commitment to excellence in environmental management. In 1999, the Code underwent a major review in consultation with key industry stakeholders. An updated 2000 Code was the outcome of this review process. Key improvements include greater emphasis on the importance of verification of Code performance and the introduction of a self-assessment protocol (the Code Implementation Survey) with the purpose of providing an industry-wide standard for gauging progress towards Code implementation.

As of October 2001, 41 companies within Australia have adopted the Code of Environmental Management representing approximately 90% of production within the Australian minerals industry.

2.0 IMPACT OF THE MINERALS INDUSTRY ON BIODIVERSITY

All recognise the value of mineral wealth, but the concept of biological wealth has been slower to be accepted (Beattie 1992). The activities of the minerals industry could potentially pose some major threats to biodiversity. Mining removes all of the biota within the active area and therefore halts the associated ecosystem functions and processes. The industry, however, has come a long way since its historical activities, and systems are now in place to ensure that steps are taken to restore biodiversity wherever possible. For example, prior to mining in Western Australia, the Departments of Minerals and Energy, and Conservation and Land Management review the resource potential and biodiversity values of pastoral leases including those where exploration and mining leases are taken out (Smurthwaite *et al.* 2000).

Following mining, the Department of Minerals and Energy and the Department of Environmental Protection ensure suitable closure techniques have been implemented.

The minerals industry is not the only one with impacts on biodiversity, and certainly it is not the industry affecting the largest area. The State of Western Australia occupies one third of the country and is the State where the minerals industry is most active. Figure 2.1 indicates that the industry occupies the least area of any of the major land uses in Western Australia, in fact occupying less than 1 percent of the land area. The area available for exploration and mining has been reduced due to their exclusion from National Parks and conservation reserves where they are no longer permitted (Trudinger and Pool 2000). By contrast, the agricultural industry, and particularly pastoral land use, affects a vast area in Western Australia, having its own impacts on the biodiversity of the land. Mining's major impacts result from clearing, excavating and waste disposal. These practices can cause erosion and soil damage, air pollution, contamination, have adverse affects on the groundwater such as salinisation and acidification, cause loss of flora, fauna and habitat and introduce disease and pests (Knight 1998).

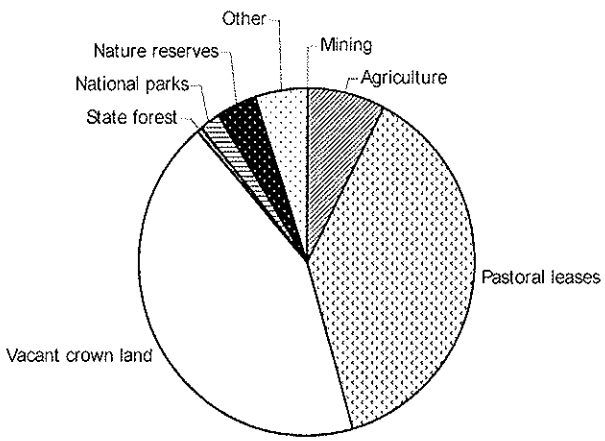


Figure 2.1. The percentage of land use in Western Australia (after Guy *et al.* 1995 and Grant 1992).

All major industries can have considerable impact on the biodiversity of Australia, particularly in the fragile arid zones, where ecosystems are already operating close to

their physiological limits (Smith 1992). Contrary to popular belief, the minerals industry is not Australia's major environmental challenge (Batini 1997), and in fact the industry is one of the leaders in ensuring that its impact on biodiversity is minimal (Anderson 2000).

There are several reviews of the impact of the minerals industry on the Australian environment (e.g. Webb 1990; Woodside and O'Neill 1995; Farrell and Kratzing 1996; and White 1997). However, with the possible exception of Woodside and O'Neill (1995), none of these reviews provide a comprehensive account of the impact of the industry on biodiversity. The following attempts to look at the positive and negative effects of the industry in relation to biodiversity.

2.1 Establishing an Ecological Context

Saunders *et al.* (1996) provide a useful review of the various processes that are threatening Australia's biodiversity. Their review clearly shows that agricultural and grazing land uses have had a far greater impact on Australia's biodiversity than the minerals industry. For instance, with regard to pressures on plant biodiversity, agriculture has been identified as the major cause of 44 presumed species extinctions as opposed to only the one attributed to the minerals industry. The pattern is similar for our faunal diversity.

Utilising ants as a measure of biodiversity integrity, Majer and Beeston (1996) calculated the relative impact of various land uses in Western Australia. In their study, agricultural clearing has had by far the greatest impact on biodiversity integrity, closely followed by rangeland grazing. The lower losses of biodiversity integrity as a result of urbanisation, roads and mining puts into context the lower impact of these relatively restricted land uses.

Agricultural and grazing land uses occupy more than half of Australia's land area. When compared to the area that is occupied by the minerals industry, it is not surprising that agricultural and grazing land uses have had a far greater impact on Australia's biodiversity. However, it is important to recognise that while the area of land affected by the minerals industry is relatively small, the magnitude of the impact can be locally significant. Mining activities often have a very rapid and devastating impact on biodiversity through the removal of vegetation and the alteration of soil profiles, topography and hydrological regimes. On the other hand, agricultural and

grazing impacts are less immediate and are often cumulative. It is plausible that Australia's previous mining legacy is the cause of only the one species thought to have become extinct, due to the relatively small area impacted by the activities rather than sound biodiversity management and rehabilitation techniques which did not exist prior to the 1970s.

2.1.1 Spatial analysis of biodiversity and mining disturbances

Due to the lack of information on mining disturbances and their impact on biodiversity, particularly at the national or industry-wide scale, a rapid assessment of the spatial analysis of biodiversity and the activity of the minerals industry was undertaken for this review using basic Geographic Information System (GIS) procedures and two readily available and relevant spatial datasets:

1. the Interim Biogeographic Regionalisation for Australia (IBRA) dataset that divides the continent into 80 biogeographic regions, which can be thought of as major environmental units (Thackway and Cresswell 1995); and,
2. the spatial representation of the mineral deposits database OZMIN developed by G. Ewers and B. Kilgour of the Australian Geological Survey Organisation (AGSO).

The results of this rapid assessment can be seen in Figure 2.2. Remembering that the individual operations have been enlarged for the purpose of this exercise, Figure 2.2 indicates that the minerals industry has impacted on the large majority (66) of the 80 different biogeographical regions identified by Thackway and Cresswell (1995). This is not surprising given the widely scattered nature of Australia's mineral resources.

Fourteen biogeographic regions were found to have no significant historical or current mining activity (Figure 2.3), and thus it can be assumed that the minerals industry has had little, if any, impact on the biodiversity that these biogeographic regions comprise. Six of these 14, however, contain significant mineral deposits, which may be the target for future mining activity. The Murray-Darling Depression biogeographic region is one of these six and is already experiencing considerable

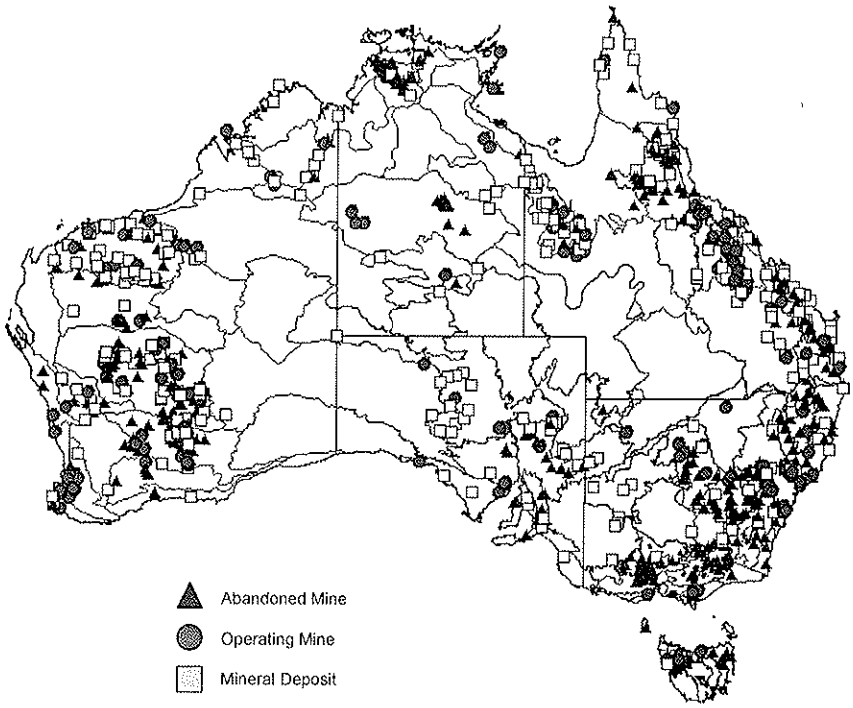


Figure 2.2. Spatial representation of IBRA and the mineral deposits database OZMIN.

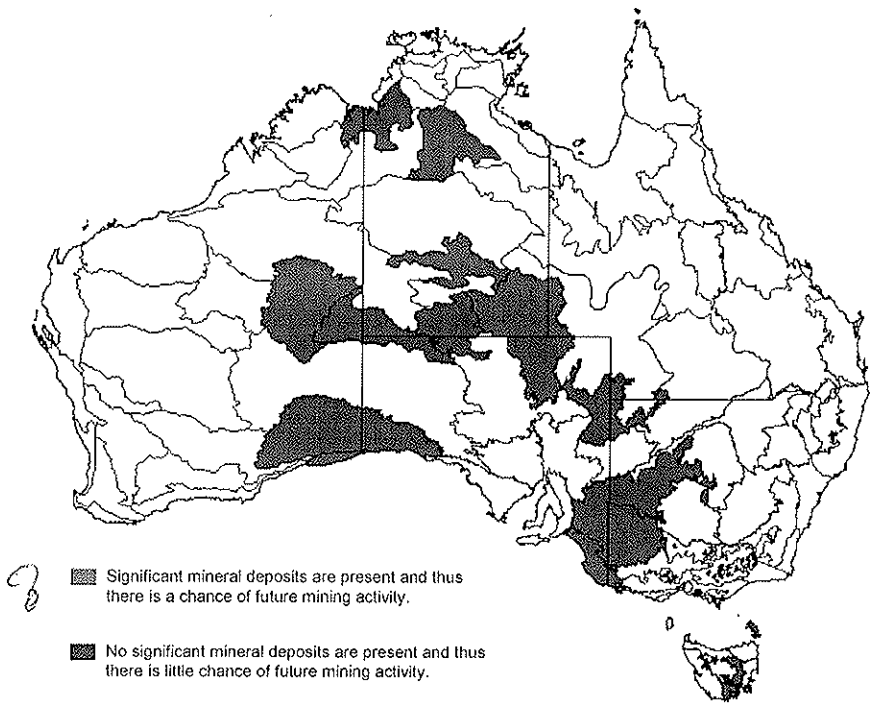


Figure 2.3. Biogeographic regions with no significant historical or current mining operations.

exploration activity and interest as a more accessible source of mineral sands than traditional coastal resources.

2.1.2 Influence on threatened biogeographic regions

Loss of biodiversity is often a result of multiple, interacting pressures (EPA QLD 1999). For this reason, it is important that the minerals industry is aware of those regions that have experienced substantial impacts on their biodiversity from other land uses. The IBRA dataset contains a qualitative and subjective assessment of the condition of each biogeographic region.

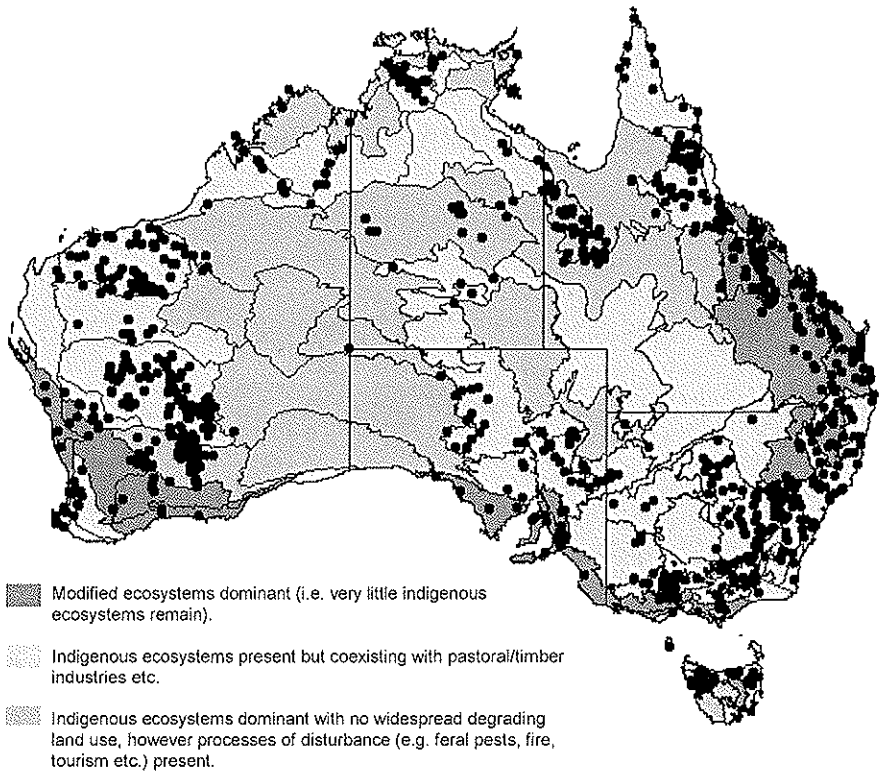


Figure 2.4. Condition of IBRA Regions and distribution of OZMIN data.

The results suggest that the most significant activity of the minerals industry in those biogeographic regions dominated by modified ecosystems (i.e. very little indigenous ecosystems remain) was in Queensland (Figure 2.4), in particular, Brigalow Belt

South, Brigalow Belt North, and South East Queensland biogeographic regions, which all have more than 60 percent of their vegetation cleared and many serious threats to biodiversity.

At the other end of the spectrum, the minerals industry has significant activity in two regions – Coolgardie and Mt Isa Inlier – where indigenous ecosystems are still dominant, and there are no widespread degrading land uses. The industry needs to guard against complacency in respect to assessing and managing impacts in these areas, because, despite the expanse of indigenous ecosystems, development can provide a conduit to other more insidious threatening processes, such as increased predation by feral animals and competitive interactions caused by introduced weeds.

2.1.3 Influence on poorly conserved biogeographic regions

Another indication of whether the minerals industry is potentially impacting on biodiversity is the degree to which the industry's activities are focussed in biogeographic regions where there is considerable bias in the existing system of protected areas.

As noted by Thackway and Cressell (1995), much of our existing system of protected areas (e.g. national parks and nature reserves) is biased toward certain environmental elements – high relief, low soil fertility, and steep rainfall gradients. Areas that are highly regarded for agriculture or other resource consumptive land uses are typically under-represented in Australia's existing system of protected areas.

Some biogeographic regions, in which the minerals industry is currently active, are poorly conserved (Figure 2.5). For instance, the Pilbara and Murchison biogeographic regions both have a high index of bias, whereby entire sub-regions, or many of the extensive ecosystems that characterise the region, are not represented in the existing system of protected areas. The minerals industry needs to acknowledge this bias in protected areas and ensure that its activities do not impact on the representativeness, comprehensiveness and adequacy of biodiversity conservation in the regions where it is operating.

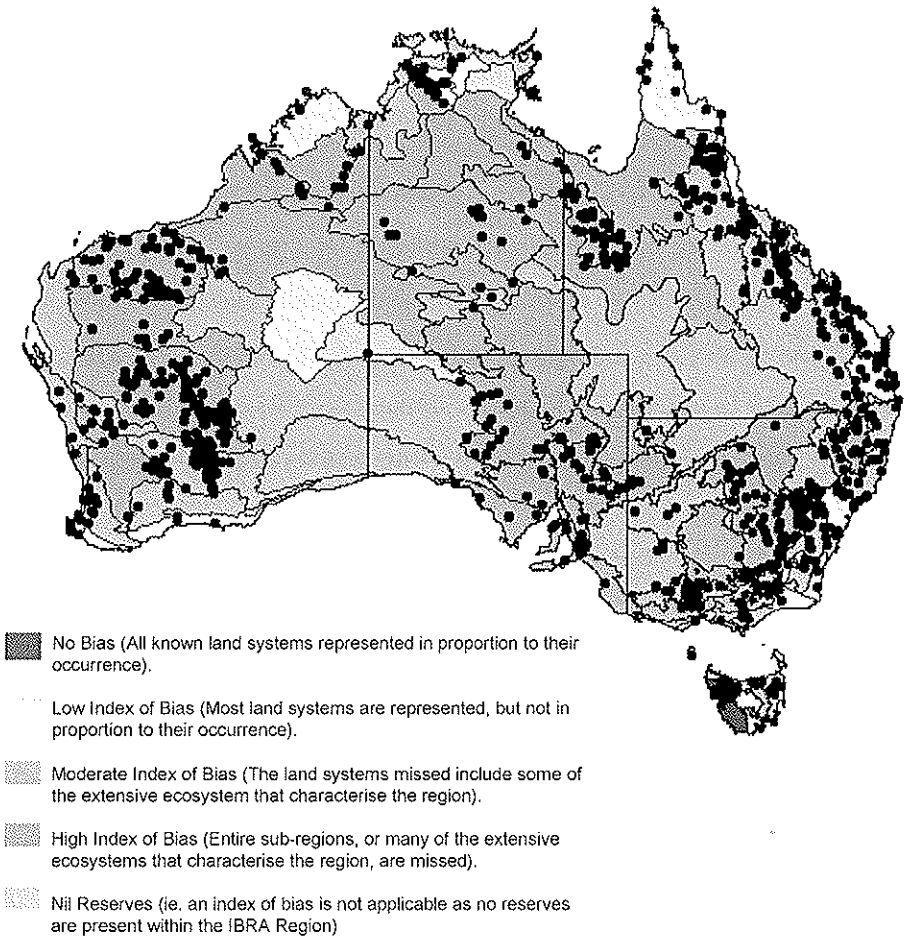


Figure 2.5. Distribution of OZMIN data and the level of bias within protected areas.

2.1.4 Influence on Australia's flora and fauna

There are few simple cause-and-effect relationships linking biodiversity loss or decline with a particular human activity (EPA QLD 1999). However, the Commonwealth Government through Environment Australia and the Australian Nature Conservation Agency, commissioned several 'Action Plans' for various fauna groups over the past decade which provide an indication of the impact the minerals industry has had on Australia's fauna. Similar information has been prepared for Australia's flora (e.g. Leigh and Briggs 1992). This information is summarised in Table 2.1.

Table 2.1 indicates that one species of plant is now presumed extinct due to the activity of the minerals industry, while 11 flora and fauna species were considered endangered by a historical threat, and a further 35 by current and future threats. However, when interpreting this information, it is important to realise that there is a certain degree of subjectivity and uncertainty regarding the extent to which the industry has played a part in the demise or decline of these species.

The plant species presumed extinct is *Euphorbia carissoides* (Bailey), which occurred in open-woodland habitat of Queensland. The causes of its extinction have been attributed to mining and domestic grazing (Leigh and Briggs 1992). This evidence appears to contradict the statement made by Woodside and O'Neill (1995) that:

' ... there is no data to suggest that any plant, animal or community has gone extinct in Australia as a result of mining.'

Table 2.1. The impact of the Australian minerals industry on the threatened status of the native biodiversity.

Lifeform	Number of presumed extinct species	Number of endangered species	
		Historical threat	Current and future threat
Plants	1	3	11
Birds		8	4
Frogs			1
Reptiles			6
Freshwater Fish			
Marsupials and Monotremes			4
Bats			9
TOTAL	1	11	35

Source: Leigh and Briggs (1992), Garnett and Crowley (2000), Tyler (1997), Cogger *et al.* (1993), Wager and Jackson (1993), Maxwell *et al.* (1996), Duncan *et al.* (1999).

Note: The column figures are the number of species affected. A species can be at risk from other threats in addition to those resulting from the minerals industry.

With regard to bats, Duncan *et al.* (1999) advise that many bat species use abandoned mines as daytime roosts and maternity sites. Often there are large congregations of

animals at these sites, and, in many bat species, the historical activities of the minerals industry has permitted their range to be extended. Duncan *et al.* (1999) advise that the closure of mines for public safety considerations, or the collapse, destruction or re-working of abandoned mines is now considered a threat to the population viability of nine threatened bat species that are dependent on these abandoned mines following the destruction of their natural roosts. The minerals industry has an important opportunity to engage other natural resource management agencies to develop on-ground management actions to protect the roosts of these threatened bats where possible. Grates have been developed that prevent human access to abandoned mines yet permit bats to enter the abandoned workings.

There appears to be no data to suggest that aquatic fauna and flora species have become threatened or endangered by activities of the minerals industry. Wager and Jackson (1993) suggest that specific sources of erosion, such as that resulting from mineral operations, can have a significant affect on aquatic biodiversity through the influence that increased sediment load has on streams and waterbodies. For example, EPA QLD (1999) has reported on the downstream impacts on biodiversity in the Dee River from contamination by the abandoned Mount Morgan Mine in central Queensland. They suggest that pollution-sensitive invertebrates, and all fish and submerged plants, were absent from sites within 20 km downstream of the mine as a result of acid rock drainage. In many cases, there is a lack of pre-mine baseline data on biodiversity to determine the impact of the minerals industry although it is presumed to be severe.

2.1.4.1 Rare or threatened species or communities

An important aspect of conserving biodiversity is the preservation of rare and endangered species, and this is particularly so in a country such as Australia which has notably high levels of endemism and species richness (Mummery and Hardy, 1994). The existence of rare or threatened species or communities has been a factor in determining whether mining is able to proceed.

A topical example is the occurrence of stygofauna and troglobitic fauna within mineral leases. These fauna reside beneath the ground surface with the former being aquatic, while the latter are terrestrial. One area where such animals abound is in the

limestone karst formations of the Cape Range province in the north-west (Pilbara) of Western Australia. These formations support one of the world's most diverse subterranean fauna with many species endemic to the area. A localised distribution of some of these species qualify them to be classified as rare or threatened under Western Australian legislation. Limestone has been mined in the Cape Range region, with proposals submitted to extend the impacted areas. In view of the capacity for mining to interfere with subsurface hydrology, this activity has been disallowed in some cases. A rich stygofauna has been detected elsewhere in the Pilbara region, and has become an issue in relation to the granting of permission for future mining in the area.

If rare species are identified within the project area, development approval is obtained only following evidence that the development will not negatively impact on the species' survival. Two examples of where development was allowed on the condition that a priority species was not threatened by the development proposal include the presumed extinct plant species *Hemigenia exilis* and the Priority fauna Pebble-Mound Mouse.

Case Studies

Hemigenia exilis

In 1995, the presumed extinct plant species *Hemigenia exilis* S. Moore (Figure 2.6) was rediscovered during vegetation surveys of the Murrin Murrin tenements of Anaconda Nickel Limited in the north-east goldfields of Western Australia. Despite thorough searching, this species had not been recorded for 100 years. The discovery of a declared rare flora species within the proposed project area required Anaconda Nickel to develop research programs and management objectives to ensure conservation of the species.

In response to this find, Anaconda Nickel commissioned surveys and conducted public and internal awareness programs to increase communication on the species. By the end of 1997, 65 populations of *H. exilis* had been discovered totalling more than 50,000 plants within the north-east goldfields.

Figure 2.6. Flowers of the Priority Four rare plant species, *Hemigenia exilis* S. Moore from the goldfields region of Western Australia.

These finds were significant in that they identified a presumed extinct plant species to have a wide distribution. By mid-1998, the conservation status of *H. exilis* was re-classified as Priority Four Rare Taxa – rare, but not threatened by identifiable factors. Government regulations require Priority Four taxa to be monitored every five years. To ensure that their activities did not impact on the survival of the populations that existed within their tenements, Anaconda Nickel conduct quarterly monitoring to record the health and biology of the populations. In addition, Anaconda Nickel commissioned research into the seed biology and genetics of *in situ* populations and redesigned the mine plan to minimise the effects of waste dumps, haul roads and mine voids on the located populations.

As a result of investigations commissioned by Anaconda Nickel, a detailed knowledge now exists of the distribution, demography, phenology, genetic diversity,

seed physiology and propagation of *H. exilis*. This knowledge will enable the expansion of populations within the project area to preserve the species within natural populations and the post-mined environment.

Pebble-Mound Mouse

The Pebble-Mound Mouse (*Pseudomys chapmani* Kitchener) (Figure 2.7) was discovered in 1980 in the Pilbara region of Western Australia. In developing its home, this species builds a mound up to 2-3 m in diameter from small pebbles. Due to a limited known population number and habitat, the Pebble-Mound Mouse was listed as a Schedule 1 endangered species. Fauna surveys commissioned by the mining company, Hamersley Iron Pty Ltd, found this species to occur within both its Marandoo development and Yandi Project Lease. Hamersley Iron commissioned surveys and research to further understand the distribution and ecology of the species.

This research, in conjunction with the State conservation agency, CALM, determined the species to have a wider distribution throughout the Pilbara than previously known. The results of research into the species ecology has allowed mining companies, that are active in the Pebble-Mound Mouse's range, to manage their rehabilitated areas in accordance with the conservation of this species. A greater understanding of population numbers, distribution and species ecology, in association with the successful translocation of the species, have enabled the Pebble-Mound Mouse to be removed from the list of endangered species.

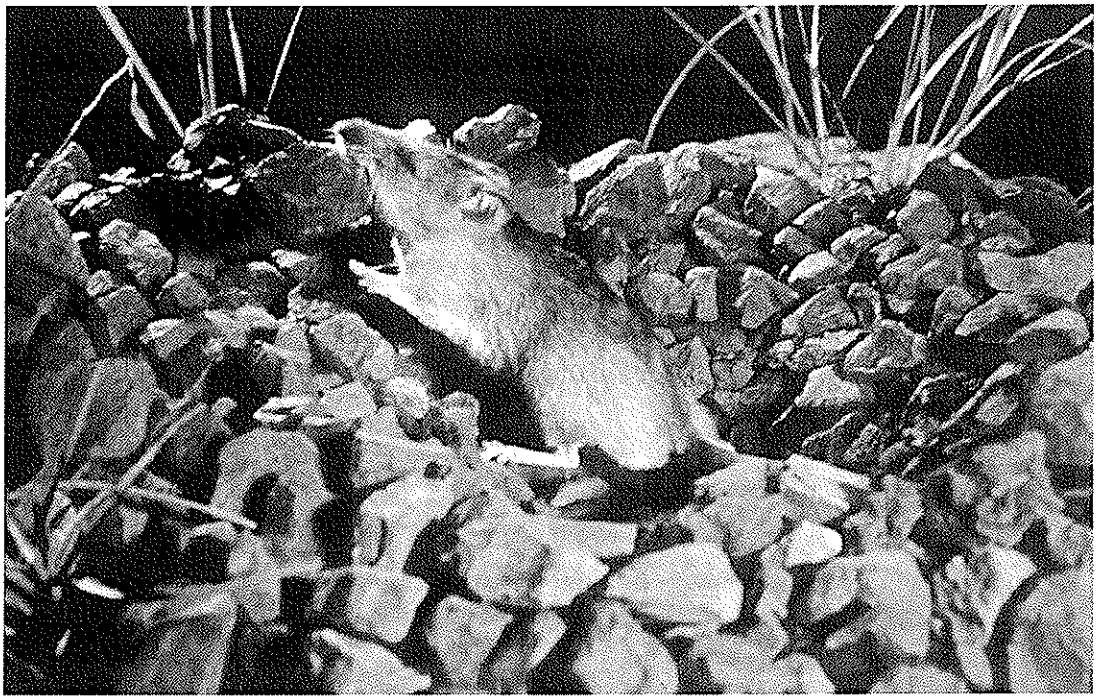


Figure 2.7. The Pebble-Mound Mouse, *Pseudomys chapmani* Kitchener, from the Pilbara region of Western Australia.

2.2 Activities of the Minerals Industry that Impact on Biodiversity

2.2.1 Terrestrial

As noted by EPA NSW (1997), most mine operators now have access to a wide array of modern technology and management approaches to account for and lessen environmental impacts. Nevertheless the immediate environment in which mines are located may be affected in a number of ways during mining, and sometimes long after mining has ceased. Woodside and O'Neill (1995) provide a good account of the wide range of environmental impacts associated with mining and exploration activities. This review, however, is more concerned with the activities that are causing specific impact on Australia's biodiversity.

2.2.1.1 Impacts of exploration and prospecting activities

Farrell and Kratzing (1996) advise that the primary impacts of exploration are the removal of vegetation and disturbance of ground in the immediate vicinity of the principal activities. Removal of vegetation obviously has an immediate impact on local biodiversity, while ground disturbance may influence the recruitment and survival within the rehabilitated areas. Recent exploration and prospecting activities are more environmentally friendly than in the past and have been outlined by White *et al.* (1996).

The use of bulldozers and graders to create gridlines in past exploration activities have left a legacy of tracks across Australia. Many of these tracks are clearly visible today despite the natural regeneration that has occurred. Whilst the tracks have increased access to remote locations, they also have the potential to impact on natural drainage patterns and, therefore, the recruitment and survival of vegetation communities (White *et al.* 1996).

Most exploration activities are now conducted with the use of non-intrusive techniques such as remote sensing and global positioning systems (White *et al.* 1996). Where field sampling is required, existing access tracks are used wherever possible to provide access for vehicles and equipment for further studies. Where existing tracks are unavailable, techniques such as folding vegetation instead of grading to minimise disturbance to root systems and enable vegetation regrowth, the use of lighter drilling rigs or helicopter-assisted drilling programs to transport the equipment into sensitive or rugged terrain are frequently being undertaken (White *et al.* 1996).

Some operators utilise wooden platforms to accommodate drill equipment and foot traffic during drilling operations. These wooden platforms are placed directly over understorey vegetation, and disperse the weight over a larger area to minimise soil compaction and allow vegetation regrowth once the wooden platforms are removed, causing minimal disturbance to local biodiversity.

Many drill holes remain uncapped and can act as pitfall traps for native wildlife (Figure 2.8) or, with time, become an area of erosion around broken drill collars. The use of buried concrete drill plugs prevent animal entrapment and are able to withstand vehicle traffic and corrosion, problems associated with the use of plastic or steel caps (White *et al.* 1996).

Figure 2.8. Uncapped drill holes can act as pitfall traps to native fauna that are unable to escape. The skeleton of previous lizards can be seen deeper in the drill hole. Photo courtesy G. Wells, Botanic Gardens and Parks Authority.

2.2.1.2 Vegetation clearance and surface impacts

The clearing of vegetation is one of the most significant impacts of mining on biodiversity. The area of vegetation affected will vary according to the type of mineral extracted and the methods employed. For instance, sub-surface (underground) mining methods cause far less vegetation disturbance than surface mining methods such as strip mining and dredging. It should also be remembered that there is often considerable infrastructure associated with mining activities such as road, rail and port developments, which will most likely result in further vegetation clearance.

Once the vegetation has been cleared, the topsoil is removed and, if possible, immediately transferred onto a rehabilitated mine pit to maximise topsoil seed viability and native species recruitment. In many cases the direct transfer of the topsoil is not possible, and the topsoil is required to be stored for later use. The storing

of topsoil is known to decrease the viability of the soil seed reserves with time, resulting in depressed levels of recruitment that may be obtained when the topsoil is re-spread at a later date (Tacey and Glossop 1980; Nichols and Michaelsen 1986; Dickie *et al.* 1988).

The excavation of the substrate materials and creation of the mine void alters the soil profile, hydrology, topography, and nutrient status of the substrate. These secondary factors have the potential to impact on the local biodiversity within the rehabilitated post-mined environment.

Extensive mining operations, mainly involving bauxite and mineral sands extraction, have historically destroyed extensive areas of vegetation and changed soil profiles and topography, but modern methods now focus on minimising this damage.

As noted by Saunders *et al.* (1996), the most serious environmental conflicts in the mining industry now revolve around access to mineral reserves in areas of significant habitat, high biodiversity or special cultural significance. Historically, sand mining has been one of the more contentious environmental issues for the industry. In the past, extensive rutile and zircon mining was undertaken on the north coast of NSW, although for the reasons mentioned above, in that State it is now confined to an area north of Newcastle (NSW State of the Environment Report 1997). Not withstanding the progress that is being made in rehabilitation and the introduction of more stringent environmental requirements in the industry, Fox (1996) has found that sand mining has historically resulted in long-term changes to biodiversity.

2.2.1.3 Impacts of mining infrastructure and development

Mining-related infrastructure developments include corridors for roads, railways, pipelines and powerlines, port developments, through to the establishment of human settlements. Large increases in local populations can place pressure on National Parks and other relatively undisturbed areas through the use of off-road vehicles, littering, illegal fishing and hunting, spread of weeds and dieback disease. The combined extent of these impacts can be far greater than those occurring within the mining lease. These kinds of infrastructure developments can have significant negative consequences for biodiversity, for example, creating opportunities for weeds

and pests, modifying natural water flows, and acting as barriers to the movement of native organisms (Noble *et al.* 1996).

Historically, mining activities have given rise to many new settlements within Australia. For example, Ballarat and Bendigo in Victoria were established during the gold rushes of the mid 19th century. However, since the 1960s, Newman *et al.* (1996) suggest that more than 25 new towns and ports have been constructed to support mine sites. They argue that four different mining settlement strategies can now be recognised – new single-industry towns, expansion of existing communities, combined industry towns, and fly-in/fly-out strategies. The first three strategies result in increases in the local population, with consequent impacts such as those described above. By contrast, the fly-in/fly-out strategy results in a lower resident population and also better control as most resident personnel belong to the workforce. The strategy is also encouraged by indigenous negotiators (Manning 1997), and hence is becoming increasingly common as it probably has the least impact on biodiversity by reducing the pressure for infrastructure development in remote areas.

2.2.1.4 Waste management and pollution impacts

The occurrence of waste dumps may be situated within the mineral lease or, in the case of mineral processing, occur at sites remote from the source of the extraction. In both cases, these deposits occur at the expense of underlying plant and animal communities. Slime dams, slurry ponds and similar impoundments are formed when the residues from ore processing are deposited in artificially dammed areas.

Hamblin (1998) advises that mining and associated processing operations require high levels of management care due to the hazardous nature of many of the materials being used, which often pose a significant environmental risk. Cyanide at toxic levels within tailings storage facilities have been responsible for the death of birds that are attracted to the expanse of shallow water (Donato 1999). Significant publicity has resulted in the minerals industry developing practical measures to minimise the effect of cyanide in tailings dams and ponds on wildlife and domestic animals.

According to Woodside and O'Neill (1995), the impacts of mining-related pollution on biodiversity have been significant in the past. For instance, the Rum Jungle Mine in the Northern Territory released 130 tonnes of copper, 100 tonnes of manganese, 40

tonnes of zinc and 13 000 tonnes of sulfate into the Finniss River in one year. A survey conducted in 1971 showed that the river was polluted to such an extent with heavy metals, acidification and radiation that it was virtually dead. Recent surveys of the same river (after the rehabilitation of the site in the 1980s) indicate that macro-invertebrates in the waterways continue to be less common downstream of the mining activities. In contrast, more recent mines such as Woodlawn in New South Wales and the Ranger Uranium Mine in the Northern Territory, operate on almost a zero water release basis, with controlled release only of collected rainwater in special circumstances.

Pollution from Australian mines in the past has primarily resulted from the oxidation of sulfidic minerals such as pyrite (iron), pyrrhotite (iron), chalcopyrite (copper), galena (lead) and sphalerite (zinc). Exposure of these minerals in waste rock dumps, tailings storage facilities, and along the walls of open pits and underground workings has resulted in the release of acid or neutral drainage with toxic metal concentrations causing loss of biodiversity within both terrestrial and aquatic environments. Surveys have found that, of the 317 active and abandoned mines in Australia that have the potential for the excavated material to be acid generating, 54 sites generate significant amounts of potentially acid-generating wastes (greater than 10% of their wastes), while a further 62 generate some potentially acid-generating wastes (less than 10% of the total waste) (Harries 1997). Rehabilitation of these sites, viz. Mt Lyell (Tasmania), Mt Morgan (Queensland), Brukunga (South Australia) and Rum Jungle (Northern Territory) have proved to be difficult.

Another source of pollution emerging from modern, large-scale mining within the Kalgoorlie and Bowen Basin regions of Australia, is salt release and erosion caused by the mining and exposure of sodic soils, subsoils and rocks (Grigg *et al.* 1997; M. Lee pers. comm.). Exposure of these materials has resulted in deep rilling, tunnel erosion, instability of mined landforms and increased concentrations of dissolved salts in streams and open pit lakes receiving discharges from these deposits.

Atmospheric pollution has caused significant loss of local biodiversity in the past, with Mt Lyell in Tasmania providing the most obvious example. As noted by Woodside and O'Neill (1995), point source anthropogenic sulphur dioxide (SO_2) emissions in Australia are dominated by mining-related smelting and roasting at Kalgoorlie in Western Australia and Mt Isa in Queensland. Yet it is considered that

the risks to biodiversity are minimal given the moderate to low acidity of Australian soils and relatively low rainfall over most of the country.

2.2.1.5 Impacts of unrehabilitated or poorly restored sites

Many abandoned mines around Australia remain unrehabilitated. In most States there is an active process of identifying those that pose the most serious threat to the environment and local biodiversity. These sites are then prioritised according the level of risk they pose, and are progressively rehabilitated, usually at public expense. In NSW alone, there are an estimated 20,000 to 30,000 derelict mines that require clean-up and rehabilitation (NSW State of the Environment Report 1997). Perhaps the most serious threat to biodiversity from these abandoned mines are from those that leak acid waste. There are several rivers in Australia that have had significant lengths of their extent rendered biologically dead from such mining impacts. For instance, some sections of the Molonglo River, which flows into Lake Burley Griffin in Canberra, were damaged to such an extent by acid leaking from the Captains Flat mine that no aquatic biota could persist. Many parts of this river remain severely damaged.

With the exception of several industry leaders, rehabilitation practices at many of Australia's current mining operations are of a relatively poor standard with respect to re-establishment of biodiversity values. This is because the restoration of landscape function and partial vegetation structure is considered by many to constitute rehabilitation success, with far less regard for the restoration of species composition. As noted by Woodside and O'Neill (1995), little if anything is done to re-establish the fauna (invertebrates or vertebrates), with many mines of the opinion that within the post-mined rehabilitation, the fauna will migrate from adjacent areas. This is quite often not the case, particularly when opportunities for secondary succession are exacerbated by excessive fragmentation or lingering pollution (Lockwood and Pimm 1999). However, due to the relatively small size of the mineral operations in comparison to alternative land uses, poor rehabilitation is only likely to impact local biodiversity.

Case Studies

New South Wales

The extensive sand dune areas north of Newcastle have been the subject of much investigation over the past 15 to 20 years. Fox and Fox (1984) studied small mammal recolonisation (two rodents - the introduced house mouse and the native New Holland mouse) during the first 10 years following mineral sand mining at Myall Lakes National Park, New South Wales. The house mouse reached its maximum 3 years after mining and was associated with bare sand, soil hardness and vegetation structure. The New Holland Mouse reached its maximum 8-9 years after mining, after appearing at the 4-5 year mark and was associated with proportion of heath plants present, vegetation structure, amount of dead plant cover and topsoil depth. Fox and Fox (1984) found that this species replacement series in rodents is stretched in time such that the post-mining environment (as opposed to the post-fire environment) more closely approximates primary rather than secondary succession. Twigg *et al.* (1989) further studied this issue using chronosequence analysis along an old mining path in Myall Lakes National Park. A 'wave of succession' moved along the mining path with early colonisation by the House Mouse (*Mus musculus*), an introduced opportunist, followed by the New Holland Mouse (*Pseudomys novaehollandiae*), a native opportunist, and finally succeeded by the Common Dunnart (*Sminthopsis murina*), as the regrowth vegetation and hence habitat became more complex. The sequence was similar to that noted for post-fire small mammal succession. Twigg and Fox (1991) studied terrestrial lizard fauna in the same area and concluded that the successional process proceeded more slowly with terrestrial lizards than the small mammals. Some species colonised as early as 5 years post mining, but other species were still absent after 15 years. They estimated a period of at least 20 years post mining for the reptile fauna to achieve its pre-mining state. Fox (1996) summarises this work as well as the response of small mammals to fire.

Several studies have demonstrated that vegetation recovery has not yet been achieved in the mineral sands industry in NSW. Buckney and Morrison (1992) compared the floristic composition of mined and un-mined sand dunes in Myall Lakes National Park from 1982 to 1990. They found that the mined area was different from both adjacent un-mined dunes and the pre-mining condition. In fact, the vegetation showed a decreasing similarity over time to the vegetation previously on the site. While

species richness and diversity increased during the post-mining period, a large proportion of this was attributed to introduced species. McNair (1993) studied post mineral sand mining regeneration with transects undertaken at Salamander Bay in the Port Stephens area and Bridge Hill and Submarine Beach in the Myall Lakes Region. He also studied post silica sand mining regeneration with transects undertaken at Tanilba in the Port Stephens area. He found little resemblance of post-mining flora to pre-mining flora and an apparent decline in the native plant species in both mineral and silica sand mining flora regeneration with no change after more than 20 years.

Other studies are attempting to compare the differential effects of different types of disturbance. Fox *et al.* (1996) compared regeneration at 44 sites at Tomago New South Wales in relation to burning, clearing and mineral sand mining. As the study is an interesting contrast between regeneration resulting from different disturbances, it is worth quoting the conclusion section of the paper at length:

'The chronosequence data demonstrates that regeneration of cleared or mined sites at Tomago is substantially slower than regeneration following the endogenous disturbance by fire'... 'The severity of the impact of disturbance on vegetation structure increases from fire to clearing with mining the most severe impact. Seventeen years of regeneration on cleared and sand mined sites at Tomago has not been enough to return the vegetation structure or the soil characteristics to the pre-disturbance state. Understorey height and the amount of vegetation on cleared or mined sites have not achieved the levels in the original forest, although canopy cover does seem to have reached pre-disturbance levels.'

Jackson and Fox (1996) used chronosequence analysis on 72 ant species from the same 44 sites at Tomago New South Wales to examine whether ant community structure varied with the type of disturbance and time since disturbance. They collected 25 habitat variables at each site covering vegetation structure, floristics, ground cover and soil variables with the key variables being percentage canopy cover of the site, percentage of bare ground and a vegetation index. Their results suggest that fire has a minor effect on the community over time, while the impact of clearing and mining is much more severe. Species composition at cleared and mined sites, after 18 years, approaches but does not match controls. The ant species succession seems to closely follow the vegetation succession and hence may be used as a bio-indicator for evaluating the extent of habitat damage and recovery after disturbance.

Past mining for gold has also resulted in extensive areas of disturbed vegetation in central and southern New South Wales, although the impact is not easily documented. It is clear though that areas impacted by gold exploration and mining activities in the middle of the 19th century retain many biological values after mining has been abandoned and regrowth initiated. An example from southern New South Wales is the Major's Creek area south of Braidwood, which was mined for gold in the middle of last century and consists almost entirely of relatively recent regrowth, but still retains a diversity of native plant species, which include the rare Araluen Gum, *Eucalyptus kartzoffiana* (M. Doherty pers. obs.). Although the composition of the area is not known prior to mining, the present composition appears to be a successful regeneration event in response to a severe but relatively short-lived impact.

Queensland

Thatcher and Westman (1975) studied succession following mining for mineral sands on North Stradbroke Island, although the study only investigated the site 5 years after mining. One interesting result was that plots closest to un-mined vegetation showed a greater rate of native vegetation re-establishment due to the availability of propagules. Hill and Phinn (1993), studying Swamp Wallabies (*Wallabia bicolor*) in the same area, found that re-vegetated sites were most heavily used for grazing at night at 2 to 4 years post-mining where vegetation was shrubby or grassy. Older sites offered little forage and were not utilised. Wallabies rested during the day in adjacent undisturbed eucalypt forest.

Further north, Unwin *et al.* (1988) studied the area around Rossville, south of Cooktown in Queensland, where alluvial tin mining has been carried out for 100 years. The evidence indicates that natural regeneration followed the cessation of mining on almost all sites. Generally, areas that were eucalypt forest reverted to eucalypt forest and areas that were rainforest reverted to rainforest. They claim an encroachment of rainforest via regeneration in the absence of Aboriginal burning into previously eucalypt-dominated areas. However, the evidence presented is not sufficiently detailed to assess the validity of this claim, nor to assess how different the pre- and post-mining vegetation types are.

Western Australia

Curry and Nichols (1986) surveyed bird species in 5 to 10 year-old rehabilitated bauxite mining areas. They found 21 species nesting in the rehabilitation, but that 11 other species only nested in adjacent un-mined areas. The main reason for the difference was a lack of suitable hollows for nesting in the young regeneration.

Majer and Nichols (1998) examined successional changes in ant communities within rehabilitated mine pits over 14 years against a forest analogue and concluded that mine pit ant community composition converged towards the forest site as time since rehabilitation increased. However, distinct differences between the mined and unmined sites remained.

Ward *et al.* (1990) discuss bauxite mine rehabilitation in Western Australia and report on research undertaken over a 10 year period. Important factors in returning mined areas to as similar a state as possible to the pre-mining condition include using freshly stripped soil rather than stockpiled soil and returning log debris as habitat. They report relatively high proportions of reptile, bird, mammal, ant and collembola species returning to rehabilitated sites. Koch *et al.* (1996) found that soil handling operations in bauxite mining in jarrah (*Eucalyptus marginata*) forest tended to drastically reduce germinable soil seed stores for rehabilitation. They traced soil seed stores at all stages from undisturbed forest that was then cleared and burnt, the soil stockpiled, re-spread and then deep ripped. The final seed content was 16 % of the original forest seed store density. Direct return of topsoil, with minimal delays at all stages, was recommended as a way of maximising retention of viability of soil stored seed. They go on to state that the jarrah forest appears to follow the 'initial floristic composition' model of succession. In this model, the composition of the vegetation in the first few years after disturbance will determine the ecosystem functioning and floristics in the long term. Therefore, it is imperative that high plant diversity is established as early as possible within rehabilitated sites.

2.2.2 Aquatic

Although most of the mining activities have only indirect impacts on aquatic biota, they can be long lasting and difficult to rectify. The common habitat stresses on aquatic environments caused by mining are:

1) land clearing and subsequent erosion, siltation, sedimentation and turbidity;
2) acid rock drainage (ARD);
3) salinisation due to disturbance of saline aquifer and dewatering;
4) dredging and extraction from lake and river beds;
5) disposal of effluents and waste water from processing operations resulting in leaching of toxins and harmful chemicals (e.g. mercury, arsenic); and,
6) direct disposal of waste mine tailings, slimes and waste rock into streams and lakes.

Most of the worst habitat degradations that have resulted in biodiversity loss today, are the results of past mining practices. Several well-investigated case studies have been chosen to catalogue appreciable mine-related impacts on aquatic biodiversity, with regard to the geographic scale and severity of their off-site effects.

Case Studies

The Mount Morgan mine site and the Dee River, central Queensland

The Dee River, in the central west region of Queensland, is an intermittent river that receives acid mine drainage from the Mount Morgan mine site. Mining operations began in the 1880s with the first copper extraction occurring in 1902. It is reasonably assumed that ARD has been entering the Dee River for a long time. Investigations on the diversity of macroinvertebrates in the 1980s showed their severe reduction over more than 15 km downstream of the mine-site, with the fauna still not completely recovering to distances of 50 km downstream (Mackey 1988). Bankside vegetation was also absent in the vicinity of the mine site, and the distribution of submerged macrophytes indicated their severe reduction over 50 km. However, some recovery in both macroinvertebrate diversity and submerged macrophyta was observed in 1985/6 that was attributed to the generally dry conditions that resulted in reduced ARD seepage into the river system (R. Jeffree pers. obs.).

More recent studies in the Dee River have confirmed the continuing nature of the pollution impacts of the mine site, following heavy rainfall events that occurred between November 2000 and February 2001 (Taylor and Howse 2001). Results

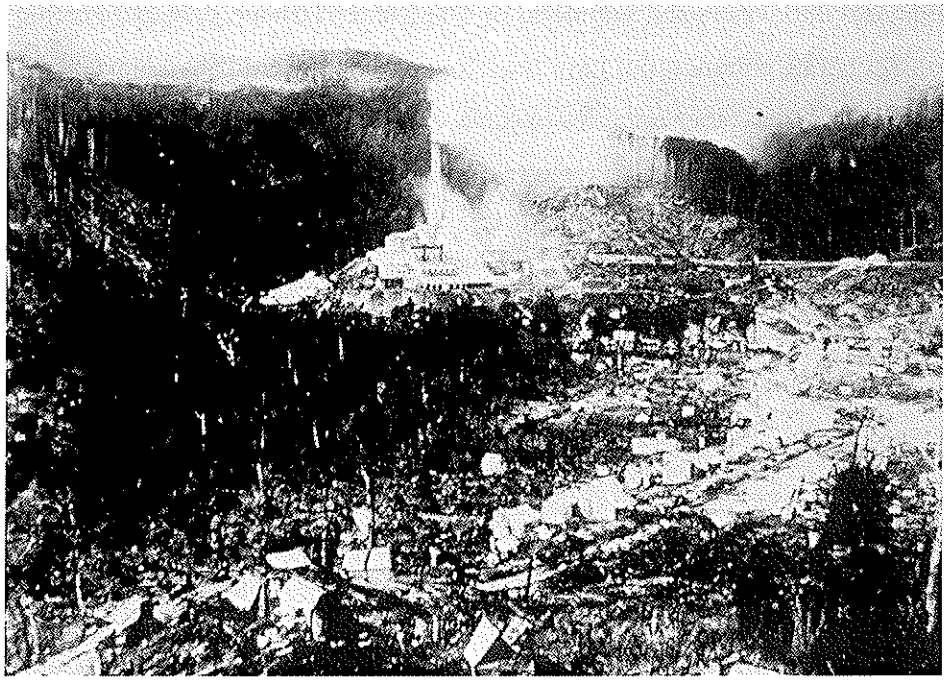
indicate that mine-site runoff water during high rainfall events can effectively bypass the mine's pumpback system, that had been designed to intercept such water before entry into the Dee River. Ongoing studies indicate that metal-rich waters from Mt Morgan have also infiltrated into the groundwater during acid flows in the river. River surface sediments are contaminated with copper, zinc and sulfur to at least 49 km downstream of the pollution source. During low flow conditions, the concentrations of various metals in surface waters were well in excess of ANZECC criteria for the protection of aquatic systems (1992) to at least 10 km downstream. During a recent high flow event, copper, zinc and aluminium concentrations in surface waters exceeded ANZECC guidelines to at least 43 km downstream. Four fish-kills were observed and monitored along the Dee River over these periods of high flows. These extended over 10-20 km downstream of the mine site, with an average density of 20 dead fish observed per 100 m of river bank.

Mount Lyell mine site: the King and Queen River catchments and Macquarie Harbour, western Tasmania

One hundred years of copper extraction at Queenstown by the Mt Lyell Mining and Railway Company, western Tasmania, has exposed large amounts of ore rich in sulfide to weathering and has resulted in the discharge of tailings into the Queen and King Rivers (Figure 2.9). Over 100 million cubic metres of mine tailings, smelter slag and topsoil have been delivered to this freshwater system, which flows into Macquarie Harbour. An evaluation of the environmental impacts of the resulting ARD on these freshwater systems has been determined by measurement of both biological and water quality conditions. Riffle macroinvertebrates were repeatedly sampled at an array of exposed and reference sites, primarily using the National River Health Program rapid assessment protocol. These investigations showed that the macroinvertebrates within reference sites were diverse and characterised by high abundances of particular taxonomic groups. All sites exposed to ARD pollution did contain a macroinvertebrate fauna, although typically of low abundance with the dominant taxonomic groups different to those found within the reference sites. All sites investigated in the Queen and lower King Rivers using this technique were highly or extremely impacted by ARD pollution, extending over more than 20 km of the river systems (Davies *et al.* 1996).

Exposed tailings on the river banks and in the delta continue to leach various metal contaminants into waters and sediments of Macquarie Harbour. Fish and benthic macroinvertebrate communities have been found to be lower in abundance and species diversity that other estuaries in south-eastern Australia. Phytoplankton in Macquarie Harbour were also characterised by low species abundance (Stauber *et al.* 2000).

(a)



(b)

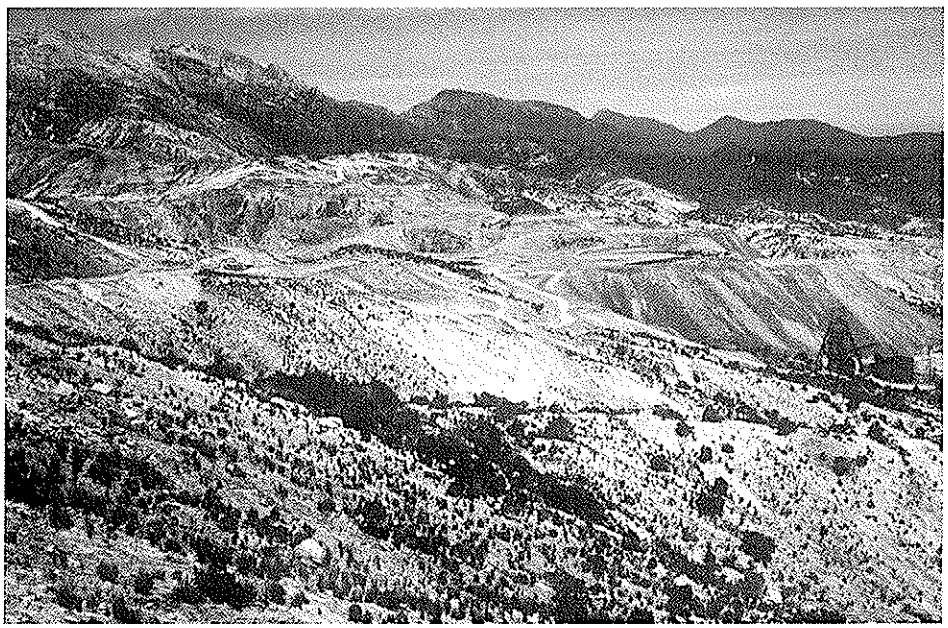


Figure 2.9. Mineral operations at the Mt Lyell the mine site in the early 20th century releasing fumes as a result of the pyritic smelting process (a). The denuded hillsides were subject to erosion as a result of mineral operations (b).

Several streams in the Queen river catchment contained populations of introduced brown trout and rainbow trout, and tributaries of the lower King River supported three species of native fish whose occurrences and abundances declined with distance from Macquarie Harbour. Recruitment of native fish into the streams of the Macquarie Harbour catchment probably occurs on an annual basis (Davies *et al.* 1996).

Mining and the salt lake systems - dewatering

The myth of impoverished biodiversity of saltlakes (salinity > 3 g L^{-1}) is being challenged by recent research work (Williams 1998a). While biodiversity of saltlakes is lower than that of freshwater, it is relatively high, except in extreme salinities. The invertebrates show a high degree of endemism. Australian salt lake fauna and flora of the semi- and arid-zones represent a wide variety of taxonomic groups including those represented in fresh water zones. These are not exclusively saltwater taxonomic groups. The heterogeneity of salt lake biota is associated with varying salinity observed at different locations even in a single lake, at different periods of the filling and drying cycle. It is not unusual to find patches of waterbodies with salinity close to 3 g L^{-1} in one part of a salt lake with many freshwater biota, whereas in another part of the lake, salinity up to 200 g L^{-1} may be observed. Lakes Lefroy, Lake Carey, Lake Dundas, Lake Cowan, Lake Yingargooda, Lake Miranda, are among a group of large saltlakes in the arid and semiarid region of Western Australia, which get filled to different levels during the rainfall events mostly in winter and summer. The duration and intensity of the rainfall determine the survival and distribution patterns of the biota.

The salt lakes in Western Australia are located along the palaeodrainage systems and basins. This saline land includes the greenstone belts of the eastern Murchison, eastern Goldfields, Yilgarn and Great Southern, which are rich in gold, nickel and other minerals. Mining has been active in these regions for decades. As the more accessible orebodies are mined, more resources have been diverted to exploring the deep sediments of the salt lakes for further mining. Associated with special problems of underground mining in these sediments, are the possible impacts on biodiversity of the salt lakes. Discharging the deep underground hypersaline water of the salt lakes into playa lake beds has been practiced for the past two decades or more. The effects

of such dewatering on a few playa lakes in the Kambalda region have been total loss of habitat of aquatic fauna and microalgae and destruction of peripheral vegetation (J. John pers. obs.). The mining industry is involved in conducting environmental assessment surveys to assess possible impacts of dewatering on biodiversity and is keen to avoid habitat loss and biodiversity.

Apart from the 'aquatic fauna and flora' which exploit the episodic rainfall events for their survival, there are also characteristic terrestrial fauna (mostly spiders, beetles and ants) which have made the lake bed their home, but unlike the former, the challenge for the latter is to endure or escape the inundation for their survival rather than exploit it. Here there are two sets of biota with conflicting needs (Williams 1998b). The impact of dewatering, physical destruction of habitat, acid drainage and the leaching of toxic chemicals on the biodiversity of salt lakes needs to be further investigated.

3.0 MANAGING IMPACTS ON BIODIVERSITY

3.1 Biodiversity Measurement

Notwithstanding the continuing scientific debate over the biodiversity concept, natural resource managers are under increasing pressure to operationalise the concept in order to report on the state of biodiversity and trends in its condition (Doherty *et al.* 2000).

It is widely recognised that there are three levels of biodiversity – ecosystem diversity, species diversity and genetic diversity – thereby forming the key units of biodiversity measurement (Beattie 1995). However, due to the complexity associated with assessing multiple levels of biodiversity, natural resource managers tend to evaluate only one or two of these levels. The majority of studies focus on species diversity as the preferred unit of biodiversity assessment and management, largely because of the ease with which different species can be recognised, but also due to the roles in ecosystem functioning that many species are now known to play. Conserving species across their range is also likely to contribute to the conservation of genetic diversity within these species.

Ecosystem diversity is considered by many as too difficult to operationalise as a unit of measurement because of problems with 'on-ground' delineation of ecosystem boundaries. Genetic diversity, while being fundamentally important as the basic unit of biodiversity, is also considered by most natural resource managers as too difficult and costly to evaluate.

Measuring biodiversity using species diversity involves a complex combination of values such as species richness, species composition and taxonomic range (Gaston 1996; Williams *et al.* 1996). Species richness alone, as a measure of biodiversity, has been used in several experimental studies investigating the functional significance of biodiversity on 'ecosystem processes' (Naeem *et al.* 1994; Tilman *et al.* 1996), but has been criticised as being inappropriate (Beck 1998), because it does not account for differences between component species. Species richness alone is consequently a poor measure of diversity, although it must be recognised that biologists have historically assessed species diversity in other more complex ways (Ghilarov 1996).

The rehabilitation of disturbed lands to an ecosystem representative of the pre-disturbed state cannot be undertaken without first assessing ecosystem complexity. Frequently undertaken as part of an environmental impact assessment, a baseline study is conducted to characterise the abiotic and biotic features of the area subject to development (see Mattiske *et al.* 1986). The outcome of this investigation has an impact on all stages of the mine life from start-up to mine closure. Should investigations determine the presence of gazetted rare species, mine planning is required to take into account the populations of these species. If relocation is not an option, then the baseline study may dictate the extent of all development and disturbance activities.

Considerable literature exists on monitoring programs to assess the impact and recovery of ecological communities following disturbance, e.g. Asher and Bell (1998; 1999). The taxa of Australian inland waters, particularly stagnant surface and subsurface water bodies, and their response to disturbance events have not been investigated to the same degree as terrestrial or marine systems (McComb and Lake 1996). Studies on the biota of temporary wetlands are rare (Lake *et al.* 1989). The 'cuddly and charismatic', 'flagship' and 'icon' species mostly hail from studies within terrestrial or marine environments. It is not surprising then, that the vast majority of

studies on the impact of the minerals industry have been on terrestrial or marine systems, with the latter mostly relating to petroleum extraction.

Diatoms and cyanobacteria often play a vital role in sustaining an array of invertebrates that, in turn, are a food source for the larger animals. Many of these microalgae are sensitive to environmental changes and hence may be utilised as biomonitors to assess aquatic health (John 1993). Indeed, the presence of diatomaceous earth within many aquatic habitats may be used as a historical record of water quality due to fluctuations in diatom species and numbers with changes in water quality.

3.2 Contribution of the Minerals Industry to Biodiversity Management

The loss of biodiversity is often related to implementation of development projects and changes in land use. The increased public concern and education in environmental issues may generate opposition to development projects with a poor environmental history. Aside from the issue of maintaining access to mineral resources, it is important that the Australian minerals industry both assesses and manages its impact on biodiversity for a host of other reasons. Biodiversity may provide, for example, ecosystem services (e.g. water purification), a source of biological resources (e.g. options for innovative technologies) and social benefits (e.g. recreation).

With an increasing community concern for environmental matters has come an increasing demand for best practice environmental management. The development of best practice techniques requires a detailed understanding of environmental issues and is directly linked to the resources that can be allocated to those pursuits. Within Australia, these resources are largely generated through revenues obtained from development processes, namely mining and agriculture (Webb 1990). Through taxes and revenues obtained from the minerals industry, the industry has become a major indirect source of funds that are available for environmental initiatives (Webb 1990).

The minerals industry conducts a considerable amount of in-house and commissioned research as part of environmental impact assessments for mining proposals and for

rehabilitation of the post-mined environment. As a result, the industry may be credited with contributing a substantial amount of knowledge on Australia's biodiversity.

Many of Australia's mineral reserves are located in remote locations and away from major population centres. The mineral leases are selected on the basis of geology rather than ecology or archaeology, and therefore are not necessarily areas that would receive thorough biological studies (Webb 1990). In many cases, biological research and surveys undertaken, or commissioned, by mineral and exploration companies, have revealed new populations or a greater understanding of the existing diversity and biology of flora and fauna of the region (Oliver 1994). For example, pre-mining vegetation surveys at Eneabba (300km north of Perth, Western Australia) over 20km^2 of the orebodies recorded 429 species, of which 15% could not be identified to species level, and 10% were classified as rare or poorly known (Brooks 1987). In addition, studies undertaken by Hamersley Iron Pty Ltd have contributed substantially to the understanding of regional flora in the Pilbara (NW Western Australia). Collections by the company have included over 50% of the known occurrences in the Fortescue Botanical District and further increased the understanding of the distribution and biology of these species (Mattiske *et al.* 1986).

Key strategies for biodiversity conservation and management should be based around the concept of bioregional planning, whereby biodiversity considerations are fully integrated with regional biodiversity management policies and programs. In some cases, such as areas impacted by rising salinity, the presence of feral animals and the proximity to sensitive environments, ecosystems that occur in adjacent areas may have an impact on the choice of management strategy occurring within the mineral lease. In many cases the minerals industry has undertaken to consider biodiversity issues of regional importance; however, in others, the minerals industry may be considered to be contributing to the loss of biodiversity and overall degradation of the landscape (Section 2.1.2). Because of its experience in landscape planning and coordination, the minerals industry can only benefit from opportunities to take a major leadership role in developing bioregional planning and management approaches to biodiversity issues in association with larger land holders such as the State governments and other industries.

The minerals industry owns a substantial amount of land within Australia despite much of this land not being mined. Occasionally, ownership of these lands is

maintained to ensure future access to the mineral resources within, when cyclical trends in the global mineral prices enable extraction to be profitable. More frequently, however, these lands are retained and managed under alternative land uses to facilitate profit for the company. The minerals industry, in managing these lands not allocated for mining, has developed management options that enable biodiversity considerations to be integrated with production.

The minerals industry has been an active participant of feral animal eradication programs and has sponsored research and practice into the application of feral eradication programs across broad scale areas of Australia. Several companies are also sponsors of initiatives to reintroduce native species into protected areas where these species were once known to exist.

Case Studies

Mount Weld Pastoral Company Pty Ltd

Mt Weld Station is a pastoral lease situated in the north-east goldfields region of Western Australia and is owned and run by the Mount Weld Pastoral Company Pty Ltd, Placer Dome Asia Pacific, Granny Smith Mine. The region has been the focus of mineral extraction for over 100 years, and hence contains hundreds of abandoned shafts, abandoned and active pits, drill holes, grid lines and infrastructure associated with mineral extraction. In addition, the Station has a long history of pastoral use and contains a number of priority flora species. In association with CSIRO Sustainable Ecosystems and the World Wide Fund for Nature, the Mount Weld Pastoral Company has developed a conservation strategy that accommodates the various ecosystems and habitats that occur on the lease, facilitates the rehabilitation of degraded areas and allows for an integration between mineral extraction, pastoral and conservation agendas.

The conservation plan allocates the paddocks within the lease to management uses to achieve a number of objectives including pastoral production, erosion control and rehabilitation, mining, catchment management and other uses.

Hamersley Iron Pty Ltd

Hamersley Iron Pty Ltd is a manager of several pastoral leases in the Pilbara region of Western Australia. It operates three pastoral stations that border Karajini National Park. In managing these pastoral stations adjacent to the conservation area, Hamersley Iron has developed a Memorandum of Understanding (MOU) with the State conservation agency, CALM (S. Anstee pers. comm.). The MOU enables the National Park to be managed in a manner similar to the biosphere reserve concept developed by UNECSO's 'Man and the Biosphere' Programme. Under this management regime, the pastoral stations adjacent to the National Park are managed as buffer zones to protect the integrity of the Park. The pastoral stations are managed on a graded system, with areas closer to the National Park managed to maximise biodiversity, while areas away from the Park are managed for cattle production. Through management of grazing, feral animal control and fire regimes within the pastoral stations, management of the National Park for biodiversity is not constrained by the Park boundaries, and multiple land use between conservation and production, or ecologically sustainable development, is shown to be possible.

Western Shield

In Australia, feral animals, namely foxes and cats, have decimated small mammal populations contributing to the extinction of 10 species of native animals and threatening extinction of 30 more. In 1996, a program initiated by CALM aims to bring these species back from the brink of extinction by controlling foxes and feral cats, on almost five million hectares of land.

The program, called 'Western Shield', introduces meat baits tainted with 1080, a naturally occurring poison found in the native plant genus *Gastrolobium*, into bushland throughout south-west Western Australia. Sponsorship, through financial assistance or in-kind support from mining companies, is essential to increase the benefits from feral predator control.

Financial support for the widespread baiting program has been obtained from the mining companies, Alcoa World Alumina Australia, Cable Sands (WA) Pty Ltd, Iluka Resources Ltd, Tiwest Joint Venture, the owners of the Boddington Gold Mine (Normandy Boddington Pty Ltd, AngloGold Australasia Ltd and Newcrest Operations

Ltd) with logistical support from the petroleum companies WAPET, Woodside and Apache (Armstrong and Batini 1998). The program has been successful in controlling fox numbers to the point where endangered native animals are beginning to thrive after being reintroduced into forest inside and outside reserves.

The provision of baits to private land owners adjacent to conservation reserves and State forests provides an extended buffer to assist in conserving native fauna populations and, to date, three forest-dwelling mammals have been removed from the State's Threatened Fauna List as a result of successful conservation management under the Western Shield program.

Other projects exist within Australia involving feral animal eradication from vast enclosed areas. Two projects in which the minerals industry is active include the Heirisson Prong Biosphere Reserve in Western Australia (Shark Bay Salt Joint Venture), and the Arid Recovery Project in South Australia (WMC Limited).

3.3 ESD and the Precautionary Principle

The conservation and appropriate management of biodiversity is a key requirement for achieving ecologically sustainable development (ESD). The precautionary principle has appeared in a number of broad policy statements, in particular, the National Strategy for Ecologically Sustainable Development (1992; p 8) and the National Strategy for the Conservation of Australia's Biological Diversity (1995, p 5). It also appears in the Intergovernmental Agreement on the Environment (IGAE) (paragraph 3.5.1):

'*If there are threats of serious or irreversible environmental damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation*'.

There are considerable gaps in the understanding of the levels of resilience in ecosystems and other aspects of biodiversity. As such, one often doesn't know how much, and what kind of, external perturbations a system can tolerate before it is significantly ecologically degraded, and for how long it will be degraded. Because of this uncertainty, there is a need for the Australian minerals industry to adopt the precautionary principle and proceed within ecologically sustainable limits if longevity

of the operations are to be maintained. Only as better information on the impacts of disturbance on biodiversity is obtained, can the need for its application be reduced.

3.4 Learning from Doing – Management 'Experiments'

One of the major constraints to the effective management of biodiversity is the paucity of scientific data regarding the way biodiversity changes in response to disturbance. However, many biodiversity researchers are now recognising that there is a significant opportunity to improve the understanding of biodiversity by maximising the learning from the significant number of large-scale management 'experiments' that are occurring on a daily basis across many of Australia's natural resource management sectors. Unfortunately to date, much of the potentially useful information from these activities has been rendered unusable due to a lack of formal experimental design such as a lack of controls and replication, changes in techniques and technologies over time, and a lack of basic before and after monitoring (see RAC 1992). In the absence of proven linkages, it is premature to suggest that there are such things as reliable 'indicators' of biodiversity, and it may be the case that certain components of biodiversity can only be assessed using direct measures.

In 1996, the minerals industry sponsored a project that would lead to the development of a technique to determine whether the underlying ecological processes that are required for the re-establishment of biodiversity occurs within rehabilitated mine-sites. Termed 'Ecosystem Function Analysis' (EFA), this technique was based on 20 years of research into rangeland ecosystems and examines a range of factors to determine the extent to which an ecosystem is achieving self-sustainability and nutrient cycling at a given time period (Bell 2001). The methodology consists of three modules (1) landscape function analysis (LFA), (2) vegetation dynamics, and (3) habitat complexity (Tongway *et al.* 1997a). Briefly, LFA examines how ecosystem resources such as water, soil, organic matter and nutrients are retained and cycled within the system, vegetation dynamics contrasts analogue sites with the rehabilitated areas, while habitat complexity is assessed on the basis of a number of features, i.e. canopy cover, shrub cover, ground vegetation cover, amount of litter, fallen logs and rocks, and free water availability (Kearns and Barnett 1999).

Industry personnel as well as other rehabilitation practitioners can benefit from the development of this technique. Through the use of simple tools, environmental

personnel will be able to assess whether the rehabilitated site is developing an ecosystem structure and habitat complexity according to the desired end-use.

This technique has been developed utilising data from 13 mine-sites from various mineral operations located across Australia and covering a range of climatic and community types. Presently the EFA technique is being assessed through further testing on mine-sites with a view to the technique subsequently being adopted by industry and government as part of an industry-financed study through the Australian Centre for Mining Environmental Research (ACMER).

3.5 Industry's Capacity to Protect Biodiversity

Mining activities and the infrastructure associated with them are just one of many land uses that impact on Australia's biodiversity. While there is much that the minerals industry can do to ensure that biodiversity is both adequately assessed and managed within its leases, the wide-ranging and cumulative nature of many of the processes that are now threatening Australia's biodiversity, demand holistic and coordinated bioregional planning approaches to the assessment and management of biodiversity. Such approaches are generally beyond the responsibility and capacity of the minerals industry alone, although there are significant opportunities for the industry to take major leadership and catalytic roles in bioregional planning and management activities.

3.5.1 Adequacy of skills and expertise

There is little accurate information available on the skills and expertise of the minerals industry with regard to biodiversity assessment and management. As such, the capacity of the industry in terms of skills and expertise can only be inferred through crude assessment of the level of education, training and networking among the industry's environmental management professionals. An industry-wide perspective has been gauged by exploring the level of access and effectiveness of the industry's key research and development providers.

3.5.1.1 Industry's environmental management professionals

As noted by Hancock and O'Neill (1996), since the mid 1990s, most new environmental management recruits in the Australian minerals and energy industries are products of the many specialist undergraduate courses now in place, and are thus qualified environmental professionals. Universities focussing on environmental management as it relates to the minerals industry include The University of Queensland, The University of Western Australia, and Curtin University, although there are many other universities around Australia offering more general environmental courses. Most of these universities have experienced high levels of student enrolment in their environmental courses during much of the 1990s; however, in the past few years, this popularity seems to have waned among some, as student interest in courses such as information technology increases.

It is clear that the skills and expertise of environmental professionals in the minerals and energy industries have greatly improved over recent years; however, accompanying this improvement, are rapidly increasing community expectations regarding the environmental performance of the operations and activities for which they are responsible. This has led to an increase in demand for environmental recruits with specialist post-graduate training in environmental science or management, in addition to the disciplinary backgrounds obtained during their undergraduate degrees. However, as the calibre of recruits required steadily rises, so too does the difficulty in filling environmental positions. As a result, there appears to be a trend for the industry to out-source some of its environmental management and R&D requirements to consultants and research providers. This is particularly true for specialist issues such as biodiversity assessment and management. In many ways, the use of consultants can provide cost-effective solutions to plan, manage or monitor environmental issues as they have access to broad-based expertise, or expertise in particular fields, e.g. specialist hydrogeologists.

Not all mining companies having followed this out-sourcing model. For instance, the Australian subsidiary of the Alcoa company (Alcoa World Alumina Australia) has developed considerable skills and expertise in the rehabilitation of jarrah forest ecosystems in Western Australia, following three decades of extensive research and development. Alcoa World Alumina Australia now possess a very competent team of environmental scientists who regularly publish the results of their studies of

ecosystem development in Australian and international scientific journals. Many consider that this research and development has now reached a point where restoration of the pre-existing jarrah forest ecosystem is a realistic target (May *et al.* 1996). In recognition of their achievements, Alcoa of Australia Ltd (Alcoa World Alumina Australia) was listed in 1990 by the United Nations Environment Program on its 'Global 500 Honour Roll'. Several other large mining companies, including Hamersley Iron Pty Ltd and Energy Resources of Australia, have also established teams of qualified environmental professionals. Other companies employ smaller teams and access the required expertise through consulting companies and/or links with Universities.

It is vitally important that environmental management professionals are able to keep pace with changing technologies and new scientific developments. Regular access to training and networking opportunities is therefore critical. Hancock and O'Neill (1996) stated that there are now significant opportunities for training within the industry through the variety of short courses and distance learning opportunities that are available. With regard to networking, the Minerals Council of Australia (MCA) has established a long tradition of annual environmental workshops, which provide an important opportunity for practitioners and researchers to get together (May *et al.* 1996). In addition, many regional groups have now formed throughout Australia, with the purpose of identifying and discussing common problems relating to environmental management of mine sites in the region (Hancock and O'Neill 1996). These groups meet more regularly than the annual MCA environmental workshops and have become a very effective way for the environmental management professionals in the minerals and energy industries to interact and liaise with government representatives, consultants and researchers within their region.

Short courses and workshops, focussing on the technical aspects of environmental issues faced by the minerals industry are also organised by the Australian Centre for Mining Environmental Research (ACMER) at locations around Australia, and these complement the more general awareness-raising national workshops conducted by MCA and the Australian Minerals and Energy Environment Foundation (AMEEF).

3.5.1.2 Industry's research and development providers

May *et al.* (1996) notes that much of the industry's environmental research is typically characterised by a single company focus on short-term site-specific needs. While this focus still dominates, there has been an increasing focus on longer-term strategic research in recent times, particularly on issues where the outcomes have industry-wide relevance. The ACMER through its research partners, the Australian Nuclear Science and Technology Organisation (ANSTO), Commonwealth Scientific and Industrial Research Organisation (CSIRO), Curtin University of Technology, The University of Queensland and The University of Western Australia, are addressing a range of biodiversity research projects relevant to the minerals industry.

3.5.2 Experience and environmental reputation

It cannot be denied that the mineral industry's reputation has suffered from some of its poor environmental performances in the past. The legacies of some of the earlier mining practices throughout Australia, for example, the denuded landscapes of Mt Lyell in Tasmania and the pollution of the Finniss River in the Northern Territory, have contributed to this. However, many of the country's eminent environmental scientists are now recognising that the modern mining practices of today's industry pose far less of a threat to biodiversity than many of Australia's more extensive land uses such as agriculture (see Beattie 1995; Saunders *et al.* 1996; and White 1997).

Professor Andrew J. Beattie, Macquarie University, Sydney, has stated in his book, 'Biodiversity – Australia's Living Wealth', which provides one of the most comprehensive accounts of Australia's biodiversity, that:

'*Australia is leading the world in many kinds of restoration technology, especially for mine sites*' (Beattie 1995, p124).

He goes on to state:

'*(researchers and the) ... environmental officers of many mining companies are beginning to understand how to re-assemble species into communities that have a chance to grow, develop and rebuild local biodiversity. This is a major advance over previous techniques, which usually involved merely covering the site with a layer of green, often exotic grasses or pine trees.*'

Further evidence of Australia's international reputation in mine-site rehabilitation is provided by the inclusion of Alcoa of Australia Ltd on the United Nations Environment Program's Global 500 Roll of Honour for its rehabilitation of jarrah forest ecosystems in Western Australia following bauxite extraction.

3.5.3 Willingness, interest and obligations

In many instances, the current operations within the minerals industry are taking the initiative to rehabilitate historical operations that occur within their mineral lease (Section 3.1: Mount Weld Pastoral Company Pty Ltd). It has been argued that, in some remote, arid areas, there has been an over allocation of resources for mine-site rehabilitation with the costs disproportional to the expected benefits (Johnston 1993). It is acknowledged however, that mining is a temporary land use and that mined land should be left in a manner that will not compromise the activities of subsequent users (Farrell 1993).

In ensuring that mined lands are effectively rehabilitated, both the industry and Government prefers self-regulation through the use of education and encouragement as opposed to the use of a 'big stick' by regulators (Fitzgerald 1993). In ensuring that self-regulation is effective, the minerals industry has produced the Australian Minerals Industry Code for Environmental Management and, in conjunction with Environment Australia, has published a number of modules containing information on a variety of topics related to best practice environmental management in Australia's minerals industry.

3.5.3.1 The Code for Environmental Management

As reported by MCA (2000), by signing the Code, signatories voluntarily commit to excellence in environmental management through seven principles:

- Accepting environmental responsibility for all actions;
- Strengthening relationships with the community;
- Integrating environmental management into the way we work;
- Minimising the environmental impacts of activities;

- Encouraging responsible production and use of products;
- Continually improving environmental performance; and
- Communicating environmental performance.

Commitment to the Code brings with it a number of obligations. In summary these are:

- Application of the Code wherever the company operates;
- Production of an annual public environmental report within two years of registration;
- Completion of an annual Code Implementation Survey to assess progress against implementation of Code principles; and
- Verification of the survey results, by an accredited auditor, at least once every three years.

However, in a review of mining environmental reports, the World Wildlife Fund believe that, as it currently stands, such a wide spectrum of environmental and social performance complies with the code (from very poor level of performance to industry leaders) and stakeholders such as WWF are unable to use compliance with the Code as assurance that an acceptable level of environmental and social performance is being met (WWF 2000).

3.5.3.2 Environmental management and regulation

Bradfield *et al.* (1996) provide an indicative list of legislation relevant to the minerals and energy industry. As noted by Brooks *et al.* (1996), the introduction of the *Environmental Protection (Impact of Proposals) Act 1974* by the Commonwealth Government was one of the most significant pieces of Federal environmental legislation, requiring the preparation of an Environmental Impact Statement (EIS) for those projects needing approval at the Federal level. However, this Act was subsumed by the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act) on 16 July 2000. Under this new legislation, actions that are likely to have a significant impact on a matter of national environmental significance are

subject to a rigorous referral, assessment, and approval process. An action includes a project, development, undertaking, activity or series of activities.

The Act currently identifies six matters of national environmental significance:

- World Heritage properties;
- Ramsar wetlands of international significance;
- Listed threatened species and ecological communities;
- Listed migratory species;
- Commonwealth marine area; and
- Nuclear actions (including uranium mining).

Because Australia's States are responsible for environmental regulation within their own borders, they have also developed their own procedures for assessing the environmental implications of projects. This autonomy has led to differing environmental legislation around the country. For instance, in Queensland the Mineral Resources Act 1994 (and subsequent amendments to) requires the holder of a mining lease to have developed an approved Environmental Management Overview Strategy (EMOS), which is an agreement between government and the mining company or leaseholder to undertake environmental protection measures and rehabilitate disturbed land. There are similar formal requirements for environmental management plans in NSW. Many of these kinds of regulations include a system of security deposits to cover the cost of rehabilitation. Unfortunately, in some cases, these security deposits have been set too low, which, coupled with inflationary pressures, has meant that some rehabilitation has been subsidised by State Governments (Lee 1999).

According to Bradfield *et al.* (1996), there have been several key trends in environmental management and regulation within Australia during the past decade. The first of these was the development of more holistic and integrated legislation in recognition of the systemic nature of most environmental issues. As such, all States have now enacted environmental protection legislation, which subsume the many previous single-issue pieces of legislation. This integrated legislation is often now administered by a single government organisation. The second important trend has been the increasing Australian and international concern for the environment, which

has led to increasingly stringent legislation being passed in a number of Australian States to further discourage activities that damage the environment.

With all this in mind, there has been a significant move in the past five years towards much greater self-regulation in the Australian minerals industry, and far less emphasis on the traditional 'command and control' approach to environmental management. The major elements of this move towards self-regulation include the development and update of the Australian Minerals Industry Code for Environmental Management, self-audit of Code signatories to ensure compliance with self-imposed standards, and the development of pollution reduction targets and incentive schemes. With regard to the latter, the NSW EPA introduced a 12-month, pilot tradeable salinity credits scheme in 1995, as a means of using economic incentives to limit saline discharges in the Hunter River during low flows to protect the river's ecosystems and the interests of irrigators (EPA NSW 1997). This innovative approach differs markedly from the traditional approach of progressively increasing the complexity and restrictiveness of license conditions.

3.5.3.3 National Parks and conservation reserves

Within Australia there are 3216 national parks and conservation reserves, the majority of which have been declared and managed according to State legislation. The Commonwealth Government declares and manages parks and reserves only on land owned or leased by the Commonwealth, in Commonwealth waters and on Aboriginal land leased to the Commonwealth, with the remaining areas being under State management. As there is no coordinating legislation between the States, over 50 designations have been developed to determine the activities of these areas, although the occurrence of national forums and joint programs do exist to ensure that these protected areas achieve a common purpose. Management of these different designations vary from strictly protected areas with limited public access, to areas where recreation is encouraged but resource development is not, to multiple use areas where resource utilisation, recreation and nature conservation are all practised.

Exploration activities have identified more than $10 billion worth of mineral deposits that are alienated in conservation reserves and other developments across Australia (Allan *et al.* 1988; Allen *et al.* 1990; Batini 1997). There are many cases both for and

against mineral extraction occurring within National Parks and other areas designated for conservation. The respective arguments often depend on whether the author is sympathetic to the minerals industry or the conservation movement, as the opposing views appear irreconcilable (Batini 1997).

Frank Batini, a leading environmental manager within the Western Australian Department of Conservation and Land Management has reported that, as of 1997, of the recognised vegetation types outlined within Beard's vegetation mapping for Western Australia;

- 21 percent were adequately reserved,
- 34 percent were represented below accepted adequate levels,
- 47 percent were not represented at all in formal reserves.

Given then that a large percentage of these vegetation types are not adequately reserved and that some of the existing reserves exist only as a result of their poor suitability for alternative land uses (Section 2.1.3), conservationists state that adequate resources exist outside of reserves to placate the minerals industry (Batini 1997). Indeed, it is claimed that given the highly fragmented nature of many of our ecosystems, excision of a mineral lease from an existing National Park is likely to have an effect on ecosystem structure and community integrity (Batini 1997).

The minerals industry recognised the need for ecologically sustainable development long before the Commonwealth government published the National Strategy for Ecologically Sustainable Development in 1992. The minerals industry cites best practice techniques that are less intrusive than historical operations and operations where the post-mined rehabilitation is comparable to analogue sites. Indeed, studies contrasting the successional patterns within the post-disturbance environment following sand mining against fire have suggested that the response to the two disturbance events is comparable. The variation between the two events occurs in the time scale, with restoration of the mature community being more rapid within the post-fire environment (Fox 1990). When operating within sensitive environments, the minerals industry has demonstrated that its activities may be compatible with conservation objectives provided that planning, monitoring and rehabilitation objectives are rigorous (Allan *et al.* 1988; Allen *et al.* 1990; Fitzgerald 1993; Lewis 1996). The area of a mineral lease that is impacted by mining is frequently very small

and, in addition, due to effective management of these areas outside of the active area (fire, feral animal control, weed control), the mineral lease may be in a more 'pristine' environment than the surrounding National Park (Webb 1990).

Examples of multiple land use compatibility between development and conservation include the Ranger uranium mine adjacent to the World Heritage-listed Kakadu National Park and absorption of the rehabilitated Bridge Hill Ridge mineral operations into the Myall Lakes National Park. While these operations may at first seem inappropriate as examples of multiple land use compatibility due to the public outcry associated with these operations, at both locations, the minerals industry undertook rigorous planning and monitoring to minimise its environmental impact.

The uranium orebodies at Energy Resources of Australia Ltd mine at Ranger were identified by an airborne radiometric survey in 1969, and open-cut mining commenced in 1980. In the 20 years that the mine has been in development and operation, there have been no environmental impacts of the mine on the values of Kakadu National Park (EIA Network 1998). Monitoring by the Supervising Scientist has determined that any elevation of radionuclides above background levels in the water or air in the World Heritage property has been within the established limits and has not harmed the surrounding ecosystem or the people that utilise these resources (EIA Network 1998).

Mineral Deposits Ltd's (MDL) Bridge Hill Ridge mine commenced mineral sands operations in the mid-1970s. Progressive rehabilitation of the mined out areas was undertaken during operations. In planning for rehabilitation that would facilitate a rapid plant cover to stabilise the dune surface and promote the return of a self-sustaining forest ecosystem similar to the pre-mined environment, MDL pioneered the direct replacement of the topsoil that is still promoted as best practice today (Lewis 1996). Reshaping of the tailings has ensured that the major topographical features of the landscape has remained intact, the vegetation is developing into mature communities, fauna have recolonised the post-mined environments and succession is proceeding along desired pathways (Lewis 1996). Relinquishment of much of the mined area has taken place, with the post-mined environment being absorbed within the Myall Lakes National Park.

Case Study

Iluka wetlands – sand mine pits into a sustainable wetland system

It is estimated that more than 70% of the natural wetlands in Western Australia have been lost since European settlement due to filling and draining. This has resulted in a shortage of habitats for aquatic biota and waterbirds, including migratory birds. Rehabilitation of mine-voids or pit-lakes into sustainable ecosystems is considered to be a positive contribution towards restoration and enhancement of aquatic biodiversity. The Iluka wetlands at Capel (formerly the RGC Wetlands Centre), 200 km south of Perth, Western Australia, are an excellent example of such a venture. In addition, it is one of the few created wetlands for conservation of biodiversity whose ecological and taxonomic components have been monitored throughout its entire history with the involvement of a multidisciplinary team of experts. The ultimate goal of the project was to create habitats to attract a maximum diversity of birds. The chronological sequence of development of the chain of wetlands at Capel has been presented by Brooks (1991), Brooks and Nicholls (1996) and Doyle and Davies (1998).

The success of the rehabilitation of the Iluka wetlands from acidic pit-lakes can be attributed to: concept plans implemented in 1987, 1990 and 1993 involving intense modifications of the physical habitat of the lakes to cater for birds, planting of appropriate emergent fringing and peripheral vegetation, colonisation of macrophytes especially Charophytes, macroinvertebrates, fish, frogs and reptiles, and continuous biological monitoring and biodiversity surveys involving all groups of biota by researchers, as part of the management strategy.

The wetlands centre has become an educational resource for primary, secondary and tertiary students. The centre is open to the general public, and there is a constant array of visitors.

3.6 Rehabilitation Progress in Impacted Areas

The effort that the minerals industry has undertaken to develop a greater understanding of native ecosystems, and hence allow improved rehabilitation of impacted areas, has dramatically improved the quality of the post-mined areas. The applicability of these techniques to broad scale areas has become increasingly relevant

as modern mining techniques impact larger areas in search of viable resources and economies of scale.

Historical operations were frequently underground and had a lower surface impact (despite being greater in number with little attempt to manage environmental impacts on surrounding surface areas). Today, large open cuts are more common within the minerals industry, and thus the impact of the mine void is greater than the smaller underground operations. Waste dumps and tailings resulting from modern mineral operations, however, are rehabilitated to a high standard, whereas, historically, a lack of community concern and understanding for environmental issues meant they often were not rehabilitated at all.

New mines may also have satellite pits with linking haul roads and infrastructure that impact on biodiversity function as previously mentioned, i.e. hydrology and fauna movement. Thus, to minimise the impact on the local environment, the new mines are required to be more rigorous in monitoring and planning.

3.6.1 Practices

3.6.1.1 Topsoil manipulation and seedbed management

The importance of viable topsoil for rehabilitation of degraded areas and mined lands has been emphasised in many publications (Iverson and Wali 1982; Koch and Ward 1994; Grant and Koch 1997; Izaurralde *et al.* 1998). It is estimated that up to 77% of species within the rehabilitated bauxite mine pits within the jarrah forest community of Western Australia are derived from topsoil seed reserves (Koch and Ward 1994; Ward *et al.* 1996). Soil seed banks are often the only source of seeds of many species that are no longer present in the vegetative biomass, particularly so for early successional species which contribute to the development of the seed bank in the initial period following disturbance (Leck and Leck 1998).

In many of the older sites where the minerals industry was active, soil seed bank dynamics were poorly understood, and the location of germinable seed within the soil horizons was equally unknown. Consequently, the upper horizons containing the soil seed reserves were mixed with soil lower in the profile upon stockpiling or return of

the soil to reformed mined pits. This resulted in a dilution of the seed bank to depth from where some species were unable to emerge.

Seed burial studies have revealed an inability of many species to emerge from depths greater than 5–10 cm (Grant *et al.* 1996; Rokich *et al.* 2000). In addition, stockpiling of soils have been found to decrease the viability of the soil seed bank (Tacey and Glossop 1980; Nichols and Michaelsen 1986). These findings would help explain the results of Tacey and Glossop (1980) and Nichols and Michaelsen (1986) who found improved recruitment and species composition amongst emergents when the upper 5 cm of the topsoil was transferred, without stockpiling, onto the rehabilitated mine pit. Although it is not always feasible to undertake direct replacement of topsoil, the majority of mineral operations utilise this technique whenever possible and minimise the time that topsoil resources are stockpiled.

There have been reports regarding the use of rock armouring to minimise erosion and facilitate microclimates which assist in the establishment of native vegetation (Williams *et al.* 2001). The use of topsoil alone on steep batters may not resist erosion and can result in lower rates of vegetation establishment if the topsoil is utilised on steep inclines (Williams *et al.* 2001). The importance of topsoil as a source of microorganisms, organic matter and nutrients is recognised, and hence techniques have been investigated to concentrate the topsoil seed bank (Ward and Koch 2000). When applied in combination with rock pitching, this technique may allow for the application of the topsoil to promote biodiversity conservation within rehabilitated areas, while retaining the protection of the rock pitching as insurance against unforseen circumstances, e.g. heavy rainfall events or fire, that may expose the surface to erosional forces.

In an attempt to enhance the establishment of rehabilitated areas, many companies utilise fertilisers to facilitate the initial growth and establishment of seedlings within rehabilitated areas until the organic matter can build up and nutrient cycling processes can begin. The incorporation of phosphorus-absorbent residues from bauxite refining to sandy soils to increase soil fertility has also been investigated with some success (Ward and Summers 1993).

Repeated passes by heavy machinery over the soil surface during rehabilitation operations may result in the development of subsurface hardpans. Deep ripping along

the contour to 1.5 m by a DC10 bulldozer and winged ripping tine can assist to break the subsurface compaction layer and enable long-term survival of deep-rooted species within the rehabilitated area (Ward *et al.* 1996). Soil scarification techniques, such as moonscaping, can assist in the development of microclimates, enable water infiltration and therefore minimise erosion, as well as assist in the development of source/sink areas to facilitate nutrient cycling.

3.6.1.2 Broadcast seed technology

The use of broadcast seed within a revegetation program can assist in the establishment of native species. The use of topsoil has been found to account for a large majority of recruits within post-mined lands in some communities (Koch and Ward 1994; Ward *et al.* 1996); however in many situations (for example where topsoil has been stockpiled or contained a high percentage of weeds), the use of broadcast seed is a vital component for re-establishment of native communities.

The importance of broadcast seed to revegetation programmes is indicative of the effort that has been undertaken into maximising recruitment from these seed reserves. A substantial effort has been undertaken by the industry leaders to maximise recruitment from this seed resource from seed handling and storage techniques, the timing of broadcast seed application, the development of dormancy-breaking techniques and engineering solutions to produce cost-effective techniques of applying broadcast seed.

The minerals industry utilises direct-seeding techniques to rehabilitate the post-mined lands, while the planting of tubestock seedlings represent a minor component of the rehabilitated landscape. In many cases, the protocols that the minerals industry has developed are applicable to the rehabilitation efforts of other groups (e.g. Landcare, bush regeneration) and hence have beneficial applications beyond the minerals industry. For example, as the success of broadcast seed applications is reliant on weather patterns and post-germination rainfall, to maximise the chances of successful establishment within arid areas, the minerals industry typically pre-treats only 30-50% of the broadcast seed to ensure that a residual seed bank remains should the initial recruits fail to become established (MCA 1998).

The minerals industry is aware of the importance of provenance in revegetation and, despite the difficulties in determining how 'local is local', the industry attempts to source seeds as close to its operations as possible.

3.6.1.3 Smoke-stimulated germination

The application of smoke to seeds has been found to promote germination in a large number of plant species from a number of ecosystems across Australia and hence, there is a great potential for smoke to assist in the revegetation of the post-mined environment (Dixon *et al.* 1995; Enright *et al.* 1997; Roche *et al.* 1997; Kintrup and Enright 1999; Read and Bellairs 1999).

Smoke may be utilised as a treatment to overcome dormancy within broadcast seed prior to sowing or storage. These effects have been reported to last for at least a year following smoke application to the seeds and storage under appropriate conditions (Brown and van Staden 1997).

Studies undertaken in *Eucalyptus* woodlands of Victoria have suggested no adverse affects of smoke on germination of species that are not smoke-responsive (Enright *et al.* 1997). This finding supports the use of smoke as a routine treatment for broadcast seed and is a technique utilised by some mining companies in Australia to improve recruitment and biodiversity within rehabilitated areas.

3.6.1.4 Long-term sustainability of rehabilitated lands

An important aspect of rehabilitation is investigation into the long-term sustainability of the restored ecosystem. This has necessitated research into many ecological processes, e.g. seed biology, fauna habitat reconstruction and completion criteria.

Following mineral extraction, the development of a mature, self-sustaining forest that fulfils all functions of the unmined forest will take a long period to develop as mineral extraction leads to a complete renewal of ecological processes within the active area (Fox 1998).

The use of fire within post-mined landscapes has been investigated and found to be successful in further enhancing species establishment (Koch 1992; Smith *et al.* 2000). More evidence of the slow development of self-sustainability was presented in the

findings that 11–15 year-old rehabilitation within the jarrah forest community in south-west Western Australia had to be treated with a different fire regime to the surrounding forest, due to different vegetation and fuel composition (Grant *et al.* 1997).

Plant disease has the potential to impede the development of community sustainability. The fungal disease *Phytophthora cinnamomi* (dieback), for example, has the potential to be dispersed across areas where the minerals industry is active through stockpile drainage, transfer of infested soil by way of vehicle movement and drainage from infected mining and rehabilitated mined areas (Crosbie and Colquhoun 1999). Studies have suggested that the total potential for dieback spread due to mine clearing (under current practices) in SW Western Australia is an average of 0.01 ha per hectare cleared for mining (Crosbie and Colquhoun 1999).

Despite setbacks such as disease, and the slow development of old rehabilitation toward sustainability, studies are indicating that functional ecosystems can develop on rehabilitated mines (Ward 1998; Tongway *et al.* 1997a). Early recognition was made of the need to develop whole ecosystems, rather than just ameliorating physical and chemical conditions and establishing vegetation (Nichols and Bartle 1986). As revegetation initiatives have become more successful, many flora studies have focussed on researching those species that are hard to germinate. To achieve maximum plant diversity, a combination of the use of applied seed, soil seed reserves and recalcitrant species are being utilised by the majority of operations.

3.6.1.5 Vegetation management

Exotic species of plants (and animals and microorganisms) have the potential to significantly alter Australia's biodiversity (BDAC 1992). In some cases this has already occurred, to the detriment of the ecosystem. Problems can also occur where native species extend their range. One of the objectives in the national strategy for biological diversity is to encourage the use of local indigenous species in rehabilitation and discontinue the use of non-local species (native or exotic) to halt the spread of species not local to areas (BDAC 1992). In the past, exotic nitrogen fixers or nurse crops were used to help improve the mine soils, provide organic matter and stabilise the soil surface (Fox 1984). Many of these species have persisted in the post-

mined landscape via recruitment from the soil seed bank with at least one (Bitou Bush; *Chysanthemoides monilifera* ssp. *Rotundata*) having become listed among Australia's top weeds of national significance as a result of being actively planted (Csurhes and Edwards 1998). The newer strategies have reduced these old practices, which helped spread exotic species and endangered biodiversity.

In areas that have been mined, the magnitude of a weed problem is a function of the availability of plant propagules (Panetta and Groves 1990). Where mines are located in pastoral land, there is ample opportunity for weed species to invade. For example, in the Ord River region, a development vegetation survey found the introduced *Calotropis* sp. growing densely along riverbanks and in the unused agricultural fields, along with the introduced *Parkinsonia* sp. (Meagher and LeProvost 1980). Often, because native seedlings remain small for long periods, they are disadvantaged when competing with introduced species (Panetta and Groves 1990). Most rehabilitation practices now refrain from using exotic species in the rehabilitation mix or utilise sterile, non-seeding species to assist in soil stabilisation to enable native species establishment. Where the species that are utilised as cover crops to assist in soil stabilisation are perennial, careful monitoring is essential to ensure that this species does not extend its range beyond the immediate area.

3.6.1.6 Habitat development

A number of mining companies are addressing the fact that some faunal habitats will take a long time to develop in rehabilitation, for example hollows in trees, logs and stumps, piles of branches, and tall perches. Many companies are constructing faunal habitats from materials pushed aside during clearing operations prior to mineral extraction. The habitats may consist of logs, log piles, stumps, rocks or various combinations of these, with soil carefully placed around them. Operational procedures describe how habitats should be constructed to provide shelter niches and opportunities for animals to burrow underneath. The reintroduction of habitat logs following mining has assisted in the succession of native fauna within the post-mined environment (Sainsbery *et al.* 1999). Various mammal and reptile species, including the rare Southern Brown Bandicoot, have been recorded utilising these constructed habitats and several other rare species are expected to do so (Nichols pers. comm.).

The use of nest boxes in rehabilitation have been undertaken by the minerals industry and have been found to be utilised by a range of small arboreal mammal and reptile species.

Consolidated Rutile Limited builds both log piles and tall perches in its rehabilitated mineral sand mines on North Stradbroke Island. The perches are designed to attract raptors and other bird species that are expected to bring plant seeds and nutrients into the rehabilitated area.

3.6.1.7 Monocultures and exotic species

Whilst rehabilitation using monocultures and exotic species was acceptable in the past, in most mines it is no longer so today. The activities of Alcoa World Alumina Australia are a good example of the differences in old and contemporary mining and their effects on biodiversity through rehabilitation procedures.

Alcoa World Alumina Australia has been mining bauxite in the jarrah forest of southwest Australia for over 30 years (Nichols *et al.* 1991). This period of time has encompassed continuing changes in the standard of mine-site rehabilitation. Up until the early 1970s, the post-mined environment was rehabilitated with exotic pines (e.g. *Pinus pinaster*), and up until the late 1980s, the post-mined environment was rehabilitated with eastern states eucalypts (e.g. *Eucalyptus resinifera*) (J. Koch pers. comm.). Exotic pine plantations were sown with no understorey seeding (Grant and Koch 1997). Pines are not conducive to the development of a native understorey, and hence few understorey species established leaving essentially a monoculture of pine trees.

Pine plantation rehabilitation ceased in the early 1970s. Non-local eucalypt species were then chosen for rehabilitation due to uncertainty regarding the susceptibility of local overstorey species to dieback disease caused by *Phytophthora cinnamomi.* Sites planted with eastern states eucalypts were seeded with an understorey of local native plants. While these may have had a higher biodiversity than the pine sites, they were not conserving the full range of biodiversity that existed in the pre-mined environment.

To help with difficulties in establishing trees in the elevated salt conditions that may be experienced when mining moves into the lower rainfall eastern jarrah forest, an

elite variant of jarrah was developed that could tolerate the elevated salt balance (Kabay and Nichols 1989). Although the genetic superiority ensured that the trees did well, these variants were propagated via clonal techniques which significantly reduced genetic biodiversity within the rehabilitated stand.

There were a number of reasons at the time for Alcoa to consider the use of exotic species and the development of monocultures, viz. resistance to dieback, future timber potential and the cheaper and less labour intensive option than establishing some of the difficult native species. Land could be stabilised and saved from problems such as erosion much quicker using species better suited to that purpose. The rehabilitation aims of the minerals industry as a whole have changed considerably over the years in response to improved knowledge, increased awareness of the importance of biodiversity (Ward 2000), and developing technology. Alcoa's current strategy of restoring a stable, self-generating forest ecosystem which maintains or enhances water, timber, recreation, and conservation values is a more suitable plan, when considering the complex, high diversity system that is the jarrah forest (Nichols *et al.* 1991) (Figure 3.1).

(a)

(b)

Figure 3.1. Alcoa World Alumina Australia operations in south-west Western Australia during mineral extraction (a) and after rehabilitation (b) showing rehabilitation that is consistent with the values of the surrounding forest ecosystem.

3.6.1.8 Aquatic habitats

Two case studies from tropical northern Australia have been selected to illustrate contrasting mine-site environmental management practices, particularly with respect to mining impacts on downstream aquatic biota and its biodiversity. The first case study describes a decommissioned and rehabilitated copper-uranium mine, Rum Jungle, that caused severe downstream ecological impact due to acid rock drainage (ARD) from the site. The second case study describes the extensive chemical and biological monitoring programs that have been developed and implemented at an operating uranium mine (Ranger), to ensure that no adverse downstream ecological impact occurs. The severe ecological impacts observed at Rum Jungle, for example, increased public awareness of environmental issues, and played an important role in the requirement for the rigorous environmental management practices currently in place at Ranger.

Both mine sites have been extensively studied over the last 25 years, and hence provide a long-term dataset to quantify both actual and predicted ecological effects in relation to measured contaminant (e.g. metal and/or pH) levels. This has led to an increasing number of studies investigating the mechanism(s) of metal-organism interactions (i.e. uptake, toxicity), particularly an understanding of biological responses in relation to aqueous metal speciation, including how water quality variables (e.g. pH, hardness, alkalinity and dissolved organic matter) may influence trace metal bioavailability. Such an approach enhances the capability of predicting the biological effects of metals across a range of water chemistry conditions. The measurement of metal speciation and bioavailability permits national numerical guidelines for protecting aquatic ecosystems to be potentially relaxed on a site-specific basis.

The Finniss river below the Rum Jungle mine has been contaminated by ARD (principally copper (Cu), zinc (Zn), manganese (Mn), uranium (U), aluminium (Al), sulfate and acidity), a product of oxidation of sulfide minerals in the mine waste, originating from multiple sources. Between 1983 and 1985, the site was rehabilitated by the Federal Government at a cost of A$18 million. This resulted in a substantial (*ca.* 70%) reduction in the annual loads of trace metals and sulfate entering the Finniss River. These reductions in contaminant loads have been followed by an

increase in the diversity and abundance of freshwater biota downstream from the mine.

Ranger is a currently operating a uranium mine that commenced mining and milling operations in 1981. It is located a short distance upstream from the floodplain of Magela Creek, and is surrounded by the World Heritage listed Kakadu National Park. In contrast to Rum Jungle, the Ranger mine is mono-metallic, with uranium being the only metal present in significant concentrations in the orebodies. Uranium is of potential ecotoxicological significance in the mine wastewaters. The highest quality water stored on site may be released into Magela Creek during the wet season (December to April), subject to strict legislative requirements and regulations. Indeed, over 57 Acts of Parliament and associated Regulations cover mining and milling operations at Ranger. It is arguably the most regulated mine in Australia. Biological monitoring of Magela Creek and toxicity evaluation of the mine wastewaters have been undertaken on a continuous basis for some 20 years. No detectable impacts of mine waste waters on aquatic ecosystems downstream of the Ranger mine have been demonstrated to date (see Section 3.6.1.8.2).

3.6.1.8.1 Rum Jungle copper-uranium mine

Rum Jungle is a decommissioned copper-uranium mine that was operational from 1953 to 1971. At the time of development, there was no legislative requirement for mining projects to undergo environmental impact assessment, so the environmental damage which subsequently resulted from mining operations at Rum Jungle was not of significant interest at the time, but was of enormous concern later on. The major environmental impact at Rum Jungle was the pollution of the East Branch and the main channel of the Finniss River (Jeffree and Williams 1975; 1980) (Figure 3.2).

It was obvious by the mid 1960s that effluent from the treatment plant and leachate from the mine waste dumps were adversely affecting the riparian vegetation at the mine site. More importantly, the aquatic biota in both the East Branch of the Finniss River, which flows through the mine site, and the Finniss River downstream of its confluence with the East Branch, were being severely impacted. The major pollutants in the East Branch and the Finniss River were copper, zinc, manganese, sulfate and acidity.

Over the period 1969/70 to 1973/74, the calculated annual loads of copper, zinc, manganese and sulfate delivered into the Finniss River system ranged from 44 to 106, 22 to 30, 46 to 110 and 3300 to 13000 tonnes, respectively (Jeffree *et al.* 2000).

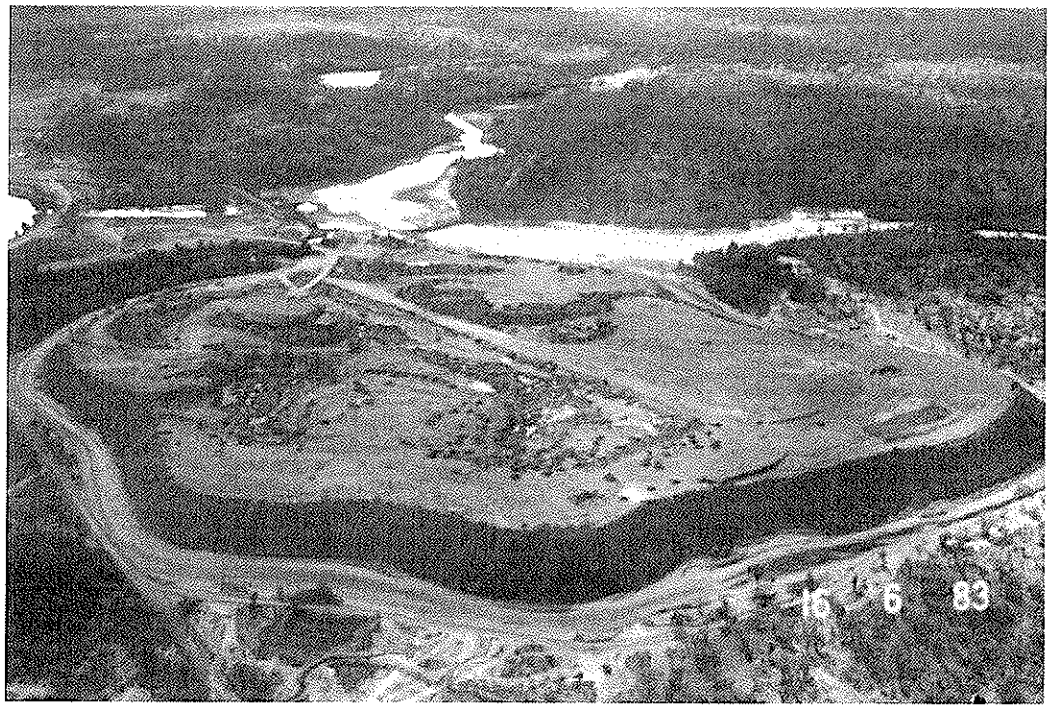
In the East Branch at the end of the dry season, when water flow ceased to the Finniss River, total concentrations of copper in the surface waters of billabongs (permanent water bodies) were as high as 250 mg L^{-1}. In the Finniss River, the highest concentrations of contaminants (copper, zinc, manganese, cobalt (Co), nickel (Ni), uranium, sulfate, and acidity) occurred when convective thunderstorms caused early wet season flows in the East Branch, and these coincided with low flows from the remainder of the Finniss catchment. During these 'first flush' events, fish-kills in the Finniss River were observed to extend between 15 km, and somewhat less than 30 km, downstream of its confluence with the East Branch. These toxic events were most likely responsible for the general reduction over the subsequent dry season in fish diversity and abundance (Jeffree and Williams 1980).

In the East Branch downstream of the mine, no rooted or submerged plants were observed, and live bank-side *Pandanus* were rare, although their dead stumps were present. The absence of the stabilising effect of bank-side roots was associated with shallow gullying, gently sloping banks and considerable deposits of sand along the banks, indicative of erosion and consequent deposition. Fish were virtually absent from the pools of the East Branch, although the physical and structural characteristics of these pools were similar to those at sites unexposed to pollution, that contained about seven species. A number of fish species were found in the unpolluted tributaries of the East Branch (Jeffree and Williams 1975).

Many groups of Insecta, that are indicators of an unpolluted state, as well as the phyla Mollusca and Porifera, were absent in the East Branch downstream of the mine.

In the Finniss River there was also an appreciable decline in fish diversity and abundance for at least 15 km downstream of the confluence with the East Branch. Sites in the Finniss River immediately downstream of the confluence with the East Branch showed lower numbers of species and individuals than those upstream, or further downstream. There was a gradual recovery in both the number of species and individuals in the Finniss River with increasing distance downstream of the confluence with the East Branch.

(a)



(b)

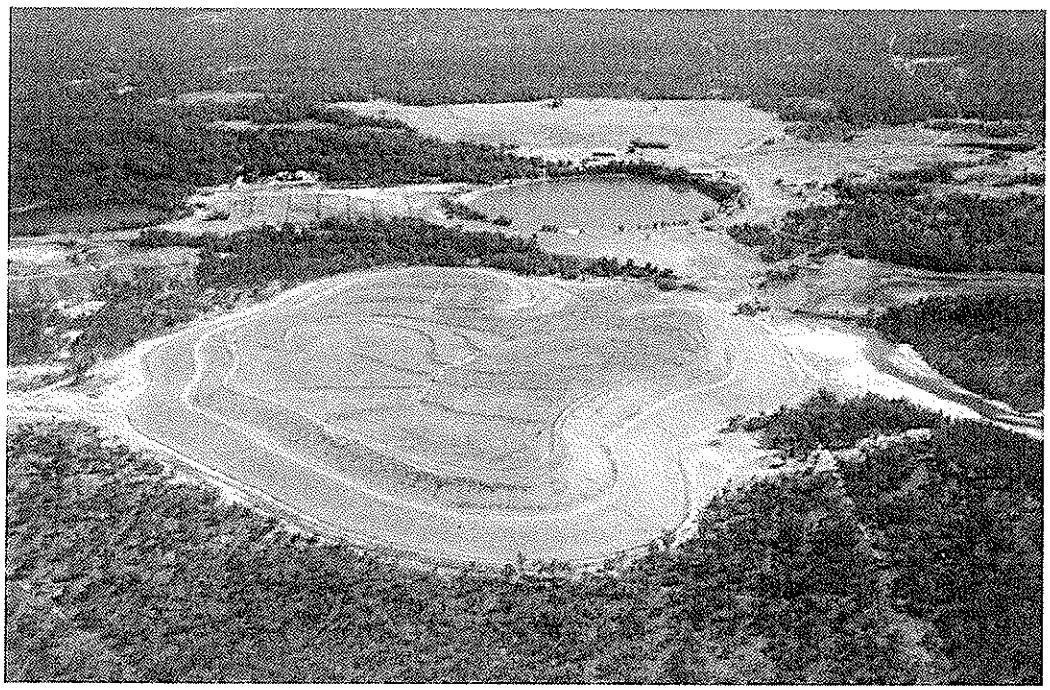


Figure 3.2. Waste dump at the Rum Jungle copper-uranium mine prior to rehabilitation (a) and after rehabilitation (b).

Although the maximum and median concentrations of copper, zinc, manganese and uranium have decreased substantially in the surface waters of the East Branch (and the Finniss River) following remediation of the Rum Jungle mine, the present concentrations still exceed the national water quality guidelines (ANZECC and ARMCANZ 1999) for the protection of freshwater ecosystems by a factor of 220, 45, 20 and 11, respectively (Jeffree *et al.* 2000).

Field studies that commenced in the early 1990s indicated that there has been only a small degree of recovery in the riparian vegetation of the East Branch following the remediation of the mine site, and associated reductions in annual contaminant loads and surface water concentrations. The description given above for the pre-remedial level of impact still generally applied.

A study of macroinvertebrate community composition and associated water quality in the East Branch, that was very similar in design to pre-remedial investigations in its choice of sites and sampling methodologies, but more quantitative in nature, was conducted. This study indicated no apparent increases in the number of macroinvertebrate taxa in the section of the East Branch receiving effluent from the mine, following remediation of the Rum Jungle mine site. However, there does appear to have been some ecological improvement in the lowest reaches, indicated by the presence of freshwater Atyid shrimps (Jackson 1993).

Sampling of benthic algae in the East Branch was undertaken from 1993 to 1995 during the period of recessional flow that occupies the early dry season. An obvious, and roughly exponential gradient of pollution (trace metals, sulfate and acidity) develops during this time. At the site of lowest pH (3.0) and highest metal concentrations (e.g. 30 mg Cu L^{-1}), the species richness was less than 10% for diatoms, and 50% for non-diatoms, relative to control sites. Collectively, the algal species richness in the East Branch downstream of the Rum Jungle mine-site was, on average, less than 50% of that typically found at control sites. Furthermore, the relative contribution of three major algal groups (blue-green algae, green algae and diatoms) to the overall number of taxa present at a site changes with position along the pollution gradient. The blue-green algae were absent from the most polluted end of the gradient leaving only green algae and a few species of diatoms. Overall, the quantitative data on benthic diatoms provide a potentially useful basis for measuring

any further improvement in the ecosystem health of the East Branch, based on primary producers at the base of the food chain (J. Ferris pers. comm.)

Within the Finniss River, ecological studies conducted following remediation have shown the following environmental improvements over the first 15 km downstream of the confluence with the East Branch:

1. the indicator fish species, eel-tailed catfish, banded grunter (*Amniataba percoides*) and archer fish (*Toxotes chatareus*), that were virtually absent, have returned;
2. fish diversity and abundance has increased to levels comparable to sites unexposed to ARD contaminants; and
3. ecotoxicologically-sensitive species of crustacea (Atyid shrimps, e.g. *Caradina nilotica*) occur within impacted regions of the river where no living specimens were found prior to remediation.

It is important to note that partial ecological recovery is occurring in the Finniss River even though the concentrations of Cu, Zn and Ni in the sediment, and Cu, Zn, Mn, U and Al in the surface waters, often substantially exceed their respective national guideline values for the protection of aquatic ecosystems (Jeffree *et al.* 2001).

Physiological and/or genetic tolerance of freshwater biota in the waters of the Finniss River to metals such as Cu and Zn is an additional mechanism to explain their partial recovery, despite concentrations in the sediments and water that exceed the recommended national guideline values (Gale *et al.* submitted).

3.6.1.8.2 Ranger uranium mine

Over the past two decades, there has been a major research effort to gather environmental data in the Alligator Rivers Region. This is largely as a consequence of the Australian Government's decision to permit the mining and milling of two (Narbalek and Ranger) of the four extensive deposits of uranium known to exist in this area. The Narbalek deposit was mined out and has recently been rehabilitated.

The Ranger deposit (Figure 3.3), located adjacent to Magela Creek, is currently being mined by a conventional, simultaneous mining/milling, open-cut operation. Mining and milling operations are expected to continue until about 2006. In 1996 the Ranger

mine was the third largest producer of uranium oxide in the Western world, accounting for 10% (4149 tonnes) of the world's production (ERA 1998).

As a consequence of heavy wet season rainfall, one of the most important environmental issues associated with the operational phase of uranium mining at the Ranger mine, has been the management of excess water that accumulates within the mine retention ponds. The options for water management are severely constrained as a result of the location of the project area upstream of the Magela Creek floodplain, which forms part of the World Heritage-listed Kakadu National Park. The Ranger mine is the most tightly regulated mine in Australia with respect to discharges of water from the site, and is required to adhere to detailed environmental requirements determined by government authorities (NT DME 1982).

Magela Creek is an ephemeral stream consisting of a series of braided sandbed channels with sandy levees and a series of billabongs and channels that expand onto a seasonally flooded black-soil floodplain. High peak flows occur in response to intense storm events in the upper catchment. High flow conditions (20–2000 $m^3 s^{-1}$) during this period impart a uniform and common water chemistry to Magela Creek. Water quality is very good, reflecting the essentially pristine nature of the catchment. For this reason, the potential for impact of waste water on aquatic ecosystems downstream of the Ranger mine is of particular concern. Hence, any mine waste water releases need to be carefully controlled to minimise ecological detriment. Uranium is the primary element of potential ecotoxicological concern.

Whilst chemical analysis of the surface water of Magela Creek indicates a slight chemical signature from the mine, no deleterious effects on aquatic biota in Magela Creek downstream of the mine have been detected since operations commenced. This record stands in marked contrast to the legacy of substantial downstream impacts from the Rum Jungle mine, and is a testament to the very high standards of environmental management on the Ranger leases.

The following description for the Ranger site describes the use of multi-tiered biological testing and monitoring (using local invertebrate and vertebrate species) to evaluate impacts downstream of the mine site, and the research that is being done to develop locally applicable numerical guideline criteria for solutes (especially

uranium) present in mine water. The framework described should be applicable to other types of mining operations throughout the world.

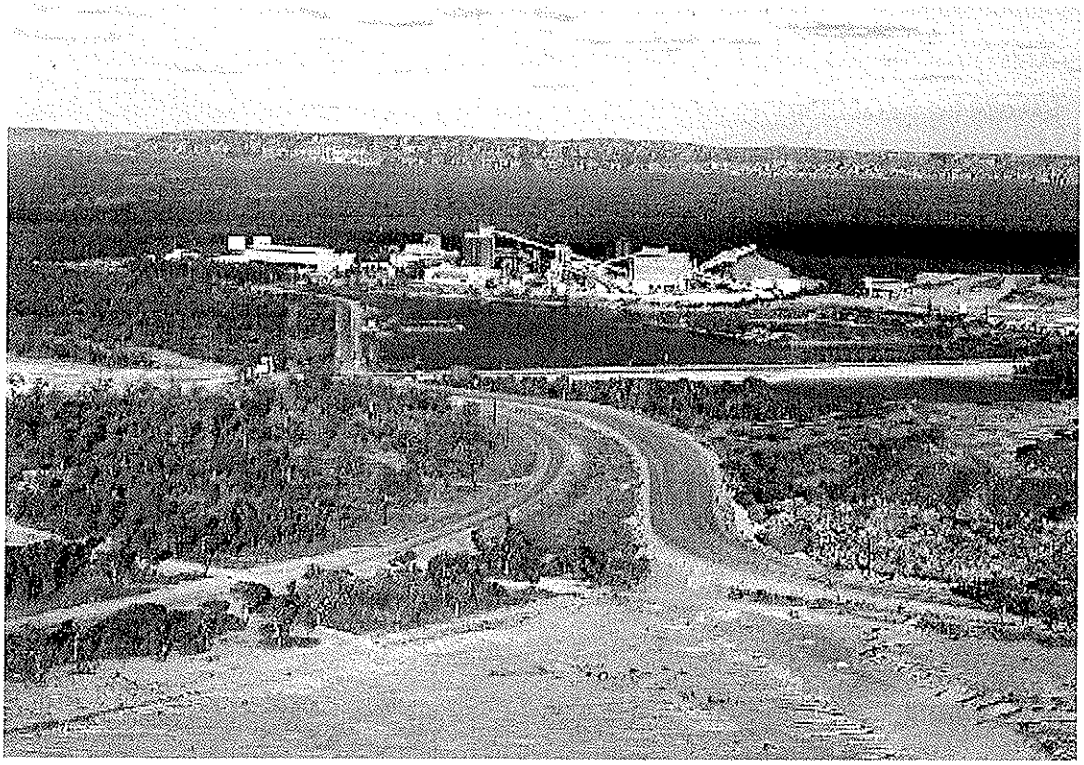


Figure 3.3. Ranger uranium mine adjacent to the World Heritage listed Kakadu National Park.

Biological assessment of mine wastewaters

During the wet season, the quality of the surface water is regularly monitored in Magela Creek, upstream and downstream of the mine-site, as well as water bodies on site, as part of the statutory water quality monitoring program. This is complemented by non-statutory, laboratory-based, pre-release toxicity testing of release water. Creek-side toxicity testing has also been conducted, during periods of water release, since the 1991/92 wet season.

The laboratory and creek-side toxicity tests are complemented by measurements of metals and radionuclides in the flesh of the freshwater mussel, *Velesunio angasi*, collected from a billabong downstream of the mine, to assess the bioaccumulation of

metals and radionuclides over time. Baseline metal concentrations in the flesh of *V. angasi* were determined throughout the Magela Creek catchment, so that future concentrations may be compared. Apart from being a suitable indicator species for biomonitoring over the longer-term, *V. angasi* is also a food source for the traditional (Aboriginal) owners of Kakadu National Park. The annual sampling and analysis of mussels for metals and radionuclides forms the only part of the biological work at the mine that is specified as a statutory (regulatory) requirement in the Authorisation to Operate (NT DME 1982). To date, there is no evidence to indicate that metal or radionuclide concentrations in mussel flesh have increased over time. Furthermore, the metal concentrations in the flesh of *V. angasi* from Magela Creek, downstream of the Ranger mine, are consistent with other minimally-polluted waters (Markich 1996).

As part of a 'whole ecosystem' approach, a program has been developed and trialed since 1994 to assess the health of aquatic ecosystems comprising the Magela Creek catchment (Corbett 1997; Sewell *et al.* 1997). This approach involves comparing the numbers of key trophic levels occupied by organisms in billabongs and riparian areas at test and control sites. It provides a tool for monitoring the health of aquatic ecosystems over long periods to determine if there are any significant ($P = 0.05$) chronic impacts from mine waste water releases. The whole ecosystem approach represents the highest tier in the biological systems approach for assessing mining impacts at Ranger. To date, this approach has not provided any evidence of deleterious impacts on the aquatic ecosystems downstream of the Ranger mine.

Pre-release laboratory toxicity testing of mine wastewaters

The management of excess waste water on the Ranger site during the wet season requires the controlled release of water in most years. Water from runoff and seepage from a waste rock dump typically contains uranium, magnesium and sulfate at levels 500, 250 and 2000 times higher, respectively, than the receiving waters of Magela Creek. The minimum dilution requirements of waste water are initially determined using pre-release toxicity testing. The actual rate of waste water discharge into Magela Creek is determined after consideration of water chemistry (Magela Creek and release water), the flow rate of Magela Creek and the results of biological testing. In short, because the impact of complex and varying waste waters on aquatic

environments cannot be predicted from their individual constituents (cf. water quality standards regulated by chemical analysis), testing the toxicity of mine waste waters on aquatic biota is desirable (Brown 1986).

An important prerequisite for the biological assessment of mine waste waters is the identification of biota in Magela Creek potentially at risk. In a comprehensive discussion on the use of freshwater biota to assess mine waste waters in the Alligator Rivers Region, it was concluded that soft-bodied and gill-breathing organisms (or their life stages) are most at risk from solutes present in the water (Johnston 1991).

The principal features of the toxicity testing approach are as follows:

1. the direct measurement of the change in toxicological responses (lethal and sublethal) of local biota selected from widely different taxa and trophic levels, to the actual wastewater as it is diluted with receiving (Magela Creek) water; and
2. the dilution of the wastewater required to render it harmless can be used as a control parameter to regulate its release; however, this must follow the application of a 'safety' factor to account for the possible occurrence of large undetected risks, additivity of undetected effects and protection of species other than those tested.

The suitability of 19 local freshwater organisms was assessed for their potential use in the pre-release tests. Several organisms were found to be useful, based on sensitivity, representation from different trophic levels (e.g. vertebrate predator, invertebrate predator and invertebrate herbivore) and suitability for laboratory culture. Three of these, [*Hydra viridissima* (green hydra), *Moinodaphnia macleayi* (water flea) and *M. splendida inornata* (chequered rainbowfish)] are currently used to assess the toxicity of waste waters following established protocols.

The principle of the pre-release toxicity tests is to compare organism response in a control water, obtained from a pristine billabong located upstream of the mine, with various dilutions [typically, 0% (control), 0.3%, 1.0%, 3.2%, 10% and 32%] of test water. From these tests, the maximum concentration at which no observed effect (NOEC), and the lowest concentration at which an effect is observed (LOEC), relative to the controls, are determined from statistical analysis (one-way analysis of variance) of the test results obtained prior to the release of waste water.

Creek-side toxicity testing of Magela Creek water

Ranger mine is currently trialing a field-based toxicity testing program to accompany the laboratory-based tests. The aim of the field-based program of testing is to check the efficacy of the pre-release tests and provide an early warning of impacts caused by the mine waste water releases. This is achieved by quantifying and assessing the relative toxicity of Magela Creek water, both upstream and downstream, before, during and after the release of waste water. Such a program is required to achieve the overall environmental protection objective, i.e. no observable effects upon local aquatic organisms in a comprehensive and sensitive biological monitoring program (Humphrey *et al.* 1995).

The results of the creek-side tests have verified the pre-release laboratory toxicity tests. These results are no surprise given that the rate of release of waste water was limited by the requirements imposed by the chemical standards (NT DME 1982) and by the dilution required on the basis of the pre-release toxicity tests. The monitoring data, therefore, confirm the adequacy of the controls imposed on the discharge of water from the mine-site.

A practical *in situ* technique used to assess the effects of mine waste water releases has been the measurement of the reproductive responses of freshwater mussels (*V. angasi*) held in mesh-covered containers buried along the creek edges (Humphrey *et al.* 1990). Adverse effects detected were of short duration only, and localised, being confined to a 'mixing zone' immediately downstream of the discharge point.

Monitoring the community structure of macroinvertebrates and fish in Magela Creek

Macroinvertebrate communities have been sampled from a number of sites in Magela Creek at the end of substantial wet season flows, on an annual basis from 1988 to the present, with the aim of developing a monitoring technique to detect any impact from mining.

Macroinvertebrate data for the 10 year period, 1987/88–1996/97, have been analysed for a site in Magela Creek upstream of the Ranger mine (control) and a site about 5 km downstream (identical to those used for the creekside tests) to illustrate

comparative changes in community structure. The results indicate that changes from year to year in community structure between the upstream and downstream sites are smaller than the natural variability that occurs at the control site. Hence, any change in community structure that has occurred downstream from the mine has not been ecologically significant and may have been due to natural variability.

Additionally, studies of fish community structure in several billabongs have shown no evidence of mining-related impacts since 1994. In one study, the fish in two billabongs, one on Magela Creek downstream of the Ranger mine and the other on a separate 'control' catchment [Nourlangie Creek (a tributary of the South Alligator River in Kakadu National Park)], have been monitored by a visual counting technique, using a canoe with a transparent bow. Multivariate measures of the dissimilarity between these two fish communities indicated that, although there were differences between streams, the differences remained relatively constant over the study period (Johnston and Needham 1999).

In summary, the biological monitoring program at Ranger has shown that operation of the mine has had no detectable impact on a range of sensitive indicators of ecological health and biodiversity.

3.6.2 Performance measures

It is difficult to obtain a national picture of the effectiveness of the Australian minerals industry in addressing issues of biodiversity, as much of the basic data that is required is not readily available throughout the industry. An agreement on completion criteria encompassing biodiversity issues needs to be considered with data collected in a regular, systematic and consistent fashion so that the potential cumulative effects of mining activities on biodiversity can be identified.

3.6.2.1 Environmental reporting and independent assessment

A fundamental obligation of committing to the updated Australian Minerals Industry Code for Environmental Management is the production of an annual public environmental report within two years of registration. In mid-2000, the World Wildlife Fund (WWF) conducted an independent assessment of 32 environmental

reports prepared by signatories to the Code (WWF 2000). WWFs major criticism of the Code as it stands, is the lack of discrimination between the varying levels of environmental performance of its signatory companies. As such, the WWF has developed reporting criteria for the purpose of assessing the reports produced by Code signatories. WWF have now undertaken two of these assessments, and see this as their contribution to improving the quality of environmental reporting.

WWFs 2nd annual scorecard on mining company environmental reports (WWF 2000) concluded that, despite a review of the MCA Code since the 1999 scorecard was produced, very little improvement in the quality of the environmental reports has been made, as assessed against WWF's criteria. In particular, WWF were concerned about the lack of progress with external verification of environmental and social performance, as well as the absence of any agreed reporting format. Unfortunately, the standard of reporting set by the industry leaders has not been extensively adopted by other companies.

In response to concerns regarding independent assessment of environmental performance, the industry has established the Australian Minerals Industry External Advisory Group (EEAG) as a new forum through which the industry can seek independent advice on how its environmental performance is perceived and can be improved (MCA 2000).

3.6.2.2 Proportion of disturbance being rehabilitated

Notwithstanding the limitations of the data on areas disturbed versus those rehabilitated, the following estimates provide some indication of the rate and extent to which ecosystems disturbed by mining are being rehabilitated. As pointed out by EPA QLD (1999), not all the area on mining leases is disturbed; the undisturbed parts of mining leases comprise buffer zones, areas included because of the linear nature of lease boundaries, prospective mine paths, proven and unproven reserves and non-surface rights to underground mines. Figures for NSW and Queensland are provided which attempt to estimate the actual area of these leases that has been disturbed and the proportion of rehabilitation that has occurred. In NSW it is believed that 14,000 hectares of land had been disturbed by mining, with approximately 2,900 hectares or 21 percent of this area being rehabilitated as of 1995 (EPA NSW 1995). In

Queensland, approximately 73,000 hectares is thought to have been disturbed by mining activities, with 23,000 hectares or 32 percent of this area having been rehabilitated as of 1997 (EPA QLD 1999).

3.6.2.3 Expenditure on the environment

There is paucity of specific data on environmental expenditure, thereby hindering ones ability to draw national conclusions regarding trends in the industry regarding expenditure on biodiversity related issues. The annual 'Minerals Industry Survey Reports' provide a useful guide, as they contain data on the amount of rehabilitation expenditure, but other substantial aspects of environmental expenditure are not considered such as research activities, pollution monitoring and control, clean-up and in capital expenditures designed to minimise the environmental impact of processing plant and equipment (MCA 2000).

When interpreting this data, MCA (2000) warns that the annual provision for rehabilitation represents the amount charged to the profit and loss statement during the period, which increases the total rehabilitation provision. Actual payments made will be made directly from the total rehabilitation provision, and will not necessarily equal the amount charged to the profit and loss statement.

Table 3.1. Rehabilitation expenditure derived from minerals industry survey reports

Item	1996/97 $ Million	1997/98 $ Million	1998/99 $ Million	1999/00 $ Million	2000/01 Forecast $ Million
Annual Provision	178	245	275	242	285
Accumulated Balance of Provision	928	975	1,208	1,396	n/a

Source: MCA (1999; 2000)

The data in Table 3.1 suggest that, apart from the 1999/00 period, there has been a steady increase in the industry's expenditure on rehabilitation. MCA (2000) suggest that this reflects an increased focus on environmental rehabilitation by the minerals industry over the past few years.

At the State level, EPA QLD (1999) reported that the Queensland mining industry's expenditure on environmental protection, including management of pollutants, land rehabilitation and change to production processes, was $53.6 million in 1993-94 and $79.5 million in 1994-95. Based on data from the Australian Bureau of Statistics (ABS 1998a;b), EPA QLD (1999) argues that about 70% of this was current expenditure, which includes personnel and material, related to rehabilitation. Nationally, about 0.5 percent of total mining industry current expenditure and about 0.7 percent of mining turnover is environment-related (EPA QLD 1999).

4.0 POTENTIAL FOR ENHANCEMENT OF INDUSTRY PRACTICES TO PROTECT BIODIVERSITY

There are a number of issues and practices that may be enhanced to ensure the protection of biodiversity whilst allowing the activities of the minerals industry to continue. The two key issues in ensuring environmentally sustainable development within the minerals industry are the transfer of existing technology to the whole industry, and continued research on improved methods to minimise environmental impacts.

In the mid 1990s, representatives of the mining industry in Australia and the Environment Protection Agency (EPA), worked together to collect information on a variety of topics that illustrate and explain best practice environmental management in Australia's mining industry. The end result was the publication of a series entitled 'Best Practice Environmental Management in Mining' containing various modules aimed at assisting all sectors of the minerals industry to protect the environment and to reduce the impacts of mining by following the principles of Ecologically Sustainable Development.

In addition, the minerals industry has access to a number of workshops and short courses designed to improve communication and technology transfer. There are a number of Centres across Australia that specialise in designing and holding courses specific to the minerals industry and rehabilitation. These courses enable the transfer of research results and best practice techniques in courses ranging from, for example,

landscape design, tailings management, acid mine drainage, revegetation on saline lands, fauna habitat reconstruction and native seed biology.

As a result, the minerals industry has the communication structure to enable dissemination of research results and best practice techniques. It is the continued investigation and research that is required to be undertaken by all mineral operations to ensure minimal impact of their activities on biodiversity that remains a concern. Over the last five years, research and development expenditure by the industry has been declining (MCA 2000). It is accepted that this is a by-product of takeovers and company mergers; however, it should not be the case that larger operations undertake research while small to medium operations do not. The public does not differentiate between the different sectors of the industry, and hence any environmental accident will impact on the credibility of the minerals industry as a whole.

There are a number of areas where the minerals industry may enhance its practices to protect biodiversity. These areas are discussed below.

4.1 Agreement on Appropriate Completion Criteria and Indicators

The minerals industry and governments are looking for criteria that determine when rehabilitation of mined areas is complete. Elliot *et al.* (1996) advise that, where these criteria have been developed elsewhere, they have usually been based on a narrow set of vegetation indices that are measured at an establishment or early development stage of revegetation. They correctly point out that achievement of these indices does not necessarily ensure that land is returned to a safe, stable and self-sustaining land use.

The Australian minerals industry has used ant species diversity (richness and composition) as an indicator of rehabilitation success for more than 20 years (Andersen 1997). In doing so, a broad level of general principles has emerged, notably that ant diversity at mine sites being rehabilitated may reflect recolonisation by other invertebrates (Andersen 1997) and soil microbial biomass (Andersen and Sparling 1997). More broadly, Andersen (1991) noted evidence that the relative abundances of ant species were strongly influenced by habitat disturbance. In mesic southern Australia, for example, opportunistic ant species will increase in abundance following fire, grazing, mining and intensive recreation.

Andersen (1997) (see also Andersen 1990; 1993) reviewed the use of ant functional groups as indicators of ecosystem restoration following mining. Despite ant species showing clear successional patterns at mine sites undergoing restoration, the link between these patterns and more general changes in ecosystems had been inadequately studied. However, Andersen (1997) gave examples of positive correlations between ant species richness and total invertebrate richness for areas mined for bauxite in Western Australia. There were also positive correlations between the composition of plant species, ordinal composition of invertebrates (on the ground and on ground vegetation) and species composition of beetles and grasshoppers in disturbed and pristine sites in and around Ranger uranium mine in the Northern Territory, and between ant species richness and soil microbial biomass at disturbed sites.

A number of studies have evaluated the effectiveness of rehabilitation of bauxite mines (Majer *et al.* 1984), sand mines (Majer and de Kock, 1992) and uranium mines (Andersen 1993). They have shown that ant community structure and composition are related to the state of revegetated areas in a number of ways following the succession of ant species through time and the build-up of species richness. These characteristics are also related to other biotic and abiotic variables.

Stage 2 of the ACMER 'Indicators of Ecosystem Rehabilitation Success' project that is being undertaken with funding from the minerals industry, is designed to further test a range of indicators that have been developed that identify the successional pathway that the post-mined rehabilitation is undergoing. This will enable the development of a predictive understanding of the ecosystem response to rehabilitation efforts in mine sites based on the theory that ultimate success is built on the firm foundation of earlier stages (Tongway and Murphy 1999).

While it is accepted that the determined completion criteria will vary according to the community perceptions of the post-mined land use, the EFA procedure (Section 3.3) will outline the variability between the rehabilitated site and that of analogue sites at a specific point in time and allow the user to determine whether the agreed completion criteria will be met without further intervention.

4.2 Improved Utilisation of Native Seed Reserves

Currently, the minerals industry is the major user of all native seed collected in Australia with reports of 70% to 80% of all seed collected utilised for mine rehabilitation (Mortlock 1999). The majority of this seed is utilised within direct seeding programs over the relatively small areas of Australia impacted by the minerals industry. With the area of degraded land within Australia, largely the result of other industries, expected to get larger in the future, it is anticipated that the supply of native seed may not be able to meet demand. For this reason, the minerals industry must address the usage of native seed resources to maximise return from this investment. There are a number of initiatives that the minerals industry can take to improve the effective utilisation of native seed.

4.2.1 Collation of existing information

The minerals industry conducts and commissions research into native seed germination and establishment. Much of this work remains as unpublished reports and is viewed only by the mine personnel. In addition, there is a substantial amount of research into overcoming seed dormancy that has been undertaken by universities, government departments and others involved in native seed research that has been published in scientific journals and conference proceedings. Finding and obtaining these published works can be difficult for environmental personnel in remote locations.

To prevent duplication of research and maximise biodiversity in rehabilitated areas, it would be beneficial for information on native seed biology to be collated into the one reference location with ready access to all rehabilitation practitioners. While not intended to become a substitute for seed tests on the broadcast seed, access to this information may, for example, provide an indication on seed viability and hence provide an estimate of the amount of seed required to be collected for a given plant community. It may also document methods of overcoming seed dormancy for a given species that have proven successful in other studies, enabling an increase in species germination within post-mined sites.

Collation of a substantial amount of this information was undertaken by Langkamp (1987) and, since then, there has been a substantial improvement in the understanding

on native seed biology. The CD-ROM publication 'Floradata' (Mortlock and Lloyd 2001) does update this information but is not complete for all species utilised in the rehabilitation of post-mined environments. Further collations that add to these publications will improve the efforts of rehabilitation practitioners through ready access to the results of research and experience of others.

4.2.2 Seed quality assurance

The term seed quality refers to the purity, viability and genetic background of a seed sample (Luscombe 1994). All are components that can impact on the rehabilitation strategy and effectiveness in the post-mined environment. In many cases, seed purchased by the minerals industry is selected on the basis of price with little regard given to seed quality (Cockerton 2000). The price per kilogram is not an accurate guide as to the true cost of the seed (Luscombe 1994). A poor quality seed batch can have an impact on the density and composition of the rehabilitated landscape.

To ensure the acquisition of a quality seed resource, the minerals industry would benefit from obtaining seed quality certificates from every seed purchase. Information on the purity, viability and germinability of the seed lot will determine the number of available, germinable units per unit weight of the seed (Dixon 1997). The slight increase in cost would be compensated by the knowledge of the quality and background of the seed supply. In this way, a greater knowledge of the broadcast seed mix can be obtained and the expected germination estimated with greater accuracy.

4.2.3 Improved dormancy-breaking techniques

Successful revegetation that incorporates direct seeding relies on the germination of broadcast seed. Seeds that remain dormant within the seedbed are at risk of predation; hence, there has been a conscious effort to understand and overcome both seed coat and embryo-imposed dormancy to induce rapid and synchronous germination following broadcast. The germination cues of many Australian native species have been determined; however, there remains a number of species for which recruitment following the broadcast of viable seed is not reliable.

To ensure that the minerals industry has minimal impact on native communities, an understanding of seed dormancy within the community that is impacted will facilitate improved revegetation. This understanding is of direct relevance in dollar terms as the application of effective seed pre-treatments will enable maximum recruitment from the broadcast seed program.

There are currently a number of groups across Australia that are capable of developing effective methods to overcome seed dormancy in native species. It is desirable that the end result of such research is the development of techniques that will enable the seeds to be treated in large numbers by revegetation personnel prior to broadcast. While it is accepted that some companies invest in technologies to maximise recruitment from their seed investments, the minerals industry will only benefit by this work being widely available for the benefit of the industry and revegetation in Australia broadly.

4.2.4 Enhancement of survival of seedlings on modified mine soils

In most areas where mining takes place in Australia, the major part of the biodiversity is usually contained in the soil. As well as seed and nutrients, healthy soil contains macro- and micro- invertebrates, bacteria and mycorrhizal fungi which together play a critical role in nutrient cycling and in the establishment and sustainability of plant communities. Topsoil management techniques which conserve these values are therefore an important component of many successful rehabilitation programs. The direct transfer of topsoil is increasingly being used in the bauxite, mineral sand and other mining industries. The technique has been shown to enhance recruitment and seedling survival in many ecosystems due to, among others, better conservation of microbes, nutrients and organic matter. In some circumstances, however, it is not always possible to utilise topsoil, e.g. where there is a risk of weed proliferation, presence of disease, and high angle of batter slope. In addition, mine soils frequently have soil conditions that are not ideal for plant growth and establishment. Corbett (1999) outlines some of the physical, chemical and microbiological properties of waste rock dump soils. Briefly, they can have low infiltration rates, subject to dispersion and erosion, may contain elevated metal and salt concentrations, pH,

electrical conductivity and cation exchange capacity, in addition to lower levels of symbiotic microorganisms than undisturbed soils.

With studies suggesting that vegetation cover is an important factor in minimising erosion (Loch 2000), further investigations into the successful establishment of native vegetation on mine soils and mine soil remediation techniques will continue to benefit the minerals industry.

4.3 Development of Artificial Seeds

Despite the latest understanding of seed biology and dormancy breaking techniques, there are a number of species that are unable to be propagated with conventional technologies. In addition, there are many species that produce few viable seeds, and hence there is a difficulty in obtaining sufficient numbers of propagules to include these species within broadcast seed programs.

Recent developments with the production of synthetic seeds have shown promise for the production of propagules to enable recruitment of species that either are rare and endangered, produce few seeds, or have resisted attempts at propagation by conventional means.

Researchers at the Botanic Gardens and Parks Authority in Western Australia are developing a method of propagating species that produce few viable seeds and for which it has not been able to restore populations with conventional methods. They are attempting to produce viable synthetic seeds through somatic embryogenesis. Simplistically, synthetic seeds are produced by applying growth hormones and other chemicals to plant cells and culturing them to develop embryos. With this method, a 2 mm piece of leaf may produce up to 200 embryos or seeds. This technique is used extensively in the northern hemisphere for commercial crops such as lucerne and certain tree species such as spruce. Work is now being undertaken on two species, one from the Restionaceae family and one from the Epacridaceae. Clearly this is an expensive method of propagation and is intended for species as a last resort to prevent them from becoming extinct.

Although genetically identical, further studies may reveal the potential of this technique to recover endangered species from other families.

4.4 Improved Understanding of Ecology of Reconstructed Landscapes

The ecology of reconstructed landscapes is of vital importance to the minerals industry in achieving its agreed completion criteria. While several researchers have found the 'initial floristics model' of succession to be the most successful in determining the structure within the mature plant community (Tongway *et al.* 1997b; Ward *et al.* 1996), it would be beneficial to establish the relationship between selected biota, habitat complexity and ecological processes that contribute to habitat quality within reconstructed landscapes (Tongway *et al.* 1997b). Hence a greater understanding of successional processes would contribute to the ability of the minerals industry to rehabilitate according to the end goals that are required for relinquishment of the lease (Grant *et al.* 2001).

There are species of algae (Charophytes) that have been reported to hyperaccumulate metals from aquatic habitats (Hutchinson 1975; John and Gayton 1994). The use of these species may assist to rehabilitate mine voids and aquatic waste waters of relatively shallow depth as they have been found to accumulate high levels of iron, manganese, copper, magnesium, zinc and phosphorous as well as providing shelter and food for macroinvertebrates, fish and waterbirds (John and Gayton 1994). Lakes colonised by charophytes (specifically *Nitella*) are often found to be crystal clear with very little turbidity (Figure 4.1). The algae develop annual spore banks within the lake sediment and, through translocation of this sediment, are able to colonise adjacent water bodies with desirable results. Due to the potential for charophytes to assist in the remediation of degraded wetlands in Australia, and the potential for mine voids to become productive wetland ecosystems, the use of these naturally occurring algae may be a cost-effective avenue for the rehabilitation of reconstructed landscapes (Ward *et al.* 1997; John 1999).

4.5 Development of Better Monitoring Techniques

In many operations, the re-establishment of a self-sustaining ecosystem is the end goal of the rehabilitation programme. The minerals industry can have many unforseen impacts on the environment, and hence extensive monitoring of the pre- and post-mined environment is required to ensure the development of the ecosystem towards

the agreed completion criteria. Comprehensive monitoring programs are now a legal requirement in all States. For example, in Queensland, companies must prepare an Environmental Management and Overview Strategy which explains in general terms how environmental impacts will be managed. A subsequent Plan of Operations describes the environmental management program in detail. Both documents are required to address monitoring. Approval to mine is given by the issuing of an Environmental Authority, which may require the preparation of an Environmental Management Program which specifically documents the mine's Rehabilitation Monitoring Program.

As a result of these requirements and the adoption by many companies of Environmental Management Systems compatible with the requirements of ISO 14001, commitment to monitoring and the quality of monitoring programs continues to improve. For example, the development and utilisation of EFA and other monitoring techniques are being used by operations to monitor and appraise whether the rehabilitated land is continuing along the appropriate successional pathway or whether further intervention will be required. Only through efficient monitoring will potential problems be detected allowing remedial action to occur before they become a major issue.

Figure 4.1. The occurrence of *Nitella* meadows within Plover Lake at Iluka wetlands (Capel) have resulted in high clarity of the water body and improved water quality.

4.6 Enhancement of Fire and Weed Management Techniques

The indigenous Australians extensively utilised fire across the country to modify habitats for hunting prior to European settlement. This facilitated the development of a number of mosaics in various states of succession that ensured the provision of habitats and niches suitable for flora and fauna utilisation. The minerals industry has begun to consider the inclusion of fire regimes within its rehabilitated areas after a suitable period of time to assist in the development of the post-fire community dynamics and long-term sustainability of the post-mined environment.

Corbett (1999) states that research emphasis should be placed on the determination of time required for the development of fire tolerance of rehabilitated areas. The accidental introduction or inappropriate use of fire within immature rehabilitation areas could be a major setback to the redeveloping system.

Non-indigenous invasive plants are a poorly recognised threat to Australia's biodiversity (Csurhes and Edwards 1998). They have the ability to impact on ecosystem function through increased competition for resources, preventing recruitment of local species, alteration of geomorphological processes, hydrological

cycles, soil nutrient content and fire regimes, or by changing the abundance of indigenous fauna that may be required for pollination or seed dispersal of local species (Csurhes and Edwards 1998).

Improved monitoring and awareness raising through environmental induction programs, progressive rehabilitation of disturbed areas and rapid response following the identification of weed outbreaks can ensure that the minerals industry is not responsible for the establishment and degradation of native ecosystems by weeds in native ecosystems (Brearley *et al.* 2001).

5.0 OPPORTUNITIES FOR INDIGENOUS COMMUNITIES TO ASSIST THE MINERALS INDUSTRY WITH BIODIVERSITY ISSUES

The 1992 High Court Mabo native title ruling and the subsequent Commonwealth Native Title Act 1993 has presented the indigenous communities of Australia with many opportunities. This Act has resulted in increased negotiations between the minerals industry and the local indigenous populations. Investigations have discovered that these opportunities are not one-sided and that benefits of the increased negotiations can be found for the minerals industry and the Commonwealth government as well as the indigenous inhabitants that surround the mineral reserves.

To accommodate the Commonwealth Native Title Act, it is estimated that the cost to the minerals industry has been an increase of one percent expenditure (Manning 1997). Industry must negotiate access to land for exploration and mining with the indigenous owners and provide royalties to the local people. In return for this investment, the minerals industry has guaranteed legal access to the mineral reserve, improvements to the local social environment, an opportunity to develop a local labour force, and contracting enterprises which may allow them to reduce their reliance on importing labour from the cities at high cost (Manning 1997).

On the whole, mining and exploration occur in remote environments away from the main population centres, and as such, the minerals industry is best positioned to improve the economic development of these isolated centres. Several companies have realised the value of negotiations and education of the local communities to assist with the management of biodiversity issues.

The ability of local indigenous communities to negotiate with mining companies has brought about greater opportunities for both the local communities and the minerals industry. This relationship has been realised by several companies that have now developed policies on the employment of indigenous people. Several case studies illustrate the opportunities that indigenous Australians have accepted in assisting the minerals industry to manage biodiversity issues.

Case Studies

Energy Resources of Australia (ERA)

Energy Resources of Australia (ERA) operates the Ranger uranium mine adjacent to the World Heritage Listed Kakadu National Park in the Northern Territory. It has developed an Aboriginal employment strategy that aims to improve employment and training opportunities for Aboriginal people. ERA has engaged both the Gagudju and Djabulukgu Associations and has established contract arrangements with Aboriginal businesses for environmental rehabilitation work at the mine and within the region.

Delegates from both organisations have been sent on an ERA-sponsored tour of indigenous development foundations in South Africa, Zimbabwe and Namibia where they were able to observe first-hand successful projects in education, trade skills development and business enterprises.

As a result, local indigenous communities have been employed for a variety of biodiversity projects within the region. These opportunities have included clearing fire breaks for fire research, the development of a native plant nursery to supply tubestock for waste rock revegetation trials, and contracts to undertake soil conservation work. Traditional owners have also directed research into the commercialisation of native flora and fauna. This research is intended to provide alternative land use possibilities for the rehabilitated mine sites following mine closure and return to the traditional owners.

Hamersley Iron Pty Ltd

Hamersley Iron Pty Ltd operates six open-cut iron ores mines in the north-west of Western Australia and employs around 2000 people. Hamersley operates on the

traditional lands of nine Aboriginal language groups in the Pilbara and works with the people from seven other language groups.

To ensure that Aboriginal people benefit from the companies activities in the Pilbara, in 1992 Hamersley Iron established the Aboriginal Training and Liaison Unit (ATAL). ATAL is a nationally registered training organisation that provides specialised training, support and resources in several key areas including biodiversity management within the exploration and pastoral industries.

The focus on developing self-reliance through the business development program has proven successful and has resulted in a number of diverse businesses becoming established. Ventures such as recycling, seed collection, weed spraying, drill hole plug manufacturing, earthworks and pastoral activities are now being successfully run by Aboriginal people.

Hamersley owns several large pastoral leases in the Pilbara region with some of these leases adjacent to the Karajini National Park. Hamersley's Memorandum of Understanding with the Department of Conservation and Land Management ensures trainees gain first hand experience in managing these pastoral leases in sympathy with the adjacent National Park.

For its achievements in Aboriginal development and cross-cultural programs, Hamersley Iron has received a number of national and State awards including the Australian Reconciliation Award in 1997 and a commendation in the National Indigenous Peoples Training Awards (1998).

Tiwest Joint Venture

Tiwest Joint Venture extracts mineral sands from the northern sandplains of Western Australia. Early mining occurred on agricultural lands, but recent activities have concentrated on extraction of the mineral reserves under native ecosystems, thus the post-mined rehabilitation requires revegetation with native species. To meet their broadcast seed needs, Tiwest assisted members of the local Billinue Aboriginal Community to develop a native vegetation seed picking enterprise employing approximately 30 people.

Tiwest engaged established native seed merchants, LandCare Services, to provide training for Billinue Community members in species identification and native seed picking and processing techniques. To assist in the business establishment, Tiwest ceded 12 hectares of its freehold farm property next to the mine to the Billinue Community and provided support for land improvement, fencing needs as well as basic services and infrastructure.

While encouraging the development of external customers, Tiwest remains as the Billinue Communities key commercial customer by contracting the Community as the sole provider to meet their rehabilitation needs.

6.0 CONCLUSION

Although the activities of the minerals industry is confined to less than 0.05% of the land surface area of Australia, its impacts on both terrestrial and aquatic environments may be disproportionately high. The long lasting nature of acid rock drainage, mobilization of heavy metals and toxic chemicals, combined with the sulfidic nature of soils in some parts of Australia, have impacted on native ecosystems and are graphically illustrated in several operations across the country. The lack of appropriate monitoring and development of completion criteria by the minerals industry in the past was consistent with community attitudes of the time (Norris 1986; Richards *et al.* 1996; Brooks 1998).

The increasing community concern for the environment has been reflected by these concerns within the minerals industry and, to their credit, the minerals industry in Australia has developed a substantial knowledge on the identification and management of biodiversity issues that are consistent with the concepts of ecologically sustainable development. Through the development of environmentally sensitive practices and ongoing research and monitoring, the minerals industry has, for the most, demonstrated it to be a responsible land manager that is aware of the temporary nature of its activities and its commitment to future land users. A number of case studies, both positive and negative, have been presented in this report to illustrate the improved awareness of biodiversity issues by the minerals industry.

The activities of the minerals industry within Australia have allowed a greater understanding of Australia's biodiversity both directly through its research activities

and indirectly through the provision of royalties that are a source of environmental funding. Given the potential for environmental disaster as a result of its operations, the minerals industry cannot afford to be complacent in the implementation and monitoring of its activities. Although well organised, the minerals industry must focus attention not on the industry leaders, but on those operations that do not utilise the extensive information that is available on best practice techniques, for it is these operations that receive the attention of the media during environmental mishaps and largely downgrade the perceptions of the minerals industry in the minds of the public.

A number of areas have been suggested for potential enhancement of industry practices to ensure that the minerals industry minimises its impacts on biodiversity. Given the applicability of many of the rehabilitation techniques developed by the minerals industry for rehabilitation initiatives by other industries, coupled with its willingness to communicate and share this knowledge, the minerals industry can be found to have contributed to the conservation of biodiversity through the reduction in the current degradation to the agricultural landscape as a whole (Hobbs 1993). With continued planning and environmental awareness, the minerals industry has demonstrated it to be a responsible proponent of ecologically sustainable development.

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Evaluating the Feasibility of Independent Third Party Certification for the Mining Sector

*Michael Rae, Andrew Rouse and Fiona Solomon**

Introduction

Mineral exploration and development can have a significant impact on the environment and its local communities. While the area of land impacted by mining in Australia is estimated to be in the order of only 0.02 per cent of Australia's land surface (Mulligan, 1996, p15), its activities can create a range of effects on local landscapes, biodiversity, habitat, marine environments, rivers and aquifers, local communities, indigenous people, and workers. Poor environmental and social performance on the part of some mining companies has created mainstream public concern about mining practice and affected the reputation of the industry as a whole.

In Australia, States have traditionally set the standards for acceptable environmental and social performance for mining and mineral processing activities. However the nature of regulation is rapidly evolving in response to rising community expectations. The Commonwealth has growing powers to intervene or impose standards on States, through Commonwealth legislation (such as the *Environment Protection and Biodiversity Conservation Act, 1999),* joint Commonwealth-State specification of standards and procedures (for example the ANZMEC Strategic Framework for Mine Closure) and the various obligations imposed by international agreements (Ameef, 2002, p48). Commonwealth and State governments are also working more regularly with industry and non-government organisations to develop guidelines for company practice, of which a prominent example is the Best Practice Environmental Management (BPEM) in Mining program (Environment Australia, 2002).

Supporting this shift is the increasing emphasis on industry self-regulation, driven by business interests and often supported by governments who may not have the resources or perhaps the inclination for regulatory enforcement. The mining industry has introduced a range of voluntary initiatives to better manage environmental and social issues arising from their operations and to communicate these to the public. These include developing industry codes of conduct (particularly the Australian Minerals Industry Code for Environmental Management of the Minerals Council of Australia, 2000), implementing environmental management systems (such as ISO14001), setting performance targets (such as reducing energy and water use and greenhouse gas emissions) and producing public environmental reports.

While there have been some significant advances in environmental and social performance on the part of some individual companies and mine sites, these have not been largely recognised nor rewarded by the market or the general public. What has been absent is a credible and independent mechanism that can differentiate companies on the basis of their environmental and social performance, rather than on the basis of management processes or company claims.

Certification

Public concern is similarly evident around the environmental performance of other major natural resource sectors such as forestry and commercial fishing. During the 1990s, "certification" schemes were successfully developed for these two commodity-based sectors, with the Forest Stewardship Council the most well-known out of the two (FSC, 2002). Independent certification has two principal objectives. The first is environmental and social: to improve natural resource management to agreed standards. The second is commercial: to provide market access and product differentiation on the basis of on-ground performance (Rae and Rouse, 2001, p15).

Certification is a process where an independent third party certifies that a company or organisation satisfies the requirements set by an environmental and social performance standard. The standard is usually a combination of prescriptive criteria and requirements that specify a benchmark level of on-ground performance. The independent third party who verifies the company's performance is usually a certification organisation that is accredited by the owner of the standard (Rae and Rouse, 2001, p14). As compliance is voluntary, certification does not seek to replace government regulation but to complement it. For example, the first principle of the Forest Stewardship Council Principles and Criteria for

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Natural Forest Management is to comply with the laws of the country of operation and respect international treaties and agreements to which the country is a signatory (FSC, 2002).

The Mining Certification Evaluation Project (MCEP) has been developed by WWF Australia and Placer Dome Asia Pacific and attracted financial and in-kind support from BHP Billiton, CSIRO, Newmont and WMC. The MCEP proposes independent, third party certification of environmental and social performance as a possible mechanism for differentiating company performance and enabling mining companies to credibly demonstrate this to their stakeholders. Commencing in August 2002 and carried out over 18 months, the MCEP aims to evaluate whether independent third party certification of environmental and social performance can be applied to the mining sector. While the project takes as its inspiration the FSC type model, the potential options will not be limited to this model in recognition of the complexities of the sector.

Key lessons from experiences in the forestry and fishery sectors suggest that the success of certification in those sectors can be attributed to the following factors. First, the standard developed under a certification scheme acts as a benchmark for on-ground performance, rather than of processes. Second, the standard itself needs to be developed by a diverse group of stakeholders in order to best represent a variety of perspectives and expertise and ensure its credibility. Third, the certification of a company should be undertaken by an independent certification organisation and a system should be established to provide appropriate accreditation. Fourth, where products are developed for particular markets, a 'chain of custody audit' can track the product from source to retail outlet and assure customers of its provenance. Finally, a trademark or logo can be used as an on-product label to differentiate products in environmentally sensitive markets (Rae and Rouse, 2001, p1).

Project description

To WWF's knowledge this project will be the first major attempt to address the issue of certifying on-ground performance in the mining sector. To enhance the range of possible input, the MCEP will form a Working Group comprised of representatives from a range of companies, NGOs and other organisations.

The MCEP will facilitate a structured and focussed debate amongst key stakeholders on the issue of environmental and social performance and the ideas of certification. Through a series of reports, it will summarise stakeholder views received on these issues, identifying points of agreement and conflict and any options developed for a certification system. The aim will be to clearly identify whether it will be feasible to develop a system with broad industry and NGO support for the independent certification of the on-ground performance of individual companies in the mining sector. If consensus on a certification system proves elusive, the project will document the reasons to enable participants and other interested parties to learn from the process.

The project team at WWF will carry out the following major activities over the 18 month period of the project:

- Form a project Working Group that is likely to comprise representatives from the mining industry, environmental and social NGOs, labour, government, academic, financial sector and certifiers;
- With the Working Group, explore options for and seek to build consensus on principles of environmental and social performance in the minerals sector;
- Explore options for and seek to build consensus on measurable and auditable standards for on-ground performance;
- Undertake an evaluation in a field trial of the model or models developed. This stage will require companies to volunteer a mining facility to be the subject of the field trial;
- Prepare and publish a report containing recommendations for further action.

Conclusion

It is the authors' hope that, should this initial evaluation project succeed in developing a model with broad support from its working group participants, the project's scope will be broadened to include a wider debate within the Australian and international community. A broader international consensus could lead to the development of a global system for independent certification of on-ground performance, allowing mining companies to credibly demonstrate their competence. This would enable competitive advantage to accrue to those able to prove their commitment to sustainable development.

To be added to a contact list for regular updates on the project, please email fsolomon@wwf.org.au

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Wildlife Notes

DEPARTMENT OF CONSERVATION AND LAND MANAGEMENT
Information Notes for the *Land for Wildlife* Scheme in Western Australia

No.9 July 2001

Photographic Monitoring of Vegetation

Keywords: management, evaluation, monitoring
Location: national
Author: B.M.J. (Penny) Hussey

INTRODUCTION

All managers need to keep records, so that they can assess how they are doing, and adjust management if necessary. Records of things like stocking rates, fertiliser application - and of course cash flow - are a necessary part of managing a farming business. But the records of how the land itself is coping with management are not always taken, because they are less easy to quantify. To do this, it is advisable to **monitor,** that is, to observe and keep a record of change in something over time. The results of monitoring can be used to **evaluate** performance.

Monitoring is the process of undertaking periodical assessments or surveys, recording the results, and periodically comparing and evaluating them to determine the effectiveness of actions or the progress of projects. How frequently this is done, and in what form, will vary according to what is being measured and the purpose of the monitoring.

Monitoring is important for two main reasons: it provides feedback on the effectiveness of management actions - and hence whether these actions need to be modified - and it enables the determination of whether the natural resource is stable, improving or declining. So that this can be done, the records need to be consistent, comparable, and easily interpreted by any interested person.

Direct seeding trial

A *January 1991*

Site prior to any work. Note cairn at right poking up above wild oats. (York)

B *September 1997*

Shrubby habitat island established.

Note:
1 the problem of lack of exact correspondonce in the photos caused by not using specific sighter posts
2 seed sown at 1.5 kg/ha, 15 species established. 78% vegetation cover.

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LANDSCAPE AND URBAN PLANNING

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Sound science or social hook—a response to Brooker's application of the focal species approach

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Abstract

Brooker [Landscape and Urban Planning, in press] recently published a worked example of the focal species approach using bird assemblages in fragmented landscapes of western Australia. We have some concerns about the scientific validity of the focal species approach. As a threat-based biodiversity surrogate scheme, the focal species approach is likely to suffer from similar deficiencies that have been identified for related forms of species-based surrogate schemes such as umbrella species and indicator species approaches. We also raise concerns that the responses of birds, which are the sole group for which the focal species approach has been applied, may not be good species-based indicators of the responses of other vertebrates (such as reptiles, amphibians and mammals) for which selected focal species purport to be a surrogate. There are also non-trivial issues regarding field sampling intensity and patch or fragment size and the potential to underestimate the conservation value of small remnants during applications of the focal species approach. Despite these scientific concerns, we nevertheless consider that the focal species approach may be an important social hook to stimulate restoration activities in degraded landscapes that are now increasingly common not only in Australia, but also in many other parts of the world. However, we make a plea for the development of additional approaches for the restoration of degraded landscapes and that these methods (as well as the focal species approach) be subject to rigorous field testing.
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Keywords: Focal species approach; Landscape restoration; Biodiversity surrogate schemes; Habitat fragmentation

1. Introduction

Brooker (in press) presented a worked example of the focal species approach to landscape restoration. The focal species approach involves the identification of a suite of species targeted for the management of threatening processes and vegetation restoration efforts. Together the set of focal species have "requirements for persistence (which) define the attributes that must be present if (the landscape) is to meet the needs of the remaining biota" (Lambeck, 1999). The focal species approach was developed some time ago (Lambeck, 1997). It has since been championed in a number of cases (e.g. Freudenberger, in literature; Lambeck, 1999; Watson et al., 2001), but also recently criticised (Lindenmayer et al., 2002a). This paper highlights what we perceive to be some of the deficiencies of the focal species approach and it has been written in an attempt to stimulate further debate that advances the science of the restoration of degraded landscapes. Our contribution builds on a previous assessment of the focal species approach by Lindenmayer et al. (2002a) and raises a number of additional points in view of the material presented by

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The application of focal species knowledge to landscape design in agricultural lands using the ecological neighbourhood as a template

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Abstract

The focal species approach to nature conservation provides a method for identifying management actions required to meet the needs of the most vulnerable species in a landscape with respect to the threatening processes responsible for their decline. For each threat, a focal species is identified as the one with the most critical requirement pertaining to the threat. With multiple threats, the resulting suite of focal species can be regarded as a "focal community", whose combined needs are used to address a range of management actions. In this paper, the critical requirements of a focal community of resident land birds are identified with respect to the threats arising from the loss, fragmentation and degradation of habitat in the central wheat-belt of Western Australia. This knowledge of their critical requirements is then used to guide a landscape design procedure for the enhancement of bird habitat in the Gabbi Quoi Quoi sub-catchment of Western Australia. The landscape design procedure is framed against the concept of the ecological neighbourhood, where the necessary size, spatial arrangement, area of influence and connectivity of existing landscape elements are used as a starting point for habitat restoration and re-vegetation.
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Keywords: Connectivity; Ecological neighbourhood; Focal species; Focal community; Fragmentation; Isolation; Landscape design; Patch size

1. Introduction

Lambeck (1997) proposed a multi-species approach for defining the attributes required to meet the needs of the biota in a landscape. "The approach builds on the concept of umbrella species, whose requirements are believed to encapsulate the needs of other species. It identifies a suite of focal species, each of which is used to define different spatial and compositional attributes that must be present in a landscape together with their appropriate management regimes. All species considered at risk are grouped according to the processes that threaten their persistence. These threats may include habitat loss, habitat fragmentation, weed invasion and fire" (Lambeck, 1997). Within each group, those species most sensitive to the threat are used to define the critical level beyond which persistence is compromised. Previously, Robert Lambeck (personal communication) has applied the approach in the Western Australian wheat-belt to define minimum remnant sizes in the Wallatin Creek sub-catchment and the Dongolocking area and to define minimum remnant sizes and the need for increased connectivity in the South Tammin and Dowerin sub-catchments. In southeastern Australia, David Freudenberger (personal communication) has used a similar methodology to provide guidelines

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New Reference
Monitoring Problems

Chapter 17

Monitoring, assessment and indicators

This chapter gives a broad introduction to monitoring and environmental assessment for the conservation of natural populations. It includes a brief outline of statistical power, management goals and endpoints, and the predictive ability of impact assessment procedures. A discussion of the use of indicators and biodiversity surrogates in monitoring and assessment is provided because of their widespread use in monitoring and the assessment of environment impacts, and also because of the increasing focus on them in conservation biology.

17.0 Introduction

An **environmental impact assessment** (or effects assessment) is a document describing the existing environment, predicting the environmental effects of a proposal, and making recommendations to mitigate its effects. Government officers and public interest groups then assess the accuracy and completeness of the document and recommend whether and how the proposal should proceed (Thomas, 1996). Thus, assessment is the process of predicting the type, risks and magnitude of the likely effects on the environment of a particular proposal. Monitoring is defined as sampling and analysis designed to 'ascertain the extent of compliance with a predetermined standard or the degree of deviation from an expected norm' (Hellawell, 1991). Monitoring may be undertaken to gauge the effectiveness of policy or legislation, to audit compliance with guidelines, or to detect changes from a reference value.

The *Australian Environment Protection (Impact of Proposals) Act (1974)* created a legal mandate for environmental protection in 'matters affecting the environment to a significant extent' (section 5A). The Australian *Environment Protection and Biodiversity Conservation Act (1999)* protects endangered and vulnerable species and endangered ecological communities. It applies where Commonwealth actions or decisions are required. There is equivalent legislation in some Australian States, for example New South Wales (*NSW Threatened Species Act, 1995*).

Where there is a legal requirement to assess the impacts of a project on the environment or its component ecosystems or species, it is necessary to be able to detect important changes in the environment, usually with reference to a standard or control. Assessment and monitoring programs fail when they are inappropriately designed (Spellerberg, 1994; Franklin *et al.*, 1999). This chapter provides an introduction to concepts relevant to the conservation of natural populations, and outlines some of the fundamentals of effective monitoring and assessment methods. However, it is beyond the scope of this book to provide a complete outline of statistical methods and experimental design protocols for environmental impact assessment (see Quinn and Keough, 2002).

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Remediation *and* Management *of* Degraded Lands

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Chapter 12

Direct Revegetation of Salt-Affected Gold Ore Refining Residue: Technology Evaluation

G.E. Ho, M.K.S.A. Samaraweera, and R.W. Bell

INTRODUCTION

Iron, bauxite, gold, nickel, diamond, and mineral sands mined and processed in Western Australia represent a significant proportion of the world's output and generate 70% of the export income of the state (Department of Resources Development, 1996). In 1994, Western Australia produced 193.6 tonnes of gold, which is 74% of Australian production and 8% of the world total. Much of the gold mined in Western Australia is in the arid hinterland of the state. A significant amount is, however, produced in the southwestern part of the state, near Boddington, in the eastern jarrah forest, 125 km southeast of Perth. Evaluation of the technology for direct revegetation of the ore refining residues from the Boddington gold mining operation is the subject of the present chapter.

Location and Site-Specific Conditions of Case Study

Gold was discovered in 1980 directly under the boundary that demarcates bauxite lease areas held by Alcoa of Australia and Worsley Alumina which have developed the Boddington Gold Mine (BGM) and Hedges Gold (HG) mines, respectively. The gold deposits are found in the surface 15 m to 60 m depth in the profile in association with highly weathered soft clay and saprolite materials. Mining for gold ore is by the open-cut method. Gold extraction produces a predominantly clay residual ore material. The refining residue is deposited into purpose-built residue storage areas (RSA) as a slurry comprising approximately 40% solids in saline process water. Residue storage areas will eventually cover 1000 ha requiring revegetation. As properties of the residue produced by the two companies were similar, BGM and HG jointly engaged Murdoch University for a three-year study to prepare a revegetation prescription to generate a sustainable ecosystem in the RSA.

A mixture of rainwater and water from the Hotham River are used in gold extraction. The water of the Hotham River is saline and the dissolved salts content can vary from 2,000 to 6,000 mg L^{-1} (electrical conductivity, 320 to 960 mS m^{-1}, with an average value of 880 mS m^{-1}), with the lowest concentrations in winter. As the process water is continuously recycled and NaOH/NaCN are added during processing, it is estimated that the final salinity of the residue soil solution will be 10,000 mg dissolved salts L^{-1} (electrical conductivity, 1,600 mS m^{-1}).

Boddington receives an average annual rainfall of 810 mm, equivalent to only half the mean annual evaporation of the area. During the period January 1989 to December 1994,

the residue storage areas experienced a large variation in the yearly rainfall from 1,079 mm in 1991 to 658 mm in 1994. The average annual rainfall during this period was 865 mm. Generally, 75–82% of the rain falls in the winter and the early spring (May to October). The rainfall pattern in 1994 was unusual, as 95% of the year's below-average total was received during the period May to October.

The natural vegetation cover of the area surrounding the mine is open eucalyptus (*E. marginata)* forest. The ecosystems that exist within the forest are self-sustaining, characterized by a high level of biodiversity and efficient nutrient cycling and water use. Relatively little water or nutrients are lost from the ecosystem. Decommissioned RSA, on the other hand, will receive more water during the rainy season than the same area would have received if it remained as forest, because of the clearing of the surrounding forest and the increased run-on. The residue is not permeable enough to allow infiltration and deep percolation of the excess incoming water, and hence the water ponds on the surface. This will be further aggravated by the buildup of upward pressure in the perched water in the RSA due to the consolidation of the lower layers of residue in the RSA. Studies by consulting engineers have indicated that the water in the upper two-thirds of the depth of the residue will remain under upward pressure for a long period.

Residue storage areas are constructed by building dikes across naturally existing valleys. Residue is piped into the RSA as a slurry comprising 40% finely ground ore, mainly non-swelling clay, and 60% water of variable salinity. The residue consolidates upon standing, increasing in density to about 70% solids. Decant water is returned to the plant and reused as process water. Surface consolidation to 55 to 60% solids typically occurs by the time freestanding water ceases to cover the residue. Vertical cracking occurs as the residue dries by evaporation to 60% solids at the surface. Cracks develop to a depth of 1 to 2 m and salt crusts form on the surface. Cracks comprise up to 15–20% of the surface area of the RSA, and occupy up to 10% of the residue of the surface to a depth of 2 m. Fifteen years after decommissioning, the residue is predicted to have a solids content of 76–84%. Because the depth of the residue increases from zero at the edge to 35 m at the deepest point, it is predicted that a natural surface gradient of 0.5 to 2% will occur when the RSA is full and the residue eventually ceases consolidating.

The environmental management programs of the two companies are somewhat similar and include the rehabilitation of the mine pit, overburden, premises of the processing plant and related debris, and the residue storage areas. Issues such as protection of the surrounding catchment from the spread of jarrah dieback (*Phytophthora cinnamomi*), stability of slopes and roads, avoidance of groundwater contamination due to spills and leakage from RSA and reservoirs, and minimizing dust formation are common to both operations and resolved after consultation with the community and relevant government agencies.

Managing RSA requires special attention and consideration. The residue in addition to being alkaline, sodic, and saline contains metallo-cyanide complexes and traces of environmentally sensitive elements such as copper and arsenic. Regular monitoring of the water quality of the nearby 34 Mile Brook, Hotham River, and a series of bore holes located around the RSA is being done in order to assess the impact of the RSA on surface and groundwater quality.

The two companies have a commitment to minimize the environmental impacts by revegetating the RSA after decommissioning. The major issues which have to be addressed during decommissioning are the hydrology of the surrounding forest, the probable influence of RSA on the quality of the groundwater, time taken for the RSA to complete its consolidation, and the success of revegetation of the RSA.

Guidelines of the Department of Minerals and Energy of Western Australia require the RSA to be covered with 500 mm of inert waste, followed by the establishment of a self-regenerating vegetation compatible with the surrounding vegetation cover. The long-term stability of any perennial vegetation established in an inert material overlying the unamended residue is questionable, as the residue poses severe constraints to plant root development. In addition, the limited availability of inert materials and their high cost made it necessary for the owners of the mines to explore other revegetation options.

In this chapter we present results of our field trials, which show that it is feasible to directly revegetate on saline gold-refining residue. Results of laboratory and glasshouse studies are added where relevant.

Constraints to Plant Growth

Salinity and waterlogging are major constraints to plant establishment and growth when directly revegetating the residue. During the winter, the RSA are waterlogged by perched water with moderate salinity. Unless appropriate ameliorations are imposed, it is likely that the combined effects of salt and waterlogging would have a highly detrimental influence on plant establishment, growth, and survival.

While the residue is moderately saline in summer and autumn, it forms a strong salt crust at the surface upon evaporation and drying. Evaporation results in the upward capillary movement of saline water from the lower layers, and leaves the salts behind on the surface. Therefore, during the summer the electrical conductivity at the surface, measured using a soil to water extract of 1:5 [$EC_{1:5}$], may be as high as 6,500 mS m^{-1}. The residue has a pH of 9.3. Generally, an $EC_{1:5}$ of 140–350 mS m^{-1} and pH above 8.5 are considered to severely limit growth of most plants. Much of the salts in the water taken from the river, and in the chemicals added during processing (NaOH and NaCN) result in sodium being the predominant cation, and consequently the residue is highly sodic.

The dried residue has a fine texture, no structure, and is lacking macropores needed for water and air movement. However, voids created by the cracks represent channels through which drainage water and air can move through the surface 1–2 m of the residue, provided they can be stabilized to avoid slumping and in-filling. In addition, the salt crust formed on the surface of the residue during the summer may be subject to wind erosion. If nothing is done to improve infiltration, the winter rains will dissolve the salt that is on the surface but not remove it by leaching. A possible sequence of events in unamended residue during winter is: slow dissolution of salt crust, filling of cracks with rainwater contaminated with salt, and surface ponding and waterlogging of residue due to stagnating water. Furthermore, the consolidated residue is brick-like in character, making it difficult for plant roots to penetrate.

Because of the processing it has undergone, the residue does not contain organic matter, plant nutrients, or any biological activity that enhances plant growth and subsequent functioning of the ecosystem.

Approach to Revegetation

Our model for revegetation is based on the following approach: (1) identification of physical, chemical, and biological properties of the residue; (2) development of methods of ameliorating the residue properties so that plant growth is possible. This involves the use of chemical amendments and leaching of salts; (3) sequential introduction of plants; ini-

tially salt and waterlogging tolerant species are selected as pioneers to be followed by more sensitive tree species once residue properties have been sufficiently improved; (4) reshaping of the residue surface to facilitate draining of excess water and drying of residue; (5) sequential revegetation of the RSA commencing on the perimeter as drying of the residue begins at the outside edges; rehabilitation is aimed at accelerating the drying process and ameliorating the periphery; and (6) avoiding the use of heavy equipment and backfilling of the residue unless it is absolutely essential, as costs involved are high.

Possible end land uses on RSA vary from an ecosystem that will blend with the surrounding eucalyptus forest at BGM, and a combination of commercial pasture and forest at HG.

A variety of rehabilitation strategies have been considered by the consulting engineers. Of those proposed, the scheme favored has the following basic features: (1) rehabilitation will follow initial consolidation and drying of the residue, 2–3 years after the termination of filling with slurry, (2) the outer edges will be reshaped and revegetated, and (3) the central areas will be covered with coarser refining residue from gold basement ore found below the present ore body at BGM. The overall revegetated residue area will have artificial drains and permanent mounds and ponds.

The main emphases of the engineering considerations for revegetation are on the methods of incorporating ameliorants into the residue, the water movement within the residue matrix, and the trafficability of the surface. Because of the nature of the constraints for revegetation, trials were directed at problems of salinity, salt migration and capillary rise, poor infiltration of water and waterlogging, surface cracking, and lack of residue structure.

Initial laboratory investigations showed that the residue consists of mainly clay and has a hydraulic conductivity of 1.9×10^{-2} m day^{-1}. When the residue was amended with 5% gypsum its hydraulic conductivity was increased by more than twofold to 5.4×10^{-2} m day^{-1}. Gypsum application increased the amount of sodium that was leached with a given volume of water and decreased sodicity. Gypsum also aided in pH adjustment by precipitating carbonate alkalinity as calcium carbonate, thus converting the residue to a calcareous material.

GREENHOUSE TRIALS

Greenhouse trials initially examined the growth of four species. The residue was amended with gypsum (rates equivalent to 0, 30, and 60 t ha^{-1}) and leached with deionized water equivalent to 75 mm or 150 mm of rain. Amounts of leaching water added were equivalent to 26% and 53% of annual May–June rainfall at Boddington. Four species, *Melaleuca halmaturorum*, *Atriplex amnicola*, *Agropyron elongatum*, and *Triticosecale* spp. cv Muir were grown. Water logging and salinity tolerances of test species were as follows:

Melaleuca	salt and waterlogging tolerant
Atriplex	salt tolerant
Agropyron	salt and waterlogging tolerant
Triticale	waterlogging tolerant

Gypsum was generally very beneficial to plant growth, as was leaching. *Atriplex* grew successfully under all conditions including the untreated residue, but the other three species did poorly in unamended residue. Gypsum also improved the residue chemical properties by neutralizing its alkalinity. With gypsum addition (2.5% by weight), the pH of

the residue was decreased from 9.3 to 7.6, and subsequent leaching further decreased the pH to 7.2.

The next greenhouse trial was initiated to determine the value of topsoiling the residue and to evaluate the optimum gypsum level required to establish triticale, the most salt-sensitive plant species. Salt was also added to the residue to increase the salinity equivalent to 500 and 1,000 mS m^{-1} in a 1:5 soil:water extract. It was found that gypsum at 6% had little effect in alleviating growth at high salinities but at the lower salinity level there was a near linear growth response to increasing gypsum levels.

The combined effect of salinity and waterlogging on plant growth was examined because when both occur they can be more damaging to plants than when present singly. In saline waterlogged residue, triticale failed to germinate directly on the residue. A surface layer of coarse sand covering the residue allowed the plants to germinate and survive for some time before they died. Saltbush, however, fared much better under salt and waterlogged conditions. An examination of triticale roots under saline conditions revealed that growth was limited to the topsoil layer and the adjacent 5 cm of the residue which had a lower salt content ($EC_{1:5}$ 137–373 mS m^{-1}) than the underlying residue ($EC_{1:5}$ 850 mS m^{-1}). This suggests that for successful growth, the rooting zone has to be sufficiently low in salt throughout the growing season.

Methods of nutrient addition were also examined by assessing the performance of *Agropyron* plants grown in the residue amended with 10% gypsum. Nutrients were supplied by one of the following ways: poultry manure; a basal chemical fertilizer mixture having the same nutrient content as that of the poultry manure; by regular application of a balanced liquid fertilizer mixture; or with an additive combination of poultry manure and the liquid fertilizer. The experiment was evaluated under both moderate ($EC_{1:5}$ 250 mS m^{-1}) and highly saline conditions (salt added to increase the $EC_{1:5}$ by 500 mS m^{-1}). The addition of salt depressed plant growth. The basal application of chemical fertilizer also had a negative effect on plant growth. Both the poultry manure and the liquid fertilizer significantly improved plant growth and their combined effects were additive. Poultry manure was superior to liquid fertilizer in relation to plant growth. The poultry manure not only provided nutrients but also improved physical, chemical, and biological properties of the residue.

These findings have very significant practical implications. First, since the wet residue is not trafficable by heavy equipment, the application of all fertilizers must be done prior to winter rain. Secondly, rehabilitation relies on the ability of the winter rain to leach salt out of the rooting zone into the deeper residue layers. If water soluble chemical fertilizers are applied in the summer with gypsum, then during the winter, most of the nitrogen and possibly sulfur and potassium will be leached from the rooting zone, making them unavailable to plants.

In another pot experiment, the effect of leaching on the availability of nutrients in poultry manure was examined. Pots filled with the residue containing 5% gypsum were mixed with varying amounts of poultry manure, up to a maximum of 100 g per kg, then leached with deionized water equivalent to 200 mm of rain (71% of May–June rainfall). A thin layer of topsoil was placed over the residue and triticale grown. While some pots received deionized water, the others received a complete liquid fertilizer mixture which supplied approximately the same rate of nutrients as supplied by poultry manure. The plants that received the combined mixture of poultry manure (100 g kg^{-1}) and liquid fertilizer showed significantly more growth than those that received poultry manure alone. The poultry manure added, 100 g pot^{-1}, would give a thickness of 2.3 cm. Since the amount of poultry manure added was equivalent to half the minimum thickness achievable with equipment

used in the field, it would be very unlikely that any nutrient deficiencies would be experienced under field conditions with poultry manure added at 1% (w/w).

From these greenhouse experiments the beneficial effects of gypsum amendment, leaching of salts, incorporation of organic matter as a nutrient source, and placing a topsoil over the residue were evident. Field trials were conducted to further assess the efficacy of the practices and amendments used in the greenhouse studies.

FIELD TRIALS—GENERAL

The goal of developing land preparation procedures was to overcome the following constraints: waterlogging; high salt content in the rooting zone; capillary migration of salts onto the dried surface; and high pH of the residue. The findings of the greenhouse experiments showed that gypsum addition and leaching of salts were prerequisites for growth of salt-sensitive species in the residue and formed the basis for development of land preparation methodologies.

Breaking Down of the Massive Apedal Clay Structure

The volume loss during drying and consolidation of the slurry leads to the formation of large apedal clay blocks. These massive clay structures have a relatively small surface area to mass ratio and as the drying is limited to evaporative water loss from the exposed surfaces it may take several years for complete drying of the first 50–100 cm of the unamended residue surface. However, the process of drying is accelerated when the exposed surface area is increased.

To increase the surface area, the residue can be ripped, plowed, and hoed during the dry part of the late summer. The initial ripping reduces the size of the clay blocks and facilitates the plowing of the residue. The process of increasing the surface area and accelerating the capillary migration of salts onto the residue surface was further accelerated by breaking the large lumps into smaller pieces by rotary hoeing.

Application of Chemical Amendments and Nutrients

The incorporation of chemical amendments and fertilizers into the residue is facilitated by the ripping and rotary hoeing process. Gypsum amendment can be effectively carried out when the surface is cultivated. Nutrients required for plant growth can supplied either by chemical fertilizers or organic fertilizers. Since the latter has a longer residence time within the rooting zone, poultry manure was used as the source of nutrients and applied immediately after gypsum in the late autumn.

Field trials were conducted in specifically-built impoundments. Results of relevant trials that demonstrate the success of direct revegetation of residue surfaces are presented here.

FIELD TRIAL—CELL A

Cell A was available for experimentation in summer of 1991/92. It had an area of 5,500 m^2 and a depth of 1.25 m and it incorporated an underdrainage system. It therefore represented the well-drained perimeter of a decommissioned RSA with a shallow depth of

residue and would provide valuable information relating to the revegetation of that part of an RSA.

Selection of Land Preparation Techniques and Planting Time

The suitability of several land preparation procedures were examined. Cracks were either filled with gravel (2,000 m^3 ha^{-1}) to facilitate rapid movement of excess rainwater, or left unfilled. Residue was either prepared by a sequence of ripping (to 50 cm depth), plowing, and hoeing or left without any preparation. Incorporation of gypsum (0, 30, 60 t ha^{-1}); incorporation of salt to achieve 10,000 mg salt kg^{-1} residue; addition of poultry manure (equivalent to a thickness of 0 or 4–6 cm); placement of topsoil (to a thickness of 0 or 8–10 cm); planting in winter or spring, and species (no plants, *Agropyron*, Triticale, saltbush) were considered.

Fourteen out of a possible 768 treatments were selected and they were biased toward favorable conditions for plant growth. Table 12.1 shows 10 of the treatments relevant to this study. The experiment was laid out in a split-plot design with blocks perpendicular to the moisture gradient. To minimize equipment movement, each block was divided longitudinally into many plots. Two major treatments (filling of cracks with gravel or not) were randomly assigned to each main plot. The other treatments were then randomly assigned to each of the fourteen 5 m by 5 m subplots within a block.

The total biomass production in winter planted plots (10.7 ± 1.2 t ha^{-1}) was significantly higher than those planted in spring (6.7 ± 0.6 t ha^{-1}). In winter, the residue was waterlogged and saline. At the beginning of spring, the residue contained less water and salts and hence the conditions were more favorable for plant growth. The net aboveground biomass accumulation rates for winter planting and spring planting were similar, 4.38 and 4.48 g m^{-2} day^{-1}, respectively. However, winter-planted species had the advantage of having a longer growth period than those planted in spring. Winter planting will not be effective if the soil conditions are not favorable. The factors that should be considered are the water content, pH and EC of the residue, and the subsurface drainage within the impoundments. Thus the selection of planting time must be based on the following two factors:

a. soil conditions: extent to which the amelioration treatments have been effective in alleviating salinity and waterlogging;
b. the length of the available growth period.

The total biomass production in gravel filled plots (6.6 ± 0.6 t ha^{-1}) was significantly lower than those in unfilled plots (9.0 ± 1.0 t ha^{-1}). This may be due to the decreased availability of water due to rapid drainage. The grasses, triticale, and *Agropyron* which have aerenchyma in their roots and hence can tolerate waterlogging grew significantly better in plots where gravel was not applied. For *Agropyron*, the total biomass was 3.9 ± 0.4 t ha^{-1} in plots with gravel filled cracks versus 12.0 ± 2.4 t ha^{-1} in unfilled plots. By contrast, saltbush which is not waterlogging tolerant, grew better in the gravel filled treatments (5.2 ± 0.8 t ha^{-1}) than in nontreated plots (1.9 ± 0.4 t ha^{-1}).

The effect of gypsum is also of interest. The grasses which were less salt tolerant performed significantly better under the high rate (60 t ha^{-1}) than the low rate (30 t ha^{-1}) of gypsum application (6.4 ± 0.7 t ha^{-1} versus 2.1 ± 0.7 t ha^{-1}). The effect of the higher gypsum rate was more prominent in the winter-planted plots (8.4 ± 2.3 t ha^{-1}) than those

Table 12.1. Selected Treatment Details and Results of Cell A Trial.

Treatment	Rotary Hoeing	Gypsum (t ha^{-1})	Top Soil Placement	Poultry Manure	Planting		Gravel in Cracks							
							Agropyron (t ha^{-1})		Triticale (t ha^{-1})		Salt Bush (t ha^{-1})			
					Win	Spr	Y	N	Y	N	Y	N		
1	N	60	Y	Y		SB	2.83		0.03		2.9			
						Agro								
						Trit								
2	Y	30	Y	Y	SB	Agro	2.9	4.43	0.43	1.61	9.7	4.13		
						Trit								
3	Y	60	N	N		SB	2.1	2.98	0	0.03	0.3	0		
						Agro								
						Trit								
4	Y	60	Y	N		Agro	3.03	4.28	0.6	0.17				
						Trit								
5	Y	60	N	Y	Agro		13.0	17.6	1.38	0.26				
					Trit									
6	Y	60	Y	Y		Agro		6.23		0.17				
						Trit								
7	Y	60	Y	Y		SB		6.10		0.36		0.94		
						Agro								
						Trit								
8	Y	60	Y	Y		SB	3.90	12.0	0.23	0.28	5.15	1.94		
						Agro								
						Trit								
9	Y	60	Y	Y	SB	Agro	2.53	5.58	0.1	0.09	0	2.63		
						Trit								
10	Y	60	Y	Y		Agro	4.60		0.73					
						Trit								
Common for all	LSD= 0.05%							4.98		2.06		4.34		

Note: SB = Salt bush; Agro = *Agropyron*; Tri = Triticale; Win = winter planting; Spr = spring planting; Y = Yes; N = No.

Table 12.2. The Effect of Gypsum Application on the pH and the EC (mS m^{-1}) [1:5, soil:water extract] Measurements of Residue Sampled in February, 1993.

Rate of Gypsum Application (t ha^{-1})	pH (±SE)	EC (mS m^{-1}) (±SE)
0	7.93 (±0.14)	657 (±138)
30	6.53 (±0.17)	147 (±25)
60	6.58 (±0.15)	211 (±48)

planted in the spring (5.9 ± 0.7 t ha^{-1}). Gypsum requirements of spring planting may be less as the combined effect of autumn gypsum application and winter leaching had ameliorated the residue. The pH and the EC measurements of the residue sampled in February 1993 confirmed that gypsum application and subsequent leaching had ameliorated the residue, creating conditions favorable for plant growth (Table 12.2). On the other hand, saltbush consistently yielded better under lower gypsum applications (8.7 ± 0.9 t ha^{-1}) than at the higher rate (3.1 ± 0.6 t ha^{-1}). All species grew significantly better when treated with poultry manure (8.6 ± 0.6 t ha^{-1} vs. 3.4 ± 0.4 t ha^{-1}).

During the period May 1992–January 1993, the area received a total of 872 mm of rain and 83% of that was received during the wet season, May–September. Thus the climatic factors throughout the growing season were favorable for plant growth. The aboveground biomass production of *Agropyron* under optimum land preparations were 17.6 and 6.10 t ha^{-1} yr^{-1} for winter planting and spring planting, respectively. These figures are higher than 1–2 t ha^{-1} yr^{-1} previously reported for *Agropyron* in ameliorated bauxite refining residues (Wong and Ho, 1994). The saltbush production in winter and spring plantings of 9.7 and 5.15 t ha^{-1} yr^{-1}, respectively, were very much higher than the average production figures, 3–4 t ha^{-1} yr^{-1} for Western Australia (Brian Warren, Department of Agriculture, Western Australia, personal communication).

Ripping, plowing, rotary hoeing, gypsum amendment, poultry manure amendment and leaching of salts during the winter prior to planting were found to be beneficial for the establishment and successful growth of the three pioneer species evaluated. Using this combination of land preparation technology and amelioration techniques to test the survival and growth of 27 species, Samaraweera et al. (1994) reported that establishment of additional pasture or crop species, and nonendemic native vegetation (including tree species) is feasible on the amended residue.

FIELD TRIAL—CELL B

Cell B had an area of 7,150 m^2 and a depth of 2.25 to 5.85 m. It had no underdrainage and was filled by residue at the end of March 1993. Cell B was prepared for trials by rotary hoeing, addition of gypsum (60 t ha^{-1}) and topsoil (10 cm layer). The salinity of residue in cell B was increased to resemble that which is anticipated in the final decommissioned RSA. Salt was added during the land preparation at a rate of 5 t ha^{-1} to give a soil content of about 10,000 mg kg^{-1} (EC 1:5 531±109 mS m^{-1}).

The revegetation strategy was to introduce species in a planned succession: grasses, then shrubs, and finally trees. The rationale was to establish vigorous growth of the pioneer grasses and shrubs for 1–2 years before introducing trees. The delay in planting trees

was to allow time for the amelioration of salinity to aid in the establishment of those species with lower salt tolerance. The hypothesis was that accumulation of soil organic matter and formation of an organic mulch on the surface would prevent capillary salt rise. It was further hypothesized that the land preparation, residue amelioration, and land reshaping practices, along with the penetration of the residue by the fibrous root systems of pioneer species would accelerate the decrease in salinity and waterlogging in the surface 0.5–1 m. The pioneer species in Cell A *Agropyron*, triticale, and saltbush all survived over two and a half years though density of the annual, triticale, declined each year. Success of this strategy requires continued investigation and examination over several years. Preliminary results from the first two and a half years growth are presented below.

Alleviating Waterlogging and Salinity

Our goal was to prevent flooding of the residue and waterlogging in the rooting zone during the rainy season. Another goal was to prevent drying of the residue surface during the summer and thereby stop the process of capillary migration of salt onto the residue surface. These were to be achieved by gravel filling the cracks, reshaping, and mounding of the residue surface.

Gravel Filling and Reshaping

The residue consolidated with a 0.5% slope. Water runoff and drainage in clayey residue at such gentle slopes is rather slow and was improved by increasing the slope to 2.5%. Four areas were created in Cell B in autumn 1993, with and without gravel in cracks and with and without reshaping.

During this exercise, large machinery, such as front end loaders and dozers, were used to spread gravel. The use of large machinery was to provide an indication of how large-scale land preparation could be carried out. Cells were further subdivided and some plots were planted with pioneer species (*Atriplex amnicola*, *Agropyron*, and triticale) and others were left as bare plots.

The mean total biomass production in all four parts of Cell B during 1993–1994 was 4.37 t ha^{-1} with a maximum of 11.58 t ha^{-1}: both values were lower than corresponding 1992–1993 production of 7.73 and 30.7 t ha^{-1} for Cell A. Indeed, the mean value for Cell A was an underestimate, as some parts were planted in spring whereas all the pioneer species in Cell B were planted in the winter. Rainfall for the two years was similar and close to average; therefore, precipitation does not explain yield differences. Analyses of Cell B residue showed that leaching over a year had not decreased its salinity to the level required for growth of nonhalophylic plants. The increased salinity of the residue in Cell B suppressed the yields of triticale, *Agropyron*, and saltbush. For example, in February 1994 while the $EC_{1:5}$ of the residue taken from unamended parts of Cell A was 121 mS m^{-1}, the corresponding value for Cell B was 1,800 mS m^{-1}. Plant growth may have also been adversely affected by compaction due to earthwork operations. There is, however, sufficient information after one year's field results to conclude that the highest biomass yields were obtained when the cracks were filled with gravel and the residue surface reshaped. Both the reshaping and gravel filling of cracks had significant positive effects on plant growth (Figure 12.1). The highest total biomass was recorded in the plots that were reshaped after gravel filling the cracks.

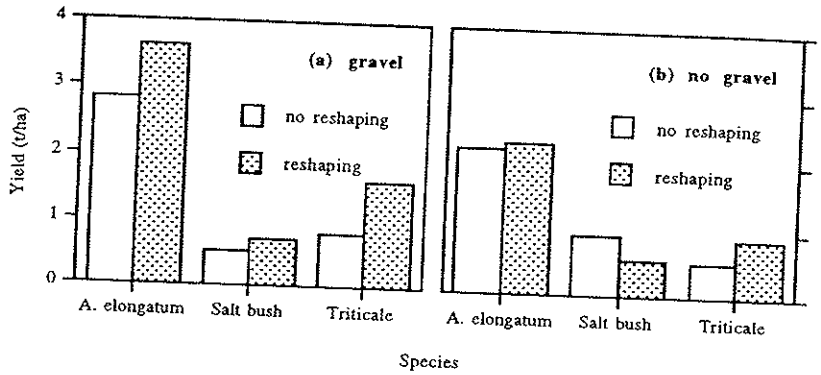

FIGURE 12.1. Effect of gravel filling of cracks and surface reshaping (no reshaping—0.5% ground slope; reshaping—2.5% ground slope) on yield (t ha^{-1}) of *Agropyron*, triticale, and saltbush in Cell B, December 1993. $LSD_{0.05}$ for comparison of means of *Agropyron*, triticale, and saltbush are 1.20, 0.52, and 0.26 t ha^{-1}, respectively.

Mounding

As the yield of pioneer species was relatively low and all tree species planted in the spring of 1993 died, niche mounding was investigated as an additional land preparation technique to sustain tree growth.

In each of the four areas of Cell B, plots were further subdivided so that one-half could be mounded. No new planting was carried out in the mounded or unmounded areas. At the time of preparation of mounds, the area was vegetated with *Agropyron* and saltbush. Triticale planted in winter 1993 had matured and died. The mounding operations destroyed aboveground shoots and damaged root systems of *Agropyron* and saltbush. The growth measurements made in the mounds in August 1994 showed that while there was no regeneration of *Agropyron* and saltbush, the production of Triticale was spectacular, in the range of 4–8 t ha^{-1} during the short growing season May–August 1994. This is particularly a significant achievement as the total rainfall for the period was about 500 mm. In contrast, the unmounded areas produced no regeneration of Triticale but production of *Agropyron* and saltbush for the period May 1993 to August 1994 was 4–5 t ha^{-1} and 1.5 t ha^{-1}, respectively. This clearly shows that biomass production can be optimized by having a mixture of mounded and unmounded areas which would enhance rapid growth of triticale during winter and *Agropyron elongatum* during the spring and summer periods.

Apart from the obvious avoidance of waterlogging and flooding, mounding decreased the salt content in the elevated portion of the mounds favoring plant growth and establishment. The residue/topsoil on the crests of the mounds in the reshaped area had $EC_{1:5}$ values of 300–900 mS m^{-1} and ESP of 2–10% in August 1994 after 480 mm of rain. The corresponding figures for unmounded reshaped areas were EC 700–1,100 mS m^{-1} and ESP 3.5–9.0%. The EC and ESP values are well within the ranges acceptable for plant growth. Yet the $EC_{1:5}$ and ESP of the residue near the base of the mounds were 26,000–46,000 mS m^{-1} and 8–21%, respectively.

The effects of mounding on plant growth are shown in Figure 12.2.

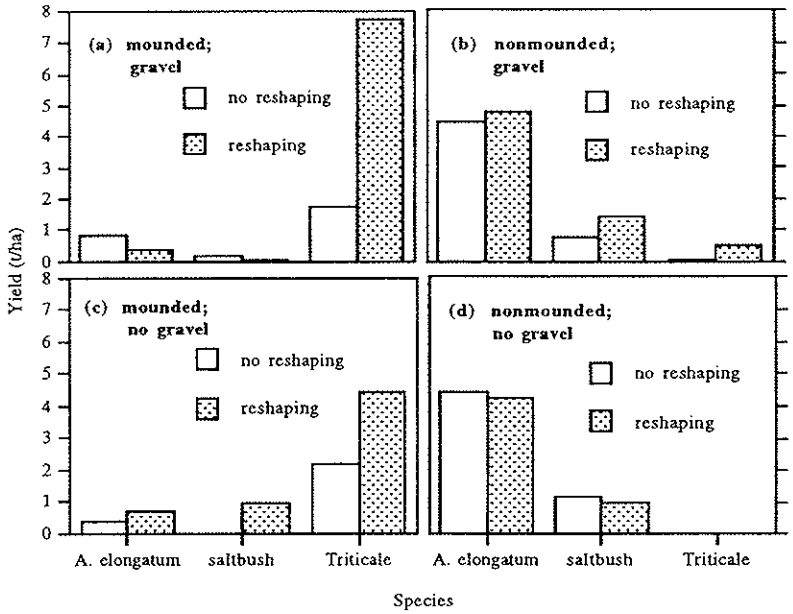

FIGURE 12.2. Effect of mounding, gravel filling of cracks and reshaping the ground surface (no reshaping—0.5% ground slope; reshaping—2.5% ground slope) on yield (t ha^{-1}) of *Agropyron*, triticale, and saltbush in Cell B, October 1994. $LSD_{0.05}$ for comparison of means of *Agropyron*, triticale, and saltbush are 1.44, 2.53, and 0.64 t ha^{-1}, respectively.

CONCLUSIONS

It is too early to conclude from research and monitoring carried out so far whether eucalyptus forest can be reestablished on the RSA, but establishment of nonendemic salt-tolerant native species and a variety of agricultural species appear to be useful rehabilitation species.

Based on research thus far, a revegetation prescription would appear to include:

- filling of cracks with gravel and reshaping of selected areas, to facilitate drainage
- ripping and plowing; to break up the massive blocks of residue into smaller blocks
- rotary hoeing: to break down the residue blocks to clods small enough to provide macropores, increase drying, accelerate surface salt migration, and facilitate subsequent leaching of salts from the surface
- gypsum addition: 60 t ha^{-1} to decrease pH, facilitate aggregation, and improve permeability and leaching of salts
- poultry manure: 2 to 4 cm thickness to give initial nutrients for plant growth and provide organic matter for aggregation of the soil, and
- topsoil placement: to provide a suitable medium for germination or establishment of seedlings.

TOPICS THAT REQUIRE FURTHER RESEARCH

A number of research questions remain unresolved but of crucial importance to the long-term effectiveness and sustainability of revegetation of gold ore refining residue. They are:

- effect of filling of cracks with gravel to resolve whether it is essential
- rate of the export of salt out of the system, as this will determine whether in the longer term, the soil will revert to surrounding conditions
- models for salt and water movement in the residue storage impoundments to determine how much leaching is required for effective removal of salt from the rooting zone
- screening of a wider range of plant species for their ability to withstand anticipated soil-related constraints in the amended residue
- effect of planting time on establishment of different species used in revegetation program
- long-term sustainability of the ecosystem including nutrient cycling, hydrological balance, and biodiversity
- finding low cost alternatives for treatments such as the fertilizers.

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The impact of plant disease on mining

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Abstract

Dieback disease caused by *Phytophthora* species is the only plant disease having a major impact on the bauxite mining operations of Alcoa of Australia Ltd and the mineral sands mining operations of RGC Mineral Sands Pty Ltd. To mine responsibly in regions of the state where dieback is a concern, both companies have implemented major dieback management programs. The financial costs associated with implementing these programs are substantial. However, estimating the total financial cost of dieback disease management on the mining operations is very difficult because dieback control procedures are integrated with mine planning and other operational procedures. Both companies recognise that these financial costs are part of the overall cost of mining in regions of Western Australia where dieback is present, and conservation of the natural vegetation communities is of high priority.

Financial considerations are not the only potential impacts of disease. Plant diseases can also impact on environmental management objectives. Dieback disease has the potential to affect three important environmental management objectives; protecting the adjacent natural vegetation, establishing key plant species in rehabilitated sites, and achieving high species richness in these sites. Achievement of these objectives is regarded as important to the success of the mining operations of both companies.

Alcoa and RGC monitor vegetation adjacent to their mining operations and rehabilitated areas for symptoms of dieback. There has been no *Phytophthora*-caused plant death found near RGC's operations in areas interpreted previously as uninfested. Monitoring of the forest surrounding a 120 ha bauxite mined area identified two new *P.cinnamomi* infestations; the total area was 0.32 ha. Establishment of key plant species in the rehabilitation (*Eucalyptus marginata* in the jarrah forest and *Banksia species* in the kwongan) has been successful despite the presence of *Phytophthora* species in the region. High species richness is achieved in the rehabilitated areas of both operations. Long term monitoring and continued research are still required on many aspects of this disease.

By developing dieback control procedures based on scientific knowledge of the pathogen and the disease process, and integrating these procedures with routine mining procedures, we believe both companies have successfully minimised the impact of their mining operations on surrounding and re-established native vegetation communities.

Introduction

Plant disease has a major financial impact on the mining operations of Alcoa of Australia Limited and RGC Mineral Sands Limited. However, there is more to impact than just financial considerations. Evaluation of the impact of disease on mining also needs to assess the impact of disease on the environmental management objectives of both companies. Achievement of these objectives is regarded as important to the overall success of the mining operations.

The only plant disease having a significant impact on Alcoa and RGC is dieback disease caused by *Phytophthora* species. This disease is a major threat to the native plant communities in the regions of Western Australia where both companies mine. Environmental management objectives have been developed for the mines; their focus is to protect the natural communities surrounding the mine and to create an ecosystem on the mined areas with vegetation similar to that which occurred before mining. The presence of dieback has the potential to prevent these objectives being achieved.

The aim of this review is to discuss both the financial impact of disease on Alcoa and RGC mining operations and the potential impact of plant disease on the environmental objectives of both companies.

Background to mining operations

Alcoa operates three bauxite mines in the jarrah forest, south east of Perth (Fig 1). Alcoa mines and rehabilitates an average of 450 ha of forest a year. Bauxite ore bodies tend to be on the mid to upper slopes and are of the order of 5-50 ha. The open cut mining procedures remove the hard laterite "caprock" and the more friable zone below the caprock. The depth of mining is usually between 3 and 5 m. Following mining the area is landscaped, the surface soil is returned and the area is ripped to remove compaction. Finally, the area is seeded with about 50 species of plants indigenous to

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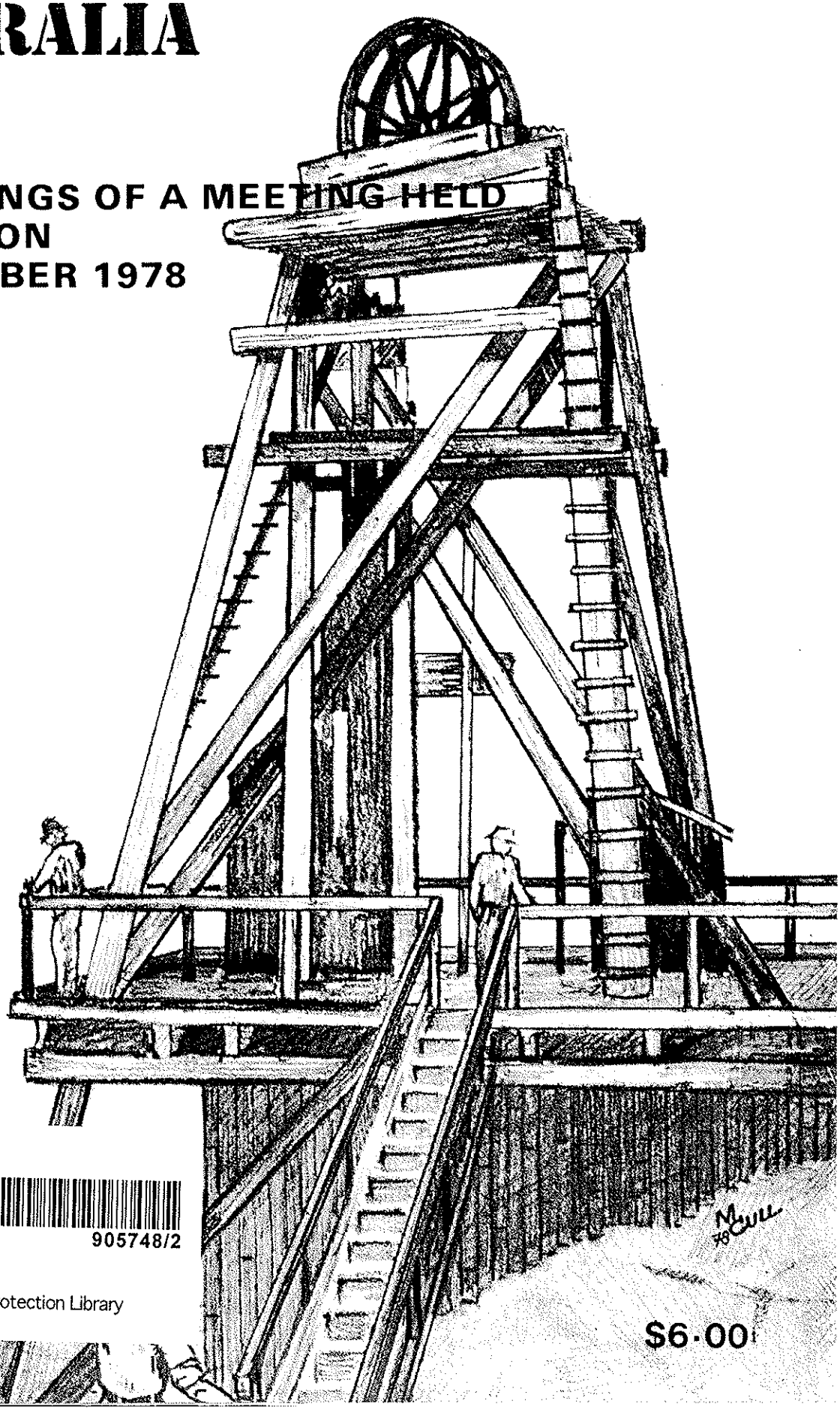
REHABILITATION OF MINED LANDS IN WESTERN AUSTRALIA

PROCEEDINGS OF A MEETING HELD IN PERTH ON 11th OCTOBER 1978



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Problems With Restoring Native Flora

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Abstract

Changing land use patterns are mentioned and early work at Broken Hill, N.S.W., briefly described for historical perspective. A comparative presentation is made of the different restoration approaches that are required for eremean regions and the sclerophyllous vegetation of the south-west. Rehabilitation techniques are considered in relation to the land management disciplinès in which they are found. Techniques used at Eneabba are briefly described with emphasis on the established practices of topsoil spreading and the use of cereal rye as a cover crop. The relevance of vegetation mapping and classification systems to rehabilitation programmes is discussed.

Social factors and future land use should be considered when planning restoration work. The accessibility of rehabilitation blocks by road increases human pressure on the developing ecosystem. Some protection from fire damage is necessary at early stages, even though controlled burning may produce benefits at a later stage of the succession. Some popular attitudes to native flora revegetation are discussed. Some benefits to botanical science and commercial horticulture are predicted.

Historical Introduction

In recent years changing economic conditions, cultural aims and land-use priorities have stimulated motivation towards the restoration of native flora and natural ecosystems. Thus we read of the abandonment of many northern pastoral enterprises, and in the south of much higher priorities being allotted to the protection of water catchments and large scale wilderness areas.

Of course the agricultural industries have by far the major role to play in these matters. However, recorded priorities required of landholders are slow to change and it is probably still true to say that taxation laws and statistical returns favour those who are clearing land of native flora, even when they have no clear aim of putting the land to productive use.

The extractive mining industry operates on much smaller areas where land may be denuded because of related activities such as exploration, transport and mining community activities, or directly in the case of open-cut mining. Because of the greater concentration of rehabilitation effort and a higher level of local motivation (particularly in arid areas), mining companies have usually led the way in the accumulation of "ways and means" knowledge on the restoration of native flora.

The first large scale Australian project began at Broken Hill, N.S.W., in 1936. The aim was to combat soil erosion and dust problems which had developed to the extent of threatening homes and mining sites. The motivation was entirely local and involved the co-operative activities of several mining companies and local government. The work was made possible by the study and experiments of Albert Morris, a local botanist and field naturalist. The use of large fenced enclosures in a semi-arid environment to keep out domestic, feral and native grazing animals was a pioneering step in rehabilitation which has since found many applications and has also become a standard method of range management assessment. Some 250 species of both indigenous and introduced plants became established in the Broken Hill regeneration areas. This is about the same number as the native species that occurred on the Allied Eneabba minesite (Lamont 1976) which we hope to regenerate in our rehabilitiation projects.

Eremean and Sclerophyllous Vegetation

There are actually three vegetation types in Australia that require separate consideration for regeneration methods. These are eremean (arid or semi-arid), sclerophyllous (predominantly hard leaved) and rainforest. The last can be ignored for most practical purposes in Western Australia.

Eremean communities contain a large proportion of pioneer species. Even those species that would not normally be considered as pioneers, are generally capable of establishment on fairly bare soils and are resistant to exposure factors such as wind and high temperatures. In spite of this, the seral stages of the particular vegetation to be regenerated is well worth study and application in the programme. Soil fertility considerations are generally minimal except where toxicity factors exist. Even where a high yielding introduced pasture species is to be used as a cover crop, work at Newman (Dames and Moore 1977) on iron ore waste dumps showed only very small response to added phosphate fertiliser.

As mentioned in the introduction, the exclusion of all herbivores may be critical for success in arid and semi-arid environments. By far the most important factor is water, but the season when this is applied is often not important.

Sclerophyllous communities contain a much smaller proportion of pioneer species, particularly those capable of establishment in the open cut situation where the whole soil profile has been disturbed. For this situation, successional studies carried out on roadsides or other cleared areas have little relevance. These communities are much more highly developed and integrated; thus seral stages, if known, are more of academic interest and rarely play an important part in rehabilitiation practices. An exception is the *Acacia* dominated subclimax often found in forest country. Generally, one has to aim much more directly at re-creating the climax community.

In spite of their hard leaves, many sclerophytes are very sensitive to exposure and sand blasting, particularly as young plants. They

often require virtual field nursery conditions for successful establishment and their seed are generally incapable of emergence from a deep setting on the soil. Thus mulch treatments become much more important.

Soil fertility is a difficult problem on which it is impossible to generalise. Each desired sclerophyllous community must be separately assessed; agricultural experience, where augmented fertility may be an important factor in getting rid of the same community, can be of great assistance if the principles learned can be applied in reverse. Some families, such as the Myrtaceae, may respond very well to added phosphates, while others, such as the Proteaceae, may do best in the most sterile medium. Added to this is the problem of applying fertiliser to establish a grass cover crop where this may be essential to prevent soil erosion and sand blasting, as in mineral sand tailings.

The exclusion of at least native herbivores is not necessary. In fact at Eneabba we are hopeful they will perform positive functions. Large numbers of kangaroos move into the rehabilitation areas each evening to crop the cereal rye *(Secale cereale* L.) cover crop. Possibly their droppings will bring in much needed native seed from the surrounding virgin areas. They also neatly crop almost to ground level competing grass adjacent to nursery raised and fertilised *Eucalyptus* spp.

Rehabilitation Techniques

It is not possible to produce a brief list of rehabilitation techniques of use in all situations where the restoration of native flora is desired. Rather, it is more profitable to group techniques in the various categories in which we find them for a wide range of land management disciplines. In theory this makes the native flora restoration specialist a "jack of all trades". In practice, it means that he, or she, must recognise the relevance to the problem of particular disciplines and then seek help from advisory institutions that happen to have the desired discipline represented on their staff.

The principal disciplines involved from the most extensive to the most intensive can be listed as follows:

Range management specialists,
Ecologists,
Agronomists or agrostologists,
Soil conservationists, and
Horticulturists.

This explains why many companies face a trivial dilemma when deciding what to call their land restoration specialist, particularly when only one person is involved. Ecologist, agronomist or horticulturist appear to be the general choice. Of course there are other disciplines whose activity spreads over the whole range and whose help to a programme may be vital. Chief among these are the botanist, plant pathologist and biometrician. To round off a programme the fauna and/or livestock factors cannot be ignored; this brings in disciplines such as zoology, animal production, entomology, microbiology, etc.

As pointed out above, effective rehabilitation techniques differ widely according to whether the vegetation is eremean or sclerophyll. A discussion on techniques suitable for arid enviroments was presented in the paper by Black and Trudinger (1976). Many publications exist on the development of suitable techniques for jarrah *(Eucalyptus marginata* SM.) forest areas where bauxite mining has occurred. These have been summarised in Alcoa of Aust. (1978 sections 6 and 7). Techniques found most useful to date in stabilising sand mining tailings at Eneabba are the spreading of topsoil and the use of cereal rye as a cover crop. The raising of native plants in a nursery and the spreading of native vegetation mulch are under trial at present and appear promising. Techniques to be investigated in the future including irrigation and the broadscale collection of large quantities of native seed.

The use of topsoil is an interesting example of the one technique used for differing reasons in both eremean and sclerophyll programmes. In arid environments topsoil often has very poor physical characteristics and may be difficult to collect and stockpile. However, it usually contains a valuable store of viable native seeds and, when collected with all vegetation material, can be used as an effective source of organic mulch (Black and Trudinger 1976). On the other hand in sclerophyll regions, the physical characteristics of topsoil is its most valuable asset, when used to cover the clay floor of bauxite pits, or sand mining tailings. Work at Eneabba has further shown that native plant establishment from seed stored even in fresh topsoil is extremely poor and that vegetation mulch, to be effective, must be collected separately from, and prior to, topsoil scraping.

Vegetation Classification

When restoring native flora some clear aims are required on the type of vegetation that should be attained at the completion of the programme. This information should include a species list, percentage ground cover, life form of the more common individual species and the general structure of the community. A survey of the vegetation prior to mining is very useful (*e.g.* Lamont 1976 for the Allied Eneabba minesite), though surveys of neighbouring areas of vegetation carried out during the programme may well suffice. However, it is important that such aims should not be inflexible,

because of unavoidable changes to the soil profile and landform.

For work in the arid and semi-arid regions of Western Australia, the vegetation survey maps of J.S. Beard are very useful (for example Beard 1975 covers the Pilbara area). Greater detail than shown on these maps is usually required for work on a particular locality; thus the preparation of a local vegetation map based on a field survey and aerial photographs is usually worthwhile.

Such a survey may include locations some distance from the work area if it is considered that these more closely match the landform and soils created by the mining disturbance. In this way a more effective approach to the restoration of native flora may include the use of species and the creation of seral communities that do not occur in the actual work locality. If a number of options are available, a field trial may be set up to compare the effectiveness of different plant communities that will develop from topsoil and mulch samples collected from the differing habitats. This approach works well in the eremean environment.

In sclerophyll climax plant communities, structure, life form and the dominant plant species tend to remain constant over larger areas. In forested areas, this constancy may spread across differing plant habitats, for example exposed ridges and certain valley floors can both be jarrah dominant. Thus vegetation maps based on the usual criteria are of little use for rehabilitation work in these situations.

A recent classification of the vegetation in the Northern Jarrah Forest into site-vegetation types (Havel 1975) should be useful for restoring native flora. The selection of this method was done with land use studies in mind and the classification is based on the floristic composition of the understorey. Using principal component analysis and applied criteria, the recorded species were reduced to some 70 indicator species from which the site-vegetation types could be defined. Further, from within and between these types, species groups were selected which showed an almost identical ecological behaviour. Thus an assessment of indicator species presence or absence allows a deduction to be made on the habitat properties of the site.

The habitat properties of a rehabilitation site may not match any natural site very closely and there could be successional factors affecting the development of indicator species on a disturbed site. However, the latter possibility would be common throughout many parts of the Northern Jarrah Forest. Thus this classification system does offer some hope for a planned approach to restoring native flora and a background of known vegetation types on which the monitoring of progress could be based.

Social and Land Management Factors

Future Land Use Potentials

In restoring native flora we hope to create a natural ecosystem which will have sufficient buffering capacity for natural regeneration to be able to maintain itself after the first three to four years of intensive effort. This is the main distinction between this work and the creation of agricultural or horticultural ecosystems. Future successional development will lead to vegetation forms of which we cannot be certain.

Nevertheless, in practically all cases there will be one or more future land uses which should be taken into account during the intensive period of land formation and vegetation encouragement. Some future land uses for the same area may be quite compatible, for instance water catchment and the conservation of flora, while others could bring some degree of conflict, such as use for pastoral rangeland and the conservation of fauna. Such conflicts should be resolved during the planning stages of the work.

One economic advantage of restoring native flora is that surface levelling and grading requirements are rudimentary compared with the requirements for an agricultural ecosystem to be produced. If the new surface is a water catchment close to streams feeding a reservoir, special attention would be required for erosion control and the construction of silt traps.

In most southern locations, transport access to the mine and the rehabilitation tracks that are required for the early intensive work, tend to leave the rehabilitation areas more open to visitors than adjacent unmined areas. This can produce a "no win situation" in which the more visitors arrive to assess the success of the programme, the less favourably the rehabilitation areas will compare with adjacent untouched areas. This factor also lends strong weight to the future use of such rehabilitation sites as recreation areas. If this land use appears inevitable, it should be planned for by providing such amenities as a circuit sealed road, a camping area, toilets and perhaps an artificial lake.

Fire Protection

Developing rehabilitation areas are fairly resistant to fire damage because of incomplete vegetation cover and a network of access tracks. The most important danger at an early stage would be the burning of a cover crop. The burning of a mulch layer could also greatly set back progress.

Those fortunate enough to be working within the jurisdiction of the Forests Department fire control organisation can plan operations within fairly stable parameters.

However, outside these areas fire control has no such expertise to guide it and a rehabilitation programme may risk as much damage from officially lit fires as from those accidentally lit. Some organisation and preparation to counter this threat is necessary with every rehabilitation team.

The careful firing of limited areas should bring some benefits in a sclerophyll community containing an appreciable proportion of *Acacia* species and members of the family Proteaceae. Such activity should be restricted to advanced stages of the succession when *Eucalyptus* trees are mature enough to resist fire damage and there are sufficient mature seed plants of the types that require fire to multiply their numbers.

Practical Problems and Influences at the Minesite

The traditional Australian land clearing complex is alive at minesites as in all other population groups. The only exceptions are certain well established groups in arid environments where dust and erosion ar ver-present hazards. Thus many people become cynical and find incongruous a deliberate programme for the re-creation of areas of native flora. Commonly with these people, the inconsistency of requiring unspoiled areas for their recreation goes unrecognised.

Thus one must expect to find at least some tacit disapproval when working on the restoration of native flora. This may express itself on and off the minesite in a variety of ways. On the positive side, the average mine worker at Broken Hill probably has a vocabulary many times larger than the average Australian when words are required for the naming of trees. Botanical plant names would be more widely used on minesites than in any other population group outside the plant professions. The level of local motivation is generally high, particularly in arid regions, and responds well to public relations efforts.

Fi. government directives on rehabilitation such as we have at our Eneabba site are probably the most important contribution towards overcoming social problems. On the contrary, in my view, official notices prohibiting the non-commercial picking of wildflowers are probably counter-productive.

All rehabilitation programmes require an in-service education and public relations component to explain what is going on and to ask for co-operation. Mine traffic is heavy footed and usually in a hurry to get to its destination. Poorly constructed roads often mean an ever widening strip of flattened virgin vegetation which may appreciably add to the rehabilitation bill. Unmarked rehabilitation tracks may lead slow tracked vehicles to a blind end where the inevitable way out is a cross country hike through what may be a carefully planted rehabilitation block or trial.

Benefits to Botany and Horticulture

To avoid too much emphasis on the problems of restoring native flora, some mention should be made of potential benefits other than the obvious ones brought about through land stabilisation and revegetation.

Premining surveys, the collection of seed for nursery propagation and the monitoring of the progress of the programme inevitably produces much data on the species and varieties of plants present on or near the site. Mining is quite frequently carried out in remote areas inadequately surveyed and collected for botanical specimens. Provided the work is done in liaison with professional botanists, a great deal of knowledge concerning new species, new varieties and their distribution may be gained. Close observations on the growth of plants in a range of new soil media may also produce valuable information on their ecology and reproductive systems.

The commercial horticulture of native plants has progressed considerably in recent years as home owners and park supervisors look for plants requiring less attention and less water. Native flora restoration programmes should produce new horticultural types, particularly where nursery activities are involved.

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Interaction of multiple disturbances: importance of disturbance interval in the effects of fire on rehabilitating mined areas

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Abstract Multiple disturbance regimes are increasingly common as novel anthropogenic disturbances are added to existing natural disturbances. However, it is generally unknown whether simultaneous or sequential effects of different forms of disturbance are predictable from the independent effects of each disturbance. This study examines the short-term effects of sequential disturbance by mineral sand-mining followed by fire in a forest community in south-eastern Australia. Four combinations of disturbance were sampled: unburned mined, burned mined, unburned forest (unmined) and burned forest (unmined, with between-fire interval matching the disturbance interval between mining and fire of the burned mined treatment). All combinations were sampled approximately 12 months following fire on the burned sites. The impact of fire after mining depended on disturbance interval. Sites burned 0.5–2.4 years since mining had fewer native vascular plant species than unburned mined sites of the same mined age, whereas sites with 10–16 years or 20–26 years between mining and fire had greater native species richness than unburned mined sites of the same age. Burning 20–26 years after mining brought native species richness within the range of burned forest. For both unmined and mined sites native seedling densities increased with burning, and with longer disturbance intervals. Weed species richness and weed seedling densities were greater on mined sites than in forest, and burning mined sites elevated weed seedling densities further, particularly for short intervals. Both disturbance interval and fire intensity are likely to have contributed to these results, as intensity on mined areas increased with interval, and at 20–26 years post-mining was equivalent to unmined forest. These results suggest that fire could be used to promote rehabilitation of these mined areas after at least 10 years, but should be excluded from earlier stages of post-mining regeneration. However, other sources of spatial and temporal variability should be considered in addition to interval and intensity, as variation among mined areas was correlated with post-fire weather conditions and available weed sources. Finally, the combined effects of mining and fire could not be predicted from knowledge of the disturbances operating separately, indicating that effects of multiple disturbance may be synergistic rather than additive.

Key words: fire, interval, mining, multiple disturbance, restoration

INTRODUCTION

Disturbance is recognized as a key process affecting levels of diversity and causing community change (Levin & Paine 1974; Connell & Slatyer 1977; Connell 1978; Huston 1979; Petraitis *et al.* 1989; Hobbs & Huenneke 1992; Tilman & Pacala 1993). Most studies of disturbance have focused on the effects of one form of disturbance, dealing either with an individual disturbance event (Specht *et al.* 1958; Purdie & Slatyer 1976; Hopkins & Robinson 1981), or recurrence of the same form of disturbance (Nieuwenhuis 1987; Cary & Morrison 1995; Taylor *et al.* 1998). Considering habitat as mosaics of patches with different disturbance histories (Sousa 1984), we recognize the importance of sequences of disturbances, and the need to deal with a disturbance event within the framework of the disturbance regime (Gill 1975; Fox & Fox 1987). However, relatively few studies have explicitly tested the simultaneous or sequential impact of more than one form of disturbance, which we have termed 'multiple disturbance' (but see Dickinson & Kirkpatrick 1987; Kirkman & Sharitz 1994; Wassenaar & van Aarde 2001) and few with fully balanced designs (but see Leigh & Holgate 1979; Collins 1987; Belsky 1992; Taylor & Fox 2001). Yet multiple disturbance regimes are increasingly common, as exogenous disturbances are frequently compounded with existing or modified versions of endogenous disturbances, or with further exogenous disturbances (Fox *et al.* 1996).

We cannot assume that the effect of multiple disturbances is the addition of the effects of the separate

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Smoke, Mulch, and Seed Broadcasting Effects on Woodland Restoration in Western Australia

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Abstract

Smoke, canopy-derived mulch, and broadcast seeds were used to maximize the establishment of *Banksia* woodland species in sand quarries in Western Australia. Smoke, particularly aerosol smoke, had a positive effect on total seedling recruitment. Pre-mined (woodland) sites showed a 42-fold increase in total germinants and a 3-fold increase in the number of species with aerosol smoke application. Post-mined (restored) sites showed only a 3.6-fold increase in total germinants and a 1.4-fold increase in the number of species. Two water-based smoke chemicals, DC10 (pH 4.5) and SC63 (pH 2.5), increased seedling recruitment at both sites but were not as effective in stimulating recruitment as aerosol smoke. Neither of the chemicals were effective in significantly increasing species richness. Application of aerosol smoke directly to seeds as a pretreatment before broadcasting had no effect on seedling recruitment. Broadcasting of seeds onto restoration sites significantly increased seedling abundance and richness. Application of a single layer of mulch from the canopy vegetation after seed broadcasting gave optimum seedling recruitment. Two layers of mulch significantly reduced recruitment, as did applying mulch before seed broadcasting. For broad-scale restoration, the application of smoke on newly restored sites would be more effectively achieved using smoke water sprayed over the soil surface. Species that do not recruit from replaced topsoil could be effectively recovered from broadcast seed rather than from mulch.

Key words: aerosol smoke, *Banksia* woodland, germination, mulch, sand extraction, seedling recruitment, smoke water, species richness, topsoil, restoration.

Introduction

The *Banksia* woodland community is widespread and common on the Swan Coastal Plain in the southwest of Western Australia. Rocla Quarry Products Pty. Ltd. operates a sand extraction quarry on *Banksia* woodlands over deep siliceous sands 30 km northeast of Perth, Western Australia. The extraction process involves removal of the *Banksia* woodland vegetation and soil profile to great depths (18–22 m). The company proposes to restore post-mined sites with an ecosystem closely resembling the pre-mined plant community. Unfortunately, the recruitment biology of *Banksia* woodland species is not well understood, providing limited opportunities for devising appropriate prescriptions for restoration and management of this plant community.

As a routine part of the restoration process the company removes the woodland topsoil, which contains soil-stored seeds, and returns the topsoil into areas to be restored. The value of fresh topsoil for increasing plant diversity in mine-site restoration is well documented (Tacey & Glossop 1980; Koch et al. 1996). For those species that do not store their seeds in the soil seedbank, replacement of species is achieved by use of broadcast seeds or through planting of green stock into restored sites. A mulch comprising canopy material of the native vegetation is occasionally applied to areas being restored after topsoil replacement. Following these practices, the diversity of recovered *Banksia* plant communities has been limited (S. Elliot and S. Fletcher, personal communication, 1995).

The recent development of aerosol smoke as a germination tool (Dixon et al. 1995; Roche et al. 1997, 1998) has provided opportunities for increasing recruitment and species biodiversity in mine-site restoration practices. The application of aerosol smoke, however, is not considered effective or practical for use in broad-scale restoration activities. After their initial smoke studies, Roche et al. (1997) tested the application of smoke water

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to the soil seedbank, a more practical method of applying this germination trigger over large areas. There was a significant germination response, with total germinants and species numbers increasing by 56 and 36%, respectively. Additionally, Roche et al. (1997) reported that treatment of seed mixes with aerosol smoke before broadcasting resulted in highly significant increases in total in situ recruitment and species numbers when compared with unsmoked seeds. Interestingly, even species normally unresponsive to smoke ex situ were responsive to smoke in situ. It was suggested that smoke may act as a deterrent against in situ seed predation (K. Dixon, personal communication, 1997). In addition, these authors found that smoke residues carried by the smoked broadcast seed may stimulate germination of seeds already resident in the soil seedbank.

Maximizing rehabilitation of the full suite of species present before mining requires an understanding of the ecology of the species involved, particularly their seed storage syndromes. Although the use of the topsoil seedbank can contribute the most seedling recruitment in post-mined sites, many species store their seeds in persistent fruits held in the vegetation canopy (serotiny). A unique feature of the kwongan vegetation (a fire-adapted shrubland) in Western Australia is that a substantial proportion of the seeds are stored in these persistent fruits (Bellairs & Bell 1993). Bellairs and Bell (1993) reported that a mulch of the canopy material was the most important source of seeds in post-mined sites, containing 85% of the total germinable seeds, whereas topsoil and broadcast seed contributed 13 and 2% of the germinable seed. On the contrary, at Alcoa of Australia's mine sites in forest areas in the southwest of Western Australia, broadcasting of native seeds provides one of the best methods of achieving high plant densities and cover (Koch & Ward 1994). It has been proposed that mulch may also ameliorate adverse conditions affecting seedling recruitment by decreasing wind and water erosion (Bellairs 1992). It would appear that the value of the topsoil, broadcast, and mulch-based seed varies with plant community, topsoil handling and storage procedures, mining type, practicality, and cost effectiveness.

This study represents part of a broader investigation on plant ecosystem restoration of *Banksia* woodlands. Specifically, we investigate (1) the effect of aerosol smoke and smoke water on the seedling recruitment from the topsoil seedbank, (2) the effect of aerosol smoke on broadcast seeds as a deterrent against seed predation, and (3) the optimum concentration of mulching and order of mulching and broadcast seeding on seedling recruitment.

Methods

Study Site

The study sites were located within an area previously mined by Rocla and an adjacent *Banksia* woodland as a reference area (Fig. 1). The *Banksia* woodland comprises a floristically rich and taxonomically diverse plant community similar to the adjacent kwongan plant communities in the northwest of the Swan Coastal Plain (Dodd & Griffin 1989). The tree canopy of the *Banksia* woodland is comprised of *Banksia attenuata* (slender banksia) and *B. menziesii* (firewood banksia), with *Eucalyptus todtiana* (prickly bark) and *Nuytsia floribunda* (Christmas tree) occurring less frequently (Beard 1989; Dodd & Griffin 1989). The understory is represented by members of the dominant woody families Myrtaceae, Epacridaceae, Proteaceae, and Papilionaceae, with dominant herba-

Figure 1. *Banksia* woodland before quarrying activities.

ceous elements represented by members of the Antheri-caceae, Stylidiaceae, Haemodoraceae, Cyperaceae, and Dasypogonaceae families (Trudgen 1997). The *Banksia* woodland habitat experiences a typically Mediterra-nean-type climate of cool mild winters and hot dry summers and is characterized by deep, well-drained, nutrient-poor, sandy soils.

Experiment a: Aerosol Smoke and Smoke Water Effects on the Seedbank

In each study site, an area encompassing 30 × 30 m was selected for in situ control, smoke water, and aerosol smoke treatments. At the post-mined site, the area was covered with topsoil that had been freshly harvested from another area of the woodland. The topsoil was applied to a depth of 10 cm, which mimicked the depth of collection. At the woodland site, the vegetation was cleared and the litter layer was carefully removed by hand so as not to disturb the soil seedbank. For each treatment, three replicate samples measuring 5 × 1 m were selected at random and pegged out in each site, with a buffer zone of 1 m between all treatment replicates.

All treatments were carried out during May (autumn) 1997 before the onset of winter rains. The smoke water treatments consisted of two commercially available concentrated smoke water products, DC10 (commercially known as "Kings Park smoke water," Kings Park and Botanic Garden, Australia) and SC63 (commercially known as "Regen 2000," Grayson Trading, Australia), each applied at three volumes (100, 50, and 10 mL/m^2) using a hand-held boom spray. The Kings Park smoke water was produced by drawing smoke from a 60-L steel combustion drum through 100 L of water for 1 hr. The aerosol smoke was applied to the soil surfaces by erecting 5 × 1 × 1-m high plastic tents over the replicates and fan forcing smoke from a 60-L steel combustion drum through a cooling pipe system into three tents simultaneously for 1 hr (Roche et al. 1997) (Fig. 2). Dry *Banksia* woodland plant material was used as fuel for the generation of smoke. After treatment the treated soil surface was stained brown. Including the control, there was a total of eight treatments per site.

Seedling recruitment of all plant species was recorded during November (late spring) 1997. Results for each species were analyzed by a one-way analysis of variance using the eight treatments as independent variables to determine smoke water and aerosol smoke effects on seedling recruitment. Fisher's protected least significant difference at $p < 0.05$ was used to determine post-hoc significance.

Experiment b: Aerosol Smoke Treatment of Broadcast Seeds

A blended seed mix of 13 species routinely used in restoration and indigenous to *Banksia* woodland was used in these trials. Species investigated were the canopy species *Allocasuarina fraseriana* (Fraser's sheoak), *A. humilis* (dwarf sheoak), *Banksia attenuata*, and *B. menziesii* and the understorey species *Acacia pulchella* (prickly Moses), *Beaufortia elegans* (Myrtaceae), *Bossiaea eriocarpa* (common brown pea), *Gompholobium tomentosum* (hairy yellow pea), *Hovea pungens* (devil's pins), *Hovea trisperma* (common hovea), *Jacksonia densiflora* (Fabaceae), *Nemcia capitata* (Fabaceae), and *Xanthorrhoea preissii* (grasstree).

For the ex situ trial seeds in seedling trays lined with Wettex perforated cloth were randomly placed in a tent receiving aerosol smoke for 1 hr (see above). Once

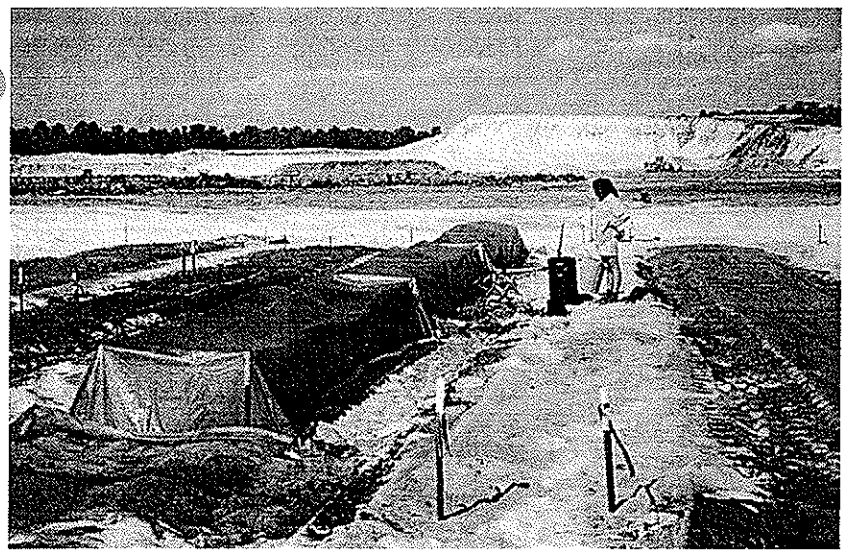


Figure 2. Smoke-filled plastic tents used to treat freshly replaced topsoil in a post-mined site at Rocla Quarry Products, Western Australia.

smoke treated, seeds were sown in plastic seedling punnets filled with white sand to a depth of 0.5 cm below the rim of the punnet. Four replicates of 100 seeds for each species were randomly assigned to each of the punnets for smoke-treated and untreated (control) seeds. Once spread, a very thin layer of the white sand was sieved over the seeds to stabilize the surface and seeds. Punnets were randomly positioned in a glasshouse and watered as required. The emergence of seedlings was recorded after 16 weeks. Percentage germination values for each species were analyzed by a *t* test.

In April (mid-autumn) 1996 an in situ trial was then conducted on seeds that were nonresponsive to the smoke treatment in the ex situ experiment. The non–smoke-responsive seed enabled the benefit of smoke as a predation deterrent in the field to be determined. For this trial aerosol smoke-treated seeds were sown directly into a 10 × 10-m quadrat in a restored site spread with 10 cm of topsoil. The seed was sown at a rate of 4 kg/ha (double the current seeding rate to facilitate scoring of germinants). A control plot of the same size was separated by a 2-m buffer strip and sown with the equivalent amount of unsmoked seed (see Roche et al. 1997 for a similar design). Three replicates of 5 × 1 m for each treatment were pegged, and germinants were scored in November (late spring) 1996. Germination values for each species were analyzed by a *t* test.

Experiment c: Mulch and Seed Broadcasting Effects

To determine the optimum thickness of mulching and the order of mulching and broadcast seeding, trials were conducted in subplots in areas undergoing restoration. The study area encompassed 0.6 ha, with the experimental design consisting of eight treatments over topsoil: control, seed mix, thin mulch (5 mm), thin mulch + seed mix under, thin mulch + seed mix over, thick mulch (10 mm), thick mulch + seed mix under, and thick mulch + seed mix over. The seed mix contained the canopy species *B. attenuata, B. menziesii,* and *E. todtiana*. The understorey species in the seed mix were *Acacia pulchella, Allocasuarina fraseriana, A. humilis, Bossiaea eriocarpa, Eremaea pauciflora* (Myrtaceae), *Hovea trisperma, Hypocalymma robustum* (Swan River myrtle), *Jacksonia densiflora, J. furcellata* (Fabaceae), *Kunzea ericifolia* (spearwood), *Melaleuca scabra* (rough honeymyrtle), *M. trichopylla* (Myrtaceae), and *Regelia inops* (Myrtaceae). A mulch of the canopy material from the adjacent native vegetation was made using a large-scale mulcher.

All treatments were carried out during May 1997 before the onset of winter rains with three 5 × 1-m replicate plots pegged in each of the eight treatment plots. Seedling recruitment of all plant species was recorded during the first week in November 1997. Results for each species were analyzed by a one-way analysis of variance using the eight treatments as independent variables. Fisher's protected least significant difference at $p < 0.05$ was used to determine post-hoc significance.

Results

Experiment a: Aerosol Smoke and Smoke Water Effects on the Seedbank

In general, the recruitment in the woodland consisted of 22 species in comparison with the 41 species recorded in the restored site. Twenty-five species that recruited in the restored site did not respond to any of the given treatments in the woodland (Table 1). Similarly, six species that recruited in the woodland did not respond to any of the given treatments in the restored site (Table 2). All treatments, including the control, had higher levels of recruitment in the restored site.

Aerosol smoke significantly increased total seedling recruitment by 42.3-fold (Fig. 3a) and species richness by 2.8-fold (Fig. 3b) in the woodland compared with the control. Nine species showed significantly higher recruitment with aerosol smoke treatment compared with the control, representing 41% of the total number of species (Table 1). Another seven species showed nonsignificant higher recruitment with aerosol smoke treatment. Six species that did not show a response to aerosol smoke occurred in low densities in both aerosol smoke and control plots. No species showed a negative effect as a result of aerosol smoke application.

In the restored site aerosol smoke significantly increased total seedling recruitment by 3.6-fold (Fig. 3a) and species richness by 1.4-fold (Fig. 3b). The lower response of this treatment, compared with the woodland, was associated with the higher level of recruitment in the control. Fourteen species showed significantly higher recruitment with aerosol smoke treatment compared with the control, representing over one third of the total number of species (Table 2). Another 13 species showed nonsignificant higher recruitment with aerosol smoke treatment. Three species (*Adenathos cygnorum* [woolly bush], *Anigozanthos manglesii* [Mangle's kangaroo paw], and *Boronia ramosa* [Rutaceae]) showed a significant negative response to aerosol smoke treatment. Eleven species did not show a response to aerosol smoke compared with control.

The liquid-based smoke agents increased recruitment at all rates in the woodland compared with the control by at least twofold (Fig. 3a). However, the only significant response (DC10 at 50 mL/m^2) was 23% of that achieved with aerosol smoking. Most species showed no significant smoke water effect (Table 1). In the restored site, all but one treatment (DC10 at the lowest rate) increased recruitment compared with the control, with most treatments demonstrating only a 1.2-fold in-

Table 1. Aerosol smoke and smoke water effects on seedling recruitment of *Banksia* woodland species in *Banksia* woodland.

		Smoke Water						
Species represented	*Control*	*DC10 (10 mL/m²)*	*DC10 (50 mL/m²)*	*DC10 (100 mL/m²)*	*SC63 (10 mL/m²)*	*SC63 (50 mL/m²)*	*SC63 (100 mL/m²)*	*Aerosol Smoke*
Acacia huegelii	0	0	0	0	0	0.33 ± 0.33	0	0.33 ± 0.33
Astroloma macrocalyx	0.33 ± 0.33	0	0	0.33 ± 0.33	0	0	0	0
Banksia attenuata	0 [a]	0 [a]	0.33 ± 0.33 [ab]	0 [a]	0.33 ± 0.33 [ab]	0 [a]	0.33 ± 0.33 [ab]	1 ± 0.58 [b]
Banksia menziesii	0.67 ± 0.67	0	0	0	0	0	0	0
Boronia ramosa	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	1.33 ± 0.33 [b]
Bossiaea eriocarpa	0.33 ± 0.33 [ab]	0 [a]	0 [a]	0 [a]	0 [a]	0.67 ± 0.33 [b]	0 [a]	1.33 ± 0.67 [b]
Calytrix sp.	0.67 ± 0.33 [ab]	0 [a]	1 ± 0.58 [ab]	4.67 ± 3.18 [b]	1.33 ± 0.88 [ab]	0 [a]	3.67 ± 3.67 [ab]	0.33 ± 0.33 [ab]
Conospermum stoechadis	0.67 ± 0.33 [ab]	0 [a]	0 [a]	0 [a]	0 [a]	0.33 ± 0.33 [ab]	0 [a]	4.67 ± 3.71 [b]
Conostylis aculeata	0 [a]	0.33 ± 0.33 [a]	0.33 ± 0.33 [a]	0.67 ± 0.67 [a]	0.33 ± 0.33 [a]	0.67 ± 0.33 [a]	0.67 ± 0.67 [a]	16 ± 5.69 [b]
Dampiera linearis	0	0	0.33 ± 0.33	0	0	0	0.67 ± 0.33	2 ± 2
Gompholobium tomentosum	1 ± 0.58 [ab]	1 ± 0.58 [ab]	0.67 ± 0.33 [ab]	1 ± 0.58 [ab]	0.33 ± 0.33 [a]	3 ± 1.53 [b]	2.33 ± 1.45 [b]	1.67 ± 0.67 [b]
Gonocarpus pithyoides	0	0	0.33 ± 0.33	0	0	0	0	0
Hibbertia racemosa	1.67 ± 0.33 [a]	7.33 ± 2.73 [ac]	14.3 ± 11.3 [ac]	7 ± 3.79 [ac]	6.67 ± 3.53 [ac]	17 ± 6.66 [ac]	22.3 ± 13.6 [c]	133 ± 32.9 [b]
Hibbertia subvaginata	0 [a]	0 [a]	0 [a]	0 [a]	0.33 ± 0.33 [ac]	0.33 ± 0.33 [ac]	0.67 ± 0.33 [c]	1.67 ± 0.33 [b]
Laxmannia spp.	0 [a]	0 [a]	1.67 ± 1.2 [a]	2.33 ± 1.86 [a]	0.33 ± 0.33 [a]	0.67 ± 0.67 [a]	0.33 ± 0.33 [a]	17 ± 8.33 [b]
Leucopogon spp.	0 [a]	1.33 ± 0.88 [a]	6 ± 5.03 [a]	10.67 ± 9.7 [ab]	0.67 ± 0.67 [a]	6.33 ± 2.85 [a]	6 ± 3.06 [a]	32.7 ± 21.3 [b]
Nemcia capitata	0	0	0	0	0	0	0	0.67 ± 0.67
Opercularia vaginata	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	6 ± 3.79 [b]
Patersonia occidentalis	0	0	0	0.33 ± 0.33	0	0	0	0
Scholtzia involucrata	0	1 ± 1	0	2.33 ± 1.86	0	2.67 ± 2.19	0.33 ± 0.33	0
Stylidium spp.	0.33 ± 0.33 [a]	2 ± 1.15 [ac]	32.7 ± 25.7 [b]	11 ± 4.16 [ab]	13.33 ± 4.9 [bc]	6.67 ± 3.18 [ab]	3.67 ± 0.33 [ab]	21.3 ± 13.8 [b]
Thysanotus dichotomus	0	0	0	0.33 ± 0.33	0	0	0	0.33 ± 0.33

Values are seedlings per 5 m² ± SE. Different letters indicate significant differences using Fishers PLSD $p < 0.05$.
Species that did not respond to any of the given treatments: *Adenanthos cygnorum, Anigozanthos manglesii, Arnocrinum preissii, Beaufortia elegans, Comesperma calymega, Conostylis candicans, C. setigera, Desmocladus flexuosus, Eremaea pauciflora, Hibbertia huegelii, H. hypericoides, Hovea pungens, H. trisperma, Jacksonia densiflora, Leschenaultia floribunda, Lomandra* spp., *Lyginia imberbis, Macarthuria australis, Melaleuca huegelei, Stirlingia latifolia, Thysanotus arenarius, T. manglesianus,* Unknown grass 1, and *Verticordia nitens.*

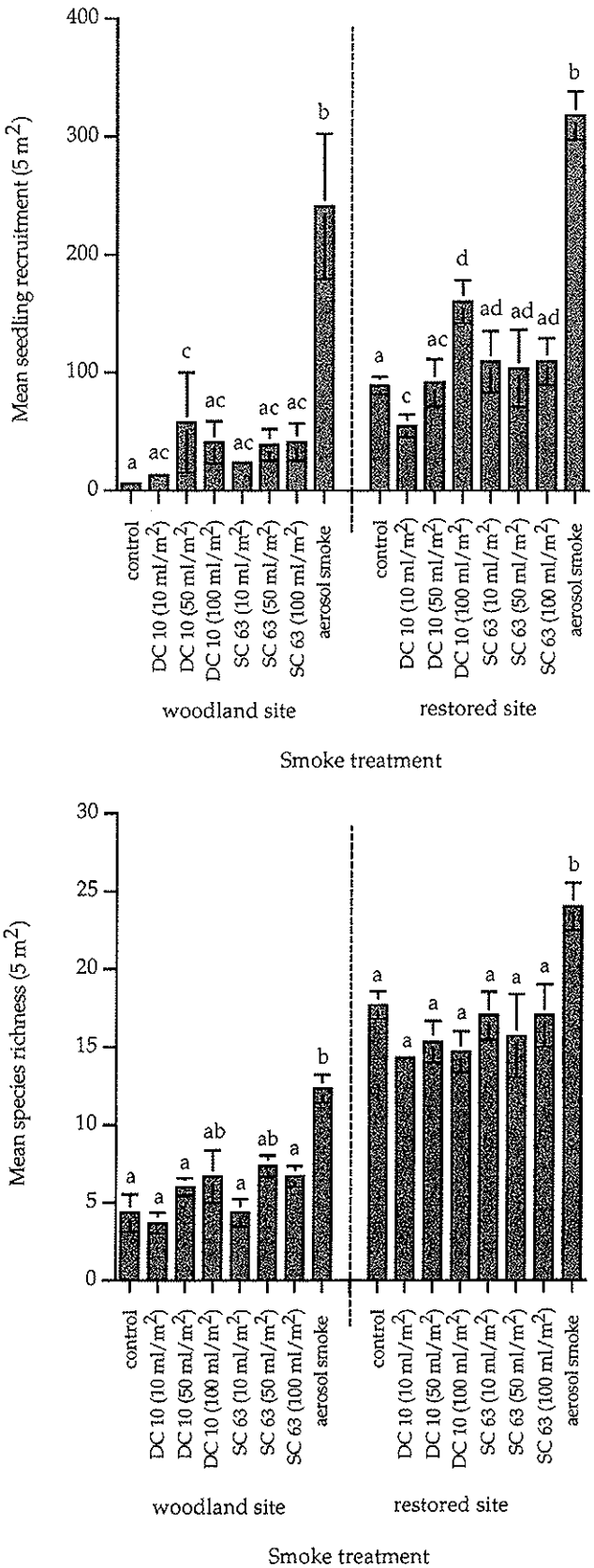
crease (Fig. 3a). However, the only significant response (DC10 at the highest rate) was 50% of that achieved with aerosol smoking. The differences in total recruitment response between control and smoke water plots and between smoke water and aerosol smoke plots in the restored sites were not as great as those recorded in the woodland. Again, most species showed no significant smoke water effect (Table 2). At both sites, application of smoke water failed to increase the number of species in comparison with the control (Fig. 3b). Higher levels of recruitment (Fig. 3a) and species richness (Fig. 3b) were recorded with the higher applications of the two smoke water agents. For example, in the restored site, *Astroloma macrocalyx* (swan berry), *Leucopogon* spp. (Epacridaceae), *Nemcia capitata* (Fabaceae), *Scholtzia involucrata* (spiked scholtzia), *Stylidium* spp. (Stylidiaceae), and *Verticordia nitens* (Morrison featherflower) showed significant effects of increasing rates of the smoke water agents (Table 2).

Experiment b: Aerosol Smoke Treatment of Broadcast Seeds

The application of smoke ex situ to a range of species did not have an effect on nine species, including six of the seven legumes, *B. elegans* (Myrtaceae), *Allocasuarina fraseriana*, and *Allocasuarina humilis* (Fig. 4). The application of smoke to these broadcast seeds did not substantially alter total in situ seedling recruitment (i.e., topsoil and broadcast species), with 127 ± 21.6 versus 143 ± 35.6 seedlings per 5 m² in the control. The application of smoke to the nine broadcast seeds that were unresponsive to smoke ex situ did not alter their seedling recruitment levels. However, four species present only as topsoil stored seed showed a significant positive germination response after application of smoked broadcast seed to the site: *Scholtzia involucrata* (0.3 ± 0.3 vs. 3.3 ± 0.9 seedlings per 5 m², $p = 0.0335$), *Hibbertia huegelii* (Dilleniaceae) (0 vs. 1.3 ± 0.3 seedlings per 5 m², $p = 0.0161$), *H. subvaginata* (Dilleniaceae) (0.7 ± 0.7 vs. 4.0 ± 0.6 seedlings per 5 m², $p = 0.0194$), and *Scaevola paludosa* (Goodeniaceae) (0 vs. 1.3 ± 0.3 seedlings per 5 m², $p = 0.0161$).

Experiment c: Effect of Mulch and Broadcast Seeds

In general, mulch not only inhibited the recruitment of topsoil and broadcast species, but applying broadcast seeds after mulch application substantially limited recruitment. Seed broadcasting over topsoil contributed 36% total seedling recruitment compared with control (Fig. 5a) and an additional seven species (Fig. 5b). Of the 16 species broadcast, only 11 species showed a response in at least one of the six treatments involving broadcast seeding. Mulch did not contribute any germinable bradysporous species. Furthermore, applying mulch over topsoil significantly decreased seedling recruitment and species richness by up to 64 and 50%, re-



spectively. The recruitment of three topsoil stored species, *Astroloma macrocalyx*, *Laxmannia* spp. (Anthericaceae), and *S. involucrata*, was significantly reduced by the application of mulch (Table 3).

Thin mulch application over broadcast seeds and topsoil did not differ in effect from seedling recruitment and species richness with seed broadcasting only (Fig. 5, a and b). However, a thick mulch application over broadcast seeds (and topsoil) decreased seedling recruitment and species richness by 40 and 30%, respectively. The broadcasted species, *Allocasuarina fraseriana*, *Beaufortia elegans*, *Eremaea pauciflora*, and *Eucalyptus todtiana*, were negatively affected by the thicker mulch layer, whereas *Bossiaea eriocarpa* was positively affected (Table 3).

Applying the broadcast seed mix over the mulch decreased recruitment by 36 to 38% (both thin and thick mulch layers), whereas there was no decreases in species richness. The recruitment of *Acacia pulchella*, *Banksia attenuata*, *Bossiaea eriocarpa*, *Eremaea pauciflora*, and *Eucalyptus todtiana* was negatively reduced by the mulch under the seeds.

Discussion

This study represents one of the first investigations into the use of smoke as a germination tool in *Banksia* woodland restoration. The findings confirm those of Roche et al. (1997) by showing that exposure to aerosol smoke enhances seedling recruitment of the topsoil seedbank, particularly in an "undisturbed ecosystem." Roche et al. (1997) reported a 48-fold increase and a 3.4-fold increase in total germinants from the topsoil seedbank after exposure to smoke in *Eucalyptus marginata* (jarrah) forest sites and newly restored sites, respectively. These results are very similar to the 42-fold and 3.6-fold increase with the application of smoke reported in this study. Aerosol smoke application also contributed to increases in species richness. Roche et al. (1997) reported a 3.9-fold and 1.9-fold increase in jarrah forest and newly restored sites, whereas our study shows a 2.9-fold and 1.4-fold increase in *Banksia* woodland and newly restored sites.

The recent development of aerosol smoke as a germination tool for many Australian native species has provided opportunities for increasing the total recruitment and the species richness in mine-site restoration through smoke-stimulated germination (Roche et al. 1997). How-

Figure 3. Aerosol smoke and smoke water effects on total seedling recruitment (a) and species richness (b) in woodland and restored sites. Bars represent standard error of the mean. Within the two sites, different letters indicate significant differences using Fishers protected least significant difference ($p < 0.05$).

Table 2. Aerosol smoke and smoke water effects on seedling recruitment of *Banksia* woodland species in restored sites.

		Smoke Water						
Species represented	*Control*	*DC10 (10 mL/m²)*	*DC10 (50 mL/m²)*	*DC10 (100 mL/m²)*	*SC63 (10 mL/m²)*	*SC63 (50 mL/m²)*	*SC63 (100 mL/m²)*	*Aerosol Smoke*
Adenanthos cygnorum	4 ± 1.53 [a]	0.33 ± 0.33 [bc]	0.67 ± 0.33 [bc]	0.33 ± 0.33 [bc]	0.33 ± 0.33 [bc]	2 ± 1.15 [b]	0 [c]	1.67 ± 0.33 [b]
Anigozanthos manglesii	5.67 ± 0.88 [a]	0.67 ± 0.33 [bc]	0.33 ± 0.33 [c]	1.33 ± 0.88 [bc]	0.67 ± 0.33 [bc]	1 ± 0.58 [bc]	1.33 ± 0.33 [bc]	2.33 ± 0.88 [b]
Arnocrinum preissii	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	1.33 ± 0.88 [b]
Astroloma macrocalyx	10 ± 4.73 [ac]	2.67 ± 0.88 [a]	19 ± 10.6 [ac]	30 ± 14 [bc]	15.67 ± 8.7 [ac]	19.3 ± 15.4 [ac]	8 ± 2.52 [ac]	68.67 ± 8.0 [b]
Beaufortia elegans	0 [a]	0 [a]	0.33 ±0.33 [ab]	0 [a]	0 [a]	0.33 ± 0.33 [ab]	0.67 ± 0.33 [b]	0 [a]
Boronia purdieana	0	0	0	0	0	0.33 ± 0.33	0	0.33 ± 0.33
Boronia ramosa	3.67 ± 0.33 [a]	0.67 ± 0.33 [b]	0.33 ± 0.33 [b]	0 [b]	0.67 ± 0.67 [b]	0.33 ± 0.33 [b]	0 [b]	0.67 ± 0.67 [b]
Comesperma calymega	0	0	0	0	0	0	0	0.33 ± 0.33
Conospermum stoechadis	0.33 ± 0.33	0	0	0	0	0	0	0
Conostylis setigera	1 ± 0.58 [a]	0 [b]	0 [b]	0 [b]	0 [b]	0 [b]	0 [b]	0.67 ± 0.33 [a]
Conostylis aculeata	1 ± 1 [a]	0.67 ± 0.33 [a]	0.67 ± 0.33 [a]	2.67 ± 0.88 [ab]	1.33 ± 0.33 [a]	2.00 ± 0.58 [a]	2 ± 0 [a]	3.0 ± 0 [b]
Conostylis candicans	0	0	0	0	0.33 ± 0.33	0	0	0
Dampiera linearis	0.67 ± 0.33 [ac]	0 [a]	0.33 ± 0.33 [ac]	2.33 ± 0.67 [b]	0.67 ± 0.33 [ac]	0.67 ± 0.67 [ac]	1.67 ± 0.88 [bc]	4 ± 1.00 [b]
Desmocladus flexuosus	0	0	0	0	0	0.33 ±0.33	0	0
Eremaea pauciflora	0.33 ± 0.33	0	0	0	0	0	0	0
Gompholobium tomentosum	3.67 ± 1.45 [ab]	1.33 ± 0.33 [ac]	3.67 ± 0.67 [b]	0.67 ± 0.67 [c]	0.33 ± 0.33 [c]	2 ± 1.15 [ac]	2 ± 0 [ac]	1.67 ± 0.88 [ac]
Gonocarpus pithyoides	5.33 ± 0.67 [ab]	3 ± 0.58 [ab]	4.33 ± 2.33 [ab]	3.67 ± .03 [ab]	2.67 ± 1.45 [a]	7 ± 2.08 [ab]	11.67 ± 4.6 [b]	7.33 ± 1.20 [ab]
Hibbertia huegelii	0.33 ± 0.33 [ab]	0 [b]	0.33 ± 0.33 [ab]	0 [b]	0.67 ± 0.33 [a]	0 [b]	0.67 ± 0.33 [a]	1 ± 0.58 [a]
Hibbertia hypericoides	0 [a]	1.67 ± 0.33 [b]	0.33 ± 0.33 [ab]	0.33 ± 0.33 [ab]	0.67 ± 0.33 [ab]	0.67 ± 0.33 [ab]	1 ± 0 [b]	0.33 ± 0.33 [ab]
Hibbertia racemosa	23.33 ± 0.3 [a]	7.33 ± 1.45 [c]	13.67 ± 3.7 [ac]	20.33 ± 3.2 [a]	10.67 ± 0.09 [c]	16.67 ± 3.5 [ac]	13.33 ± 3.9 [ac]	45.67 ± 1.2 [b]
Hibbertia subvaginata	2.33 ± 0.88 [a]	2 ± 1 [a]	2 ± 0.58 [a]	2.33 ± 0.67 [a]	3.33 ± 0.33 [a]	3 ± 0.58 [a]	3 ± 0.58 [a]	7.33 ± 1.45 [b]
Hovea pungens	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	1 ± 0 [b]
Hovea trisperma	0	0	0.33 ± 0.33	0	0	0	0	0.33 ± 0.33
Jacksonia densiflora	0.33 ± 0.33 [ab]	0.33 ± 0.33 [ab]	0 [a]	0.33 ± 0.33 [ab]	0 [a]	0.67 ± 0.67 [ab]	0 [a]	1 ± 0.58 [b]
Laxmannia spp.	1 ± 0.58 [ab]	0 [a]	0 [a]	0.67 ± 0.33 [ab]	0.33 ± 0.33 [ab]	0 [a]	0 [a]	1.67 ± 0.33 [b]
Lechenaultia floribunda	0.33 ± 0.33 [a]	0 [a]	0.33 ± 0.33 [a]	0.67 ± 0.67 [a]	0.67 ± 0.67 [a]	0.67 ± 0.67 [a]	1.33 ± 0.67 [ab]	2.33 ± 0.67 [b]
Leucopogon spp.	13.67 ± 4.3 [ad]	13.00 ± 4.0 [a]	18.33 ± 4.3 [ad]	28 ± 3.06 [d]	16.67 ± 5.3 [ad]	14.33 ± 6.0 [ad]	16 ± 3.46 [ad]	73.33 ± 5.8 [b]
Lomandra spp.	0	0.33 ± 0.33	0.33 ± 0.33	0	1 ± 1.00	0	0	0
Lyginia imberbis	1.33 ± 0.33 [ab]	0.67 ± 0.67 [ab]	0.33 ± 0.33 [ab]	2 ± 0.58 [a]	1.33 ± 0.33 [ab]	0.67 ± 0.33 [b]	1.33 ± 0.88 [ab]	1.67 ± 1.20 [ab]
Macarthuria australis	0.33 ± 0.33	0	0	0	0	0	0	0
Melaleuca heugelii	0 [a]	0 [a]	0 [a]	0 [a]	1 ± 1 [ab]	0 [a]	0 [a]	2 ± 0.00 [b]
Nemcia capitata	0.67 ± 0.33 [a]	0.67 ± 0.67 [a]	0.67 ± 0.33 [a]	2.67 ± 0.33 [b]	2.33 ± 0.33 [ab]	1.67 ± 0.88 [ab]	1.67 ± 0.88 [ab]	2 ± 1.00 [ab]
Patersonia occidentalis	0.67 ± 0.33	0.33 ± 0.33	0.67 ± 0.33	0	0.67 ± 0.33	0	0.33 ± 0.33	0.67 ± 0.33
Scholtzia involucrata	4.33 ± 1.33 [a]	5 ± 2.52 [a]	11 ± 3.79 [ad]	23.33 ± 5.0 [bd]	8.33 ± 2.33 [a]	11 ± 2 [a]	9.67 ± 1.67 [a]	30 ± 10.21 [b]
Stirlingia latifolia	0	0	0	0	0.33 ± 0.33	0	0	0.33 ± 0.33
Stylidium spp.	4.67 ± 1.86 [a]	12.33 ± 3.8 [ac]	9.67 ± 1.86 [ac]	34.67 ± 2.9 [b]	37 ± 9.07 [b]	17.67 ± 2.4 [c]	28.7 ± 14.9 [bc]	41 ± 5.13 [b]
Thysanotus arenarius	0	0	0	0	0	0	0.33 ± 0.33	0
Thysanotus dichotomus	0	0.33 ± 0.33	0	0	0	0.33 ± 0.33	0	1.33 ± 1.33
Thysanotus manglesianus	0 [a]	0.67 ± 0.33 [b]	0.33 ± 0.33 [ab]	0 [a]	0 [a]	0 [a]	0.33 ± 0.33 [ab]	1 ± 0.58 [b]
Unknown grass 1	0	0	0	0	0	0	0	0.33 ± 0.33
Verticordia nitens	0 [a]	0.67 ± 0.33 [ac]	3.67 ± 0.33 [b]	4 ± 0.58 [b]	1.67 ± 0.33 [c]	1 ± 0 [c]	4.67 ± 2.73 [b]	11 ± 1.15 [b]

Values are seedlings per 5 m² ± SE. Different letters indicate significant differences using Fishers PLSD ($p < 0.05$).
Species that did not respond to any of the given treatments: *Acacia huegelii, Banksia attenuata, Banksia menziesii, Bossiaea eriocarpa, Calytrix* sp. and *Opercularia vaginata*

ever, this method is not practical for broad-scale restoration because of difficulties in application of aerosol smoke. In comparison, smoke water application is more practical but is not as effective in stimulating seedling recruitment. Indeed, in our study none of the smoke water concentrations was as effective in stimulating seed germination as aerosol smoke. This may be due to higher concentrations of active agent(s) in the aerosol smoke compared with smoke water (Roche et al. 1997). The concentration effect was apparent, with higher recruitment achieved with higher concentrations of smoke water.

Interestingly, once a physical disruption of the soil seedbank occurs, as in topsoil replacement, the increase in seedling recruitment resulting from the disturbance phenomenon alleviates the full benefit of smoke application. This was demonstrated by a greater response to smoke in the woodland compared with the restored site. Smoke may behave in the same manner as physical disturbance of the soil (J. Koch, personal communication, 1997), which is well known to promote germination of native species.

Roche et al. (1997) found that species unresponsive to smoke ex situ are responsive to smoke in situ on the ba-

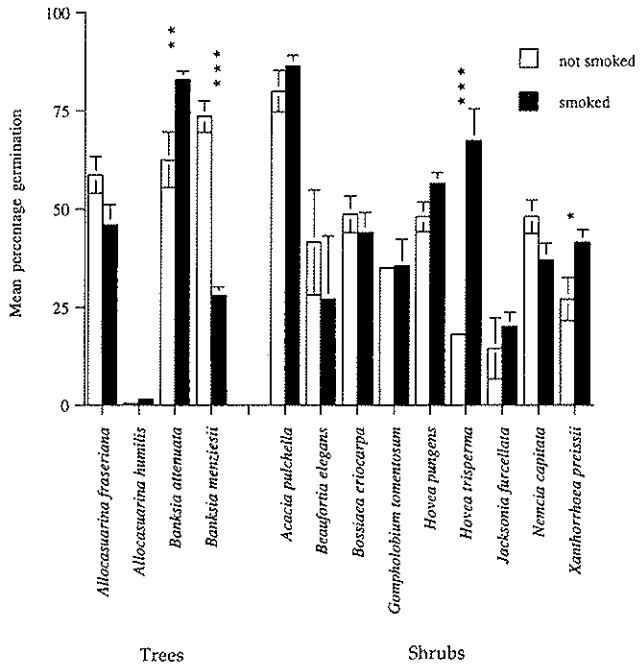
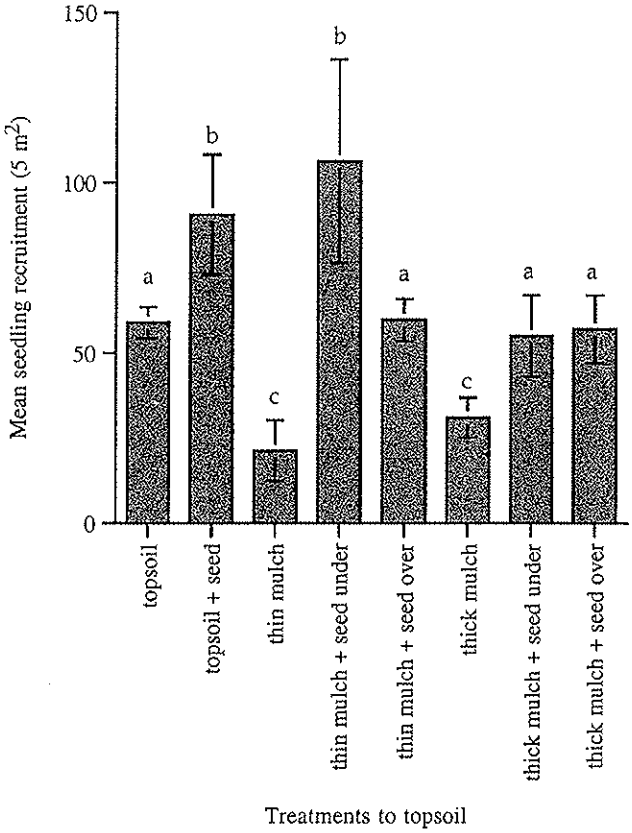


Figure 4. The effect of aerosol smoke on a range of *Banksia* woodland species used in broadcasting at Rocla ® Quarry Products. Bars represent standard error of the mean. Significant differences using *t* tests are as follows: $^{*}p < 0.05$, $^{**}p < 0.005$, $^{***}p < 0.0005$.



sis that smoke may act as a deterrent against predation, that is, untreated seed in control plots may be more palatable to predators than smoke-treated seed. However, our study found that prebroadcast smoke treatment of seeds as a deterrent against predation did not alter or benefit the total in situ seedling recruitment (topsoil and broadcast species). Indeed, application of smoke did not alter or benefit the seedling recruitment of the nine species that are normally unresponsive to smoke ex situ. Under the prevailing environmental conditions of the *Banksia* woodland, smoke may simply act as a germination cue rather than provide any antipredation activity. Indeed, smoke residues carried by the smoked broadcast seeds, once leached into the topsoil after rainfall, stimulated germination of seed from the soil seedbank (Roche et al. 1997).

For mine-site restoration, topsoil is an important and, for some *Banksia* woodland species, the sole source of seeds and germinants, for example, *Adenanthos cygnorum*,

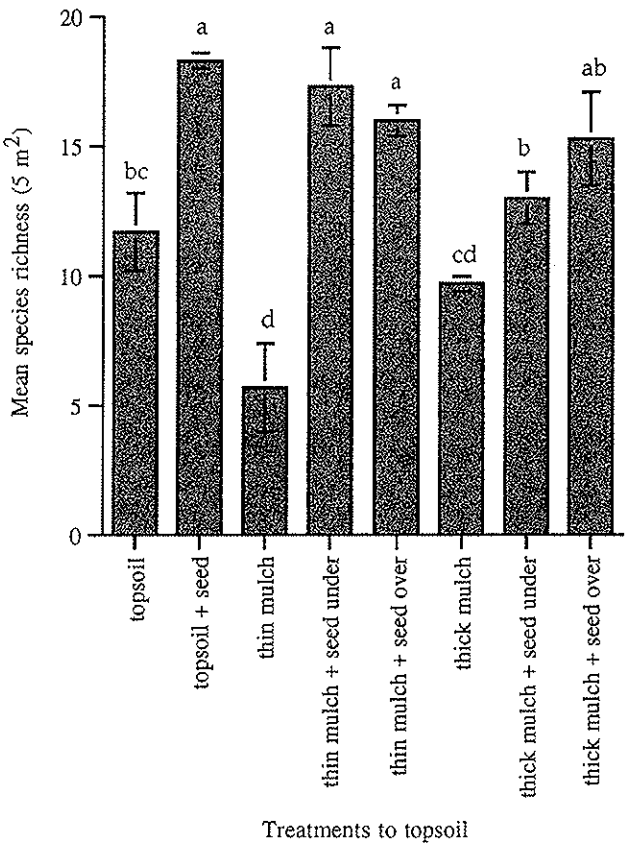


Figure 5. Mulch and seed broadcasting effects on total seedling recruitment (a) and species richness (b) in restored sites. Bars represent standard error of the mean. Different letters indicate significant differences using Fishers protected least significant difference ($p < 0.05$).

Table 3. Mulch and seed broadcasting effects on seedling recruitment of *Banksia* woodland species in restored sites.

Species Represented	*Control*	*Seed*	*Thin mulch*	*Thin mulch with seed under*	*Thin mulch with seed over*	*Thick mulch*	*Thick mulch with seed under*	*Thick mulch with seed over*
Acacia huegelii	0 [a]	0 [a]	0 [a]	0 [a]	0.3 ± 0.3 [b]	0 [a]	0 [a]	0 [a]
Acacia pulchella	0 [a]	1.3 ± 0.9 [b]	0 [a]	3.3 ± 1.2 [b]	0.3 ± 0.3 [a]	0 [a]	1.7 ± 0.3 [b]	0 [a]
Acacia sp.	0.3 ± 0.3	0	0	0	0.3 ± 0.3	0	0	0
Adenanthos cygnorum	2 ± 1.2	2.3 ± 0.9	2.0 ± 0.6	1.0 ± 0.6	2.0 ± 1.2	1.3 ± 0.7	3.7 ± 1.3	3.3 ± 1.5
Alexgeorgea nitens	0.3 ± 0.3 [b]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]
Allocasuarina fraseriana	0 [a]	1 ± 0.6 [b]	0 [a]	0.3 ± 0.3 [ab]	0.3 ± 0.3 [ab]	0 [a]	0 [a]	0.3 ± 0.3 [ab]
Anigozanthos manglesii	1.3 ± 0.9	4 ± 1.7	1.3 ± 1.3	3.3 ± 0.9	3.3 ± 0.3	3 ± 1.7	2 ± 0.6	4 ± 2.1
Astroloma macrocalyx	21.7 ± 3.2 [ac]	33 ± 7.1 [a]	6.3 ± 2.3 [b]	16.0 ± 2.0 [c]	22.0 ± 1.2 [ac]	7.0 ± 3.0 [b]	5.3 ± 1.8 [b]	8.7 ± 3.7 [b]
Banksia attenuata	0 [a]	4.3 ± 1.9 [b]	0 [a]	4.3 ± 1.5 [b]	2.3 ± 0.7 [bc]	0 [a]	3.3 ± 0.3 [b]	0.7 ± 0.7 [ac]
Banksia menziesii	0 [a]	2.0 ± 1.5 [b]	0 [a]	1.7 ± 0.3 [b]	0.3 ± 0.3 [ab]	0 [a]	1 ± 1 [ab]	2 ± 1 [b]
Beaufortia elegans	0 [a]	2.0 ± 1.5 [b]	0 [a]	0.3 ± 0.3 [ab]	1 ± 0.6 [b]	0 [a]	0 [a]	2.3 ±1.2 [b]
Boronia purdieana	0.3 ± 0.3	0.7 ± 0.7	0	0.3 ± 0.3	0.3 ± 0.3	0	0	0
Boronia ramosa	2.3 ± 0.3 [b]	0.3 ± 0.3 [a]	0 [a]	1.7 ± 0.7 [ab]	1.3 ± 0.3 [ab]	1.7 ± 0.9 [ab]	1.7 ± 0.9 [ab]	2.0 ± 1.5 [ab]
Bossiaea eriocarpa	0 [a]	3 ± 0.6 [c]	0 [a]	16.0 ± 3.0 [d]	1.0 ± 0.6 [b]	0 [a]	9.0 ± 3.1 [d]	1 ± 0 [b]
Conostylis aculeata	0.3 ± 0.3	0.3 ± 0.3	0	0	0	0	0	0.7 ± 0.7
Conostylis setigera	0 [a]	0 [a]	0 [a]	0 [a]	0.3 ± 0.3 [b]	0 [a]	0 [a]	0 [a]
Dampiera linearis	0 [a]	0 [a]	0.3 ± 0.3 [b]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]
Eremaea pauciflora	0 [a]	1 ± 0 [b]	0 [a]	0.3 ± 0.3 [a]	0.3 ± 0.3 [a]	0 [a]	0 [a]	0.7 ± 0.3 [ab]
Eucalyptus todtiana	0 [a]	1 ± 0.6 [b]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]
Gompholobium tomentosum	3.0 ± 1.7 [a]	6.0 ± 0.6 [ac]	3.7 ± 1.3 [a]	7.7 ± 0.9 [bc]	4.7 ± 0.9 [a]	3 ± 1.0 [a]	10.7 ± 1.8 [b]	9 ± 2 [b]
Gonocarpus pithyoides	2.3 ±0.9 [a]	4.0 ± 1.2 [a]	0 [b]	2.0 ± 0.6 [a]	3.0 ± 1.2 [a]	4 ± 1.0 [a]	0.7 ± 0.3 [b]	4 ± 0.6 [a]
Hibbertia hypericoides	0	0.3 ± 0.3	0	0	0.3 ± 0.3	0	0	0
Hibbertia racemosa	9.0 ± 3.2 [a]	5.0 ± 2.1 [a]	1.7 ± 1.7 [b]	5.7 ± 1.5 [a]	6.0 ± 1.2 [a]	6 ± 2.0 [a]	5.7 ± 0.7 [a]	7.3 ± 2.3 [a]
Hibbertia subvaginata	0.7 ± 0.7	0	0	0.3 ± 0.3	0.3 ± 0.3	0	0	0
Hovea trisperma	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0 [a]	0.7 0.3 [b]
Jacksonia densiflora	0	0	0.3 ± 0.3	0.7 ± 0.7	0.3 ± 0.3	0	0	0.3 ± 0.3
Laxmannia spp.	2.7 ± 0.7 [a]	2.3 ± 0.3 [a]	0 [b]	1.3 ± 0.3 [a]	1.7 ± 0.7 [a]	0.3 ± 0.3 [b]	0 [b]	1 ± 1 [ab]
Leschenaultia floribunda	0	0.3 ± 0.3	0	0	0	0	0	0
Leucopogon spp.	0.7 ± 0.7 [a]	5.3 ± 1.7 [a]	0.7 ± 0.7 [a]	31 ± 24.7 [b]	1.7 ± 0.3 [a]	1 ± 0 [a]	4.7 ± 2.7 [a]	0.3 ± 0.3 [a]
Lomandra spp.	0.3 ± 0.3	0	0	0	0.3 ± 0.3	0	0	0
Lyginia imberbis	0	0.3 ± 0.3	0	0.3 ± 0.3	0	0.3 ± 0.3	0.3 ± 0.3	0.3 ± 0.3
Melaleuca scabra	0 [a]	0 [a]	0 [a]	0.3 ± 0.3 [b]	0 [a]	0 [a]	0 [a]	0 [a]
Nemcia capitata	5.0 ± 0.6 [a]	4.7 ± 2.4 [ab]	4.7 ± 4.2 [ab]	5 ± 0.6 [a]	5 ± 2.5 [ab]	2.3 ± 0.3 [b]	2 ± 1 [b]	5 ± 1.5 [a]
Opercularia vaginata	0.3 ± 0.3	0.3 ± 0.3	0	0	0	0.3 ± 0.3	0	0.7 ± 0.3
Scholtzia involucrata	6.3 ± 0.7 [a]	5.7 ± 3.2 [a]	0.3 ± 0.3 [b]	3 ± 0.6 [ab]	0.7 ± 0.7 [b]	0.3 ± 0.3 [b]	3.3 ± 1.5 [ab]	2.7 ± 0.3 [ab]
Stylidium spp.	0	0.3 ± 0.3	0	0.3 ± 0.3	0	0.3 ± 0.3	0	0

Values are seedlings per 5 m² ± SE. Different letters indicate significant differences using Fishers PLSD ($p < 0.05$).
Data values are presented are seedlings per 5 m² ± SE. Different letters indicate significant differences using Fishers PLSD ($p < 0.05$).

Boronia ramosa, Gonocarpus pithyoides (Haloragaceae), *Hibbertia hypericoides* (yellow buttercups), *H. racemosa* (stalked Guinea flower), *H. subvaginata, Leucopogon* spp., and *Scholtzia involucrata.* Most of these species tend to have very low seed viabilities or complex dormancy mechanisms. Purchasing sufficient quantities for effective levels of recruitment through broadcasting is not cost effective. Although the use of the topsoil seedbank is important, seed broadcasting is a necessary tool in *Banksia* woodland restoration for canopy-stored species. Broadcasting increased seedling recruitment by contributing 33 germinants per 5 m², representing just 7% efficiency of the broadcast mix of diaspores delivered to the site.

A mulch of the canopy material from the native bushland did not contribute any canopy-stored species. Bellairs and Bell (1993) reported that a mulch of the canopy material provided 85% of the total germinable seed fraction in rehabilitation of kwongan vegetation in southwest Western Australia. Further studies in *Banksia* woodland may seek to investigate the removal of canopy vegetation during different times of the year, apart from May (this study), to determine if the time of removal during the year affects the germinable seed content.

It is usually assumed that mulch may also ameliorate adverse conditions for seedling recruitment by decreasing wind and water erosion (Bellairs 1992). However, the results of this study showed a negative effect of mulch on recruitment of seedlings resulting from topsoil or broadcast seed. Therefore, for achieving effective restoration of *Banksia* woodland species, mulch is not providing sufficient benefits to justify the significant costs associated with harvesting and distribution.

The cost-effective establishment of *Banksia* woodland species in a mining context depends on topsoil han-

dling and storage procedures that in themselves can represent the major source of species richness. In turn, recruitment of biodiverse *Banksia* woodlands can be enhanced by the application of smoke products and broadcast seed to restored sites that have been subjected to the best handling of replaced topsoil.

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Essays

Indicators of Biodiversity for Ecologically Sustainable Forest Management

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Abstract: *The conservation of biological diversity has become one of the important goals of managing forests in an ecologically sustainable way. Ecologists and forest resource managers need measures to judge the success or failure of management regimes designed to sustain biological diversity. The relationships between potential indicator species and total biodiversity are not well established. Carefully designed studies are required to test relationships between the presence and abundance of potential indicator species and other taxa and the maintenance of critical ecosystem processes in forests. Other indicators of biological diversity in forests, in addition or as alternatives to indicator species, include what we call structure-based indicators. These are stand-level and landscape-level (spatial) features of forests such as stand structural complexity and plant species composition, connectivity, and heterogeneity. Although the adoption of practices to sustain (or recreate) key characteristics of forest ecosystems appear intuitively sensible and broadly consistent with current knowledge, information is lacking to determine whether such stand- and landscape-level features of forests will serve as successful indices of (and help conserve) biodiversity. Given our limited knowledge of both indicator species and structure-based indicators, we advocate the following four approaches to enhance biodiversity conservation in forests: (1) establish biodiversity priority areas (e.g., reserves) managed primarily for the conservation of biological diversity; (2) within production forests, apply structure-based indicators including structural complexity, connectivity, and heterogeneity; (3) using multiple conservation strategies at multiple spatial scales, spread out risk in wood production forests; and (4) adopt an adaptive management approach to test the validity of structure-based indices of biological diversity by treating management practices as experiments. These approaches would aim to provide new knowledge to managers and improve the effectiveness of current management strategies.*

Indicadores de Biodiversidad para el Manejo Ecológicamente Sostenible de Bosques

Resumen: *La conservación de la diversidad biológica se ha convertido en una de las metas importantes del manejo de bosques de manera ecológicamente sostenible. Los ecólogos y manejadores de recursos forestales necesitan medidas para juzgar el éxito o el fracaso de los regímenes designados para mantener la diversidad biológica. La relación entre las especies indicadoras potenciales y la biodiversidad total no ha sido bien establecida. Se requiere de estudios diseñados cuidadosamente para determinar las relaciones entre la presencia y abundancia de especies indicadoras potenciales y otros taxones y el mantenimiento de procesos críticos del ecosistema en los bosques. Otros indicadores de diversidad biológica en bosques, además de las especies indicadoras o como alternativas de las mismas, incluyen lo que llamamos "indicadores basados en la estructura". Estas son características a nivel de rodal y a nivel de paisaje (espacial) del bosque tales como la complejidad estructural del rodal y la composición de especies de plantas, la conectividad y la heterogeneidad. A pesar de que la adopción de prácticas para mantener (o reproducir) las características clave de ecosistemas forestales aparentan ser intuitivamente sensibles y ampliamente consistentes con el conocimiento actual, se*

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Indicators for monitoring minesite rehabilitation: trends on waste-rock dumps, northern Australia

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Abstract

Minesite rehabilitation needs to be monitored with easily measured indicators so that trends can be plotted and assessed over time to address closure criteria. Our aim was to evaluate two traditional vegetation indicators used in woodlands and forests (tree composition and size), and two new landscape surface indicators (integrity of rip-lines and a nutrient cycling index), as well as a new habitat complexity index. We measured these five indicators on rehabilitated waste-rock dumps of differing ages on two mines in northern Australia, and compared these measures to those for nearby natural savannas. Our results confirm that tree composition and size were useful traditional indicators of vegetation development on rehabilitation because trends were towards that expected for nearby savannas. Surface roughness, as indicated by rip-lines, was also a useful indicator of the potential for a landscape to retain resources because these rip-lines persisted until vegetation was well established to assume this role. As this vegetation developed, soil surface condition, as indicated by a nutrient cycling index, also progressed towards values found on nearby natural savannas. A habitat complexity index suggested that older rehabilitation sites were developing the structural features needed by fauna. Although our findings need to be confirmed for other mines and for older sites, they do suggest that these five ecological indicators are useful for monitoring minesite rehabilitation.
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Keywords: Habitat; Landscapes; Restoration; Revegetation; Savannas; Soil condition; Vegetation

1. Introduction

Monitoring and assessing rehabilitated minesites is a global problem (Cairns, 1995). Rehabilitation must be monitored and assessed against a goal, which for many minesites in Australia is to restore waste-rock sites to stable, self-sustaining ecosystems that are similar to surrounding landscapes (Bell, 2001). For example, on Ranger mine in the Northern Territory, the goal upon completion of mining is to reform waste-rock dumps and revegetate these dumps to blend-in with nearby tropical savannas (ERA, 1997). These rehabilitated sites will have to meet criteria established by Ranger's supervising authorities and Aboriginal owners (McNally, 1987).

Traditionally, vegetation indicators such as plant species composition and growth rates are monitored on minesites (e.g. Chambers et al., 1994). These simple vegetation measures meet the criteria for indicators in that they are easy to measure and they respond in sensitive, robust and predictable ways over time to stresses and management actions (Dale and Beyeler, 2001). When rehabilitating waste-rock dumps to resemble nearby savannas, a simple vegetation indicator

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is the mix of trees species. Natural savannas are typically composed of large, fire-resistant *Eucalyptus* trees (Gillison, 1994). However, if *Acacia* trees are dominant, this indicates poor rehabilitation because most of these trees are easily killed by fires (Fawcett, 1995b), which are common in savannas. Thus, the proportional mix of *Eucalyptus*, and other fire-resistant trees, in relation to fire-sensitive *Acacia* trees can be used to indicate whether rehabilitation sites are developing into savannas. Another simple indicator of vegetation development is tree growth, which is easily assessed by measuring trunk diameters. With time, rehabilitation sites should grow trees of a similar mean size to those found in surrounding savannas.

The use of landscape ecology principles and indicators to monitor and assess ecosystem change is considered a new, innovative, and useful approach (O'Neill et al., 1992). These new ecological indicators relate spatial patterns to ecological processes operating over landscape surfaces (Bell et al., 1997; Bradshaw, 1997; Dale and Beyeler, 2001). For example, surface roughness and condition, along with the cover and pattern of vegetation patches, serve as indicators or surrogates for complex and difficult to measure landscape processes that operate to retain vital resources such as rainwater, soil sediments, and organic matter (Ludwig and Tongway, 1992; Tongway and Ludwig, 1997). Hence, landscape surface roughness and condition are simple measures that may provide useful indicators for monitoring rehabilitation trends on minesites.

Waste-rock dumps were prepared for rehabilitation by first re-shaping piles of rock into gently sloping landforms with heavy earthmoving equipment and then ripping these compacted surfaces to prepare them for planting and to create rough surfaces consisting of rip-line troughs and side-mounds, which may be separated by flat areas depending on the spacing of ripping equipment travel over the surface. These rough surfaces facilitate the process of capture of rainfall by slowing runoff and increasing infiltration. These physically induced landscape processes are critically important to rehabilitation until biological systems (e.g. vegetation patches) begin to play a major role in retaining water and nutrients on the site (Fig. 1). Thus, the degree to which rip-lines remain intact over time may be a useful indicator to monitor rehabilitation. When rip-lines erode and fill-in, they will no longer effectively capture runoff water, sediments and litter.

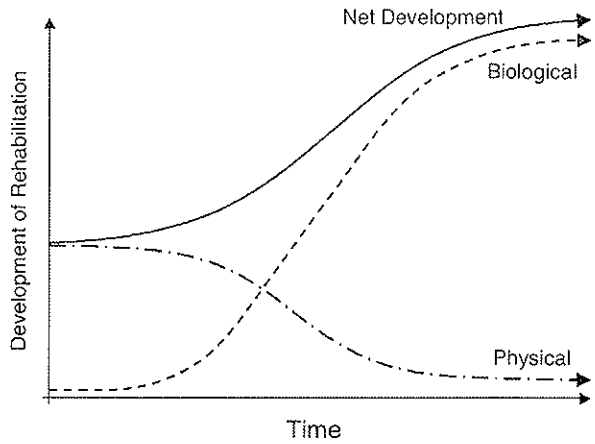

Fig. 1. As ecosystems develop with time on minesites, biological structures and processes (e.g. vegetation patches that retain water and recycle nutrients) largely take-on the role of rehabilitation from physical structures and processes (e.g. rip-lines that retain litter and water for decomposition and infiltration) (redrawn from Tongway et al., 1997).

A procedure for assessing indicators of surface condition in rangelands and minesites has been developed (Tongway and Hindley, 2000), where landscape units occurring along transects running downslope are delineated. For example, on rehabilitated minesites, rip-lines, as surface roughness features, are identified (the rip trough and its two adjoining earth banks) and assessed separately from intervening flats. Natural landscapes do not, of course, have these rip-lines, but they do have other distinctive landscape units such as vegetation patches (tree 'islands', log-mounds, shrub thickets and grass clumps). The roughness and condition of these landscape units can be easily measured, meeting this and other important criteria for indicators (Dale and Beyeler, 2001; Kurtz et al., 2001).

Another potentially useful new indicator for monitoring minesite rehabilitation is habitat complexity, that is, the mix of plant life forms (trees, shrubs, etc.), and other structural features (logs, rocks, etc.) that provide habitats suitable for animals (Catling and Burt, 1995). This habitat complexity index is a simple landscape structure metric based on the ecological principle that more habitats for animals will develop as vegetation complexity and landscape integrity increase (Andreasen et al., 2001). Recent studies in Australia have demonstrated a strong relationship between habitat complexity and the abundance of different ground-dwelling mammals (e.g. Coops and

Catling, 1997; Catling and Coops, 1999). Ideally, with time, rehabilitated minesites in northern Australia should develop the vegetation complexity attributes that are found in nearby natural savannas.

In this paper we evaluate two traditional and three new ecological indicators that appear to be useful for monitoring and assessing rehabilitation on minesites in northern Australia. Such indicators are needed to provide the monitoring data required to meet mine closure criteria (Waggitt and McQuade, 1994).

2. Study areas

The Pine Creek minesite is located 200 km south east of Darwin (Fig. 2), at latitude 13°49′ S, longitude 131°50′ E, Northern Territory, Australia. Beginning in the 1870s, gold was mined by panning creeks and shaft extractions, but in 1985 Renison Goldfields Consolidated (RGC) Ltd., operating as Pine Creek Goldfields Ltd., began open-cut mining, which created waste-rock dumps. Although this gold mine did not fully close until 1995, some rehabilitation of waste-rock dumps began in the wet-season of 1991–1992 (Fawcett, 1995a).

The Ranger minesite is located 250 km east of Darwin (Fig. 2), at latitude 12°40′ S; longitude 132°54′ E. Open-cut uranium mining operations have created two waste-rock dumps (north and south). As specified in Ranger's rehabilitation plans (ERA, 1997), revegetation has progressed over those parts of the waste-rock dumps no longer used in current mining operations.

At the Pine Creek minesite, waste-rock dumps were first re-shaped into level or gently sloping landforms (Fawcett, 1995b). The next step was to shallow rip or scarify the surface immediately prior to seeding (Fig. 3a). Scarification was accomplished by multi-tine ripping to a depth of 30 cm. This shallow depth was used to avoid digging into deeper toxic materials present on some sites. Scarification was done on the day of sowing so that seed was sown into freshly prepared surfaces. This avoids problems that can occur if an area receives rain before sowing and surface crusts form so seeds blow away. At Ranger mine, prior to revegetation with seed and nursery stock (ERA, 1997), site preparation involved deep ripping (depth > 50 cm) along the contours of waste-rock dumps (Fig. 3b).

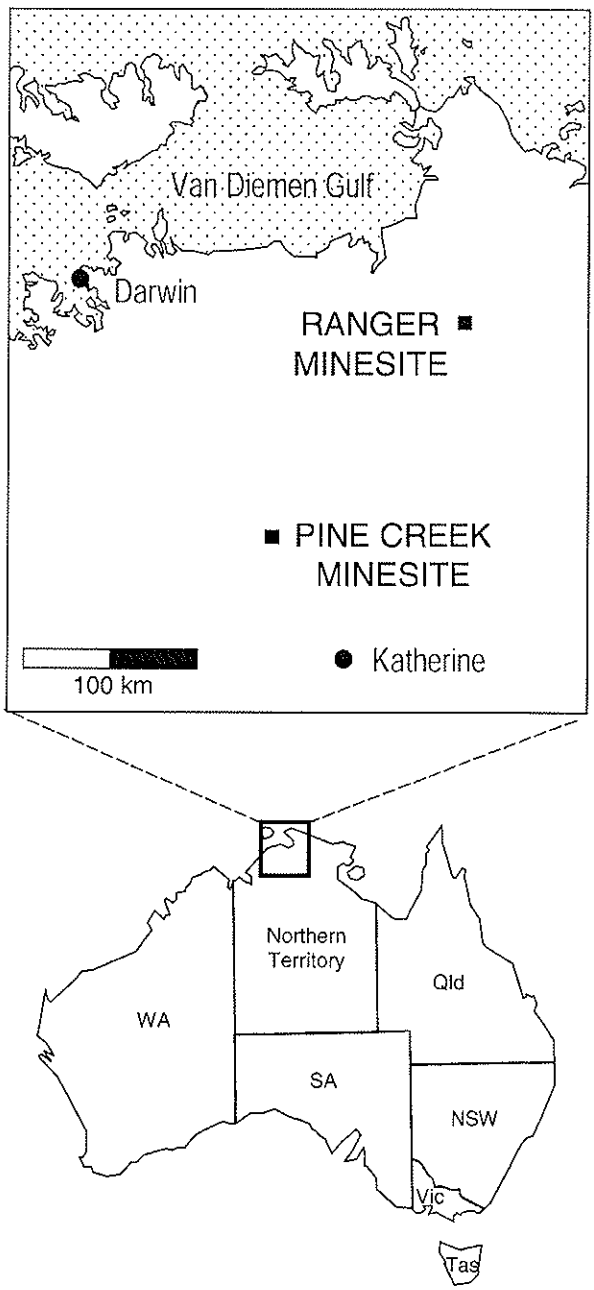

Fig. 2. Location of the Pine Creek and Ranger minesites in the Northern Territory of Australia.

3. Methods

3.1. Site selection

On the Pine Creek minesite, in collaboration with Mike Fawcett, Rehabilitation Consultant, we selected

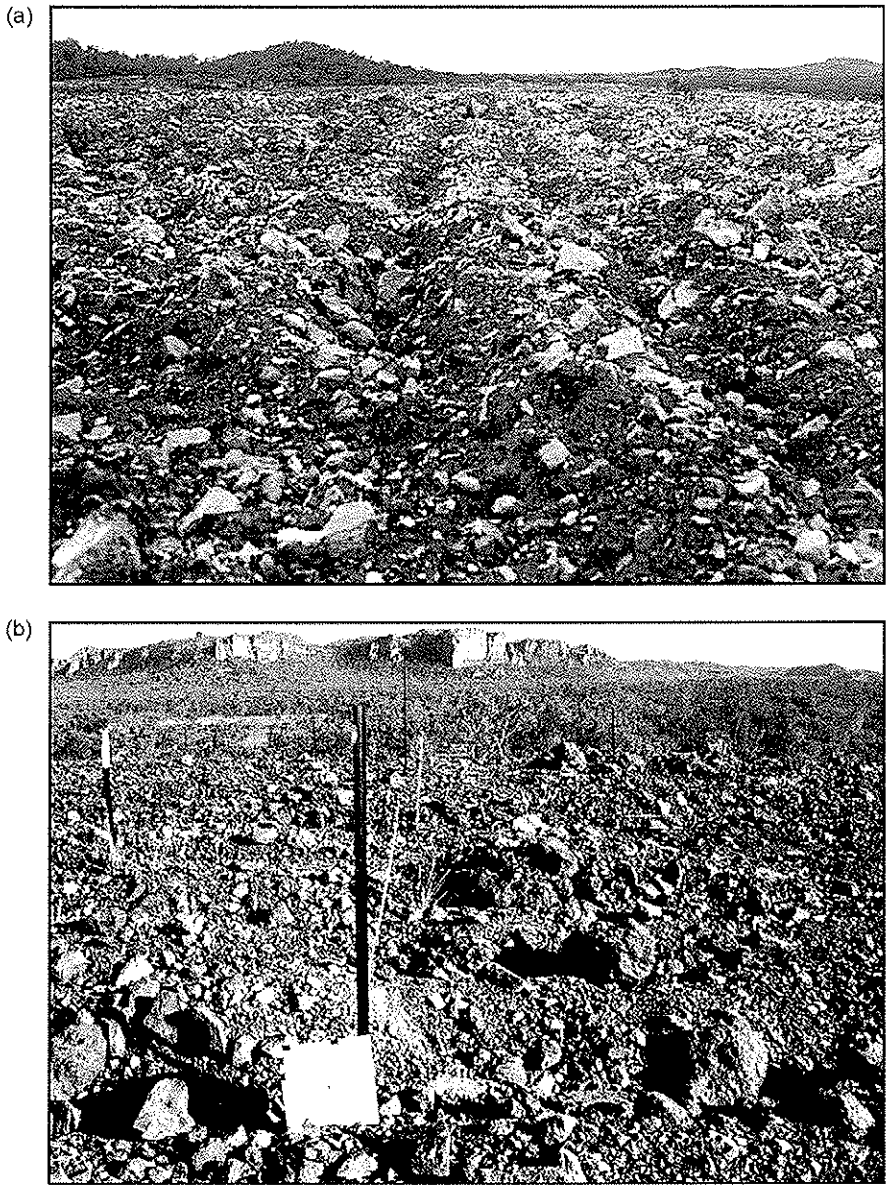

Fig. 3. Recently ripped surfaces on waste-rock dumps at: (a) Pine Creek and (b) Ranger minesites. At Ranger, savannas and the Kakadu Escarpment are in the background.

three sites that ranged in rehabilitation age from 1 to 4 years. These sites were rehabilitated in the 1992–1993, 1993–1994, and 1995–1996 wet-seasons and were measured in the 1996–1997 wet-season. Although with time, revegetation techniques have undergone some small refinements (Fawcett, 1995b), most changes are likely to be due to the age of the rehabilitation.

On the Pine Creek minesite, a natural savanna site was selected as an unmined 'reference' (Fig. 4a). This savanna site was located about 0.5 km northwest of the waste-rock dumps. The savanna vegetation type is salmon gum (*Eucalyptus alba*)—northern box (*E. tectifica*) low woodland (Wilson et al., 1990). Ironwood (*Erythrophleum chlorostachys*) and round-leaved bloodwood (*Eucalyptus latifolia*) trees also occurred

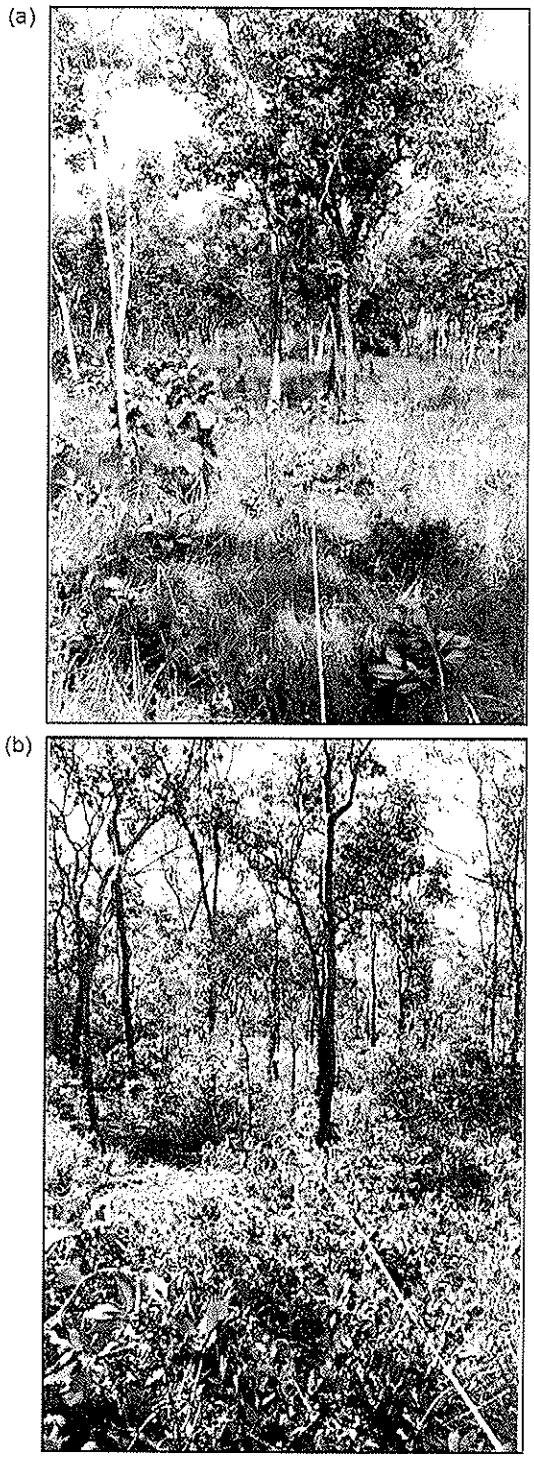

Fig. 4. Natural savanna sites at: (a) Pine Creek and (b) Ranger minesites.

on the site. Clumps of *Pandanus spiralis* occurred along a creek defining the bottom edge of this savanna site.

On the Ranger minesite, in collaboration with rehabilitation personnel, Geoff Cramb, Hazel Nisbet and Annie Lane, we selected three sites that varied in rehabilitation age from 2 to 12 years. These sites were rehabilitated in the wet-seasons of 1984–1985, 1992–1993, and 1994–1995, and were measured in the wet-season of 1996–1997. Over the rehabilitation period of 1984–1985 to 1994–1995, slightly different revegetation techniques were used (e.g. Reddell and Milnes, 1992). Thus, some changes on Ranger sites may be related to revegetation technique, but age of the rehabilitation is likely to be the dominant factor.

On Ranger, a natural savanna site was also studied (Fig. 4b). This site was located 1.3 km in from the Ranger entrance gate. The vegetation type was a stringybark (*Eucalyptus tetrodonta*)—Darwin woolly butt (*E. miniata*) savanna (Wilson et al., 1990). Ironwood (*Erythrophleum chlorostachys*), northern box (*Eucalyptus tectifica*), rusty barked bloodwood (*E. bleeseri*), and grey bloodwood (*E. porrecta*) trees also characterized the site. Clumps of palm (*Livistona humilis*) and perennial grasses (e.g. *Heteropogon triticeus*) were abundant.

3.2. Transects

On the 1- and 3-year-old rehabilitation sites on Pine Creek, we located three transects of 25–30 m in length. These transects ran directly downslope from near the top of the revegetated slopes to near the bottom within the rehabilitation. On the 4-year-old site, two 20 m transects were positioned to cross (downslope) two 10 m wide terraces, which were constructed to drain excessive runoff from adjacent waste-rock slopes. On the Pine Creek natural savanna site, a single 100 m transect was positioned to run downslope.

On Ranger 2- and 4-year-old sites, we positioned three 30 m transects to run across the rehabilitation in a downslope direction. These sites were located on the tops of flattened and ripped waste-rock dumps, and were of low slope (<0.5%). The 12-year-old rehabilitation site was located on a small terrace below a higher and younger section of the north waste-rock dump. This site was burnt in 1986 to control *Acacia* spp. that had established on the terrace. Because the

terrace was narrow, we used three 15 m transects. On the Ranger savanna site, a single 100 m transect was positioned to run downslope, which was very gentle (<0.5%).

3.3. Measurements

Tree composition and size were sampled at points spaced at 10 m intervals along the transects on each site using the point-centered quarter (PCQ) method (Bonham, 1989). The tree nearest the point in each of four quarters about the point was identified as to being in one of three tree groups: *Eucalyptus*, *Acacia*, or 'other'. The diameter at breast height (DBH; about 1.3 m on the tree trunk) of each tree species was recorded for each PCQ, and these data were used to determine mean tree DBH and the percentage composition of the three tree groups for each site.

The amount of surface covered by distinct rip-lines (trough and earth banks) was measured along each transect using the line-intercept method (Bonham, 1989). Also along each transect, and for each landscape unit, 11 soil surface condition features were assessed and used to compute three indicators: (1) soil surface stability or resistance to erosion; (2) soil water infiltration potential; and (3) nutrient cycling status, as described by Tongway and Hindley (2000) and on the web (http://www.cse.csiro.au/research/Program3/efa/index.htm). In this study we focused on the nutrient cycling index because it varied greatly across waste-rock dumps on Pine Creek and Ranger, whereas surface stability and infiltration potentials were always high. The nutrient cycling index largely depends on litter cover and depth, and on the degree of decomposition (i.e. the index is lower if litter is not decomposing, only accumulating) and type of litter (e.g. soft, easily decomposed tree leaves versus hard grass leaves). The amount of surface cover of perennial grasses and cryptogam crusts (mosses, lichens, algae) also factor into the nutrient cycling index.

The habitat complexity index (HCI) is based on five site features: (1) tree canopy cover; (2) shrub cover; (3) ground vegetation cover; (4) amounts of litter, fallen logs, and rocks; and (5) a potential moisture rating (Catling and Burt, 1995). These five features are scored on a scale of 0–3 (four-class system). For example, Coops and Catling (1997) visually scored a forest site with no logs and other debris a 0 (i.e. no habitat for ground-dwelling small mammals), whereas a site with >70% cover of these features was scored a 3. They scored a dry site as a 0 (i.e. low soil moisture), whereas a site with wet soils was a 3. On each landscape unit along each transect on our sites, all five features were scored by one of us (G.B.). These scores were then summed and averaged across landscape units and transects to give a HCI for the site (minimum = 0, maximum = 15).

4. Results

On Pine Creek and Ranger rehabilitation sites, *Eucalyptus* spp. have successfully established (Table 1). Although the composition of other native trees was, in most cases, still lower than that expected for nearby savannas, and *Acacia* spp. were present, this strong establishment and high proportion of eucalypts is the desired trend. Further, mean tree DBH increased with age on rehabilitation sites. For example, on the 12-year-old Ranger site, mean DBH was already about half that for a nearby natural savanna (3.5 cm versus 6.9 cm). The largest eucalypt tree measured on this Ranger site was 18.5 cm DBH. Assuming an annual tree growth rate of 3 mm per year for undisturbed savannas (Garry Cook, CSIRO, personal communication), this tree would be 62 years old. On Pine Creek, the largest eucalypt tree was 16.2 cm DBH (approximately 54 years old). Although eucalypt trees can grow faster on cleared unburnt sites (Mucha, 1979), cyclones, termites and frequent fires cause eucalypt trees to be short lived in northern Australia (Gillison, 1994).

As savanna trees developed on rehabilitation sites, the proportion of the landscape surface covered by rip-lines tended to decrease, as expected, with site age (Fig. 5a). Although some of these site differences are due to a slightly different rip-line spacings used on some sites, our findings clearly document that rip-lines persisted for at least 12 years (on Ranger), which is the important indicator.

On Pine Creek and Ranger, mean nutrient cycling indices tended to increase with rehabilitation site age (Fig. 5b). In fact, the 12-year-old Ranger site had a higher mean nutrient cycling index than the nearby savanna site. This high index was largely due to the accumulation of litter on the surface of this older

Table 1
The percentage composition of *Eucalyptus*, *Acacia* and other tree species, and the mean tree diameter at breast height (DBH in cm), on: (a) Pine Creek and (b) Ranger rehabilitation sites, and on natural savannas at these sites

Minesite location and time of rehabilitation	Age	Tree composition (%)		Other spp.	Tree DBH
		Eucalyptus	*Acacia*		
(a) Pine Creek					
1995–1996 wet-season	1	45	53	2[a]	0.1
1993–1994 wet-season	3	73	27	0	1.2
1992–1993 wet-season	4	56	38	6[b]	1.4
Natural savanna	54[c]	41	0	59[d]	10.6
(b) Ranger					
1994–1995 wet-season	2	23	56	21[a]	0.3
1992–1993 wet-season	4	79	17	4[a]	2.1
1984–1985 wet-season	12	78	18	4[a]	3.5
Natural savanna	62[c]	50	32	18[d]	6.9

[a] Largely *Grevillea pteridifolia*.
[b] Largely *Calytrix exstipulata*.
[c] Ages are estimated for these savanna sites based on their largest eucalypt trees (see text).
[d] For example, ironwood, *Erythrophleum chlorostachys*, green plum, *Buchanania obovata*, and cocky apple, *Planchonia careya*.

rehabilitation site, and litter factors strongly into the SSC index for nutrient cycling.

On both Pine Creek and Ranger rehabilitation sites, habitat complexity rapidly developed with site age towards that expected for natural savannas (Fig. 5c). Much of this complexity on rehabilitation sites was due to a high cover of shrubs and annual grasses and weeds. Therefore, these results must be interpreted in conjunction with plant composition data.

5. Discussion

The long-term goal for many mines in northern Australia, such as Pine Creek and Ranger, is to rehabilitate waste-rock dump sites to be similar in vegetation structure to surrounding natural savannas (Fawcett, 1995b; ERA, 1997; Bell, 2001), where these savannas are self-sustaining ecosystems that function to regulate resources. Our results confirm that tree composition and size are two useful traditional indicators of vegetation structure; their rehabilitation trends are towards that expected for natural savannas. Our findings also suggest that three new ecological indicators also have useful trends, although these indicators and trends need to be evaluated over other minesites with a greater range waste-rock materials and rehabilitation ages. Such studies are currently being conducted around Australia on bauxite, mineral sand, coal, iron ore and other hard rock minesites as part of an "Ecological Indicators Project" funded by a consortium of mining companies (see Tongway et al., 1997).

The nutrient cycling index, a landscape or soil surface condition indicator, was easy to measure on waste-rock dumps because it strongly relates to the cover and decomposition of surface litter (Tongway and Hindley, 2000). Rip-lines rapidly collect litter (Fig. 6a) and, during the wet-season, this litter quickly decomposes. This decomposition was evident from fungal, microarthropod, and earthworm activities in litter layers, and by the presence of dark humic layers next to the mineral soil surface (Fig. 6b). As vegetation develops, litter will cover most of the rehabilitation site (Fig. 6c). Natural, undisturbed savannas in northern Australia have a good consistent cover of litter from trees and grasses (Gillison, 1994), which was confirmed by high nutrient cycling indices for the savanna sites at both Pine Creek and Ranger. This litter cover protects the soil surface from rain-splash erosion (Greene et al., 1994), and provides habitat for invertebrates (Whitford et al., 1992). Thus, as simple indicators of nutrient cycling, litter cover and decomposition are useful measures.

The habitat complexity index clearly indicated that older rehabilitation sites were rapidly developing a more complex vegetation structure. This index was

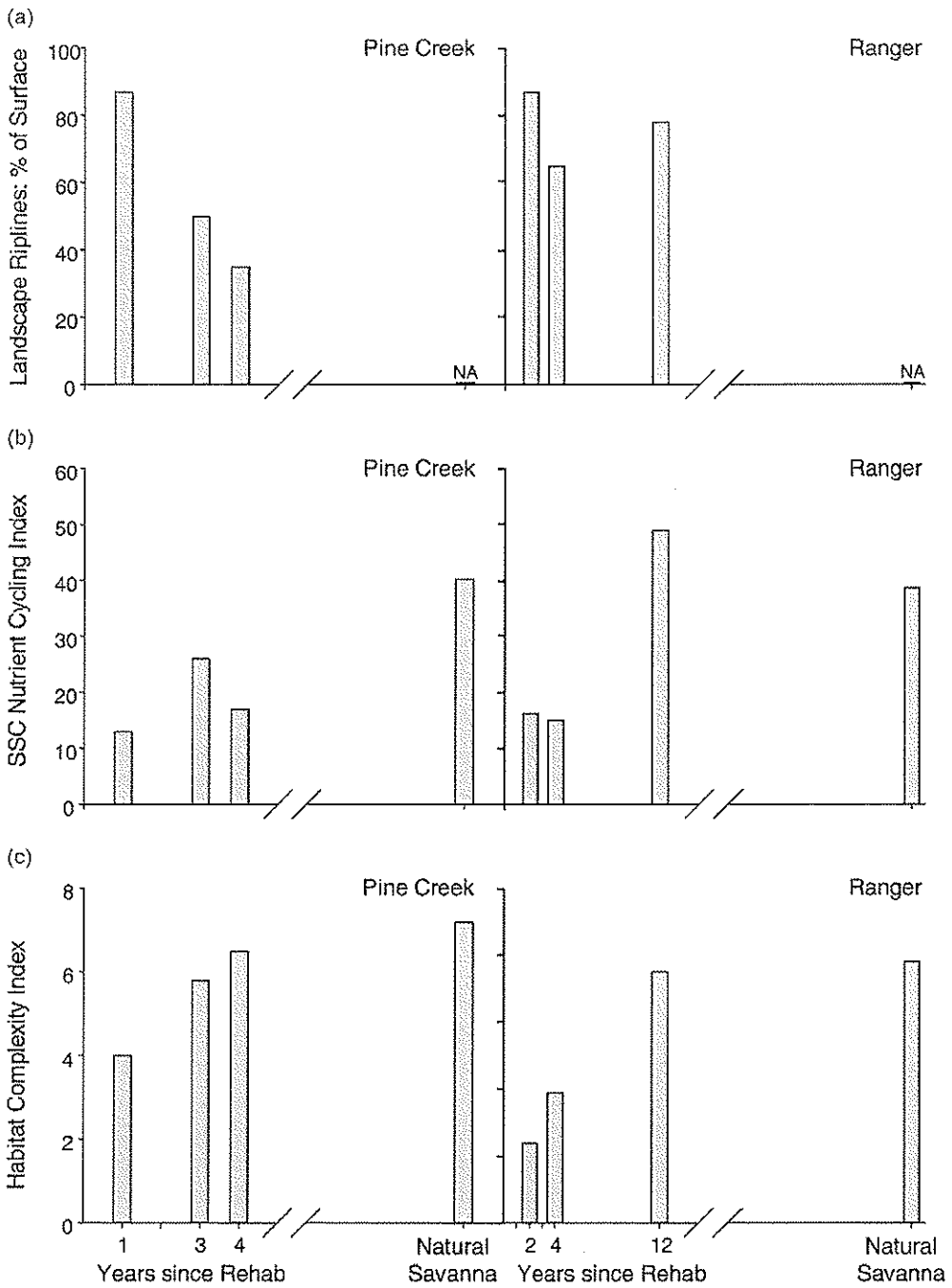

Fig. 5. Changes in: (a) the proportion of the landscape surface covered by rip-lines; (b) the nutrient cycling index; and (c) the habitat complexity index on Pine Creek and Ranger rehabilitation sites of different ages and for nearby natural savanna sites. NA, not applicable.

specifically developed to reflect the habitat requirements of ground-dwelling mammals in the eucalypt forests of southeastern Australia (Catling and Burt, 1995). Therefore, it may be useful to adjust this index to improve its application to minesites in northern savannas, and for other animal groups. For example, savanna birds are strongly correlated with the shade and shelter provided by savanna trees and grasses

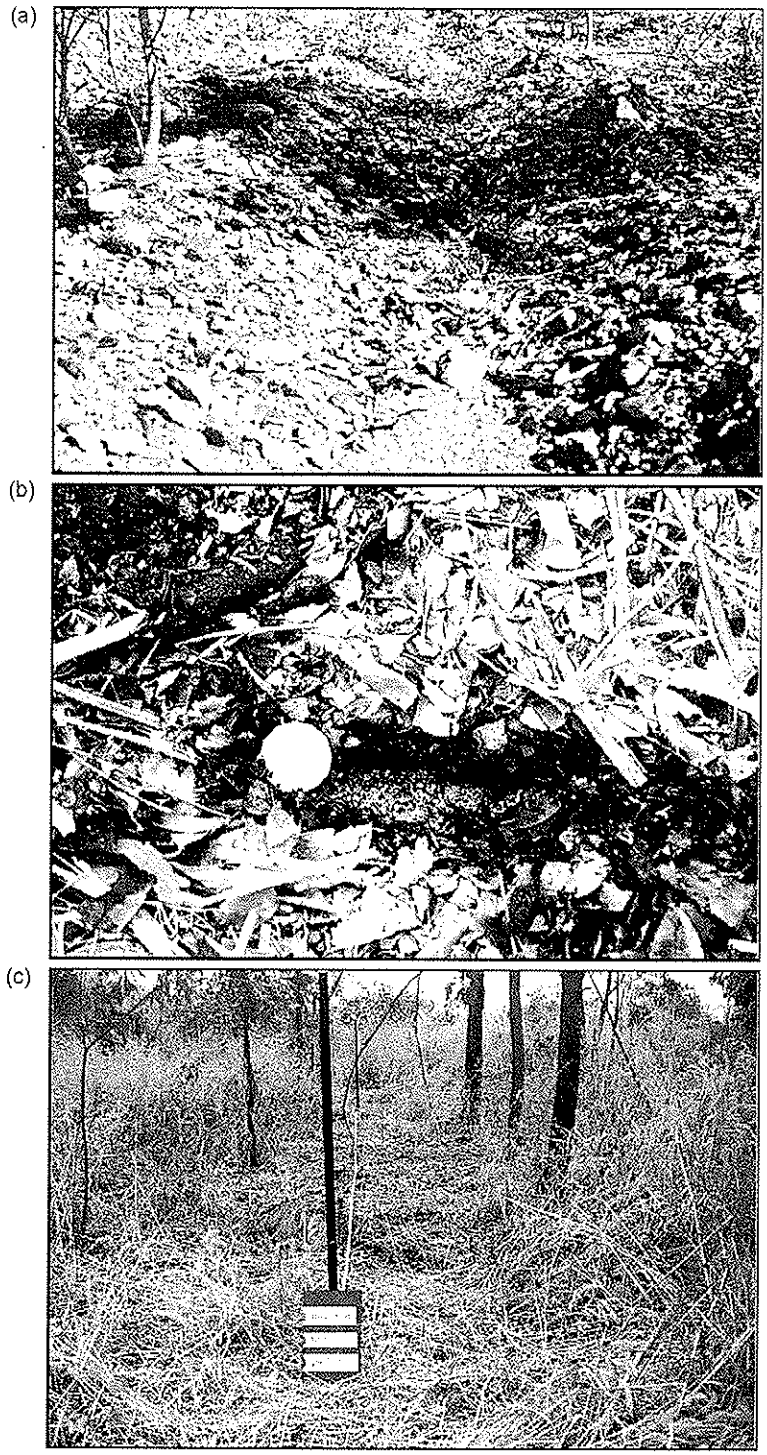

Fig. 6. On young rehabilitation sites, (a) rip-lines trap litter, where (b) it decomposes to form soil humus, and after 12 years on Ranger (c) vegetation has developed so that litter covers the site.

(Woinarski et al., 1988; Ludwig et al., 2000), and the habitat complexity index could be adjusted to more strongly reflect these vegetation features.

Of course, ecological indicators are only surrogates for a complex of difficult to measure ecosystem structures (e.g. vegetation patches, species diversity), and processes and functions (e.g. runoff, infiltration) (Dale and Beyeler, 2001). Biological indicators for some of these ecosystem attributes have been identified for minesites in northern Australia. For example, mycorrhiza can be used as indicators for plant–nutrient relations (Reddell and Milnes, 1992). In this study we evaluated two traditional vegetation indicators (tree composition and size), two new landscape surface indicators (integrity of rip-lines and a nutrient cycling index) and a new habitat complexity index. These five indicators were relatively easy to measure in the field, hence providing immediate information for assessing rehabilitation. They also had sensitive and predictable trends with time, criteria that make them useful indicators for monitoring and assessing environmental conditions (Kurtz et al., 2001), as was the case for the rehabilitated waste-rock dumps on the two mines we studied in northern Australia where this monitoring is needed to assess closure criteria (Waggitt and McQuade, 1994).

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Monitoring rangeland biodiversity: Plants as indicators

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Abstract As well as being important components of biodiversity in their own right, plants reflect the physical environment, are the primary target of many of the pressures acting on rangelands, and are relatively amenable to measurement. Hence, measurements based on plants have considerable potential to be efficient indicators of the response of rangeland biodiversity to land use. A recent report commissioned by the National Land and Water Resources Audit recommended a core set of 11 indicators, six of which relied on measurements of plants. These were trends in (i) the extent of clearing; (ii) the cover of native perennial ground-layer vegetation; (iii) the distribution and abundance of exotic plant species; (iv) the distribution and abundance of fire-sensitive species; (v) the distribution and abundance of grazing-sensitive species; and (vi) the distribution and abundance of listed threatened entities. Most indicated responses of plants to pressures acting on them. Only two (clearing and exotic plants) related to pressures. We recommend that the set be expanded to include two additional pressure indicators, one for grazing and another for fire, in recognition of their extent and potential influence on rangeland biodiversity. We also recommend that benchmark sites be included in all ground-based monitoring programmes to provide reference standards for those biotic indicators about which little is known. Assessments of the current state of knowledge about these indicators for two case-study regions, the Gascoyne–Murchison strategy area and Cape York Peninsula, have shown that it would be possible to monitor most of them directly at regional scales, but that current monitoring programmes fall short of achieving this.

Key words: benchmarks, Cape York Peninsula, clearing, fire, Gascoyne–Murchison strategy area, grazing.

INTRODUCTION

Plants as indicators of rangeland biodiversity

Biodiversity indicators are environmental or biotic attributes whose measurement signals the effects on biodiversity of pressures caused by human activities. For measurement, indicators need to be embedded in a well-developed interpretive framework and have meaning beyond the individual attributes being measured (Saunders *et al.* 1998). There are many attributes of plant assemblages, populations and species whose measurement can meet these requirements for rangeland biodiversity.

At the broadest scale, attributes of vegetation communities provide considerable information about the environment in which they occur. This is because vegetation communities reflect physical environmental variables, such as climate, weather, topography and soils. Their response to the physical environment flows on to influence biophysical variables, such as pasture biomass, biotic richness and flammability. Productivity, as monitored by greenness indices derived from remotely sensed data, is an example of one such vegetation attribute that is a powerful indicator of the state of the environment.

Plants are also useful indicators of the response of biodiversity to pressures associated with land use. This is especially so for rangelands, where plants are the primary target of many of the most potentially damaging pressures: grazing, clearing, weed invasions, inappropriate fire regimes and global change, which act most directly on the composition and structure of plant communities, and only secondarily on other elements of biodiversity (Woinarski 2001a). Indicators of response to pressure can be assessed in terms of vegetation communities (e.g. extent of clearance, trends in woody or ground-layer vegetation, wildfire scars) or individual plant species or populations (e.g. trends in threatened species or populations of grazing- or fire-sensitive plant species).

Plants are also important elements of biodiversity in their own right; hence, indicators of changes in the composition of vegetation communities or the relative abundance of their constituent plant species provide important primary information about these components of biodiversity. In addition, because plants are sensitive to many of the processes threatening rangeland biota, and are also relatively amenable to monitoring, plant indicators have considerable potential as

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Woodland Restoration in the Western Australian Wheatbelt: A Conceptual Framework Using a State and Transition Model

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Richard J. Hobbs[1]

Abstract

Woodlands dominated by *Eucalyptus* spp. in temperate southeastern and southwestern Australia have been extensively cleared for agriculture and are often badly degraded by livestock grazing. This has resulted in the loss of biodiversity and widespread land degradation. The continuing decline of these woodlands has become a concern for the conservation of biodiversity, and there is a growing interest among farmers, land managers, and researchers in developing techniques for restoring them. Currently few scientific guidelines exist for undertaking woodland restoration programs. We use a state and transition model to develop hypotheses on restoration strategies for salmon gum (*Eucalyptus salmonophloia*) woodlands. We consider that this approach provides a suitable framework for organizing knowledge and identifying areas where further information is needed, and hence provides a useful starting point for a restoration program. The model has the potential to provide a tool for land managers with which they can assess the action and effort needed to undertake woodland restoration in agricultural landscapes.

Introduction

Woodlands dominated by *Eucalyptus* spp. in temperate southeastern and southwestern Australia have been extensively cleared for agriculture (Carnahan 1990; Hobbs et al. 1993; Prober & Thiele 1993). For example, Beard & Sprenger (1984) estimated that in the Western Australian wheatbelt only 10% of the *Eucalyptus loxophleba* (York gum) woodlands and 20% of *Eucalyptus salmonophloia/Eucalyptus salubris* (salmon gum/gimlet) woodlands remain. Furthermore, a substantial proportion of the remnant woodlands in Australia's cereal-growing regions occur as isolated patches that are often degraded by livestock grazing, with no understory remaining and little or no regeneration of the tree species (Figs. 1 & 2). In the central wheatbelt in Western Australia approximately 40% of remnant *E. salmonophloia* woodlands had little or no understory, 57% had no tree species in the size class of 0–10 cm stem diameter at breast height (dbh), and 28% had no tree species in the 10–20 cm dbh size class (Norton et al. 1995). Similarly, in a large proportion of the New South Wales wheatbelt, little regeneration of the dominant tree species was found in remnant *Eucalyptus albens* (white box) woodlands, and less than 0.01% of these woodlands remained relatively unmodified (Prober & Thiele 1993). These observations suggest that in Australia's agricultural regions many of the ecosystem functions that sustain woodlands have been modified, and that degradation and loss of biological diversity will continue, unless some form of management aims to restore them.

Further loss of woodlands from the landscape seems counterintuitive in regions where over-clearing has resulted in widespread land degradation and loss of biodiversity, especially when many farmers are currently investing considerable resources in revegetation programs and landscape engineering to mitigate these problems. As a consequence there is a growing interest among farmers, land managers and researchers in developing techniques for restoring degraded remnant *Eucalyptus* woodlands.

Management of Remnant *Eucalyptus* Woodlands

Currently, there are few scientific guidelines for farmers and land managers wishing to undertake a woodland restoration program. Recommendations are generally limited to excluding livestock grazing from the woodland by erecting a fence. This has been seen as a relatively simple and easy method of restoring plant species diversity and structure to the woodland (Beckman & Davidson 1990). But the limited data available suggest that fencing alone may not be enough to restore some degraded remnant *Eucalyptus* woodlands. Where grazing pressure has been eased, different responses have been observed (Hobbs 1989; Cluff & Semple 1994; Petit et al. 1995). In some woodlands there has been no significant increase in species richness while in others there has been an increase in the abundance of a limited

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Long-Term Trends in Faunal Recolonization After Bauxite Mining in the Jarrah Forest of Southwestern Australia

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Abstract

Fauna serve a key role in many forest ecological processes. Despite this, few studies have considered long-term faunal recolonization after mining and rehabilitation of forest ecosystems. In the jarrah forest of southwestern Australia, permanent fauna monitoring sites have been established in bauxite mined areas rehabilitated in 1990 and in a range of representative unmined forest control sites. At each site mammals, birds, reptiles, and ants were surveyed in 1992, 1995, and 1998. The aims of the monitoring were to develop a better understanding of faunal recolonization trends, to produce recommendations for promoting fauna return, and to consider which techniques and fauna groups are best suited for monitoring recolonization. The results showed that successional trends varied between fauna groups. Generalist foraging mammals recolonized rapidly, whereas small predators took longer. Feral mice were initially abundant and then declined. Birds gradually recolonized, and after 8 years bird communities were very similar to those in unmined forest sites. Reptile species took longer, and after 8 years numbers of species remained lower than in unmined forests. Species richness and diversity of ants in 8-year-old rehabilitation were comparable with those of unmined forest in some rehabilitated sites but were lower in others. The composition of ant communities was still different from that of unmined sites. Ant species that only use disturbed forest declined rapidly in abundance as rehabilitation aged. The results suggest that although the rates of faunal recolonization will vary, with time most or all mammal, bird, reptile, and ant species should inhabit rehabilitated bauxite mines. The densities of many are likely to be similar to those in unmined forest, but for others it is too early to know whether this will be the case. Techniques for promoting fauna return are discussed. This study demonstrates that no single fauna group is suitable for use as an overall "indicator" of faunal recolonization; different fauna species and groups reflect different aspects of faunal succession.

Key words: bauxite mining, faunal recolonization, jarrah forest, rehabilitation.

Introduction

Most of the world's forests have declined significantly in area due to clearing for a variety of purposes. The conservation value of the remaining forest areas is now widely acknowledged, and in many countries legislation exists to minimize further clearing. Where clearing for specific land uses is unavoidable, there is a moral and often a legal obligation to rehabilitate the forest where practical.

Mining is one land use in which rehabilitation after disturbance is often possible. Numerous studies have examined techniques for reestablishing forest vegetation communities after mining in Australia (Koch & Ward 1994), England (Tewson & Broome 1998), Brazil (Parrotta & Knowles 1999), and elsewhere. However, few studies have investigated the processes of faunal recolonization after post-mining rehabilitation of forests. It is important to consider this, not only for conservation of overall biodiversity, but also because fauna serve a key role in many forest processes, including pollination, nutrient cycling, soil aeration, grazing, predation, and control of insect pests.

The jarrah forest of southwestern Australia is a tall, dry, sclerophyll forest. The main tree species are jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*). There is a relatively open midstory of species such as bull banksia (*Banksia grandis*) and *Persoonia longifolia*. Understory is usually less than 1 m high and includes a high diversity of shrub and herb species from a variety of families, including the Liliaceae, Leguminaceae, Epacridaceae, Dilleniaceae, Restionaceae, and Cyperaceae (Bell & Heddle 1989). In stream zones and nearby areas the vegetation is structurally and floristically quite different. Other tree species are present, including bullich (*Eucalyptus megacarpa*) and *Banksia littoralis*. The stream zone understory in higher rainfall areas of the forest is often very dense and up to 2 to 3 m tall.

Studies have shown that the jarrah forest supports 29 mammal, 150 bird, and 45 reptile species (Nichols & Muir 1989). In the region to the east of the jarrah forest, where habitat has been cleared for agriculture, a number of these species have declined to the point where they are officially

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Recovery of shrubland communities on abandoned farmland in southwestern Australia: soils, plants, birds and arthropods

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Passive recovery of land formerly used for agricultural production may be an inexpensive and rapid method of ecosystem recovery, and may provide an alternative method to active revegetation. Passive recovery may also contribute to sustainable agriculture (soil salinity). For undisturbed and disturbed areas of the central wheatbelt of Western Australia, this paper reports the effects of farming history (clearing only, cultivation, duration of farming, and time since farming ceased) on the soil nutrient content, plant floristics (richness and composition) and structure, and the abundance, species richness and species composition of birds and arthropods. Only one site was cultivated for >6 years.

We summarize as follows: (1) Previous clearing and cultivation has left no residual effects on the nitrogen or phosphorus content in the sandy soils. (2) There were no significant differences in terms of plant species richness but some differences in cover of woody plants, grass cover and plant species composition for farming history or time since farming ceased. (3) There were no significant differences in bird species richness but differences in species composition for time since farming ceased. (4) Arthropods showed few (and low) significant differences in their abundance, richness or species composition across different farming histories and time periods since farming ceased.

Farming of these shrublands has left only minor changes in the composition and structure of the vegetation, and in the abundance, species richness and species composition of the passerine bird and arthropod assemblages. Abandoned parcels of land on the sandy soils which support shrubland may yield useful conservation benefits with relatively little input.

INTRODUCTION

EXTENSIVE clearing of native woodlands, heathlands and shrublands in the last 100 years, has resulted in broadscale land degradation from rising water tables and salinity in southwestern Australia (McFarlane *et al.* 1993). The restoration of a hydrologic balance that reduces runoff and recharge is essential for sustainable farming over much of the Western Australian wheatbelt. This requires the use of deep-rooted crops and pastures and replanting of part of the landscape with woody vegetation, especially in recharge areas (Nulsen 1993). At least 20% of the landscape needs to be covered with deep rooted perennial vegetation to achieve a reduction in recharge rates and address the hydrological imbalance. However, there is often less than 4% in remnant native vegetation in the study area (Arnold and Weeldenberg 1991; Saunders *et al.* 1993). In addition, there is increasing recognition of the need for extensive revegetation to conserve the existing biological diversity (Hobbs 1993a). The costs of achieving this goal are high, and additional strategies need to be explored (State Salinity Council 1998). One possibility is to abandon farming on less productive areas and allow secondary succession to occur. Information on degrees of vegetative recovery of abandoned farmland for Australia is scarce, however (but see Lianzhong and Whelan 1993), and absent for recolonization of vertebrates and invertebrates. Yet, both groups are likely to have an important influence on ecosystem processes necessary for self-sustaining restoration of disturbed ecosystems, such as pollination, seed dispersal and seed predation, herbivory and decomposition (Bradshaw 1980; McDonnell and Stiles 1983; Curry and Good 1991; Norton *et al.* 1995; Majer 1997). Studies on recovery processes on other continents, with their different biota, evolutionary histories and clearing rates are unlikely to provide useful guidelines for southwestern Australia (e.g., Angelstam and Arnold 1993).

In the central wheatbelt of Western Australia, land abandonment took place in the period 1940–1975. This was a period of rapid land clearance and some land proved unsuitable for farming. Land that was too rocky, or too infertile, to use for cropping was abandoned immediately after clearing, or after a few years of cultivation. Using a space-for-time approach to examine ecosystem recovery, we compare soils, plant, birds and macroinvertebrates in abandoned farmland and in undisturbed vegetation

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Bird Utilisation of Rehabilitated Bauxite Minesites in Western Australia

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ABSTRACT

Alcoa of Australia Limited rehabilitate bauxite mined areas in the southwest of Western Australia. One of the aims of the revegetation programme is to promote the return of fauna species which inhabited areas prior to mining.

This paper discusses the extent to which bird species utilise revegetated bauxite minesites. It was shown that revegetated areas as young as 4–5 years can support similar bird species numbers, densities and diversities as unmined forest. However, the avifaunal populations occupying any rehabilitated area depend largely on the rehabilitation technique used. Factors which may be used to increase further avifaunal utilisation of minesites are discussed.

The effects of jarrah dieback and fire on forest birds are compared with results for bauxite mined areas. From this, it is possible to assess the combined impact of dieback, fire and mining on the longer term conservation of jarrah forest bird species.

INTRODUCTION

In recent years wildlife conservation has become recognised as an important land use in many of the world's forested areas. In practical management terms this has meant that many areas have been set aside exclusively for the purpose of conserving flora and fauna. Obligations have been placed on land users to maintain as much compatibility with

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Long-term trends in avifaunal recolonisation of rehabilitated bauxite mines in the jarrah forest of south-western Australia

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Abstract

Alcoa of Australia Limited rehabilitates areas in the jarrah forest of south-western Australia following bauxite mining. One of the aims of the company's rehabilitation programme is to re-establish the faunal communities which existed prior to mining. In the summers of 1981, 1987 and 1993, company biologists conducted bird surveys as part of a case study to determine the extent of avifaunal recolonisation in representative mined areas.

Analysis of monitoring data collected during the three surveys of the same areas showed considerable variation in the avifaunal communities of three rehabilitated sites. Numbers of bird species and bird diversity in a site seeded with dense understorey continued to increase, and by 1993 all avifaunal community parameters measured in this site were equal to or greater than those recorded in healthy unmined forest. Bird species numbers increased in a site given fresh topsoil but not seeded between 1981 and 1987; however, all parameters showed little further change by 1993. The oldest site, which was not seeded or given fresh topsoil, showed the greatest degree of change. In 1981, it supported only one bird species, but by 1993 all bird community parameters were similar to those recorded in the site which received fresh topsoil but no seed. Species composition of all rehabilitated sites' avifauna also changed. The similarity of bird community composition increased over time. By 1993, the healthy forest communities were more similar to those of some rehabilitated sites than to those of forest degraded by dieback disease. The study demonstrated that development of the vegetation community is a key factor controlling the composition of bird communities in rehabilitated mine sites. Recommendations are given to further promote avifaunal recolonisation following mining. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Bauxite mining; Bird recolonisation; Forest; Rehabilitation; Succession

1. Introduction

Conservation of biodiversity is now given a high priority in forest ecosystems. In many countries, commercial land uses are managed in such a way as to minimise their impact on a forest's biodiversity. Where clearing of forest is a necessary part of a land use, there is now a moral and, in many cases, a legal obligation to minimise the area cleared and maximise the rehabilitation of floral and faunal communities.

Western Australia's jarrah forest is a dry sclerophyllous forest dominated by the jarrah tree, *Euca-*

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Adequacy of rehabilitation monitoring practices in the Western Australian mining industry

By Scott A. Thompson and Graham G. Thompson

Scott Thompson *is a PhD student and* ***Graham Thompson*** *a senior lecturer in the Centre for Ecosystem Management, Edith Cowan University (Joondalup Drive, Joondalup, WA 6027, Australia. Email: ecosystems@iinet.net.au; g.thompson@ecu.edu.au. Tel: (+61) 407 385239). This project is part of PhD research that has developed a rehabilitation index and monitoring protocol to assess the success of developing ecosystems in disturbed lands and mine sites.*

Summary Currently in Western Australia (WA) there are no mandated standards for assessing rehabilitation success for the mining industry. We argue a case for focusing on the creation of near-natural, self-sustaining, functional ecosystems as the primary objective for rehabilitation programs and where this is not achievable, mines should plan to be 'environmentally neutral' by undertaking some improvements to degraded landscapes. We suggest that flora and fauna monitoring are appropriate tools for managers and regulators to obtain information on the extent to which a rehabilitated area has achieved a near natural, self-sustaining, functional ecosystem similar to that in the adjacent undisturbed area. This monitoring can also be used for completion criteria and closure plans. We report results from a short questionnaire sent to mine site environmental managers to assess the extent of flora and fauna monitoring in rehabilitation areas. Survey results highlighted the need for a more systematic and consistent approach to the monitoring of flora and fauna in rehabilitated mining areas in WA. Of 36 respondents, 23 mines monitored flora, three monitored fauna and two monitored both.

Key words *fauna, flora, mining, monitoring, rehabilitation.*

Introduction

The mining industry, government authorities and the community acknowledge the need for criteria to determine when active involvement in rehabilitation of mined lands is complete (Elliott *et al.* 1996; Tacey *et al.* 1993; Department of Minerals and Energy 1996; Muir 1996; Nichols 1997; Bellairs 1998; Osbourne & Brearley 1998; Minerals Council of Australia 2000; Chamber of Minerals and Energy 2000; Grant *et al.* 2001). A decade ago, the focus of most mine site rehabilitation programs was to establish good density and cover of vegetation across the disturbed area. While this resulted in rehabilitated sites that were essentially stable and may have looked aesthetically pleasing, it did not necessarily mean that the rehabilitated sites were moving towards the establishment of functional ecosystems (Bellairs 1998). Because of this focus on vegetation, a narrow set of vegetation indices were established to measure early developmental stages of revegetation (Department of Minerals and Energy 1996).

Currently the Department of Industry and Resources (DoIR; formerly Department of Minerals and Energy – DME) in WA requires rehabilitated areas to be safe, stable and non-polluting landscapes (Anderson 2001), but says little about the re-establishment of sustainable, functional ecosystems. For example, the DME (WA) Department of Minerals and Energy (1996 p. 37) states that '... the re-establishment of self-perpetuating vegetation, which is integrated with the surrounding ecosystem, is the most common land use objective ...' for mine site rehabilitated areas. Such criteria could be met by a vegetation community of non-local species, which probably would not provide the range of niches necessary for the fauna in the adjacent undisturbed area to colonize the rehabilitated area and eventually create a self-sustaining, functional ecosystem. The use of non-local species also creates potentially serious weed problems. Specific monitoring protocols and rehabilitation standards are currently ill defined (Australian and New Zealand Minerals and Energy Council and Minerals Council of Australia 2000) and mining companies in WA are being released from their environmental obligations before they can demonstrate the creation of, or significant progress towards, a near natural, self-sustaining, functional ecosystem (Anderson *et al.* 2002). A number of researchers have argued that there is a compelling case for industry and regulators to adopt an approach that focuses on creating functional ecosystems rather than just revegetation (Andersen 1994; Bisevac & Majer 1999). Bisevac and Majer (1999) suggested that the goal for rehabilitated mined land should be to restore the structure, diversity, function and dynamics of the undisturbed ecosystem. In many circumstances this will mean the creation of a self-sustaining, functional ecosystem similar to that in the adjacent undisturbed area from which the rehabilitated area will recruit most of its fauna. The need for 'high-quality' rehabilitation has become necessary with the phasing in of performance standards for assessing the development of rehabilitated areas (Bisevac & Majer 1999). Mine site rehabilitation should be viewed as managing succession processes towards the creation of ecosystems that are functionally compatible with that which existed before the disturbance or in the adjacent undisturbed areas.

Thirty of the respondents (83%) to a questionnaire sent to all mine site environmental managers in WA indicated that some rehabilitation had been completed on their mine since operations began, with 8 years being the average age of the oldest rehabilitated area. There was considerable variability in the meth

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REHABILITATION INDEX BASED ON REPTILE COMMUNITY STRUCTURE FOR MINE SITE MONITORING

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ABSTRACT

Biotic integrity is possessed by an ecosystem in which its composition, structure, and function appear not to have been adversely affected by human activities; perhaps a final objective for a rehabilitated mine site. Biotic integrity or functionality of an ecosystem is difficult to measure and efforts to restore the integrity of, or develop a functional ecosystem in a disturbed site have been dominated by non-biological measures such as chemical and physical parameters. The approach I have adopted is based on the presumption that the biological community will follow if the chemical and physical parameters in a rehabilitated area are appropriate.

A preliminary rehabilitation index based on the assemblage of small reptiles is presented as a method for measuring the progress towards rehabilitation success. The rehabilitation index uses a series of parameters of the reptile assemblage to provide an estimate of the biotic integrity or functionality of an ecosystem. Parameters used include; species richness, abundance, trophic composition, habitat preference and predatory strategy. Reptile community data comes from undisturbed and rehabilitated waste dumps in the area approximately 50 km north of Kalgoorlie.

INTRODUCTION

Currently in Western Australia (WA), the Department of Mineral and Petroleum Resources (DMP - formerly Department of Minerals and Energy) and the Department of the Environment (DoE) advise mining companies to use the guidelines outlined in the *Best Practice Environment Management in Mining* series when developing rehabilitation programs. The onus is on the mining company to describe what its rehabilitation program will be and when enough is enough in rehabilitating the mine site; completion criteria are ill defined. Mining companies negotiate with staff from the DMP to determine when they have done sufficient rehabilitation at a particular disturbed site for them to be released from their environmental obligations, but the absence of guidelines means that the decision to return environmental bonds for rehabilitated lands is subjective.

Restoring an ecosystem to its original condition is ideal, but often not achievable (Recher, 1993). Restoring ecosystem structure and function to approximate pre-undisturbed conditions, without perfect replication, is a more achievable goal. Rehabilitation programs put in place appropriate soils and vegetation, hoping that natural processes and the invasion of invertebrates and vertebrates from adjacent areas will progress toward the establishment of a functional ecosystem (Majer, 1989).

Succession

Natural development of the ecosystem, or succession, involves moving from a simple, low complexity biotic state, towards a more complex state (Majer, 1989). Mine site rehabilitation can be viewed as managing a succession from the surface produced by the mining activity to a self-sustaining ecosystem.

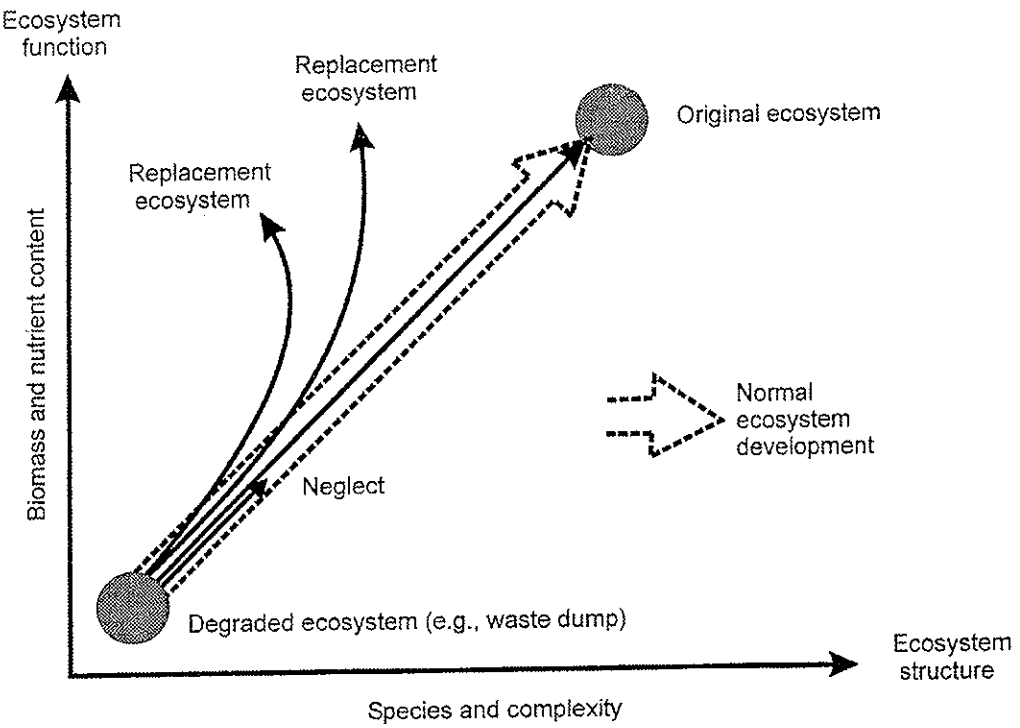
In primary succession, vegetation and associated animals colonise a bare inorganic surface, and once this 'pioneer community' is established, there are a series of replacement communities ('seral stages') that gradually modify the microenvironment until the climax community is established. The seral stages, and community composition are often predictable. Primary succession occurs after a major disturbance event, such as glaciation, or volcanic eruption or, more recently, mining activity (Werner & Wigston, 1989). In secondary succession, a disturbance such as fire, or a cyclone, modifies an existing climax community such that a residual community is left, which will then undergo a series of changes until the climax community is re-established.

Large mining operations are likely to initiate both primary and secondary successions. As far as biotic community development is concerned, it is of little consequence if the disturbance event initiating change is natural or anthropogenic. Thus, if we monitor and analyse the results of disturbance events in natural ecosystems, the information may be applied to the rehabilitation of an operation such as a mine-site (Werner & Wigston, 1989). Fauna respond to changes and succession in vegetation communities rather than just a change in time (Fox, 1997). Figure 1, developed from Bradshaw (1984), shows the process of rehabilitation from a completely degraded ecosystem towards the original ecosystem. The natural ecosystem development involves the movement from a simple state in the bottom left of the graph towards a more complex state in the top right (Bradshaw, 1984).

Closure and completion criteria

It is neither practical nor desirable to rigorously define a single comprehensive set of rehabilitation standards that will apply across all types of terrain and all manner of operations. It is necessary to look at the specific requirements of each individual operation and determine rehabilitation standards on a site-by-site basis (Hollands, 1993). Because mining operations tend to develop in unpredictable ways, planning for rehabilitation needs to be adaptable (Department of Mines and Energy, 1997).

Figure 1: Process of succession from a degraded ecosystem to the original ecosystem (Bradshaw, 1984)



Once the decommissioning phase of a mine has started, the difficulty faced by the owner or operator is when to finish the process. Without a clear understanding of the endpoints, which can determine effective closure, there will be considerable uncertainty.

A commonly accepted goal of rehabilitation of mined land is to restore the structure, diversity, function and dynamics equivalent to that in an undisturbed ecosystem (Bisevac & Majer, 1999). The need to carry out high-quality rehabilitation has become more acute in the industry with the phasing in of performance standards for assessing ongoing rehabilitation, and success indicators or completion criteria for assessing the end product (Bisevac & Majer, 1999).

Fauna as bio-indicators for rehabilitation progress and success

The goal of land rehabilitation programs has shifted from just revegetating an area, towards developing a functional ecosystem (Andersen, 1994). The aim of a bio-indicator based study is to use the living components of the environment under study, as the key to assess the transformations and their effects. In the case of landscape reclamation, they are used to monitor the remediation process in different parts of the landscape over time (Paoletti, 1999). This shift towards developing functional ecosystems in rehabilitated areas has lead to investigations that use invertebrates as indicators of ecosystem restoration programs (Andersen, 1993; Majer, 1989; Majer & Nichols, 1998). The use of vertebrates as bio-indicators is relatively new.

Indicator taxa are species or higher taxonomic groups whose parameters, such as diversity, presence or absence, or infant survivorship, are used as proxy measures of ecosystem conditions (Hilty & Merenlender, 2000). A *biological indicator* might be defined as "a species or group of species that readily reflects: the abiotic or biotic state of an environment; represents the impact of environmental change on a habitat, community or ecosystem; or is indicative of the diversity of a subset of taxa, or of wholesale diversity, within an area" (McGeoch, 1998, p. 185). Chase, *et al.*, (2000, p. 475) defined a *biodiversity indicator species* as "those whose presence is correlated with high species richness or with the presence of a threatened biological community or sub-association".

Since lower trophic levels must have appropriate representation before higher trophic levels can colonise rehabilitated areas, it follows that the presence of higher trophic levels indicates that lower trophic levels are functional. However, it cannot necessarily be assumed that because lower trophic levels appear functional that higher

trophic levels will automatically evolve. If secondary and tertiary consumers are not present, it can be assumed that one or more of their requirements are lacking (Knight, 1998). Secondary and tertiary consumers may therefore be seen as an appropriate group to focus on, in the search for indicators of ecosystem functionality and rehabilitation success (Knight, 1998). Many small vertebrates, such as reptiles and small mammals are secondary or tertiary consumers and are also relatively easy to sample and identify (Knight, 1998).

Vertebrates can be herbivorous, omnivorous or carnivorous (including invertivorous) and are unlikely to recolonise rehabilitated areas unless the vegetation structure (food and shelter) and invertebrates (food) necessary for their survival are present. For example, the absence of decomposing logs will mean there will be few or no termites, so lizards that only eat termites are unlikely to colonise an area that has few decomposing logs. If the composition of lizard communities in undisturbed, ecologically functioning biotopes is understood, then this knowledge can be used to develop an index of rehabilitation progress/success, to indicate progress towards the development of a functional ecosystem on either the biotope or landscape scales.

Small lizards can be excellent indicators of rehabilitation progress because they are relatively easy to sample and identify, display a good community structure based on dietary requirements (e.g., herbivorous, omnivorous, invertivorous, carnivorous), foraging strategy (e.g., sit-and-wait versus widely-foraging predators), activity period (e.g., nocturnal and diurnal), body size (small to large) and variety of habitat requirements (e.g., arboreal, terrestrial and fossorial). Most small lizards have a defined activity area, only move in response to competitive pressures, and have a life span long enough to recolonise rehabilitated areas and establish new communities. Most mining areas in arid and semi-arid regions in Western Australia have an abundance of lizards; normally about 40 species (Pianka, 1989). For example, a research site near Bungalbin Hill studied for the past 12 years has 47 reptile species (P. Withers, pers comm.), and a study site in the Great Victoria Desert studied for the past 27 years has 47 species of lizards (E. Pianka, pers. comm.).

The primary purpose of this study is to develop a rehabilitation index based on knowledge of small lizards' community structure that will enable mine site environmental officers and government regulators to monitor progress toward the development of a functional ecosystem in rehabilitated mine sites. This index could then be used as one of the primary tools in assessing when a mine can be released from their rehabilitation obligations.

DATA SOURCE

Pit-traps were placed in undisturbed biotopes to determine lizard species richness and abundance. Similar sampling was conducted on five rehabilitated waste dumps of various ages. This enabled comparisons to be made between lizard species richness and abundance in the undisturbed biotopes, the transition area (area between the undisturbed area and top of the rehabilitated waste dumps) and rehabilitated areas on top of the waste dumps at various stages of development.

A total of 888 pit-traps were used for sampling (≈ 50 000 pit-trap nights to date). Undisturbed areas have 48 pit-traps in 8 arrays of 6 pit-traps, and the transitions or slope of the waste dump and the top of the waste dumps have 36 pit-traps each; 6 arrays of 6 traps. All the sites were surveyed 5 times a year in January, April, June, September and December for 2 years.

REHABILITATION INDEX

Since Karr (1981) first wrote about his *Index of Biotic Integrity* (IBI) many research projects have used his methods of assessing ecosystem health, particularly in fish communities (Karr, *et al.*, 1986). Biotic integrity is defined as the ability to support and maintain "a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Karr & Dudley, 1981). In general, biotic integrity is possessed by ecosystems in which composition, structure, and function have not been adversely affected by human activities. In the context of Bradshaw's (1984) diagram (Figure 1) it is represented by the filled in circle in the top right hand corner.

The *Rehabilitation Index* that I am developing is based largely upon the IBI that was designed by Karr *et al.*, (1986) to measure the health of riverine ecosystems, most of which were being degraded by pollution and land management practises. My rehabilitation index measures the reverse; i.e., progress toward the establishment of a self-sustaining functional ecosystem in a rehabilitated area. Karr *et al.*, (1986, p.1) explains that the "strength of the IBI is its ability to integrate information from individual, population, community, zoogeographic, and ecosystem levels into a single ecologically based index". If these aspects can be captured, then this makes any rehabilitation index a powerful tool for monitoring rehabilitation success.

Karr *et al.*, (1986) used 12 parameters in the IBI, which fall into three main categories: species composition, trophic composition and fish abundance and condition. Data were obtained for each of these parameters at a given site and evaluated in light of what might be expected at an unimpacted or relatively unimpacted location in a similar geographical region (Karr *et al.*, 1986). A numerical rating was then allocated based on whether its evaluation deviates

"strongly from", "somewhat from", or "approximates expectations". The sum of the 12 ratings then determines the overall site score (Karr *et al.*, 1986).

My index is in the initial stages of development and is likely to be extensively modified as it is more thoroughly examined and analysed. What follows is a preliminary outline of the general structure of the index and how it might operate.

Thus far I have used five parameters to develop the Rehabilitation Index; species richness, abundance, trophic composition, preferred habitat and predatory composition. Use of this assessment system assumes that my monitoring strategy samples the entire reptile community in a biotope. I have extensively sampled eleven undisturbed habitats, five of which are adjacent to the five rehabilitated waste dumps I have also sampled.

The values obtained for each parameter in a rehabilitated area are compared to the value expected at a site located in a similar geographical area where human influences have been minimal in this case the adjacent undisturbed area. Specifically, I have compared the pit-trappable reptile assemblage structure in the rehabilitated area with the adjacent undisturbed area and other undisturbed areas in the vicinity. A rating system has been assigned to each parameter and the scores used to judge the extent to which the rehabilitated area approximates that of the adjacent undisturbed habitat. The sum of these scores from each of the five parameters is then used to provide an overall index score for the waste dump (see Table 1).

Formulation of most parameters still to be fully worked out and the numbers used here are to show how the index is calculated.

The data presented for 'waste dump' calculations is from the sampling on the top of waste dumps. Data are also available for the waste dump slopes and in future these data will be combined with that for the top, to provide an overall index for the waste dump. I will use two waste dumps (Gimlet South and Rose) to demonstrate how the index will work.

Gimlet South waste dump was rehabilitated in 1993 with some extra earthworks being conducted during 1997. The waste dump is primarily constructed of blue rock with a thick layer of clay topsoil. Rose waste dump was rehabilitated in the 1994 and has a good coverage of soil and vegetation. Both waste dumps are about 50 km north-west of Kalgoorlie (Gimlet South UTM 313518, 6635628 and Rose UTM 333205, 6619667).

Parameter 1 – Species richness

This parameter has two subsets: the total number of species for the area and the number of species within the skinks, geckoes, dragons, goannas, legless lizards, blind and elapid snakes groups expressed as a proportion of the total number of species present. The expectation is that species richness will be higher in undisturbed areas than rehabilitated waste dumps, and as the rehabilitated area progresses towards the self-sustaining functional ecosystem the number of species will increase. The proportion of species in each group in the rehabilitated area will progressively shift as the site progresses towards mimicking the undisturbed functional ecosystem.

The total number of species is scored out of a possible score of 50 (Table 2). The undisturbed area always gets the highest score and the waste dump score is calculated based on the proportion of species found, compared to the number of species in the undisturbed site (Table 2). This way, the waste

Table 1: Index scores, integrity classes and the attributes of those classes.

Total index score (sum of ratings)	Attributes
141-180	Comparable to the best situation without human impact; regionally expected species for habitat type; species present with a full array of age (size) classes; balanced trophic structure; self-sustaining functional ecosystem.
121-140	Species richness approaching expected levels; not all late succession species present, some species present with less optimal abundances or size distribution; trophic structure incomplete.
85-120	Species richness below expectation, some groups not well represented, some specialists not present.
51-85	Lack of specialists; fewer species, skewed trophic structure and relative abundances.
31-50	Few vertebrates present; only early colonisers present, lack of community structure.
0-30	Early succession species only. No community structure. Only opportunistic early colonisers are present.

dump score is always relative to the undisturbed area. The extent to which the proportion of species in each of the groups (skinks, geckoes, dragons, goannas, legless lizards, blind snakes and elapids) matches that in the undisturbed area is scored out of 40. The index scores are assigned based on how well the proportion of the various reptile groups in the rehabilitated area matches that in the undisturbed site. If the proportion of reptiles in each of the groups mimics that of the undisturbed site the score is a 5; if the rehabilitated site deviates strongly from the expected proportion in the undisturbed area then it can score as low as 1. An overall score for species richness is obtained by summing the two scores; species richness and proportional representation of species. A similar method is used to score each of the five parameters.

Parameter 2 – Abundance and diversity

There are three subsets to the abundance and diversity parameter; absolute abundance or density, relative abundance or evenness and diversity. The density score is determined by the number of reptiles caught per 100 pit trap days. Density is an important measure of the capacity for the ecosystem to support individuals. It would normally increase as the environment approaches a climax community. Both Rose and Gimlet South waste dumps have fewer specimens than at the undisturbed site (Table 3).

Relative abundance can be measured in two ways - evenness (Tramer, 1969) or a measure of the slope of the graphical presentation of the relative abundance data (see figure 2 and 3). The pattern of relative abundance and evenness for Rose waste dump is similar to the undisturbed site (Table 3). This demonstrates that this waste dump is starting to develop a community structure similar to the undisturbed area. Gimlet South waste dump has one dominant species rather than an even spread of species; this is not representative of the undisturbed area (Figure 2 and 3). This dominant species probably occupies niches of other species in the undisturbed community structure. There are also about half the number of species found on the waste dump compared to the undisturbed area. Rose therefore scores more highly in the relative abundance parameter than Gimlet South. Simpson's (1949) diversity index scores are similar for both the undisturbed areas (≅0.90). Rose waste dump has a more comparable Simpson's diversity index than Gimlet South respective to their undisturbed areas; resulting in a higher score. On preliminary analysis, for abundance and diversity, Rose waste dump scored a 24, and Gimlet South waste dump would score 16 compared a possible 40 in the undisturbed area (Table 3).

Parameter 3 –Trophic composition

The trophic composition parameter examines how the reptile assemblage is segregated into dietary niches. Species are assigned to a trophic group based on their adult feeding patterns. The subsets in this parameter include the proportion of dietary specialists, and proportion of omnivores, carnivores (vertebrate only), and invertivores within the community. An omnivorous species is defined as one that consumes significant quantities of both plant and animal material. A 'significant quantity' is defined as greater than 25% animal and 25% plant material. An invertivore is defined as a species whose diet consists of at least 85% invertebrates and a carnivore is a species whose diet consists of at least 85% vertebrate prey. A species is defined as being a 'dietary specialist' if greater than 85% of its diet comes solely from one food source (i.e., ants or termites). Data for diets were obtained from the published literature and from discussions with field biologists/ecologists.

Table 4 shows that both of the undisturbed areas have similar trophic compositions with about 84-90% of the reptiles being invertivores, 6-8% are omnivores and the remainder vertebrate carnivores. In both of the waste dumps there are no carnivores and more omnivores, suggesting that carnivores might be later colonising species. The proportional representation of species in each of the trophic groups compared with that in the undisturbed area is scored on a scale of 1 to 15 (Table 4).

There are no dietary specialists for Rose waste dump compared to 20% in the undisturbed site (Table 4). Dietary specialists are scored on a scale of 1 to 10 (Table 4). The summary of scores for Rose and Gimlet South waste dumps are out of a possible maximum of 25.

Parameter 4 – Preferred habitat

The preferred habitat parameter is a measure of the proportion of arboreal, terrestrial and fossorial species in the rehabilitated area compared with the undisturbed area. Data on the ecology and biology of the species was obtained from published literature, discussions with biologist/ecologists and from personal experience in the field. Arboreal species are those that primarily forage and inhabit trees or under bark. Fossorial species are those that primarily forage and inhabit loose soil, leaves and organic matter and terrestrial species are those that primarily forage and live in open areas, generally digging their own holes, living in other species holes or sheltering under bushes and shrubs. This parameter is scored on a scale of 1-15. Rehabilitated areas with reptile assemblages that closely mimic that in the undisturbed area get a higher score.

The proportion of species living in preferred habitats is very similar for Gimlet South and Rose undisturbed sites with terrestrial species dominating the habitat composition (Table 5). Gimlet South waste dump proportions are very similar to the undisturbed sites, whereas, Rose has a higher proportion of terrestrial species and no fossorial species in the rehabilitated sites (Table 5). On a preliminary analysis, Rose waste dump scored an 8, and Gimlet South waste dump scored 12 compared with a maximum of 15 in the undisturbed area (Table 5).

Table 2: **Summary of parameter scores for species richness at Gimlet South and Rose undisturbed areas and waste dumps**

	Sites	Gimlet South				Rose			
	Maximum score	Undisturbed		Waste dump		Undisturbed		Waste dump	
		(#) / %	Score	(#) / %	Score	(#) / %	Score	(#) / %	Score
Total number of species (#)	50	30		11	18	25	50	11	22
% skinks		30		18		28		9	
% geckos		20		55		36		36	
% dragons		13		18		12		36	
% goannas		7		0		4		9	
% legless lizards		7		0		0		0	
% blind snakes		7		9		8		0	
% elapids		17		0		12		9	
Proportional number of species	35				15				20
Parameter 1 score	85		**85**		**33**		**85**		**42**

Table 3: **Summary of parameter scores for abundance and diversity at Gimlet South and Rose undisturbed areas and waste dumps**

	Sites	Gimlet South				Rose			
	Maximum score	Undisturbed		Waste dump		Undisturbed		Waste dump	
			Score		Score		Score		Score
Number of captures/100 trap nights (density of captures)	20	8.59	20	2.78	6	9.20	20	3.43	8
Simpson's diversity	10	0.91		0.68	4	0.90		0.82	7
Evenness (J)	10	0.82		0.68	6	0.81		0.80	9
Parameter 2 score	40		**40**		**16**		**40**		**24**

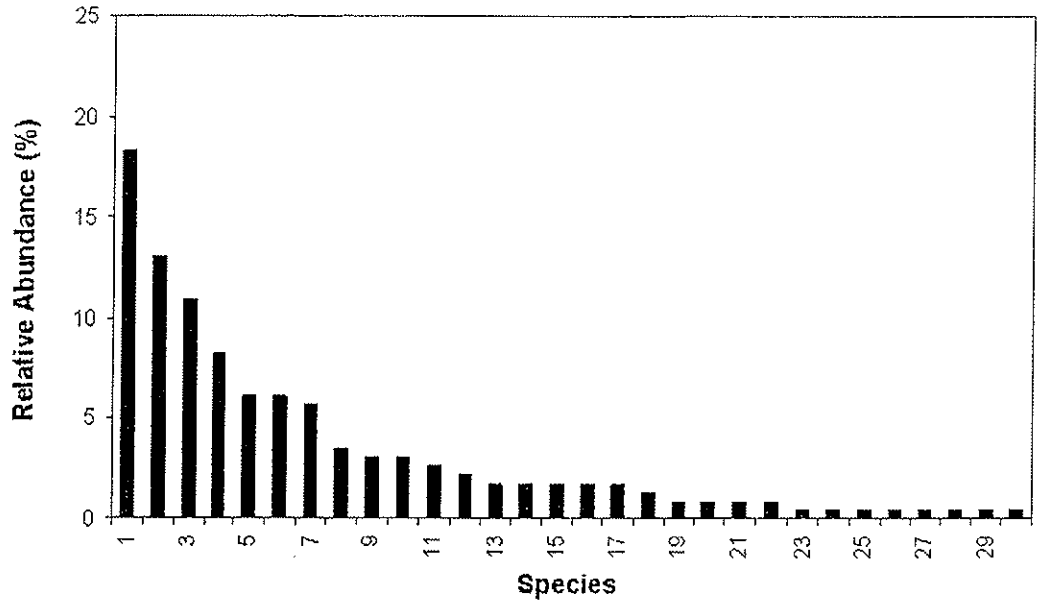


Figure 2: Relative abundance of reptiles at Gimlet South undisturbed

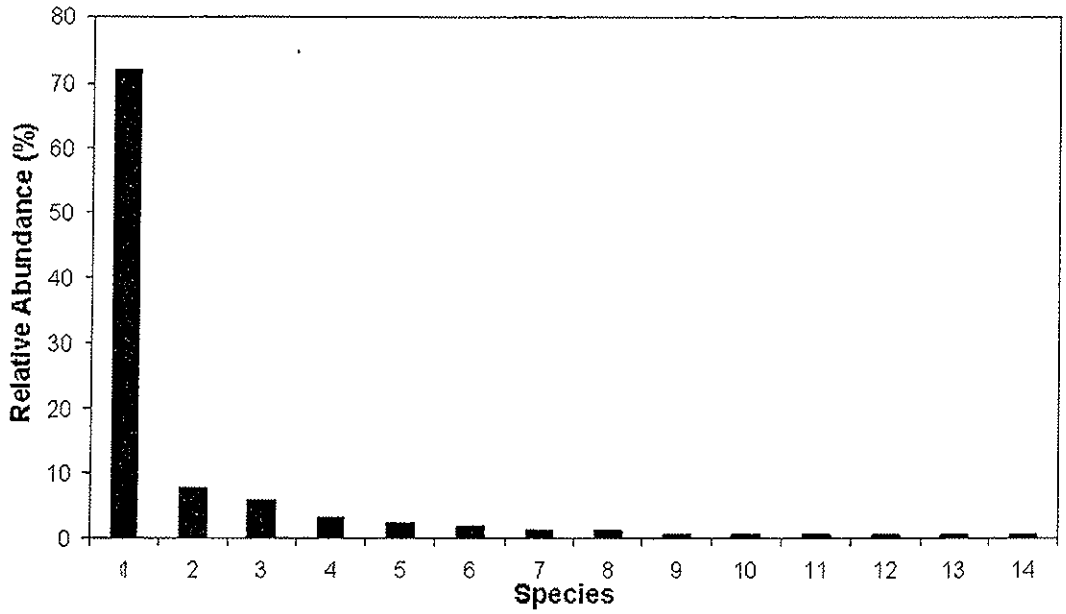


Figure 3: Relative abundance of reptiles at Gimlet South waste dump

Table 4: **Summary of parameter scores for trophic composition for Gimlet South and Rose undisturbed areas and waste dumps**

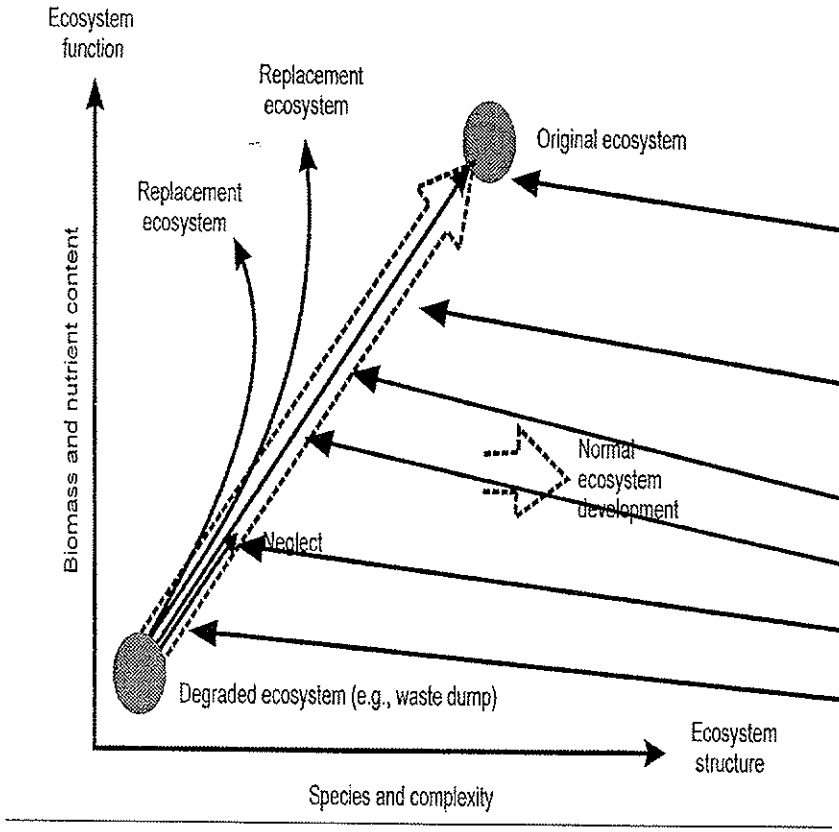
	Sites	Gimlet South				Rose			
	Maximum score	Undisturbed		Waste dump		Undisturbed		Waste dump	
			Score		Score		Score		Score
% dietary specialists	10	23	10	9	3	20	10	0	1
% omnivores	5	7		9		8		18	
% invertivores	5	90		91		84		82	
% vertebrate carnivores	5	3		0		8		0	
Summary score for trophic composition					13				8
Parameter 3 score	25		**25**		**16**		**25**		**9**

Table 5: **Summary of parameter scores for preferred habitat at Gimlet South and Rose undisturbed areas and waste dumps**

	Sites	Gimlet South			Rose		
	Maximum score	Undisturbed	Waste dump		Undisturbed	Waste dump	
		Score	Score		Score		Score
% arboreal	5	17	18		16	18	
% fossorial	5	23	18		20	0	
% terrestrial	5	60	64		64	82	
Parameter 4 score	15	**15**	**12**		**15**		**8**

Table 6: **Summary of parameter scores for the predatory strategy at Gimlet South and Rose undisturbed areas and waste dumps**

	Sites	Gimlet South			Rose		
	Maximum score	Undisturbed	Waste dump		Undisturbed	Waste dump	
		Score	Score		Score		Score
% sit-and-wait species	5	7	27		8	18	
% widely foraging species	5	57	18		68	55	
% actively foraging species	5	37	55		24	27	
Parameter 5 score	15	**15**	**7**		**15**		**10**



Total Index score (sum of ratings)	Attributes
141-180	Comparable to the best situation without human impact; regionally expected species for habitat; species present with a full array of age (size) classes; balanced trophic structure; self-sustaining functional ecosystem.
121-140	Species richness approaching expected levels; not all late succession species present, some species present with less optimal abundances or size distribution; trophic structure incomplete.
85-120	Species richness below expectation, some groups not well represented, some specialists not present
51-85	Lack of specialists; fewer species, skewed trophic structure and relative abundances
31-50	Few vertebrates present; early colonisers, lack of community structure
0-30	Early succession species only. No community structure. Only opportunistic early colonisers present

Figure 4: Linkage between Bradshaw's (1984) process of ecosystem development and the rehabilitation index scores

Table 7: Summary of overall index scores for Gimlet South and Rose undisturbed areas and waste dumps

Sites	Undisturbed sites	Gimlet South Waste dump	Rose waste dump
	Maximum score	Score	Score
Species richness	85	33	42
Abundance	40	16	24
Trophic composition	25	16	9
Preferred habitat	15	12	8
Predatory strategy	15	7	10
Overall Score	180	84	93

Parameter 5 – Predatory composition

The predatory composition parameter is a measure of the proportion of sit-and-wait species, widely foraging species, and actively foraging species. Actively foraging species are those that forage over a broad area, relatively quickly looking for a dispersed food source. Widely foraging species are those that forage over a wide area but feed from a concentrated food source (i.e. termite mound or ant nest). Sit-and-wait predators are ambush predators and may move from one location to another in search of sites from which they can rapidly move to catch their prey.

The proportion of sit-and-wait species in both the undisturbed sites is low compared to the actively and widely foraging species (Table 6) and both waste dumps have a higher proportion of sit-and-wait predators compared to the undisturbed sites. On a preliminary analysis, Rose waste dump scored a 10, and Gimlet South waste dump scored a 7 compared with a maximum of 15 in the undisturbed area (Table 6).

Overall Index

The scores for each of the five parameters are summed into a table (Table 7) to provide an overall index of the site. Scores can range from 0-180. Based on an understanding of the succession of reptile species into rehabilitated areas and a preliminary analyses of the available data, the range of scores (0-180) can be divided into six classes (Table 1) and a general description of the reptile assemblage at each class can be developed. This table can then be linked to Bradshaw's (1984) process of succession from degraded ecosystems to a climatic community (Figure 4).

There are other possible parameters that can be added to the index such as lizard size and activity times. Like all the parameters described above these will be subjected to further investigation. When fully developed it may be feasible to indicate an index score that can be used to determine when the mine can be released from its rehabilitation obligation. As this index only deals with the small herpetofauna it will probably need to be considered in conjunction with a measure or index of stability [(e.g. Landscape Function Analysis (Tongway, 2001)] and vegetation structure. It is also possible the index can be developed to incorporate small mammals and birds and maybe some groups of invertebrates. I have data on the small mammals for all my research sites and this aspect of the index will be further developed.

For this index to have a wider application than the sites from which it has been developed it should be tested in a range of other habitats, e.g. sandy coastal heath, inland red sand dunes covered with spinifex, woodlands and forest in the south west of WA.

This Rehabilitation Index is not the total solution to monitoring rehabilitation success, but it can provide an indication of the progress of the small faunal recolonisation onto rehabilitated areas. It is hoped that this index in combination with a vegetation monitoring and soil/landscape structure monitoring scheme (e.g. Ecosystem Functional Analysis) will provide an overall quantifiable measure of rehabilitation success.

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A comparison of the passerine avifaunas of a rehabilitated minesite and a nearby reserve in south-western Australia

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Abstract. The passerine birds at two sites on a former minesite rehabilitated about 20 years earlier were compared with those at two sites in a nearby banksia woodland reserve near Capel in south-western Australia from July 1996 to February 1997. The same assemblages of common gleaning insectivores were present in both areas and in similar numbers. However, although the same species of honeyeaters occurred at both areas, the former minesite had many more larger honeyeaters, but fewer small spinebills. It is suggested that these differences were not only the result of differences in floristics, but also the result of clumping of replanted species on the minesite, which allowed large aggressive honeyeaters to defend these rich nectar sources, but largely excluded small nectarivores. Those wishing to encourage small honeyeaters to recolonise rehabilitated areas may need to include areas where planting density ensures dispersed nectar sources.

Introduction

The rehabilitation of land following mining is often aimed at restoring components of the original ecosystems or emulating those of undisturbed areas nearby (Collins *et al.* 1985; Majer 1989). Plant communities are often the primary focus, but recolonisation by vertebrates and invertebrates may also be considered. The selection of plants used for rehabilitation is often dictated by abiotic factors such as climate and soil type, as well as the availability of species. However, there may also be a desire to use plants that provide a resource for native biota, preferably plants present in the area before disturbance (Nichols and Watkins 1984). Honeyeaters (Meliphagidae) are one of the most diverse, abundant and widespread families of birds in Australia and one group whose distribution and abundance may be influenced by both the floristic and structural composition of rehabilitated vegetation. This study aimed to contribute to our understanding of these relationships by a comparison of the passerine birds, particularly honeyeaters, in vegetation at a rehabilitated minesite, and those in natural bushland in a nearby reserve within south-western Australia.

Methods

The study was conducted near Capel (33°35′S, 115°30′E), 200 km south of Perth, Western Australia, in an area of deep sandy soils. It used the Renison Gold Consolidated Wetlands Centre, now Capel Wetland Centre, a rehabilitated mineral sands minesite, and the nearby Capel Nature Reserve; these will be termed the minesite and the reserve, respectively. The minesite was cleared of its original vegetation over 100 years ago, then planted with pines, later cleared for mining. The rehabilitated minesite comprises a chain of wetlands surrounded by vegetation planted between 1977 and 1983. The 300-ha minesite was sampled using a 8-ha site at its southern end and a 5-ha site 3 km further north. The 94-ha reserve was 4–9 km from these sites and was also sampled using two areas, a 9-ha site at its southern end and a 10-ha site in its northern half. The reserve comprised banksia woodland with mixed heath, presumably similar to the original vegetation of the minesite.

A 1.1-km fixed transect was established in each of the four areas, two at each site, and used to census birds for 30 min between 0800 and 1000 hours twice monthly between 21 July 1996 and 24 February 1997 following the methodology of Newland and Wooller (1985), Loyn (1986) and Allen (1989). Every attempt was made to conduct the census under similar weather conditions throughout this period. All birds within 25 m each side of the transect line were recorded by direct observation or from their calls, trying not to count individuals twice. There were therefore 56 censuses overall, 14 in each of the four areas and 28 at each site. The results of both sets of censuses in the reserve were very similar, as were both sets in the minesite, and so data have been pooled for each site to aid comparisons between them. In addition, birds were captured in all four areas using mist nets, positioned to sample all parts of the habitat, then measured and pollen samples taken from them in order to assess the plants they visited (Wooller *et al.* 1983*a*, 1983*b*). Visual censuses probably overestimate large birds, whereas mist-netting may overestimate small and more secretive birds (Ford and Paton 1982). Use of the two techniques was, therefore, thought to give a more balanced picture of the bird assemblages at the sites compared.

Two 45-m-long transects through each area were employed to obtain measures of relative plant density and projected foliage cover, using the wandering-quarter plotless method (Brower *et al.* 1990). Vegetation less than 0.5 m high was rarely used by honeyeaters and was not included in these transects. Every two weeks throughout the study, records were made of the most abundant species in the Proteaceae and Myrtaceae that were in flower, scoring flowering as prolific, moderate or sparse.

Results

The bird assemblages

Overall, 603 birds of 36 different species were recorded during censuses in the reserve, compared with 533 birds of 33 species recorded at the minesite; 28 of these species were

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EARLY REGROWTH IN REHABILITATED BAUXITE MINESITES AS BREEDING HABITAT FOR BIRDS IN THE JARRAH FOREST OF SOUTH-WESTERN AUSTRALIA

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Summary

A survey was undertaken to assess 5 to 10 year old rehabilitated bauxite minesite vegetation as a resource for birds while breeding in the jarrah (E. marginata) and marri (E. calophylla) forest at Jarrahdale, Western Australia. Minesites have been rehabilitated and revegetated with local shrub species and a mix of local and eastern eucalypts resistant to the root rot fungus Phytophthora cinnamomi. Quadrats in minesites and adjacent forest were searched for evidence of birds breeding at the peak of the spring nesting period.

Twenty-one species of birds used minesite rehabilitation for foraging or hunting while breeding; twelve of these were found nesting or rearing fledglings in the revegetation. Among the breeders were four understorey dwelling species which were otherwise uncommon in surrounding upland forest or (in two cases) confined to stream zone undergrowth. Total densities of all pairs or groups breeding in minesites were similar to those in nearby forest. Breeding territories often spanned areas of both minesite and forest around the edges of pits.

Eleven other species found breeding only in forest quadrats all nested in sites as yet unavailable within minesites, such as tree hollows, fissures, bark crevices and crown foliage. The possible use of nest-boxes to aid in the early provision of sites for hollow-nesting species is being investigated.

Introduction

In parts of the Darling Range in Western Australia, areas of native forest dominated by jarrah Eucalyptus marginata and marri E. calophylla have been mined for bauxite. Minesites are mostly between two and twenty hectares in extent, and remain surrounded by forest. These bauxite pits have been revegetated in various ways (Tacey 1979, Tacey and Glossop 1980). Currently, planting involves establishing eucalypt and understorey associations resistant to the root rot fungus Phytophthora cinnamomi, a disease that kills jarrah and many understorey species. Recolonisation of minesites by local species of plants and invertebrates has been described by Fox et al (1981) and Nichols and Bunn (1980). Among vertebrates, 67 species of birds have been found making use of rehabilitated minesites within 10 years of their establishment (Kabay and Nichols 1980, Nichols and Watkins 1984). The present study was undertaken to assess the early potential of minesites to provide breeding habitat for birds.

Preliminary field work at Jarrahdale carried out between 1978-81 indicated that most birds of the jarrah forest bred only in spring, with a sharp peak of activity in October-November. This regular breeding season co-incides with the onset of the warm, dry months November to April and is consistent with a wider pattern of spring breeding by forest-dwelling birds in the southwest (Davies 1980).

Study areas and methods

Minesite rehabilitation and adjacent plots in jarrah/marri forest at Alcoa of Australia Limited Jarrahdale No.2 minesite (32°07'S, 116°05'E) were searched daily for evidence of breeding during the period 12th October to 8th November 1981. A total of twelve one-hectare quadrats were marked out to span parts of four minesites and adjacent forest areas. Minesite and forest quadrats were sampled as equally as possible, with similar amounts of time being spent to locate birds and their breeding sites in the two broad habitat types. The six hectares of minesite plots sampled were composed of three distinct types of rehabilitation:

1. Two hectares of a 6-year-old stand of marri 5-10m high. This plot had a patchy but sometimes dense understorey of mainly leguminous shrubs. Plant diversity was comparable to that recorded in unmined forest (Tacey and Glossop 1980).

2. Three hectares of a 10-year-old stand of eastern Australian tallowood, Eucalyptus microcorys, 8-12 m high, with sparse understorey.

3. One hectare of a 5-year-old plantation of mixed Western Australian and eastern Australian eucalypts 5-8 m high, entangled with a thicket-like understorey of 2 m tall Acacia spp. and other, mostly leguminous shrubs.

Quadrats extended up to 300 m from the minesites and about 500 m of forest/minesite edges were included. Three of the six hectares of forest contained many dead trees and had been subjected to a low intensity burn during the previous autumn.

Results and Discussion

At least 21 species found to be using the minesite quadrats during the study period bred during that time (Table 1). Twelve of these were found nesting or rearing fledglings in the minesites. Old nests of the New Holland Honeyeater Phylidonyris novaehollandiae and Silvereye Zosterops lateralis, two species that did not breed in the study area at the time, were also found in the minesites.

Only eight species in a total of 23 breeding species nested in both the minesite and forest quadrats (Table 1). Nine others that nested in nearby forest hunted or foraged in minesite rehabilitation while breeding: these were the Little Eagle Hieraaetus morphnoides, Port Lincoln Ringneck Barnadius zonarius, Elegant Parrot Neophema elegans, Tree Martin Cecropis nigricans, Western Thornbill Acanthiza inornata, Little Wattlebird Anthochaera chrysoptera, White-naped Honeyeater Melithreptus lunatus, Striated Pardalote Pardalotus striatus and Australian Raven Corvus coronoides. In many instances, occupied territories spanned boundary areas between forest and minesite. Forest-nesting birds often visited nearby

minesites to feed, some from blossoms (e.g. White-naped Honeyeater, Little Wattlebird), or amid leafy shrubbery (Striated Pardalote, Western Spinebill Acanthorhynchus superciliosus, Western Thornbill and Port Lincoln Ringneck) or on the ground (Elegant Parrot).

Some bird species e.g. Grey shrike-thrush Colluricincla harmonica, Western Rosella Platycercus icterotis and Varied Sitella Daphoensitta chrysoptera are commonly observed in the jarrah forest but were not recorded breeding in the present study. Reasons for this might be the relatively small areas sampled, and that some of the forest had been burnt the previous autumn. It is unlikely that any diurnally active breeding birds were missed in the quadrats given the intensity of coverage.

Three of the four species confirmed as breeding in rehabilitation but not in the forest quadrats are primarily understorey dwellers. Of these the White-breasted Robin Eopsaltria georgeana and Red-winged Fairy-Wren Malurus elegans are otherwise confined to dense streambed vegetation when breeding in the Jarrahdale area. Their early colonisation of understorey thickets in hillside minesites indicates an ability to disperse and to adopt novel and perhaps seral habitats that resemble creekline associations. The fourth species, the Brown Honeyeater Lichmera indistincta, is an example of a species that was abundant in mixed shrubbery and middle-level eucalypt foliage typical of forest-minesite boundaries rather than continuous forest.

Each of the eleven species that nested in nearby forest but not in rehabilitation vegetation was a user of nest sites in mature, senescent or dead trees: either hollows (e.g. parrots), fissures (Rufous Treecreeper), bark crevices (Western Thornbill), forks (Scarlet Robin) or canopy foliage (White-naped Honeyeater). No such sites yet exist in the minesites.

The birds found nesting in the minesites represent virtually all of the locally-occurring species that could be expected to find suitable nest sites in early stages of regrowth. These correspond to the second in a sequence of four growth (or successional) stages envisaged by Kabay and Nichols (1980) to occur in the rehabilitation process. Later stages are characterised by the maturation of trees and their production of bark, litter and eventually hollows and spouts. Such growth may take many more

years to accomplish. Bird species most likely to commence breeding in minesites in the shorter-term are those that nest in the forest canopy foliage (e.g. White-naped Honeyeater) or on high forks (e.g. Scarlet Robin Petroica multicolor).

Early recolonisation of minesites by hollow-nesting species cannot be anticipated. However, nest boxes have been successfully used in eastern Australian eucalypt woodlands where hollows are limiting (Golding 1979). Nest boxes have been placed in rehabilitated areas to see if it is possible to promote the early return of hollow-nesting species.

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Boyd Wykes correctly predicted the peak time of year to look for birds breeding at Jarrahdale. Comments on the manuscript were contributed by I. Colquhoun, S,J.J.F. Davies, D. Kabay and G. Slessar while typing was done by Mrs. D. Pratt.

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TABLE 1

Comparison of species proved (*) or suspected breeding in 6 ha of rehabilitated minesites and 6 ha of adjacent forest at Jarrahdale, October to November, 1981.

	No of Breeding Pairs/Groups	
SPECIES NESTING IN BOTH FOREST AND MINESITE QUADRATS.	Minesites	Forest
Fan-tailed Cuckoo	1*	1
Western Yellow Robin	2*	1*
Golden Whistler	3*	1
Grey Fantail	4*	1*
Splendid Fairy-wren	2	1
Western Gerygone	4*	2*
Inland Thornbill	4*	1*
Western Spinebill	3*	2*
SPECIES NESTING ONLY IN MINESITE QUADRATS		
White-breasted Robin	2*	
Red-winged Fairy-wren	2*	
White-browed Scrubwren	1*	
Brown Honeyeater	1*	
SPECIES NESTING ONLY IN FOREST QUADRATS		
Brown Goshawk		1
Port Lincoln Ringneck		1*
Elegant Parrot		2*
Sacred Kingfisher		1*
Scarlet Robin		1*
Western Thornbill		4*
Rufous Treecreeper		1*
Little Wattlebird		1*
White-naped Honeyeater		3*
Striated Pardalote		5*
Australian Raven		1
MINIMUM NUMBER OF PAIRS/GROUPS	29	31

TERRESTRIAL FAUNA SURVEYS: WHY, WHEN AND HOW MUCH?

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ABSTRACT

Surveys of small vertebrate terrestrial fauna provide regulators and mine site environmental staff with a valuable source of data to plan, monitor and obtain closure on rehabilitated mine sites as well as preparing an EIA. Presuming the objective of a rehabilitation program is to create a functional ecosystem, then it is important that a comprehensive survey of the terrestrial fauna is undertaken in the rehabilitated area, and the undisturbed area that the rehabilitated area is designed to mimic. The survey effort required to capture a nominated proportion of the species on a site can be determined from the species accumulation curve. Twenty litre buckets are superior to PVC pipes as pit-traps, but pipes catch species unlikely to be caught in buckets and therefore should be used in conjunction with buckets. Pit-traps with drift fences are better than Elliott traps, but should be supplemented by Elliott and funnel traps for a complete survey of an area. Pit-traps should be placed 5 –7 m apart, arranged in a straight line, with shelter in the bottom of buckets. Ideally, surveys should be undertaken during spring, summer and autumn and in more than one year if the data are to provide an understanding of biodiversity and ecosystem function.

INTRODUCTION

As land managers our aim should be to leave the site without anyone knowing we have been there. This can be difficult in some mining environments as the scale and impact of the disturbance is such that mining companies would argue that it would make the mine economically non-viable, particularly as overseas competitors are not required to meet similar environment standards. However, as stewards for the environment it behoves us to work towards this objective. In circumstances where it is neither reasonable nor feasible to achieve such an objective, then being 'environmentally neutral' may be the next best option. For example, it may not be feasible to fill a mining void, however, to offset the blight that this void has on the landscape and the environment, the mining company might negotiate with the regulators to improve another part of the degraded environment under their stewardship by de-stocking an overstocked rangeland and letting the natural fauna and flora return, eradicating feral pests, or supporting environmental research.

If we accept the objective of *leaving the site without anyone knowing we have been there*, then our rehabilitation programs should be designed to recreate the self-sustaining functional ecosystem that existed before the disturbance. It is appreciated that this will be difficult for some sites and will have to settle for recreating 'similar' functional ecosystems.

Bradshaw (1984) provides a very useful model that represents the development of ecosystem function and structure as a 2-dimensional graph (Figure 1). In a rehabilitation context, natural ecosystem development, or succession, involves moving from a disturbed and degraded environment through a series of stages to a more complex state at the top right of the graph (Figure 1). Rehabilitation involves planning a process that assists the environment to move toward this final state. Incomplete or inappropriate rehabilitation will result in an ecosystem that differs from that which originally existed. If our objective is to leave the site without anyone knowing we have been there, then the objective of our rehabilitation programs will be clear. The recreation of functional ecosystems will require that we stabilise the substrate, ensure appropriate topsoils are in place and revegetate the site as near as possible to the vegetation community structure that existed before the disturbance. We then let nature take its course and hope that the invertebrates and vertebrates from the surrounding areas will move in and colonise the area.

If we consider the broad scale successional process, an appropriate floral community structure will only develop on suitable soils. Invertebrates will only colonise an area if suitable habitat (shelter and food source) is available. Small vertebrates are either herbivorous and thus require the appropriate vegetation, invertivorous and are dependent on the availability of invertebrates which depend on the vegetation, which depend on the soils, or are carnivorous, feeding on small vertebrates. It is therefore obvious that small terrestrial vertebrates (e.g., reptiles, frogs and mammals) are generally the last to move into an area, and seldom are these introduced by human intervention. Small terrestrial vertebrates can therefore provide a useful indicator of the extent to which an ecosystem is moving toward the recreation of the original system, or a replacement system, as they are the last stage in the rehabilitation process. If we understand the structure and dynamics of the small terrestrial vertebrate community in the original ecosystem then we can use this knowledge to develop a rehabilitation index, which can tell us how far we are along the path to recreating the original ecosystem (see Thompson, S. 2002, Rehabilitation index based on reptile community structure for mine site monitoring, this volume).

Why do we need to survey the terrestrial fauna? Well the answer is obvious; we need to understand the structure and dynamics of the small terrestrial fauna in an undisturbed ecosystem if we intend to recreate it. If we intend to use fauna as a measure of rehabilitation success (e.g., rehabilitation index) then we need to collect the input data.

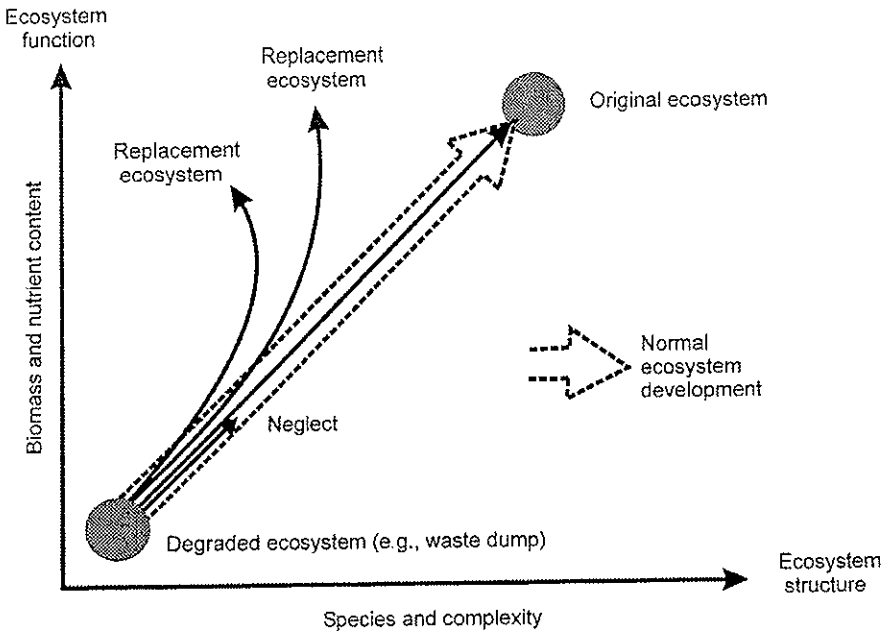

Figure 1: The process of ecosystem development (Bradshaw, 1984)

There are also legislative reasons for undertaking small terrestrial vertebrate fauna surveys at mining sites. The Commonwealth Environment Protection and Biodiversity Conservation Act 1999, section 18, indicates a person must not take an action that will impact on threatened, critically endangered, endangered and vulnerable species. Similarly, a person must not take an action that will significantly impact on a listed threatened or endangered ecological community. Clearly, before disturbing an area, appropriate fauna surveys need to be undertaken to ensure compliance with this Act. More recently the Environmental Protection Authority (EPA, 2002) has issued Position Statement No. 3, *Terrestrial Biological Surveys as an Element of Biodiversity Protection.* In this publication the EPA has grouped the Interim Biogeographic Regions (IBRA) in Western Australia (WA) to define the level of biological survey expected for the preparation of an environmental impact assessment (EIA). Depending on the scale and nature of the impact, proponents of a disturbance are required to either undertake; a) a desktop study and reconnaissance survey, or b) a desktop study, reconnaissance survey, and a comprehensive flora and fauna survey. No detail is provided of what the EPA expects in regard to a 'comprehensive' fauna survey. However, the EPA does indicate it expects proponents to ensure the terrestrial biological surveys provide sufficient information to address biodiversity conservation and ecological function. Biodiversity value is to be considered at the genetic, species and ecosystem level, and ecological function, at the ecosystem level. Clearly, to assess issues related to biodiversity and ecological function a near complete list of fauna species for the site is required. The importance of 'rare' species in the context of ecosystem function will determine whether a complete list of vertebrate species should be required.

Current level of terrestrial fauna surveys in Western Australia mining industry

We surveyed 88 mines in WA seeking information on the extent to which mines had developed completion criteria and to ascertain the current status of flora and fauna monitoring in the mining industry in this state. Thirty six companies responded to the questionnaire, of these 23 systematically monitored flora and only three monitored fauna. Although, we failed to capture information for all of the mines in WA, for those we did, few are systematically surveying the fauna.

Current level of fauna surveying for the preparation of EIA

Fraser (2001) assessed the extent of fauna surveys and data analysis undertaken by consultants during the preparation of 15 of the most recent EIA in the Coolgardie IBRA. Of these, only three searched the Department of Conservation and Land Management (CALM) and Western Australian Museum (WAM) databases, but most reviewed the published literature. Only three of the 15 consultants undertook field surveys in both spring and autumn, most undertook a single survey in either spring or autumn, and only one consultant surveyed the site during summer.

Fauna surveys for a biotope were sampled at one location only by all consultants; the mean number of pit-traps, Elliott traps and cage-traps used was 8, 11 and 1 respectively. The mean trapping effort within a season was 138 pit-trap nights, and for all seasons was 164 pit-trap nights for a biotope (Fraser, 2001). For fauna surveys covering a landscape scale impact, the mean number of pit-traps, Elliott-traps and cage-traps used was 76, 92 and 8 respectively, and the mean number of pit-trap nights for a single season was 1 211, and for all seasons it was 1 507 (Fraser, 2001). Only 12 of the 15 consultants had a component of their field survey designed to search for rare/endangered or priority-listed fauna. Fraser (2001) concluded there was insufficient field survey effort to provide anything other than a cursory understanding of faunal diversity and little effort was made to detect the presence of species requiring special protection under existing legislation.

Based on the findings of Fraser (2001) it is apparent that insufficient field survey effort was undertaken by most of the consultants to enable mine site environmental officers to plan and monitor rehabilitation programs in the Goldfields region if we are to use Bradshaw's (1984) approach.

SURVEY PROTOCOLS AND ISSUES

Well, what should be the protocols for surveying small terrestrial fauna? For the remainder of this paper we review the literature to see what advice others offer and we summarise some of our data from an extensive pit-trapping program that we have undertaken in the Ora Banda region over the last two years to provide guidance for terrestrial fauna survey protocols.

We consider the following questions:

- How much survey effort is required?
- What sort of trapping protocol should be used?
- What strategies can be used to search rare and range restricted species?

How much?

We can answer the 'how much' question by reviewing species accumulation curves. Species accumulation curves simply graph the rate at which new species are identified for the amount of trapping effort.

A species accumulation curve typically rises steeply as the common species are caught during the initial trapping effort and then the slope of the curve flattens as considerably more effort is required to capture the rare or difficult to trap species (Figure 2). Trapping effort can be measured as either the number of pit-trap days (or Elliott-trap days) or the number of specimens caught.

The relative abundance of species at most sites takes on a characteristic concave to the origin curve if species abundance is arranged in an array from highest to lowest (Figure 3). The graph of relative abundance can be related to the species accumulation curve, as the more abundant species are normally caught first and a much greater trapping effort is required to catch the less abundant species. For example, the species accumulation curve in Figure 2 commences to plateau between 42-44 species, which corresponds with the species being represented less than 0.15% in the population (Figure 3).

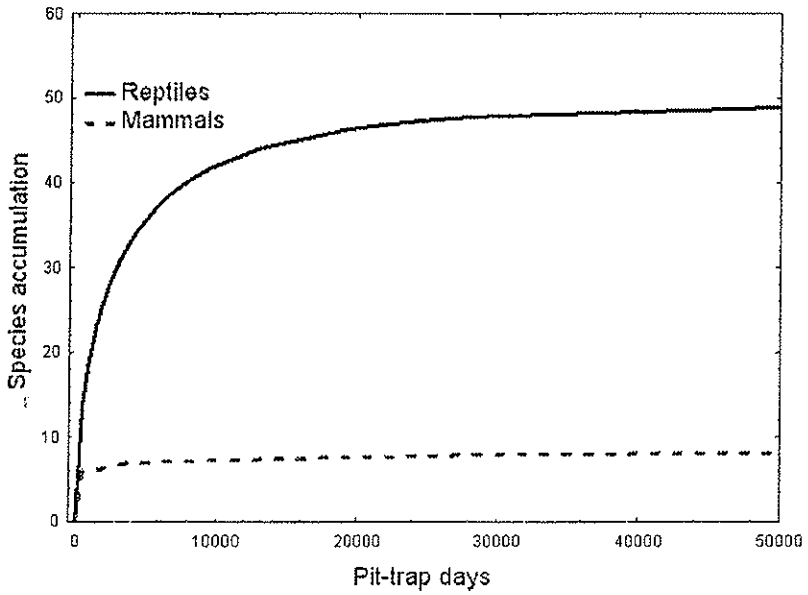

Figure 2. Species accumulation curves for reptiles and mammals for Ora Banda study sites

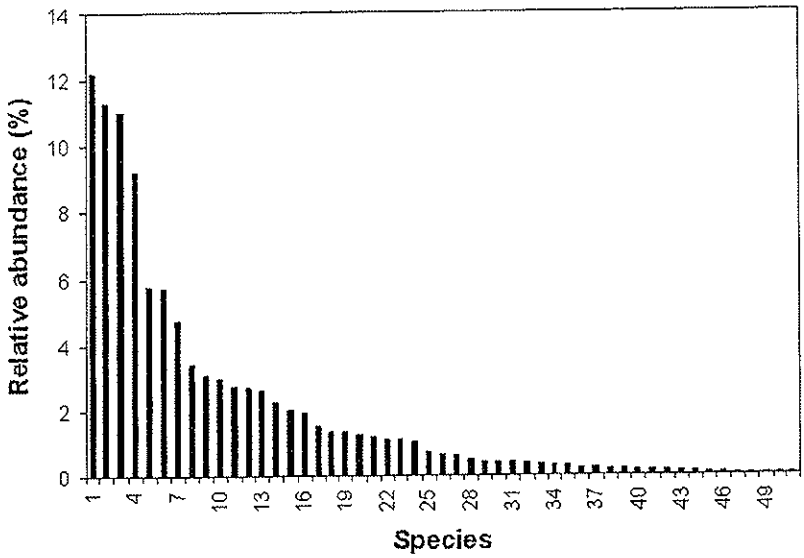

Figure 3: Relative abundance of reptile species at Ora Banda

The shape of species accumulation curves is influenced by species richness, diversity and evenness (Thompson and Withers, unpub. manuscript). Therefore, species accumulation curves are likely to differ among sites (Thompson, Withers, Pianka and Thompson, unpub. manuscript).

How then can species accumulation curves be used as a tool for predicting species richness and what is the survey effort required to assess diversity at a particular site? Determining the survey effort is a two-stage process. The first stage involves a survey to capture sufficient specimens for the species accumulation curve to commence to plateau. Based on our data for five heterogenous or landscape scale sites (two in the Great Victoria Desert, Bungalbin sand plains, Barrow Island and Ora Banda; Thompson, Withers, Pianka and Thompson, unpub. manuscript) this is about 1 000 - 1 500 specimens, and for a homogenous biotope it is about 400 specimens. The species accumulation curve can then be used to estimate species richness and the effort required to catch a nominated percentage of the trappable species (e.g., 75%, 95%, etc). To determine the full extent of the survey required, two measures of effort must be used; a) the number of specimens caught will indicate the trapping effort required to catch a nominated percentage of species at the site, and b) pit-trap days will indicate the field time and costs of trapping. For a given number of species to be caught, the number of specimens or pit-trap days can be read off the species accumulation curve.

Two cautionary notes are in order here. Species accumulation curves are not always smooth and the earlier plateauing of a species accumulation curve may provide an under-estimate of species richness. For example, Thompson, *et al.* (Thompson, Withers, Pianka and Thompson, unpub. manuscript) show the species accumulation curve for a Bungalbin sand plains site does not plateau even after 40 000 pit-trap days (Figure 4). For a heterogenous site in the Great Victoria Desert, after 11 000 pit-trap days species richness was estimated to be 43, but as trapping continued and more rare species were captured, after 25 000 pit-trap days species richness was estimated to be 55 species.

Fraser (2001) established a panel of experts consisting of six academic and government researchers, and six consultants that had extensive experience in terrestrial fauna surveys to develop criteria of 'best practice' for terrestrial fauna surveys. This panel of experts were asked to nominate the pit-trapping effort required to adequately survey:

a) a biotope, which was defined as a homogenous habitat, and

b) a landscape, which was defined as a heterogenous habitat containing ten biotopes.

For the biotope, these experts considered 137 trap-nights per season and 409 trap-nights for all seasons, and for the landscape scale, 1 371 trap-nights per season and 3 630 trap-nights for all seasons was sufficient to adequately survey the area. When these data are plotted to species accumulation curves for a biotope and landscape scale sites in the vicinity of Ora Banda it is apparent that the level of surveying effort recommended is inadequate to catch 95% of the species in the area (Figures 5 and 6).

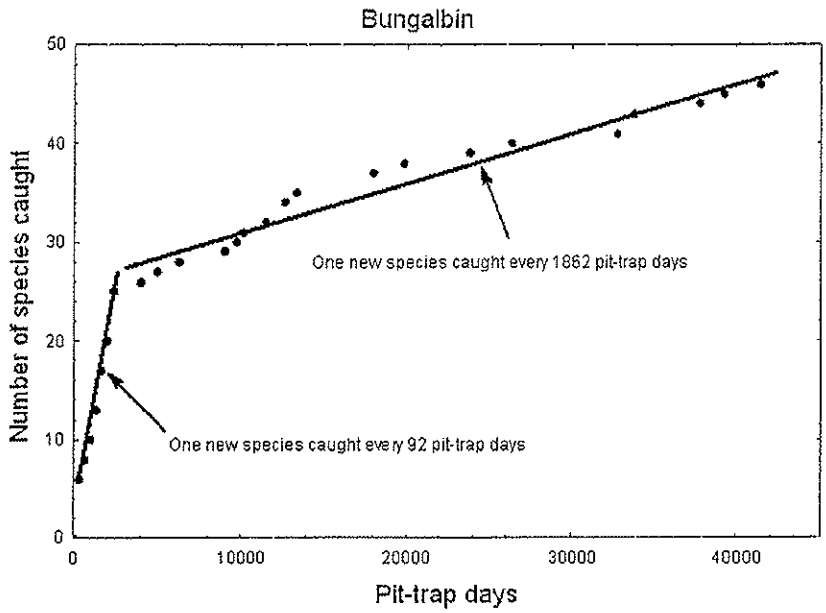

Figure 4: *Species accumulation curve for a Bungalbin sand plains site (Thompson, Withers, Pianka and Thompson, unpub. manuscript)*

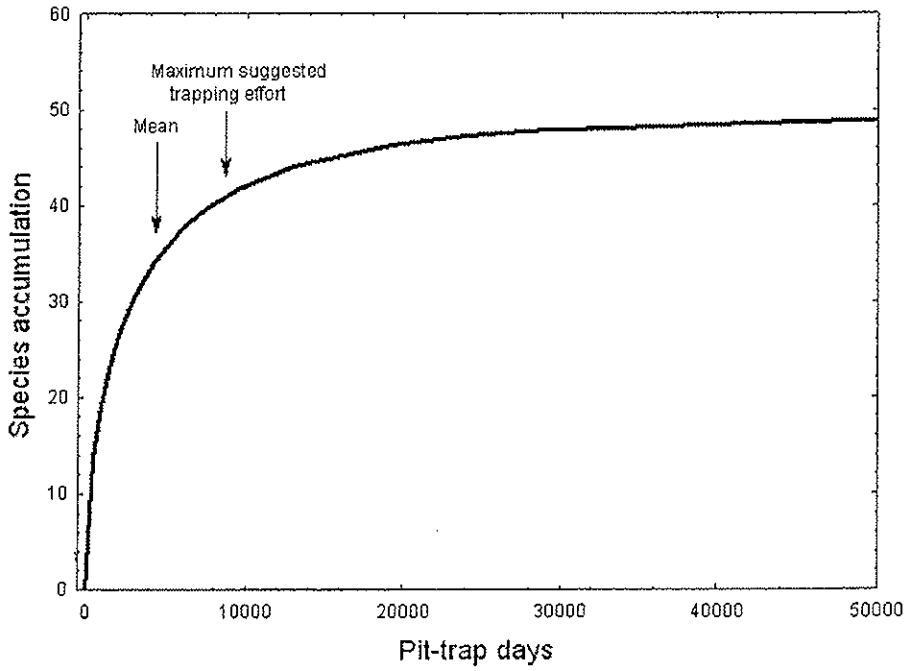

Figure 5: *Species accumulation curve for Ora Banda at landscape scale showing the trapping effort suggested by experts*

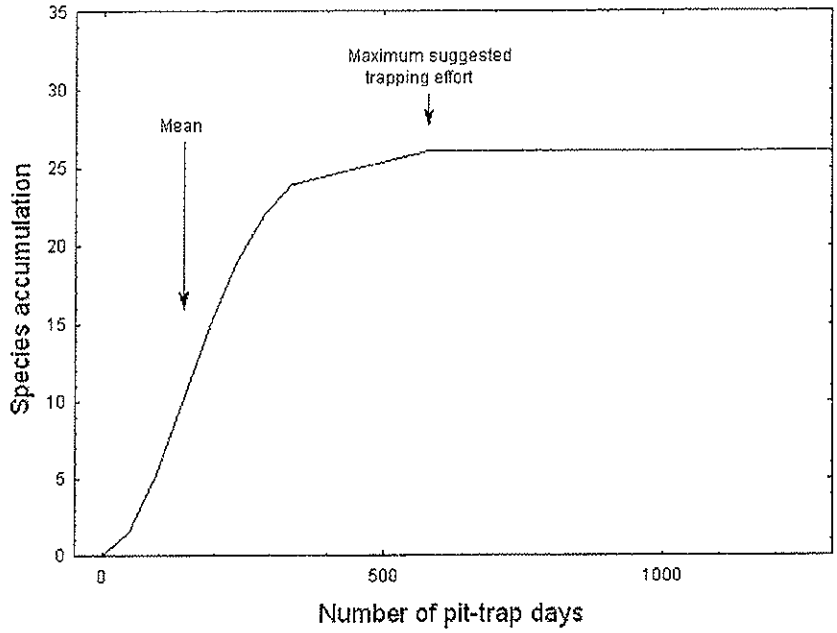

Figure 6: Species accumulation curve for site at biotope level scale showing the trapping effort suggested by experts

Trap design and layout

Cage vs pit-traps

Cockburn *et al.* (1978) report pit-traps with drift fences to be more effective than baited Elliott traps for catching small reptiles and mammals. Greenberg *et al.* (1994) report pit-fall traps captured fewer species than single and double ended funnel traps, but yielded more individuals of many species and higher average species richness than funnel traps. Cockburn *et al.* (1978) reported funnel traps were much more successful in catching large snakes.

In October, 2001 we placed 90 Elliott traps 100 m apart along a disused track on the Bungalbin sandplains, an area where we had previously caught 47 species of reptiles and 8 mammal species. 180 Elliott-trap nights yielded only one *Ctenotus* skink and no mammals. During September/October, 2001 we set 90 Elliott traps for a period of six days then moved and set them again for another six days in 11 undisturbed and five rehabilitated areas around Ora Banda. Overall capture rates for mammals and reptiles were significantly higher for pit-fall traps than Elliott traps (Table 1). More species of reptiles were caught in pit-fall traps than Elliott traps and there was little difference in the species capture rate for mammals. These data indicate pit-fall traps are superior to Elliott traps for terrestrial fauna surveys.

Table 1: Reptile and mammal capture rates using Elliott and pit-fall traps in the vicinity of Ora Banda (1 260 Elliott-trap nights and 6 230 pit-trap nights)

		Captures / 1000 Elliott trap nights	Captures / 1000 pit-trap night
No specimens	mammals	23.0	36.9
	reptiles	2.3	52.8
No Species	mammals	1.6	1.2
	reptiles	2.4	5.3

Buckets vs pipes as pit-fall traps

Both 20 L PVC buckets and 150 mm x 600 mm PVC storm water piping are used for pit-fall traps. Which is better? Friend *et al.* (1989) and Morton *et al.* (1988) report buckets caught more animals than pipes. A closer examination of the data by Friend et al. (1989) indicates that large lizards and geckos showed highly significant biases for large pits; frogs, small lizards, snakes and legless lizards less so, while small mammals were caught equally often in both pit types.

We have 888 pit-fall traps installed on waste dumps and undisturbed areas in the vicinity of Ora Banda. A line of pit-fall traps consists of 3 buckets and 3 pipes that alternate and are joined by 30 m of fly-wire drift fence. Based on approximately 50 000 pit-trap days of data, buckets caught more reptiles than pipes, whereas pipes caught more mammals than buckets. Some large lizards (e.g., *Varanus gouldii, Ctenophorus cristatus*), snakes (e.g., *Pseudonaja modesta*) and the larger mammals (e.g., *Notomys mitchelli*) are more likely to jump out of buckets than pipes. We conclude from this that if only one type of trap is to be used, buckets would be preferred, however, there is a case for using buckets and pipes because of the additional mammals and larger reptiles that are caught. Based on the data of Cockburn *et al.* (1978) there is a case for using funnel traps, perhaps placed along pit-fall trap drift fences.

Drift fences

Drift fences are generally made of fly-wire (300 - 350 mm high). They stand vertical and run between pit-fall traps, and are used to guide animals into the pit-fall traps. As they are expensive to purchase and install; are they worth the effort?

Bury and Corn (1987), Morton *et al.* (1988) and Friend *et al.* (1989) report pit-fall traps joined by drift fences were more effective than those without in capturing small vertebrate fauna. The ideal distance between pit-fall traps seems to be between 5 and 7 m. Morton *et al.* (1988) report the optimal design to maximise captures was to arrange the pit-traps and drift fences in the form of a cross, however, a more recent investigation by Hobbs *et al.* (1994) suggest the most effective design was a straight line of pit-traps and drift fences with buckets 7 m apart.

Friend *et al.* (1989) report small mammals, geckos and small lizards showed significant bias towards permanently erected fences, while frogs, snakes and legless lizards showed the opposite response. Some reptiles are attracted to fresh diggings (e.g., *Varanus eremius*) and data from one trapping period to the next is unlikely to be comparable because of 'digging in' effects.

Table 2: Seasonal difference in the number of specimens and species caught in a pit-trapping program in the Tanami Desert (Morton *et al.* 1988)

	Season / Design	Big pits	Cross arm	Fence	No fence
No specimens	Oct. / Nov.	226	173	121	83
	March / April	91	85	70	49
No species	Oct. / Nov.	21	23	17	7
	March / April	14	12	9	7

Table 3: Seasonal differences in the number of reptile specimens caught in a pit-trapping survey around Ora Banda

	Sept. 2000	Dec. 2000	Jan. 2001	April 2001
Total No geckos	276	322	386	127
Total No skinks	149	155	197	60
Total No dragons	38	16	44	19
Total No goannas	0	12	14	3
Total No legless lizards	6	8	5	2
Total No snakes	29	28	29	2
Total No reptiles	498	541	675	213

Data from 10 undisturbed and 4 rehabilitation sites for September, December, January and April 2000/01; 5 376 pit-trap days for each season.

Covers on pit-traps

During summer, the interior of pit-traps often heat to above the critical thermal maximum of most mammals and reptiles. To avoid heat-stress of captured animals different types of covers have been tried. Hobbs and James (1999) placed external covers over the pit-traps to reduce thermal radiation and heat-stress. Insulation foil covers reduced the core temperature in pit-traps, but these above-ground foil covered traps caught 39-43% fewer small vertebrates.

We have used both white styrene 'meat' trays and egg cartons placed in the bottom of PVC buckets to provide shelter for captured animals. As the animals cannot detect this material in the bottom of the pits until they are in the pit-trap, their presence is unlikely to affect capture rates. Our experience is that few species die from heat stress in 600 mm deep, 150 mm diameter pipes.

Seasonal differences

Morton *et al.* (1988) report significant differences in the total number of specimens and the number of species caught in a pit-trapping survey in the Tanami Desert between October/November and March /April (Table 2).

Our data for Ora Banda also demonstrates significant seasonal differences in the number of reptiles and mammals that were caught across 10 undisturbed and four rehabilitated habitats (Tables 3, 4 and 5).

In addition to seasonal differences there are also temporal differences. We surveyed the same 10 undisturbed and four rehabilitation sites in September 2000 and 2001. The daily maximum temperature was generally higher in September 2000 than in September 2001, the consequence was a significantly higher number of reptiles and mammals were caught in September 2000 (Table 6).

These data provide clear evidence that a terrestrial fauna survey in a single season in one year is inadequate to provide an understanding of species richness, diversity or abundance at an individual site or at a landscape scale. Data are not available to enable us to understand year-to-year natural variations in the population structure over a successive number of years, however, it is apparent that there are considerable changes from year-to-year.

Table 4: Seasonal differences in the number of dragon lizards caught in a pit-trapping program around Ora Banda

	Sept. 2000	Dec. 2000	Jan. 2001	April 2001
Ctenophorus cristatus	0	0	5	0
Ctenophorus reticulatus	10	2	17	7
Ctenophorus scutulatus	3	0	3	0
Moloch horridus	5	1	2	0
Pogona minor	15	9	17	12
Tympanocryptis cephala	5	4	0	0

Data from 10 undisturbed and four rehabilitation sites for September, December, January and April 2000/01; 5 376 pit-trap days for each season.

Table 5: Seasonal differences in the number of mammals caught in a pit-trapping program around Ora Banda

	Sept. 2000	Dec. 2000	Jan. 2001	April 2001
Cercartetus concinnus	32	30	22	3
Mus musculus	112	118	55	144
Ningaui spp.	4	0	1	7
Notomys mitchelli	0	1	0	0
Pseudomys bolami	0	46	19	21
Pseudomys hermannsburgensis	14	22	5	2
Sminthopsis crassicaudata	35	116	48	66
Sminthopsis dolichura	11	45	12	22
Total	208	378	162	265

Data from 10 undisturbed and 4 rehabilitation sites for September, December, January and April 2000/01; 5 376 pit-trap days for each season.

How useful are desk studies?

The EPA advocates (*Terrestrial Biological Surveys as an Element of Biodiversity Protection. Position Statement No. 3., 2002*) that where a project is likely to affect biodiversity then one of two levels of investigation be undertaken. For projects in some IBRA with either low or moderate potential impacts (Level 1) a desktop study and reconnaissance survey is required, for impacts assessed to be higher in scale or nature then a desktop study, reconnaissance survey and a comprehensive flora and fauna survey are required. No guidance is provided on what might be expected of the desktop study, but it is presumed that a search of the literature be undertaken to identify fauna species likely to occur in the area, with particular attention to rare, range restricted or endangered species. It might also be expected that consultants or proponents would consult both the CALM and the WAM databases, however, we know from Fraser's (2001) study this does not always occur.

How useful are searches of CALM and WAM databases, and literature reviews in identifying species likely to be found in a particular area? The Ora Banda area of the Goldfields was one of the WAM/CALM regional survey areas. Our Ora Banda study sites fall within the Kurnalpi - Kalgoorlie study area and are closest to the Black Flag Homestead study area (McKenzie *et al.* 1992). Specimens caught during the CALM/WAM survey were vouchered with the WAM and species lists along with a description of the vegetation can be found in McKenzie *et al.* (1992).

McKenzie *et al.* (1992) also searched the WAM database and species not captured but likely to be in the area were listed in their report. It might therefore be concluded that the fauna for the Ora Banda region is well documented and a desktop study and literature review would identify all possible species likely to be pit-trapped at our Ora Banda study sites.

Table 7 lists species of reptiles we have caught after approximately 50 000 pit-trap days from July 2000 to April 2002 that were not recorded in the field survey of CALM/WAM (McKenzie *et al.* 1992) for the Black Flag Homestead area. We also indicated those reptile species the WAM/CALM staff thought should be in the area based on WAM records (Table 7). A similar analysis was undertaken for the mammals in the area with a surprising result. We recorded one reptile, and two or possibly three species of mammals not likely to be identified being in the area based on a desktop study comparable with that likely to be undertaken by a diligent consultant. It is therefore evident that our relatively scant knowledge of the distribution of some species is such that erroneous conclusions might be drawn about biodiversity and ecological systems, even in areas where our knowledge of the fauna is considered reasonable.

Table 6: **Temporal differences in the number of specimens of reptiles and mammals caught in a pit-trapping program around Ora Banda**

	Sept. 2000	Sept. 2001
Total No geckos	276	172
Total No skinks	149	45
Total No dragons	38	22
Total No goannas	0	3
Total No legless lizards	6	1
Total No snakes	29	19
Total No reptiles	498	262
Total No frogs	11	0
Cercartetus concinnus	32	37
Mus musculus	112	50
Ningaui spp.	4	18
Pseudomys bolami	0	8
Pseudomys hermannsburgensis	14	6
Sminthopsis crassicaudata	35	41
Sminthopsis dolichura	11	20
Total No mammals	208	180

From 10 undisturbed and four rehabilitation sites during September 2000 and 2001; 5 376 pit-trap days for each season.

Table 7: **Species caught in during 55 000 pit-trap days not recorded in earlier surveys or thought to be in the area (McKenzie *et al.* 1992).**

Identified in the Ora Banda survey but not recorded by the WAM/CALM survey	Suggested as likely to be in the area by the WAM/CALM study from other sources	Not anticipated in the area
Reptiles		
D. assimilis		
E. richardsonii	X	
E. striata		X
P. minor	X	
T. cephala	X	
V. tristis	X	
D. butleri		
D. fraseri		
P. lepidopodus	X	
D. fasciata	X	
P. australis	X	
R. australis	X	
R. bituberculatus	X	
R. hamatus	X	
S. bertholdi		
S. semifasciata		
Mammals		
Ningaui ridei	X	
Ningaui yvonneae	X	
Notomys mitchelli	X	
Pseudomys hermannsburgensis	X	
Pseudantechinus woolleyae		X
Antechinomys langier		??
Pseudomys albocincereus		X

Strategies for detecting rare, range restricted or endangered species

Within the Coolgardie IBRA there are two reptile species with a priority status: Woma Pythons (*Aspidites ramsayi*) and Carpet Pythons (*Morelia spilota imbricata*) are both listed in Schedule 4, of the Wildlife Conservation Amendment Regulations 2001. We have a confirmed sighting (photograph) of *M. s. imbricata* within 1 km of the Ora Banda town site in 2000, and WAM records suggest that there are likely to be other specimens of this python in the IBRA.

WAM records also indicate that the Woma python was relatively abundant in the wheatbelt up until the 1970s (Maryan, 2002). Recent WAM records show Womas to the east of Kalgoorlie, and there is a 1966 WAM record of a Woma being caught south of Menzies.

Maryan (2002) reports a recent sighting of a Woma at Bandya homestead north of Kalgoorlie. The preferred habitat of Womas appears to be myrtaceous heath on sand plain (Smith, 1981), of which there is considerable in the IBRAs covering the Goldfields. In the right habitat it is therefore probable there are few remaining specimens of this Schedule 4 species.

If we consider rare or range restricted mammals in the IBRAs around the Goldfields, and consult Strahan's distribution in *The Mammals of Australia*, it is evident there could be a number of rare, endangered or range restricted species in the region. Pit-traps, Elliott traps and the other search strategies commonly used by consultants are unlikely to turn up either of the Schedule 4 pythons and most of the listed mammals if they were in the area.

Table 8: Possible search strategies to identify priority-listed and range restricted taxa.

Example taxa	Strategy	Reference
Pythons	Road searches, tracks	
Woolley's Pseudantechinus	Hair traps	Lobert, Lunmsden, Brunner & Triggs 2001
Marsupial moles	Examination of carnivore scats	Paltridge, 1998
Large snakes	Funnel traps	Greenberg *et al.* 1994
Arboreal varanids and mammals	Tree nooses, snares	Reed 2000
Mallee fowl nests, burrows of Mulgaras, Bettongs and Bilbies	Aerial photograph searches	
Mammals	GIS searches based on habitat	Catling & Coops 1999
Mammals and birds	Airborne, videography	Catling & Coops 1999

It is evident that to identify many of the rare and priority-listed species alternative search strategies should be adopted. To date little has been documented on how we might approach this issue. We currently have a PhD student investigating suitable cost-effective terrestrial fauna search strategies that might be used for the purposes of preparing an EIA, complying with the Environment Protection and Biodiversity Conservation ACT 1999 and the thrust of the EPA guidelines. Detection of priority-listed species will be an aspect of this study. There is a range of search options that could be useful and will be explored, and these are shown in Table 8.

SUMMARY

There is evidence to suggest that a poorly planned and executed terrestrial fauna surveys or ones that apply inadequate effort are a waste of time and resources. Inadequate terrestrial fauna surveys are unlikely to provide useful data on the presence of species in a particular site, and are unlikely to be useful in planning or monitoring rehabilitation sites. Therefore, the adage 'do it right or don't waste your time and money' seems most appropriate when considering terrestrial fauna surveys. Desktop studies cannot be relied upon to provide comprehensive species lists for sites, particularly in areas that have been poorly surveyed.

The question of how much survey effort is required to identify a nominated proportion of the species at a site can be estimated in a two stage process; a preliminary survey to establish a species accumulation curve to the point the plateau commences to consolidate, and using this curve to predict the number of species for a site. The survey effort can be determined from a species accumulation curve based on the number of specimens caught, and the time and cost for the survey from a species accumulation curve based on pit-trapping (or Elliott trapping) effort.

Existing data suggest that pit-traps are better than cage traps but, to capture the entire range of trappable species, pit-traps should be supplemented by cage and funnel traps. Drift fences enhance captures. The optimal distance that pit-traps should be apart along drift fences is between 5 - 7 m, and a line of pit-traps is as good as the alternative configurations. Shade, in the form of egg cartons or styrene trays placed in the bottom of PVC buckets provides shelter for captured animals and reduces heat stress. Pit-traps should be dug in and left until the 'digging in' effects have disappeared to ensure trapping data are comparable between different survey periods.

Ideally, areas should be trapped/searched in late spring, summer and early autumn to obtain an understanding of small terrestrial vertebrate diversity and abundance at a site. Single surveys in January are likely to provide the highest species and specimen counts, closely followed by December, and to a lesser extent late spring and early autumn. Year-to-year variation in species abundance and overall diversity, suggests that surveys need to be undertaken over a number of years if a complete understanding of the dynamics of the ecosystem are to be understood. The presence of many of the rare and priority-listed species is unlikely to be identified using the search strategies commonly applied for the purposes of preparing an EIA. More innovate approaches are required.

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A Review of the Concepts and Application of Rehabilitation Following Heavy Mineral Dune Mining

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Dune mining has had an enormous impact in some coastal regions. This paper reviews: the concepts of rehabilitation, restoration and revegetation as applied to dune mining for heavy minerals, the basic principles of dune rehabilitation, some case studies of rehabilitation and its success, the methods for measuring the success of rehabilitation, the process of producing a dune rehabilitation plan, and makes recommendations for pilot studies and guidelines for the implementation of the rehabilitation plan. Studies in Australia and South Africa in particular, provide successful models to draw upon in formulating the criteria and principles discussed in this paper. © **1999 Elsevier Science Ltd All rights reserved**

Keywords: restoration; rehabilitation; revegetation; reclamation; heavy minerals; dune mining; beaches; dunes; wetlands; soil fauna and flora.

Introduction

Heavy mineral dune mining has been carried out for more than sixty years in Australia (Allen *et al.*, 1990) resulting in much experience in rehabilitation of dune systems. Along the Australian east-coast, studies of previously mined areas in 109 localities revealed an extensive range of land uses. These include recreation, urban, conservation, agriculture, forestry and water supply areas amongst others (Allen *et al.*, 1990). Because human populations and heavy mineral sand deposits are both concentrated along the coastline, it is important to consider future land uses following mining. Early mining on the frontal dunes in Australia led to techniques of dune stabilization rather than rehabilitation of the ecosystem (McDonald, 1964). However, with the extension of mining into most of Australia's eastern coastline and subsequently world sites (Fig. 1) rehabilitation technology has advanced to a level whereby native ecosystems can be re-established (Brooks, 1986).

Australia's experience in dune and other mining rehabilitation led to the establishment of the Australian Centre for Minesite Rehabilitation Research in order to meet the needs of industry, governments and the community for sustainable systems for land affected by mining (Bell, 1996). In Africa dune mining for the last twenty years at Richards Bay in KwaZulu–Natal has resulted in many studies being conducted on rehabilitation (Moll, 1992; Mentis and Ellery, 1994; Van Aarde *et al.*, 1996; Lubke *et al.*, 1996; Van Aarde and Smit, 1997). These and other rehabilitation studies provide baseline information for any proposed dune mining rehabilitation project. The emphasis in this paper is on rehabilitation after dune mining for heavy minerals using the dredge pond process, although techniques of rehabilitation from other case studies may also be applicable.

Because of the confusion about the use of the terms replacement, rehabilitation, restoration and revegetation, the concept of rehabilitation needs to be clarified. Rehabilitation of ecosystems has been the focus of many ecological studies over the last few years (e.g. Buckley, 1989; Jordan *et al.*, 1990; Berger, 1991; Cairns, 1993). Harris *et al.* (1996) give a number of definitions of terms, and favour reclamation rather than rehabilitation, a term rarely used in the United Kingdom in these studies. How one measures the success of rehabilitation or restoration of the degraded ecosystem has been examined by Westman and O'Leary (1986), Westman (1991), Ewel (1990) and Harris *et al.* (1996). The latter authors also assess why restoration projects fail.

It is important to have information on vegetation and habitat types of the region prior to mining, and baseline studies on the soil, fauna, flora and vegetation are therefore essential. The establishment of vegetation and the restoration of the ecosystem will vary according to site-specific environmental conditions, the nature of the soil and various other biotic influences peculiar to the region. Since these aspects are site specific, one has to consider them independently for the various sites.

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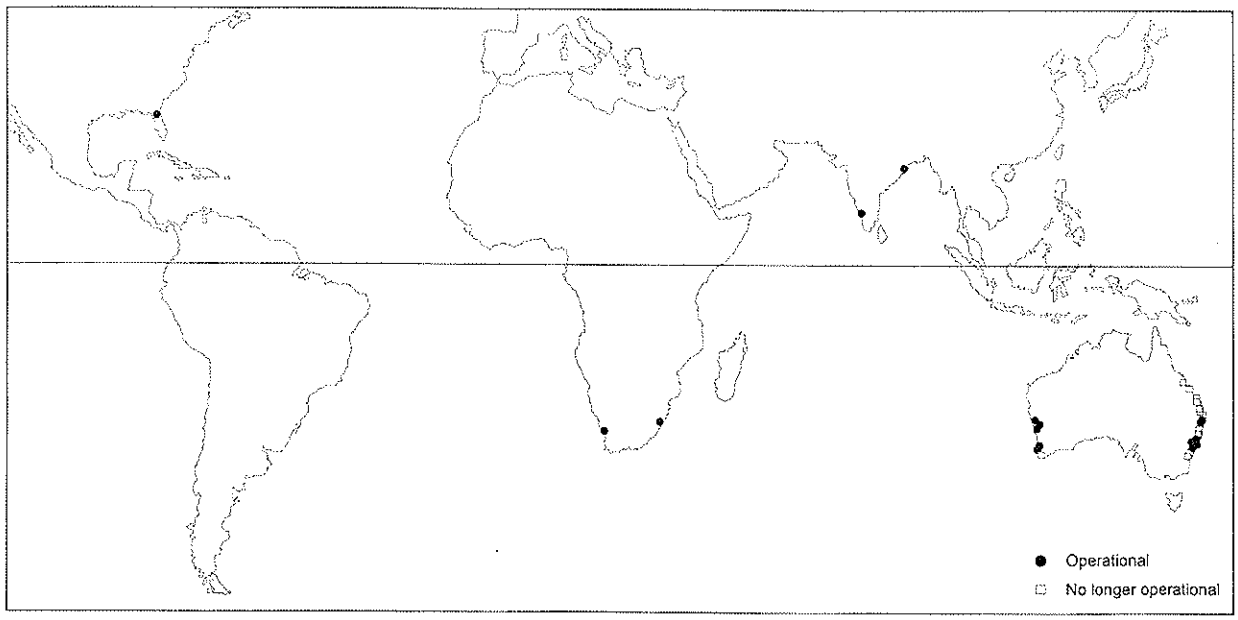

Fig. 1 The world location of heavy mineral dune mining in coastal environments. (Source Allen *et al.*, 1990; Allen, 1999, pers comm.; Brooks, 1999, pers. comm.).

In carrying out a rehabilitation study, experimental work is necessary, and the information presented here considers the potential for rehabilitation to provide a conceptual plan which can be tested by means of experimental studies or field trials at sites where dune mining is taking place. For the well-being of indigenous people, it is also important to rehabilitate or establish vegetation which is the preferred land use of the local people, provided it fits in with the constraints dictated by the environment. In view of the necessity to restore ecological and cultural diversity to the landscape (Naveh, 1998), a mosaic of vegetation types along with cultivated crops or woodlots is the most likely preferred long-term goal in regions where dune mining has taken place in southern Africa (Coastal and Environmental Services, 1999). This requires a broadening of conceptual and methodological scope from the natural sciences to the humanities (Naveh, 1994); an approach which is followed in modern environmental impact assessment (e.g. Coastal and Environmental Services, 1999).

With these points in mind, the goals of this paper are therefore:

- To review the concepts of dune rehabilitation, restoration and revegetation.
- To present the basic principles of rehabilitation following dune mining.
- To examine rehabilitation studies and their success in various habitats especially following dune mining.
- To present methods of measuring the success of dune rehabilitation.
- To discuss the formulation of a conceptual rehabilitation plan, which includes both ecological and cultural landscape restoration.

Rehabilitation, Restoration or Revegetation?

Mining has been an activity of man for thousands of years. The development of more sophisticated mining methods and tools has caused greater impacts on the environment. It is only in this century that serious consideration has been given to the rehabilitation of areas before the actual mining takes place. Bradshaw and Chadwick (1980) reviewed the state of restoration of land and did much to formulate ideas and concepts on rehabilitation. They, and in later papers, Bradshaw (1983, 1984, 1990) identified the major problems of derelict land as being high concentrations of toxic metals and salts, acidic conditions, coarse particles and lack of moisture and essential nutrients. These factors combined to create a hostile environment which is not conducive to the establishment of plants. Fortunately, in dune mining, only mobile sand and lack of nutrients and moisture are important. Early work on rehabilitation of derelict land concentrated more on the removal of the nuisance value of pollution and unsightly mine dumps (e.g. Thatcher, 1979). More recently, the emphasis has been on the reclamation of land in a way that allows it to be re-used. Thus, Bradshaw (1984), like Harris *et al.* (1996), prefers the general term of reclamation to relate to the return of the land to a functional use.

Bradshaw (1984) produced a diagram to show the various concepts in terms of changes in biodiversity and structure of the vegetation with time. Lubke (1993) modified it to show how it applied to dune mining on the south-east coast of southern Africa (Fig. 2). After any mining, one starts off with a degraded environment low in biomass, nutrient content and species

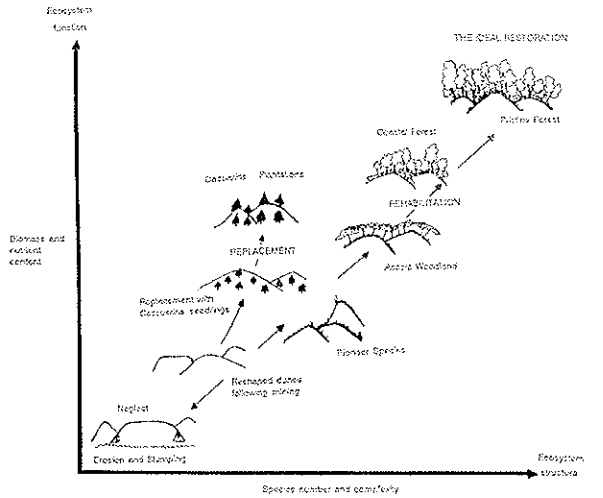

Fig. 2 Restoration, rehabilitation and revegetation in dune mining, as applied in Richards Bay in Kwa–Zulu–Natal (adapted from Bradshaw, 1984).

diversity. From this point one needs to consider the objectives of the rehabilitation programme before starting any stabilization or revegetation. The process most simple and easy to accomplish is *revegetation*, which is the establishment of vegetation on the degraded environment. If this vegetation is not the same type as that which originally covered the area, then this is referred to as *replacement*. Replacement may be desirable if one wishes to introduce agriculture or forestry to a region. However, if the intention is to return a degraded system to its original state, one must look to a process of *restoration*. Vegetation will be accompanied by the return of animals and other organisms to the environment, and may lead to a process of ecosystem development. Restoration implies returning the system to what it was prior to the disturbance. It is unlikely that complete restoration will be achieved, and thus it is more appropriate to use the term *rehabilitation*. Rehabilitation may be thought of as a process somewhere between the extremes of degradation and restoration (Fig. 2). These concepts have been reviewed by Bradshaw (1984), Westman (1991) and more recently by Lubke (1993) with respect to dune mining in KwaZulu-Natal. In the latter case, plantations of *Casuarina* trees were the replacement process and restoration of the coastal forest was the ultimate goal for the rehabilitation of natural or indigenous vegetation (Fig. 2). Whatever the end point, rehabilitation or replacement, the process must be viewed as a practical land use management technique in which the end product is borne in mind. Aronson *et al.* (1993) also presents a model for ecosystem degradation and restoration which is similar to that proposed by Bradshaw (1984). They identify two states of *restoration*, viz. *sensu stricto* and *sensu lato* and consider *rehabilitation* will only achieve the latter state, which is the alternative to the original ecosystem. Their term *reallocation* is a different term for *replacement* as discussed above.

Conway (1985), in an analysis of agricultural systems, proposed a number of agricultural models of productivity, stability and sustainability which could be subjected to disturbance or stress. Mining would be a perturbation or disturbance causing an impact on the system, thus bringing about a decrease in biomass. Once the destructive force is removed, rehabilitation or reclamation may be achieved resulting in an increase in biomass to the original level. Harris *et al.* (1996) apply stress and disturbance to degraded environments in a similar model showing change in biomass over time. Van Andel (pers. comm., 1998) stressed the importance of measuring the correct parameters in the application of such a model. He showed how different authors measured different components such as productivity or species richness over time in the study of rehabilitation or restoration.

Principles of Rehabilitation

The type of mining and the characteristics of the particular mineral deposit both affect the degree to which mining disturbs the landscape (Environmental Protection Agency, 1995). Surface mining will result in the destruction of the existing vegetation and soil profile, and can significantly change the topography and stability of the landscape. Because toxic overburdens or waste slimes are not produced in dune mining, the rehabilitation process is somewhat simpler than it is for other types of mining. Nevertheless, the basic principles of rehabilitation shown in Table 1 still apply.

Rehabilitation Studies and Success

Rehabilitation and restoration projects on degraded environments, especially wetlands and tropical ecosystems, are presently given great emphasis in ecology. In recent years, thorough reviews of rehabilitation (e.g. Bradshaw and Chadwick, 1980; Jordan *et al.*, 1990; Harris *et al.*, 1996) and also journals specifically related to restoration ecology and degradation of the environment have been produced (viz. *Applied Vegetation Science*, *Ecological Engineering*, *Land Degradation and Development* and *Restoration Ecology*). We review some of the studies most relevant to rehabilitation or replacement of vegetation in dune mining.

Beaches and foredunes

In only limited cases in Africa has mining been carried out along the beach and in the foredunes. Such mining has, however, been undertaken in Australia, with successful rehabilitation of the dunes which were formed above the mined beach (Brooks and Yeates, 1980). The closer one comes to the beach, the more

TABLE 1

Principles of rehabilitation (adapted from Environmental Protection Agency, 1995).

- Prepare a rehabilitation plan prior to the commencement of mining
- Agree on the long-term post-mining landuse objective for the area with the relevant government departments, local government councils and private landowners. The landuse must be compatible with the climate, soils, topography of the final landform and the degree of management available after rehabilitation
- Progressively rehabilitate the site, where possible, so that the rate of rehabilitation is similar to the rate of mining
- Prevent the introduction of noxious weeds and pests
- Minimize the area cleared for mining and associated facilities to that absolutely necessary for the safe operation of the mine
- Reshape the land disturbed by mining so that it is stable, adequately drained and suitable for the desired long-term landuse
- Minimize the long-term visual impacts by creating landforms which are compatible with the surrounding landscape
- Reinstate natural drainage patterns disrupted by mining wherever possible
- Minimize the potential for erosion by wind and water both during and following mining
- Remove or control residual hazardous materials. Identify any potentially toxic overburden or exposed strata and manage them so as to prevent environmental damage
- Characterize the top soil and retain it for use in rehabilitation. It is preferable to reuse the topsoil immediately rather than storing it in stockpiles. Only discard if it is physically or chemically undesirable, or if it contains high levels of weed seeds or plant pathogens
- Consider spreading the cleared vegetation on disturbed areas
- Deep rip compacted surfaces to encourage infiltration, allow plant root growth and key the topsoil to the subsoil, unless sub-surface conditions dictate otherwise
- Ensure that the surface one or two meters of soil is capable of supporting plant growth
- If topsoil is unsuitable or absent, identify and test alternative substrates, e.g. overburden that may be a suitable substitute after addition of soil improving substances
- Revegetate the area with plant species consistent with the post-mining landuse
- Meet all statutory requirements
- Make the area safe
- Remove all facilities and equipment from the site
- Monitor and manage rehabilitated areas until the vegetation is self-sustaining and meets the requirements of the landowner or land manager, or until their management can be integrated into the management of the surrounding area

dynamic the system, consequently, more effort has to be put into the rehabilitation process to ensure stability of the land surface as soon as possible. At Richards Bay in KwaZulu-Natal, rehabilitation of the dunes has mainly taken place a few hundred metres from the high tide mark. However, where mining has approached closer to the high tide, slumps of the steep dune system have occurred (Moll, 1992). This problem has led to detailed studies of the water flow within the dune, and of methods of overcoming slumping (Davies, Lynn and Partners, 1992). Degradation of the dunes after mining can be likened to damage caused by natural processes such as large storms and shifting sands in blow-out areas. These areas also require stabilization and management; accounts of such processes in dune systems is given for the UK by Ranwell and Boar (1986), and for South Africa by Tinley (1985) and Lubke and Avis (1991). The dune stabilization process has been reviewed by Avis (1989) for the Cape Province.

Use of indigenous or native species

The problem affecting the stabilization of sand dunes is that there is a deficiency of water and nutrients and strong maritime winds coming off the ocean may cause extensive erosion and changes in the dune pattern before vegetation can become established. Indigenous plant species are better adapted to these adverse conditions, although they may take some time to become established. Using indigenous species to stabilize dunes is generally encouraged in South Africa (Avis, 1989), North America (e.g. Chambers *et al.*, 1994; Corbett *et al.*, 1996) and Australia (e.g. Ford and Langkamp, 1987; Ward *et al.*, 1996), especially if a natural ecosystem is to be formed. Indigenous plants facilitate the natural succession of dune communities.

Heavy mineral dune mining in southern Africa

Following dune mining at Richards Bay (Fig. 3a), a successful rehabilitation programme has been developed by Camp (1990). The process involves the removal of vegetation from the mine path in front of the dredge pond (Fig. 3b), and the deposition of the topsoil on the tailings behind the pond. The tailings are reshaped into their original contours prior to topsoiling and revegetation. When indigenous vegetation is to be established, a cover crop of annuals such as paola grass (*Pennisetum americanum*), sun hemp (*Crotalaria juncea*) and *Sorghum* sp. together with indigenous species are used as a cover crop (Fig. 3c). These annuals are short lived and will eventually be replaced by indigenous grasses and woody shrubs and trees (Camp and Weisser, 1991). In that region, *Acacia karroo* is a predominant early successional forest tree, and it dominates the rehabilitated areas. (Figs. 3d and e). Being a legume with nitrogen-fixing bacteria in root nodules it has competitive advantage in the nitrogen-poor soils of the rehabilitation sites. A dominant species can become restrictive in the successional process, retarding the influx of other species. Suggestions have been made to thin out the *Acacia karroo* trees to speed up the succession at this rehabilitation site (e.g. Moll, 1992; Lubke *et al.*, 1996). The increase in legumes in

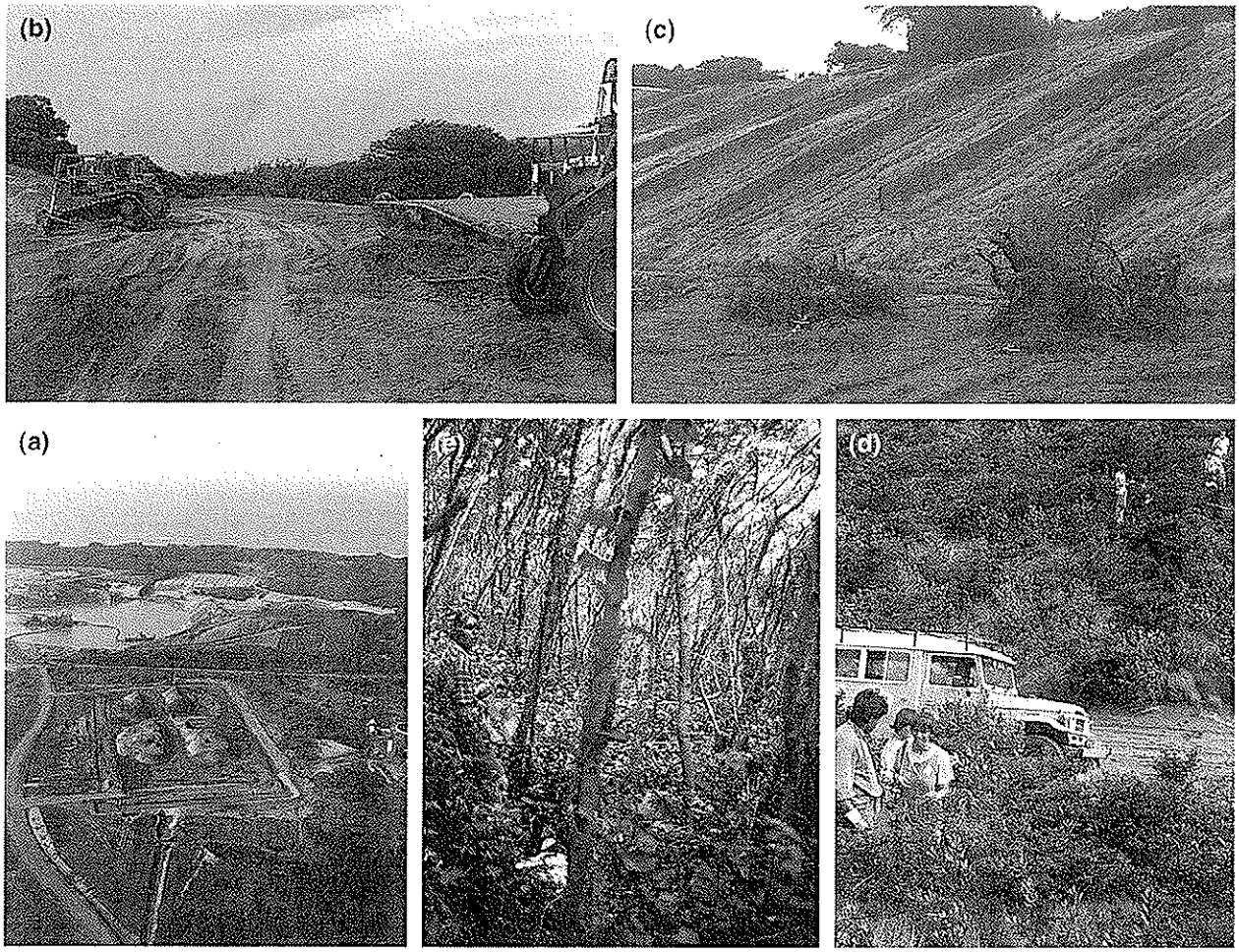

Fig. 3 Mining and rehabilitation of the dunes at Richards Bay, South Africa, by Richards Bay Minerals. (a) The mine pond with dredger and extraction plant on a cleared dune field. Heavy mineral concentrate is stockpiled in the foreground. (b) Clearing the vegetation in the mine path. (c) Establishing a cover crop of annuals amongst the windbreaks on reformed dunes. Note the old stumps and branches as a habitat for birds and animals in the foreground. (d) A two-year-old rehabilitated stand of grasses and *Acacia karroo* (sweet thorn) trees being sampled in 1989. (e) An eight-year-old rehabilitation site with *Acacia karroo* as dominant.

other reclamation sites where the soil is deficient in nitrogen is also recorded in North America (Chambers *et al.*, 1994) and Australia (Chapman *et al.*, 1996). The latter authors also suggest that to maintain species diversity it may be necessary to thin dominant legumes.

Heavy mineral dune mining in Australia

On north Stradbroke Island in Queensland, mining covered about 117 ha along the foredunes. A rehabilitation programme was based on using *Spinifex* grass, which has the ability to bind the foredunes and is a natural foredune pioneer. Brush matting, brush fences and cover crops were used to provide the initial stability while the *Spinifex* was becoming established. A large-scale tree planting programme was also initiated, and areas were fertilized during the rehabilitation (Brooks and Yeates, 1980). After frontal dunes were recreated and stabilized with *Spinifex* grass, *Pandanus* palms, oaks and *Banksia* trees were planted within the dune system. A number of depressions and dune slacks were created with a high water table so that these could be re-established as wetlands. This required hand-planting of sedges along the banks of the wetlands, twelve native species being used to colonize the depressions. Wetlands created in this way are quickly invaded by other indigenous species as they are visited by birds and other wildlife. Although it is somewhat more difficult to create a diverse community in this way, wetlands can readily be restored.

Rehabilitation of wetlands

Wetlands in the form of mangrove and freshwater swamps, pans or vleis and hygrophillous grasslands are an important component of the coastal environment, and it is important that wetland restoration is achieved satisfactorily. A number of studies have been done on the creation and restoration of ponds, marshes and other wetlands, especially in the USA (Charbreck, 1989; Holloway, 1994), and in Australia (Brooks, 1988, 1989). In the United States, extensive degradation and loss of wetlands has occurred in many of the states, with California and Ohio being those which have lost more than 90% of their wetlands (Cairns, 1993). As a result of the National Research Council Institute (1992) initiative, restoration of aquatic ecosystems, such as freshwater and salt marshes, in the USA has become a major priority. Kusler and Kentula (1989) describe the recreation of wetlands in the USA that have been destroyed by mining and various other processes and Brooks (1989) gives an account of the reconstruction of wetlands after dune mining in Australia. Where dune slacks or swales need to be rehabilitated, the clay layer may need to be reconstructed.

Rehabilitation in arid environments

In arid environments such as the south-west coast of Africa, rehabilitation presents many problems, especially when indigenous species are used (Lubke *et al.*, 1999b). Field trials to establish the best species and methods of rehabilitation were necessary in a phosphate mine site near Langebaan, north of Cape Town. The cultivation of the plants from seed and cuttings and their establishment in such adverse environments requires many innovative methods (van Eerden, pers. comm.) Studies where invasive alien plants species have been removed and replaced with native plants on the Northern Californian dunes required similar horticultural expertise (Pickart and Sawyer, 1998).

Introduction of terrestrial fauna, and soil fauna and flora

It is not usual to reintroduce the fauna into the area, as it returns naturally once the vegetation has become established. Many accounts have been given of the return of fauna to areas in Australia. Osborne and Walker (1989) recount how vertebrates have recolonized waste dumps. At Richards Bay, the return of the birds, ants, mammals and other vertebrates to the system has occurred naturally and has recently been described by Van Aarde *et al.* (1996). Scullion (1992) studied the return of macrofauna and microorganisms in the topsoil following surface coal mining in the UK. He found that aerobic bacteria remain present in stored topsoil in sufficient quantities to respond once the soil is re-instated. However, vesicular arbuscular mycorrhizal fungi and earthworms are lost from all but the surface layer of stored soil. Therefore, a proportion of inoculum of the surface layer of topsoil should be placed as strips across the disturbed sites to speed up recolonization by earthworms (Scullion *et al.*, 1988). Harris *et al.* (1987) suggested that the material from the surface of topsoil stores may also provide an inoculum for recolonization by the fungi. The importance of introducing a mycorrhizal inoculum in rehabilitation studies is emphasized in a number of recent studies (e.g. Corbett *et al.*, 1996).

Pilot Studies and Field Trials

In many cases of dune mining a pilot plant is established to extract the heavy minerals to determine the quality and quantity of minerals that may be extracted. (Fig. 4a). After the initial bulk sampling it is then possible to carry out field trials on the settlement ponds or tailing dumps (Lubke *et al.*, 1999a). In this case the tailings site was divided into three different areas and trials were carried out to determine the effectiveness of replacing topsoil, the desirability of using a brushwood mulch and on evaluation of the rate of establishment of the various species from seeds. A mixture of seeds of twenty-seven species of grasses, herbs and woody plants were used in the latter treatment (Fig. 4b). Although the results of this study are still being assessed other field trials (e.g. EEU, 1990, Lubke *et al.*, 1999b) have shown that the success of these studies can be used to determine what techniques shall be applied.

Costs of Rehabilitation

Many accounts of successful rehabilitation of dune mining sites are outlined above. The major limitation in any rehabilitation process is the amount of money made available by the mining company. There is a balance between relatively successful rehabilitation and a reasonable cash outlay to achieve a sustainable ecosystem. If insufficient funds and expertise are applied, rehabilitation may fail. Conversely, extensive funding does not ensure success unless the process is well planned and managed.

At Richards Bay, the cost of rehabilitating indigenous vegetation by Richards Bay Minerals in 1999 was in the order of R25,000 to R30,000 (US $4,000–5,000) per hectare. These costs include the spreading of topsoil, establishment of windbreaks, seeding and limited ongoing management (Fig. 3). Recontouring costs are regarded as part of mining costs (Camp, pers comm.) Similar costs have been incurred for roadside rehabilitation in South Africa. At the Chemfos site (Lubke *et al.*, 1999b), costs are approximately R12,000 (US $2,000) per hectare, including indigenous seed collecting, which helps to cross-subsidize the rehabilition

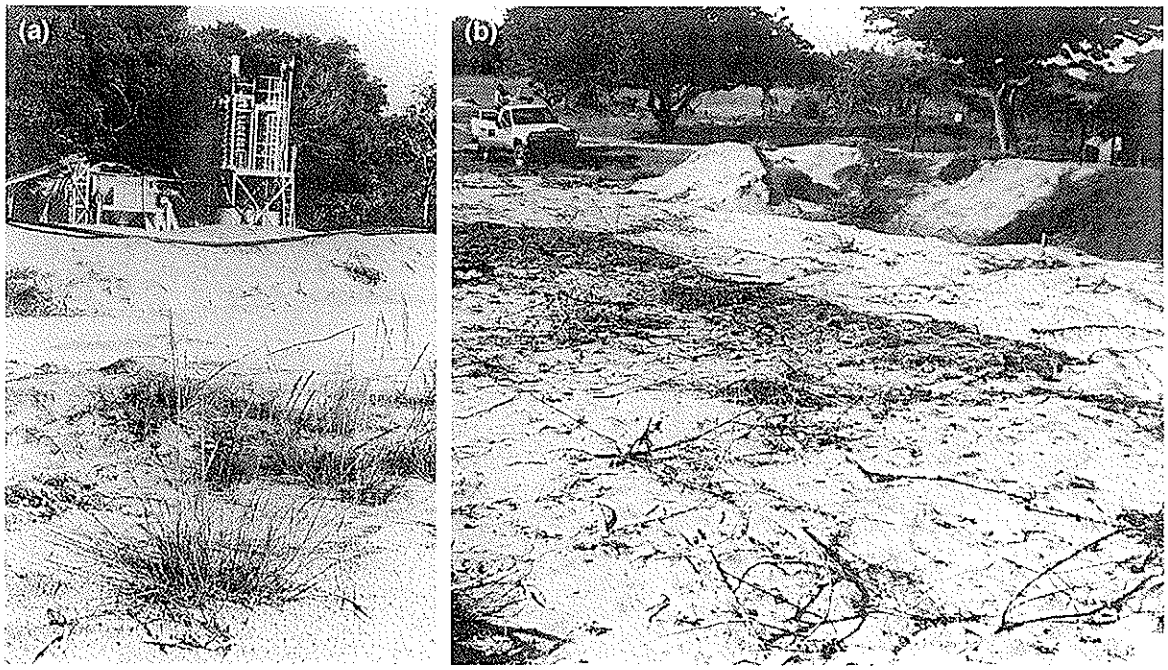

Fig. 4 (a) The pilot heavy mineral extraction plant established by Genbique at the Tigen project site in Mozambique. (b) Field trials established on the tailings produced by the pilot plant.

programme by sale of seed to other rehabilitation projects.

Measuring the Success of Rehabilitation

It is not simply the re-establishment of vegetation and the creation of an environment similar to the original that is a criterion for the success of rehabilitation. The initial objective must be to create a self-sustaining land surface which is stable against the erosive forces of water and wind, thus preventing localized environmental change (Bell, 1987). The longer term task, however, is to return the mined land to a condition suitable for other forms of land use. It may not be necessary to return the land to natural vegetation for wildlife conservation or recreation (rehabilitation), but to re-establish human settlements that were removed prior to mining. The alternative is replacement, the establishment of agricultural crops or forestry plantations for the benefit of the local population.

Whether a replacement or rehabilitation strategy is implemented, the goals or criteria for successful rehabilitation can be measured. Martinieck and Associates (1984) consider that rehabilitation can be assessed using the following criteria for creation of a "self-sustaining" land surface.

- At least three years of stability of land form.
- Stability of dunes should have resulted from the successful establishment of self-perpetuating vegetation or the management of a commercially viable crop.
- Perennial plant species should have survived for at least three summer seasons.
- The vegetation must have established a density of plants that allow for long-term stability of the restored land forms.

However, rehabilitation needs to be assessed in terms of the return of indigenous vegetation and plant and animal species. Ford and Langkamp (1987) estimated that a rehabilitation program can be considered successful if the landform is stable and resistant to environmental deterioration, and if the vegetation is self-generating.

What is evident from these studies is that monitoring is necessary to show changes in plant community type and structure. The rehabilitating dune area needs to be monitored over a long time period in order to establish the parameters that may be useful in determining whether or not rehabilitation has been successful. (Table 2). Thus, Westman (1991) uses various parameters from the rehabilitated environment to measure criteria of performance that can be used to determine whether the goals of rehabilitation have been achieved. These criteria should be measured using quantitative measures of the vegetation, and not simply qualitative statements, as is very often done in rehabilitation projects. For example, Yu Zuoyue (1995) in evaluating the rehabilitation success of tropical forests in China, assessed the soil fertility of the mixed forest which had been established, the ground water in the different vegetation types, the effects of temperature and humidity and the

TABLE 2

Some parameters which may be used to assess the success of rehabilitation (from Westman, 1991).

Ecosystem structure	Ecosystem function
Biota	*Biota*
Composition	Productivity/growth rates
Absolute and relative abundance	Nutrient flux (biota, litter)
Pattern of local and regional distribution	Pollutant flux
Gene Frequencies	Natality/mortality rates
Height	Migration
Density	
Biomass	*Physical Habitat*
Nutrient pools	Nutrient flux (air, water, soil)
	Soil movement
Physical Habitat	Radiation flux
Topographic features	Hydrological flow
Water quality	
Water quantity	*Social values*
Energy content (air, water soil temperature)	Recreational opportunity
Soil structure	Historical or archaeological interest
Soil/litter nutrient pools	Educational or scientific interest
	Aesthetic qualities
	Health and safety
	Psychological environment
	Consistency with government plans and policies

recovery of vegetation. The project was judged successful. Chambers *et al.* (1994) examined both soil and scrub/grass vegetation on spoil from phosphate mines in Idaho after rehabilitation. They found that after four years there was no difference in biomass and species composition between the spoil treatments and the control treatments on different soil types. However, after fourteen years seeded and mulched reclamation treatment plots had a better cover and biomass than other treatments and controls. Enumeration of a number of soil properties showed that the reclamation methods were responsible for soil differences and consequently the differences in plant species composition and biomass. By contrast, Partridge (1992) in measuring the recovery from sand mining in New Zealand, found that rehabilitation was not successful, mainly due to the introduction of alien species which smothered the natural vegetation, preventing the recovery of natural plant communities.

PERL (1990) used ecosystem attributes in assessing how well-constructed wetlands replaced the functions of natural wetlands. They listed 28 attributes which could be assessed, with the highest priority components being: hydrology (all aspects), elevation, organic matter of the soils, inorganic nitrogen in sediments and soil water, algal and higher plant cover, invertebrates, fishes and birds. Similarly, Aronson *et al.* (1993) discuss various vital attributes related to ecosystem structure which they believe should be measured for several successive years for evaluating stages of degradation and ecosystem rehabilitation or restoration.

Long-term environmental consequences of management must therefore be considered when analysing rehabilitation programmes. Dynamic indicators such as ecosystem resilience (the ability to recover from a disturbance) are important indicators of rehabilitation success over time (Berger, 1991). Berger presents a holistic and comprehensive evaluation system which uses multi-attribute decision analysis for appraising restoration and conservation projects. This model considers ecological, economic and social project impacts. He attempts to show what information is known about a project and how a decision is reached. Each project requires a post-project audit to provide parameters for the model, thus eliminating bias and providing a standardized system for making a decision as to whether a project has been successful or not. His model is fairly complex and would require the parameters to be determined for each particular system or project action.

A simpler way of determining the success of rehabilitation projects, although rather more difficult to measure, is that of Ewel (1990). He states it is necessary to determine whether the reconstituted community resembles the original, and whether it contains the same dominant species and a similar structural physiognomy. He provides six criteria against which the success of rehabilitation can be judged:

- *Sustainability*, which implies that the community must be capable of perpetuating itself without continuous management. This would mean assessing to see if the species are reproducing and whether the soil build-up is providing nutrients for the restored system.
- *Productivity*, which means that the system must show an increase in abundance or biomass over the growing period. This is measured in terms of biomass and the production of the plants or animals in the system.
- *Invasibility*, which means that there must be evidence of the invasion of new species into the newly formed

community. This is easily achieved by carrying out counts of the species in the new community.

- *Nutrient retention*, which implies that there must be a satisfactory build-up of a soil system and recycling of nutrients so that the nutrients are not simply lost from the system. This is difficult to measure as detailed analysis of the soil is required as well as information on nutrient recycling.
- *Biotic interaction* would complete the picture of restoration as invasion of later successional species in a fully integrated ecosystem should be the ultimate goal of the rehabilitation programme.
- *Restoration status*. Studying the interaction between species within the ecosystem would require detailed studies to see if it had reached the same level of interaction found within the natural ecosystem.

Long-term monitoring of rehabilitation is therefore necessary in order to establish if the process has been successful. Many environmental and biotic variables need to be measured. However, usually a balance can be established between measuring the strict ecological criteria and those which merely indicate that vegetation has been successfully established and the stable system has been achieved.

A Conceptual Rehabilitation Plan

The process of rehabilitation should be achieved in the shortest possible time period following mining. Information on the mine programme, the vegetation and other aspects of the site provide the basis for the establishment of a rehabilitation plan (Table 3). The plan should be flexible so as to allow for changes in the mining procedure, and indicate the potential for rehabilitation in the different areas where mining is to take place. The plan should build on the results of base-line studies of the vegetation and flora and of preliminary field trials. When there is great variation in the landscape and techniques to be applied, we have found that the use of GIS in mapping the region is most useful (Lubke *et al.*, 1999b). In this way the size of the landscape units, quantity of plant material and other methods can be summarized and mapped to provide predictions for future rehabilitation as sites become available (van Eerden, pers. comm.).

Discussion

Arising from the analysis of many studies on rehabilitation, a number of general recommendations can be formulated regarding dune mining rehabilitation:

- Reshaping and reconstruction of the dunes needs to be carried out prior to the establishment of vegetation. The creation of favourable environments are important components of the rehabilitation plan. This depends on the mine plan which should be worked out well in advance and planned in consultation with the rehabilitation consultant. Some dunes should be allowed to assume their own form and pattern before stabilizing with vegetation if this can be done without leading to extensive wind erosion or blow-outs.
- Top soil should be transported to rehabilitation sites as soon as possible after the sites have been prepared, because if stockpiled for any length of time, the soil loses nutrients, microfauna and flora and its potential for vegetation growth is significantly reduced.
- At a later stage, animals such as earthworms can be reintroduced to enhance the topsoil, but this may require more detailed study. Reintroductions or modifications of habitats for animals should be carried out in ways that do not infringe on the rehabilitation programme or destroy or damage vegetation.
- The rehabilitation process must be monitored closely, so that the process can be enhanced or corrected should die-back of vegetation occur. For example, alien weeds should be removed from the rehabilitation sites as they are potentially invasive in disturbed areas.
- Infrastructure for mining such as roads, buildings, etc should be sited away from rehabilitation areas and such mining activities planned so that they do not impact on rehabilitation sites.
- Mine visitors should be limited and controlled in rehabilitation sites so as not to cause an impact on these areas.

TABLE 3

Information to be included in a detailed rehabilitation plan (from Lubke *et al.*, 1998).

• Planning and mapping of the area for future land use
• The proposed mine routes, mine path and infrastructure need to be established early on so as to locate the various rehabilitation sites
• Planning of research studies required to assess the techniques and feasibility of the rehabilitation plan
• More detailed studies on the physical and chemical properties of the soils to determine the variability of the soils for the different vegetation types
• More detailed studies on the plant species which are to be used, e.g. seed production, seed harvesting and propagation of suitable species from cuttings
• Investigate the potential of various annual species to provide a cover crop
• More detailed studies on the physical and chemical properties of the soils to determine the variability of the soils for the different vegetation types
• Planning of field trials to assess the potential rehabilitation program. These should take place in the same area of the proposed mining. Trials would include looking at alternative techniques of rehabilitation
• Matching of rehabilitation techniques to the type of terrain, soil conditions, plants to be used. For example, the siting of correct plants in the wetland areas

- The return of the local communities to rehabilitated areas should be controlled and only allowed when rehabilitation has been satisfactorily achieved so as not to cause damage to these areas.
- Rehabilitation planning with the use of GIS is useful if carried out from the initial stages. A record of techniques, whether a success of failure, can be recorded and conceptualized for future application during the course of the mining process.

There are a number of recommendations which can be made regarding the development of a rehabilitation programme:

- Soils and vegetation types need to be surveyed to determine the desired community types that are to be established.
- The range of vegetation types in the area should not be completely destroyed. Pockets of plant community types should be maintained in adjacent areas to allow for the migration of species into the rehabilitation sites and improve species diversity.
- Top soil for any particular community should come from a similar community so that it contains the same types of seed and propagules.
- Only seeds collected from the area should be added to the topsoil so as not to result in the invasion of weeds and alien species.
- The seed mixtures should be balanced and not contain an excess of one species which could dominate the community.
- If a species becomes particularly dominant in the community, it should be selectively thinned out to allow other species to invade.
- Attempts should be made to transplant individual plants of important species from the native vegetation. Alternatively, seeds should be obtained from these plants to establish new populations in the new communities.
- Continual management of the communities for 5–10 years may be necessary to maintain these communities, and increase diversity as well as to prevent invasion by alien species.

Many of these suggestions may only become meaningful following practical study and field trials of rehabilitation in the area to be mined.

Summary

Studies on rehabilitation provide useful information on how a site can be restored and/or revegetated to provide useful and sustainable crops. Burton (1987) points out that there is an interaction between resource development, land use and environmental management which must all be taken into account in the mining process. It is this which one must strive for in formulating a conceptual rehabilitation plan.

Successful rehabilitation projects have been carried out following dune mining in many different countries, and there is no doubt that a successful rehabilitation program can be formulated and carried out in most areas where dune mining is proposed. It is however important to synthesize information contained in the relevant specialist studies in order to achieve successful rehabilitation. It should not be overlooked that natural ecosystems have evolved over an extensive period and contain a great diversity of species which form very complex interrelationships. When disturbed, the ground is invaded by colonizing species and these bring about natural changes through a series of successional stages, resulting in a stable community. Management of the ecosystems should strive to speed up this process and establish a viable and self-sustaining community.

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Adaptive restoration of sand-mined areas for biological conservation

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Summary

1. Adaptive management approaches to ecological restoration are current best practice. The usefulness of such an approach was tested in this study by implementing repeated experiments that examined restoration options for derelict sand mine sites dominated by *Imperata cylindrica*. Reclamation of degraded land that is dominated by *I. cylindrica* is a common problem throughout the tropics.

2. Initially, the hypothesized barrier to regeneration was limited seedling establishment because of *I. cylindrica* competition. After burning the grassland, woody weed control and planting of seedlings were implemented in factorial combination.

3. Seedling survival 28 months after planting averaged 26%, with < 1% of all seedlings establishing to a height > 1 m. The hypothesis that a transition barrier comprising solely biotic interactions restricted regeneration of native woody cover was rejected after seedlings and natural regeneration failed to thrive in this experiment.

4. A revised hypothesis, that the transition barrier comprised a combination of abiotic limitations (soil deficiencies) and biotic interactions (*Wallabia bicolor* browsing and *I. cylindrica* competition), was developed. A second experiment tested this hypothesis by removing *W. bicolor* (fencing), slashing the *I. cylindrica*, adding organic mulch and planting a mixture of native pioneer and secondary successional woody species in factorial combination.

5. Seedling survival was 61% in the second experiment and mulching significantly enhanced the survival and growth of all planted species. Planting alone reduced the regeneration of *I. cylindrica* after slashing. Native woody cover establishment was maximized by planting seedlings in mulched treatments.

6. *Synthesis and applications*. Taken together, these experiments support the hypothesis that there is a barrier restricting regeneration of native woody cover, and the barrier probably comprises both abiotic and biotic components. By adopting an adaptive management approach to the ecological restoration of sites, significant insights into their management requirements have been gained, supporting the current best practice restoration framework. Insights gained through monitoring and adaptation will be used to update the reserve plan of management, enhancing restoration of this severely degraded area and promoting connectivity of native woody cover within the conservation estate.

Key-words: *Imperata cylindrica*, mineral sands mining, organic amendment, restoration barriers, restoration ecology, restoration framework

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SOME CONSIDERATIONS IN PLANNING FOR MINE DECOMMISSIONING

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INTRODUCTION

Decommissioning is becoming an important consideration in the planning and operation of any mine and mineral processing plant in Australia today. However sometimes the particular requirements for decommissioning are considered an afterthought, to be addressed properly when the operation is effectively shut down. This paper considers the evolution of contemporary thinking on decommissioning requirements, some points for consideration in planning decommissioning, and areas where more work is needed.

Even today, many mine operators question the need to even consider decommissioning. After all, AMIC claims that mining takes up less than 0.02% of the land surface of Australia, and many mines and processing plants are located in relatively remote areas where any subsequent redevelopment is highly unlikely. At the same time however mining is promoted as a temporary land use, and there is an expectation that mined land will be left in such a way that subsequent uses are not compromised. Proper decommissioning of mines and infrastructure, and rehabilitation of disturbed land in an integral part of this requirement.

THE GOOD (BAD?) OLD DAYS

The development of decommissioning practices in Australia over the last 30 or 40 years parallels the rise of environmental awareness.

Before the Environment Boom

Prior to the green explosion in the early 1970s, there were few requirements placed on mining companies when they wanted to relinquish a lease, beyond a few conditions aimed mainly at safety issues. Thus there were requirements for shafts and adits to be capped or sealed, and open cut mines to be stabilised or made safe. The fate of buildings was often not a real issue, as the common practice was to relocate buildings from one site to another, or to conduct a great auction sale of buildings and equipment.

Thus we have the situation from the early 1900s that little remains of the township of Kuridala in north-west Queensland, because most of the buildings were taken to Mount Isa to form the nucleus of the new boom town. For example, the original hospital at Mount Isa was a building moved in its entirety from Kuridala. This practice of course continues to this day. I am sure that, on inquiring at many current mining operations around Australia, especially gold mines, the origin of some of the equipment would be found to be another operation.

In general, there was no consideration of the existing or future environmental impact as a result of abandoning or closing the operation. It must be remembered also that these early mining operations were usually much smaller than those of today, and thus their impacts may be fewer and smaller. For example, in all the controversy over the proposal to develop a new mine at the former mining location of Coronation Hill in the Northern Territory, there was little or no discussion of the scars on the hillsides from the mining in the 1950s. There are exceptions to this however, and the former mining and processing operations at Brukunga in South Australia, Captain's Flat in New South Wales, and Rum Jungle in the Northern Territory immediately spring to mind. The controversy over the impacts of these operations in fact led to the development of many of the standards and practices which we apply today in decommissioning and rehabilitating mine sites.

This former practice of simply abandoning a site once its commercial value had been exploited was not unique to the mining industry. There have been numerous examples of manufacturing plants outliving their value and being closed without rehabilitation. the difference here is that usually such plants were located close to or within expanding metropolitan areas and were quickly recycled for other industrial operations or residential housing. This practice has, nevertheless, led to further environmental problems, and these will be touched upon in a subsequent section.

The late 1970s and early 1980s

With the advent of greater public awareness of the environment, and the growth of the conservation movement, there was considerably more focus on the long-term effects of mining and mineral processing operations on the environment. The introduction of the various systems of environmental impact assessment focussed attention on the end use of mined land. As a counter to often outlandish claims made about the potential environmental impact of mines, and to ensure continuing access to land for exploration, the Australian mining industry began promoting the concept of mining as a temporary land use. This in turn resulted in a concentration on rehabilitation of the minesite and considerable research effort into the process and practice of mine rehabilitation.

It was rare to find an EIS produced in this era which did not, in at least general terms, discuss the end use of the site and rehabilitation practices to be employed. However, as pointed out by Buckley (1991), subsequent practices at many of these mines bore no relation to what was included in the EIS. Also, partly because the science was relatively young, some mistakes were made. For example, the EIS for the Woodlawn mine, prepared in the late 1960s, pointed out that there would be no acid drainage from the waste rock dump because the pyrite content of the ore and waste was low. Within two years of the opening of the mine, acid drainage was clearly evident from the waste rock dump, and the mine had to spend considerable time and money controlling this problem. This issue resulted partly because the geologists had a different concept of what 'low in pyrite' meant compared to environmental scientists, and partly because at that time other factors which could influence the acid generating capacity of the rock were not realised.

At the same time, regulatory authorities began to make existing mines more accountable. They did this by reviewing lease conditions to ensure that rehabilitation requirements were included. While many mines already had comprehensive rehabilitation programs, the aim of these changes was apparently to make all mines aware of developing community standards, and perhaps to ensure that the public statements of industry spokesmen were reflected in actual practice.

Notice that the emphasis to that time was on rehabilitation. This usually meant that the focus was on land disturbed by mining, or waste rock dumps and tailings dams. There was frequently little reference to other parts of a mine, particularly the processing plant and support buildings.

The attention of the regulatory authorities, and especially their governments, also turned to abandoned mining areas. Considerable sums of money were committed and ultimately spent on cleaning up so-called environmental legacies of past mining. Again, Captain's Flat and Rum Jungle immediately spring to mind. The aim of both these exercises was to reduce or eliminate a definite environmental issue, coincidentally in each case the pollution of a river system by acid waters and dissolved heavy metals.

To this time, environmental control and decommissioning requirements of mining operations usually remained with the government department responsible for regulating the industry, that is the Department of Mineral Resources or its equivalent.

CONTEMPORARY REQUIREMENTS AND PRACTICES

In the mid to late 1980s we saw a move towards expanding the environmental regulatory responsibilities of mines' departments, accompanied by reviews of Mining Acts around the country. These reviews invariably enhanced the control systems imposed by the Departments, and a variety of management processes with acronyms such as MREMP and EMOS were developed. At the same time,

departments or statutory authorities charged with environmental protection functions, such as the Environment Protection Authority, also obtained clearer controls over mining and mineral processing operations.

Legislative Requirements

It is not the purpose of this paper to cover in any detail the various legislative mechanisms in Australian states which control the operation and decommissioning of mines and processing facilities. The mechanisms currently in place in New South Wales, which are similar to those in most other states, will be outlined in another paper at this Workshop. However, for the sake of completeness, a brief review will be presented here.

Most mining operations have in recent years had their lease conditions reviewed and updated to reflect modern attitudes towards environmental management. These conditions almost invariably now include requirements to consider decommissioning strategies and final land uses during the life of the operation. Frequently these issues must be specifically addressed in Environmental Management Plans, or their equivalents. In addition, these conditions now usually require the operation to be conducted in accordance with other environmental legislation, such as an Environmental Protection Act. Operations must obtain licences to emit air, water and solid wastes, and demonstrate an on-going commitment to reduce these waste streams.

The recent concerns about contaminated land, resulting in the passage of land contamination legislation, also impact on the operation and decommissioning of mining and mineral processing operations (see later). It is almost certain that all tailings dams and many waste rock dumps in Queensland will be classified as contaminated land.

Operators must now be aware of all these requirements when planning for decommissioning of a mine or plant. The ultimate incentive for proper decommissioning and rehabilitation is the now common requirement for operators to lodge security deposits or bonds which are meant to cover the potential costs should the operator default, or not meet appropriate criteria for lease relinquishment.

Guidelines and Codes of Practice

The industry too has not been idle in developing its own guidelines and codes of practice for good environmental management, including decommissioning and rehabilitation. The fact that this paper is being presented at the Eighteenth annual AMIC Environmental Workshop is evidence of this. Chambers of Mines around Australia have been active in developing and promoting environmental policy statements as well as various specific codes and guidelines.

My experience though has been that, while these documents may not be overly prescriptive, they are not necessarily widely known among smaller operators in the industry. The mining industry must be able to demonstrate a considerable degree of self-regulation before its policies and documents are fully accepted as potential alternatives to regulatory proscription.

PLANNING FOR DECOMMISSIONING OR SHUT-DOWN

Good decommissioning of a mine does not just happen - it requires considerable planning to succeed. Some practical aspects of this planning are outlined below.

When Should You Start Planning, and Why?

The timing of decommissioning planning is often regarded as a vexed question by personnel closely associated with an operation. Many say that this planning is best done very close to shutdown, and others say within the last few years of operational life. A compromise view often promoted is that the timing of this planning is dependent on the life of the project - for a short project (ie. less than 5-10 years) planning should begin at or near the commencement of the mine, while for a long project it should begin about 5 years before the end.

In contrast to these views, I believe that planning for decommissioning of a mine or processing facility should begin at the project planning stage. This may appear to be a conundrum - that shutdown planning, at least in conceptual terms, should begin even before the project has commenced operations. My reasons for this view are many, and some of these will be outlined below.

Many long-term aspects of a mine, and to a lesser extent of a processing plant, are decided upon during the project planning phase. These may be the location of key parts of the operation such as waste dumps and tailings dams, or construction techniques such as whether to include a toe drain or not in a dump. Proper planning of a site, with a view to considering possible end uses and future rehabilitation needs may save considerable expenditure in the future.

For example, conventional planning generally aims to minimise haul distances for waste rock, while tailings dams may be located in a suitable valley. This may mean that these two facilities are some kilometres apart. But what if the only suitable cover material for the tailings is the waste rock? The waste will have to be re-mined and carted considerable distance at or neat the end of line life, not a cheap exercise. Similarly, planning a single tailings dam which will not be completed until the end of mine processing means that there can be no progressive rehabilitation, usually a far cheaper alternative. At Pasminco's Elura mine near Cobar a rock quarry for underground fill was located adjacent to the existing tailings dam, so that it was used for tailings when the dam was filled. While logical for

operational purposes, this has also meant that the area for final rehabilitation is lessened.

Failure to characterise waste material has often resulted in future environmental problems at a mine, as shown by the Woodlawn example above. Similarly, even the handling techniques for waste material should be considered early in the planning phase. For example, the special advantages of separately storing topsoil were not recognised at many mines, with the result that the topsoil has ended up at the bottom of stockpiles where its value is destroyed. Characterisation of the various overburden and interburden materials at Queensland coal mines in the early 1980s resulted in selective stripping and placement such that the revegetation success was enhanced (Kelly and Kelly, 1987).

Environmental planning can often reduce future problems at processing plants. While initially expensive, complete paving of a plant area can reduce the potential for spills to cause land contamination which might lead to a large clean-up bill at the closure of the plant. Strategic location of facilities can reduce duplication and solve future problems. For example, the quite fortuitous siting of a process water supply dam downstream of the processing plant, main waste rock dump and tailings dam at Pine Creek Goldfields has resulted in the water dam becoming an integral part of the site's water management system (Matthews, 1988), and it will continue to act as a collector for any contaminated seepage from the waste rock and tailings after mine decommissioning.

What Should You Consider?

Many of the factors which should be considered in mine and mineral processing plant decommissioning have been mentioned in the previous section. These include:

- the siting of major facilities;
- the interrelationship of these facilities and environmental protection measures;
- the proposed end use of the land;
- characterisation of waste materials to identify potential environmental problems;
- identification and characterisation of disused waste dumps, particularly those for chemicals and other potentially polluting substances;
- the location of any waste materials removed from the site;

The aim of a decommissioning plan should be to minimise operational and post-operational environmental problems. As well as laying out a logical plan for demolition of facilities and infrastructure, and rehabilitation of the site to the agreed

final land use, it should also identify all potential sources of on-going environmental contamination and how they are to be controlled or eliminated.

Strategic Planning

There are many advantages in setting out clearly what the objectives are in decommissioning a site (ie. a **strategy**), and then in preparing a detailed prescription for how the decommissioning is to take place (ie. a **plan**). The preparation of a strategic plan allows an operator to focus on what the issues might be in decommissioning, and then to obtain the approval of the relevant regulatory authorities for the plan. The establishment of a plan some time before decommissioning allows problems which might develop to be circumvented (as discussed above), as well permitting the operator to schedule decommissioning activity into the normal production process of the site.

It must be stressed that such a plan, in particular, must be dynamic. This is to ensure that changing circumstances can be easily incorporated into the plan. One way of doing this is to conduct a review of the strategy and plan on a regular basis, perhaps in conjunction with preparation of an annual environmental management plan.

SPECIAL CONSIDERATIONS

As well as these standard requirements for good practice in decommissioning, there are some special considerations which now should be included when developing strategies and plans.

Heritage and Mining History

When we usually hear of heritage in relation to a mining operation, the most frequent context is in relation to aboriginal or european heritage sites which may be placed at risk by mining, and therefore which need to be protected. However there is another context which has special meaning for decommissioning. The Institution of Engineers has, in recent years, become more vocal about preserving our industrial engineering heritage for future generations. This is because many fine examples of this heritage have been destroyed in the name of progress. The same could be said for mining operations which are totally rehabilitated.

In many parts of Australia our mining heritage has become a tourist attraction. Consider Sovereign Hill, adjacent to the site of the 1989 Environmental Workshop, and towns such as Burra in South Australia and Ravensworth in Queensland. people today are interested in their heritage and want to understand what happened in the past. But most of these attractions are the results of mining carried out in the nineteenth and early twentieth centuries. What about the more recent mining? When I last visited Captain's Flat, I was interested to know what the

various large concrete structures were used for during the operation. However there was nothing there to tell me. This is the case at almost every recently rehabilitated minesite in Australia.

We must ensure that, while decommissioning minesites to contemporary standards, the future heritage value of the site is also recognised. I was interested to review the Guidelines for Rehabilitation and Environmental Management Plans for Broken Hill Mines recently (Poseidon Mining Investments, 1992). The Guidelines specifically state that the planning of rehabilitation and of any subsequent land-use at Broken Hill will need to recognise and take into account the tourist, educational, scientific and historical significance of Broken Hill's geological features and mining structures.

Complete removal of all infrastructure in an attempt to return a mine to a greenfield site, such as at Goldsworthy, could be regarded as an attempt to eliminate a valuable part of our heritage. One compromise in this could be to ensure that the site and the operations carried out on it are fully documented, as I believe is happening at Goldsworthy. This will at least allow the heritage value to be documented at a future date, and also provide good records should any problems arise at the site in the future (see next section).

Interestingly enough, heritage requirements have, in some instances, impeded mining companies in their desire to rehabilitate a former operational site. The saga of the Lambton Colliery just south of Newcastle is a case in point (Davies, 1993). FAI Mining, the last of many owners of the colliery, ceased operations at the end of 1991. Standard conditions on the mining lease required rehabilitation of the mining area and removal of buildings, machinery, plant equipment, constructions and works as may be directed by the Minister. Approval to close the Lambton Colliery was granted conditional on the submission of a program for rehabilitation and restoration of the site and completion of the program within 6 months. However, when the company applied to Lake Macquarie Council for approval to demolish the buildings and structures, the Hunter Regional Environmental Plan 1989 (Heritage), in which the Lambton Colliery was listed as an item of local heritage significance, was invoked and the matter became the subject of an extended case in the Land and Environment Court. While ultimately the company was given permission to demolish the site, the case is a warning to mining companies that the heritage value of a site may be a consideration at both the commencement and completion of operations.

Land Contamination

Many states of Australia have recently enacted, or about to enact, legislation requiring the identification and possible clean-up of contaminated land. Contaminated land is identified by either direct investigation or by reference to a standard list

of operations which may lead to contamination. Mining and mineral processing operations are usually included on these lists. This means that, as well as acknowledging the requirement to rehabilitate a minesite, a company now must consider the potential for the site to be contaminated and recognise that there may be some liability associated with this. With the increasing popularity of environmental audits of minesites and mineral processing operations the potential for contamination arising from the operations is becoming well recognised. However less well recognised is the possibility for waste rock dumps and tailings dams to also be regarded as contaminated land.

Yet to be resolved with any certainty is the issue of who carries the liability for such land once a mining lease is relinquished. A mining company could receive a nasty shock if it is found to be liable for contamination some time after it has relinquished title to land.

UNRESOLVED PROBLEM AREAS

Despite the great advances in environmental and rehabilitation science over the last 20 or so years in Australia, a number of problem areas still remain. These potentially have great impact on the issue of when is decommissioning and rehabilitation of a site completed, and therefore when can security deposits be recovered and leases relinquished. The major issues facing the mining industry are discussed below.

Rehabilitation Completion Criteria

I believe the greatest issue in rehabilitation yet to be resolved to the satisfaction of all parties is that of completion criteria. For the purposes of this discussion I define completion criteria as *'those measures, either qualitative or quantitative, against which the success or otherwise of rehabilitation can be measured, so that the objectives of the decommissioning plan can be achieved and the lease relinquished'.*

The Queensland Department of Minerals and Energy has recently prepared draft technical guidelines for environmental management for mining in Queensland. One of these guidelines covers site decommissioning. This guideline states that the objective of decommissioning a minesite should be:

To ensure that areas which have been mined and/or affected by mining operations are left:

(a) tidy and safe, with public safety risks reduced to acceptable levels;
(b) stable and resistant to erosion;
(c) suitable for the agreed post-mining land use;
(d) at or better than the agreed levels of contamination; and
(e) within agreed waste discharge levels.

Other states already have, or are moving towards, similar objectives for decommissioning.

Completion criteria are most frequently invoked when the effectiveness of revegetation needs to be assessed. For areas which are being rehabilitated to agricultural or pastoral purposes, they are typically couched in terms of productivity. For areas being returned to natural vegetation, the criteria frequently include measurements of species richness and diversity, or vegetation structural analysis.

There are two problems associated with the use of completion criteria. The first is the question of whether the particular criteria used are appropriate for assessing the rehabilitation. The second is whether the criteria are relevant to the desired end use and community. For example, if the desired end product is a mature eucalypt forest which may take over 100 years to develop, how can we assess 10-year old rehabilitation to satisfy all interested parties that the end product will be achieved without waiting the 100 years.

Until these questions are resolved, there will always be difficulties in applying completion criteria to rehabilitation.

The Ultimate Void

Open cut mining commonly leaves an open pit (final or ultimate void) remaining at the end of operations. While the volume of the void can be minimised by careful mine planning, this is frequently not done and thus often quite large voids can result. Such voids represent a potential danger to people and livestock, as well as a potential source of environmental pollution resulting from water accumulation. The environmentally preferred rehabilitation method is backfilling to create at least a self-draining surface, if re-formation of original contours is not feasible. This option is frequently not economically feasible and a satisfactory alternative must be sought.

Alternative uses sometimes considered for voids include:

- water storage;
- development as a wetland/wildlife habitat; or
- waste disposal.

There are excellent examples of innovative rehabilitation of mine voids, such as the RGC Wetland Project at Capel in Western Australia (Brooks, 1992). However, in many cases the void is simply made as safe as possible and abandoned.

The potential for water contamination, especially of groundwater, still exists at some abandoned mine voids.

Waste Dumps

Because the primary aim of any mining operation is to maximise the amount of ore mined, the characterisation, mining and placement of waste rock can sometimes receive less attention than is required. This has resulted in unanticipated environmental problems arising at mines during (eg. at Woodlawn) and after mining. As with all other aspects of the mining operation, proper planning of waste dumps can minimise or even eliminate such problems in the future.

The disposal of waste materials from smelters has also often led to environmental issues. Smelter slags have been used for landfill, and as a material for reclamation of swampy areas. Because these slags contain residual levels of heavy metals, they are often targeted in land contamination issues. The problems that have arisen because of such used of slags from the Pasminco smelters at Port Pirie and Boolaroo are typical examples of this. Other smelter waste materials such as boghouse dusts, which again can contain relatively high levels of metals, can also lead to problems.

Particular issues to be considered when planning for decommissioning include the siting of waste dumps, the possible future use of the material either as backfill or cover for tailings dams, construction of dumps to minimise infiltration of water, the potential for hazardous leachates to develop, and the long-term stability of the dump. Many of these issues are still to be resolved.

Tailing Dams

Tailings dams or dumps are effectively a special subset of the waste dump situation. Because tailings are usually very finely ground and contain residual amounts of chemicals used in the processing operation, they can be an important source of environmental contamination. Tailings can also be a potential source of acid drainage. Because of their physical and chemical nature, some tailings may be particularly difficult to revegetate.

While revegetation of more benign tailings has sometimes been successful, at least in the short term, the long term stability of this vegetation has not yet been demonstrated. More hazardous tailings materials have been rehabilitated by isolating the vegetation from the tailings by means of impermeable clay cappings and capillary breaking layers. The chemical processes taking place within these tailings are not clearly known and may lead to environmental problems in the future.

Acid Mine Drainage/Acid Rock Drainage

The production of acidic water within abandoned mines, waste rock dumps and tailings dams as a result of oxidation of pyritic material is now commonly referred to as Acid Mine Drainage (AMD)

and Acid Rock Drainage (ARD). Many environmental scientists in the mining industry regard the production of AMD/ARD as the greatest environmental issue yet to be solved.

Techniques now exist to enable the characterisation of various materials with regard to their acid producing potential, and many innovative techniques are now being introduced at minesites to control AMD/ARD. However we still do not fully understand the exact processes which can take place within a dump or dam, and how to minimise or eliminate the generation of ARD/AMD in the long term.

CONCLUSIONS

Decommissioning of mines and mineral processing facilities should now be an integral part of environmental management of these operations. In most cases it should be a consideration even from the early stages of the project planning. The development and regular review of dynamic decommissioning plans and strategies will enable potential issues to be recognised, and action plans to be developed to address them. By doing this, many potential environmental issues can be avoided or eliminated. There can also be large cost advantages to a mining company in pre-planning of waste disposal.

Recent events have shown that two areas must now receive special consideration - those of heritage value and land contamination. Although many advances in knowledge and practice for decommissioning have been made over the last 20 years or so, not all the answers have yet been obtained. In particular, completion criteria, ultimate voids, waste dumps, tailing dams and acid mine drainage are identified as areas requiring considerably more investigation.

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Second Edition

Rehabilitating Damaged Ecosystems

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CHAPTER 1

Restoration Ecology: Protecting Our National and Global Life Support Systems

John Cairns, Jr.

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I. INTRODUCTION

Two factors will markedly affect the future of human society: (1) the burgeoning human population presently increasing at nearly 100 million per year and (2) the unprecedented rate of ecological destruction by a single species, *Homo sapiens*. Society has three choices in its relationship with global ecosystems: (1) continue the present rate of destruction until the inevitable collapse occurs, (2) establish a no-net-ecosystem-loss policy so that each ecosystem degraded or destroyed accidentally or deliberately is replaced in kind, and (3) use the no-net-loss policy as an interim phase and establish a

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policy that the rate of ecosystem repair exceeds the rate of destruction so that per capita ecosystem services will not diminish precipitously as the global human population grows to 10 billion in the next century. If the human population continues to grow at its present rate, ecologically repaired ecosystems will probably not survive because of the pressures of the increased human population on natural systems. Even so, even temporary repair would improve things for the present inhabitants of the planet. However, if the goal is to have the largest number of people on earth possible for many centuries into the future (that is, to look at the number of humans possible not only at the present moment but possible over a long period of time), then both stabilizing the human population or reducing it gradually with a concomitant strategy for ecological restoration and diminished ecological destruction make sense. The spotted owl controversy has exacerbated the polarization between those who wish to protect the environment and those who wish to protect jobs. This and many other similar controversies have produced some bumper stickers in such poor taste that it is a service not to repeat them. Regrettably, ecological restoration and environmental protection are inseparable because restoration will be less necessary if the rate of destruction is markedly altered, and restoration itself, however well funded, will be only temporary if environmental protection is inadequate. The root of the problem seems to be that some segments of human society are sufficiently environmentally illiterate to believe that humans can exist in totally artificial technological life support systems on this and other planets, with a few selected domesticated species from Earth. Ecologists have not been notably helpful in correcting this misapprehension because their ecosystems of choice for research are generally far from anthropogenic effects or where anthropogenic effects are carefully controlled, such as the Galapagos Islands. Before the era of acid rain, a thinning ozone layer, and atmospheric transport of toxicants, the term "pristine ecosystems" would have been used to describe these areas; however, it now seems unlikely that any area of Earth is unaffected by the activities of human society. In this case, ecologists who think they are studying ecosystems unstressed by anthropogenic effects are either deluded or are basing judgments on inadequate information. There is a regrettable tendency to characterize environmentalists as anti-people/anti-jobs. Persons who favor jobs are labeled self-serving, insensitive slobs who would destroy nature for material gain. This polarization, with regard to environmental or ecological decisions, cannot continue in human society, despite the efforts of many politicians and other interest groups to polarize these issues. None but the most extreme environmentalists wish people to be without jobs or unable to feed their families; and, all but the most far out "jobs first" people wish their children and grandchildren to enjoy natural systems.

Individuals should be able to favor environmental protection and restoration without being callous to the needs of the average citizen for a paying job and financial security. Similarly, those preoccupied with earning a living

should not be viewed as anti-environmental, although regular attempts are made by both groups to demonize persons who describe themselves as environmentalists or those who describe themselves as industrialists. Actually, most individuals in both groups are probably interested in the amenities and security which accompany a position with adequate financial rewards and simultaneously wish a good environment in which to spend their leisure time. The term "environmentalist" should not denote someone who puts other organisms ahead of humans, and those wishing the members of human society to enjoy good health and diet should not be viewed automatically as against natural systems. Quinn[1] feels that the word "environment" or "environmentalist" should not be used because of the connotations just described.

Cairns et al.[2] have developed a suite of indicators to monitor regional ecosystem health, similar in concept to management use of "leading economic indicators." One of the major thrusts in this approach is the high probability that desirable properties of indicators of environmental health vary with their specific management use. Different indicators are required when collecting data to assess the adequacy of the environment, monitor trends over time, provide early warning of environmental degradation, or diagnose the cause of an existing problem. Trade offs between desirable characteristics, costs, and the quality of information are inevitable when choosing indicators for management use. Decisions about what information to collect for which purpose can be made more rationally when available indicators are characterized and matched to management goals. If one finds these assumptions palatable, then it is not particularly profitable to attempt to find a single, all-purpose indicator or to try to extrapolate from a structural attribute to a functional attribute, but rather to select the suite of attributes and methods necessary for each particular decision.

II. DEFINING ECOLOGICAL RESTORATION

Since *Rehabilitating Damaged Ecosystems*[3] was first published, much has happened in the field of restoration ecology. Events such as the Exxon *Valdez* oil spill in Prince William Sound, AL, have called attention to both the need for restoration and the uncertainties about both natural recovery and human assisted recovery. The symposium on biodiversity,[4] jointly sponsored by the National Academy of Sciences and the Smithsonian Institution, documented the dramatic global loss of species and the habitat necessary for their continued existence. Cairns[5] indicated in that volume that ecological restoration, together with preservation of remaining habitat, would be a means of arresting this frightening rate of species extinction. Additionally, the term "ecosystem services" — that is, the activities of ecosystems that benefit humans, such as regulating atmospheric gas balance, improving water quality, providing genetic material for developing new crops in a period of global climate change,

developing pest resistance, and the like — is receiving increasing attention. A more detailed description of ecosystem services has been beautifully furnished by Ehrlich and Ehrlich,[6] and readers are referred to that book for a fuller understanding of this vital point. Again, the rapidly developing field of restoration ecology has the potential, if vigorously utilized, of maintaining these ecosystem services.

Ecosystem services may well be the most persuasive justification to the general public for ecological restoration. The importance of the ecological life support system is now becoming more and more apparent, although almost certainly at too slow a rate. The basic assumption, however (namely, that natural systems can provide services less expensively and more reliably than technological systems), is finally reaching the consciousness of the general public and even some of the world's leaders. Created wetlands have now been shown in many instances to be far less costly and as, or more, efficient than technological systems for treating a variety of wastes of anthropogenic origin. It is difficult to visualize a technological system that would regulate the atmospheric gas balance as effectively, inexpensively, and reliably as natural ecosystems. There is an inextricable link between human population growth, ecosystem services, and ecological restoration. Global human society has five options with regard to these crucial relationships.

1. Continue the present rate of ecological destruction and human population growth until some catastrophic event causes a re-examination of present policies — The joint statement by the Officers of the Royal Society of London and the U.S. National Academy of Sciences[7] *Population Growth, Resource Consumption, and a Sustainable World* is not sanguine about control of human population growth in the near future. They feel the rapid rise from 5.4 billion in 1991 to 10 billion in 2050 *may be unavoidable*. If the present rate of ecological destruction continues with no significant counteracting ecological restoration, ecosystem services per capita will dramatically, and possibly irreversibly, decline. Although the field of restoration ecology is in its infancy, there is already abundant evidence that the rate of ecological restoration and recovery lags far behind the rate of human population increase. To continue on the present course does not seem prudent.

2. Replace damaged ecosystems with fully functional restored ecosystems at a rate resulting in no-net-loss of ecosystem services — i.e., rate of destruction offset by rate of repair — This would still result in a reduction in ecosystem services per capita if, as seems highly probable, the world's population continues to expand dramatically. There are other serious problems as well. The relationship between global biodiversity and ecosystem services is not well understood. However, a growing body of evidence indicates that organisms placed in loosely defined functional groups do not, in fact, function

similarly (e.g., Nelson Hairston, Jr., Douglas Distinguished Lecture at Rocky Mountain Biological Laboratory, 1992). The no-net-loss approach, however, would provide more time to generate the needed scientific information before funding for large-scale restoration is made available. It is, however, worth emphasizing that robust information on these complex issues will not be generated quickly.

3. Restoring damaged ecosystems at a rate that will produce a net gain of ecosystem services globally — This would still not increase human services per capita in the face of a dramatic increase in human population, but the loss of ecosystem services per capita would be substantively lower than in relationships 1 and 2.

4. Control human population growth to match the rate of ecological restoration so that ecosystem services per capita remain constant — This would, of course, require draconian measures in human society globally. At the present time, such attitudinal changes seem highly improbable but so, a few years back, did the events that have occurred in what was formerly the USSR and the eastern block of communist countries. Even this monumental effort might not dramatically alter the ecosystem services per capita from their present level. However, attitude changes would buy more time for a number of restoration efforts to provide useful scientific information and, at the same time, since human population cannot continue to grow at its present rate forever, provide some evidence of paths toward stabilization of human population least disruptive to human society.

5. Gradually reduce human population size, reduce the rate of ecological destruction, and increase the rate of ecological healing so that sustained long-term use is enhanced — This would, of course, require even more draconian measures. However, if the total number of humans the earth is capable of supporting is to be maximized, the length of time that this is possible should be a major factor. Therefore, reducing human population numbers in the short term will probably increase the total number of humans inhabiting the earth for the entire existence of the species.

III. RESTORING AQUATIC ECOSYSTEMS

The most important event for me in restoration ecology since the first edition of *Rehabilitating Damaged Ecosystems* was chairing the National Research Council (NRC; the operating arm of the National Academy of Sciences and the National Academy of Engineering) committee that produced *Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy*.[8]

The definition of "restoration" agreed upon by that committee was "the return of an ecosystem to a close approximation of its condition prior to disturbance."[9] This definition, although prepared for aquatic ecosystems, applies equally well to other types of ecosystems. The committee agreed that it would be extraordinarily difficult to restore ecosystems to their precise predisturbance condition since:

1. Each ecosystem is the result of a sequence of climatic, biological, and other events unlikely to be repeated in precisely the same way
2. Commonly, the structural and functional attributes of the ecosystem before disturbance are not precisely documented
3. The species that once inhabited the ecosystem may no longer be available or, if available from other ecosystems, may not be physiologically identical with the race, subspecies, etc. formerly inhabiting the damaged ecosystem

The NRC publication[8] espouses viewing all restoration in a landscape context and recommends that restoration include integration of the restored patch into the larger ecological landscape of which it is a part. Thus, restoration as an ideal model would consist of replicating the structural and functional attributes of the damaged ecosystem at the time the ecosystem was damaged — this might be viewed as resetting the ecological clock to the time before damage occurred. Under this definition, one would assume that the subsequent successional processes would be essentially those that would have occurred had the ecosystem not been damaged. This view is probably tenable as an ideal model if the ecosystem has only been recently damaged and restoration is fairly rapid and effective. It would be especially defensible if recolonization from the larger ecological landscape surrounding the damaged patch was likely to be effective. However, ecosystems are dynamic rather than static, and, if the larger ecological landscape of which the damaged ecosystem is a part has undergone substantive change since the damage occurred, it might be more appropriate to restore the damaged portion to the state it would have been in *at the present time* had no damage occurred. This might be markedly different from the predisturbance condition. Estimating the ecosystem condition if normal successional processes had occurred (and no damage) is even more difficult than determining predisturbance condition.

IV. REHABILITATION/RESTORATION

The first edition of *Rehabilitating Damaged Ecosystems* did not use the term "restoration" in the title because, at that time, it was difficult to find an example of full ecological restoration. (Regrettably, the NRC committee also failed to find robust examples of full ecological restoration in 1992, possibly because sufficient time had not elapsed since the restoration efforts began.) As

a consequence, the title *Rehabilitating Damaged Ecosystems* was used to emphasize that only selected attributes had been replaced in the majority of cases in the existing literature. The term "rehabilitation" used in this context appeared first at a symposium sponsored by the Ecological Society of America (ESA) well over a decade ago.[10] Since the symposium was held in conjunction with the annual meeting of the ESA, was well attended, and was free of opposition to the definitions provided by Magnuson and his colleagues, it seemed appropriate to use the term in the title of the first edition of *Rehabilitating Damaged Ecosystems* to emphasize that restoration was an aspiration and that rehabilitation was an achievable goal.

V. DOCUMENTING SUCCESS OF ECOLOGICAL RESTORATION PROJECTS

Ecological health, as is the case for human health, is an on-going process. Therefore, while each restoration effort will have some unique components, some criteria could be commonly, perhaps universally, applied. A broad criterion would be how closely the repair of the damaged ecosystem has brought it to the predisturbance condition in both structural and functional attributes. An additional criterion would be whether the resiliency (ability to bounce back after ecological harm or displacement) has been maintained. Since the objective is to keep the ecosystem in robust condition, this will necessarily be measured indirectly as it is for human health. Above all, the criteria for success and failure should be established before the restoration project begins, although experience during the process itself might lead to modification. In all cases, both the selection of criteria for success and failure and the modifications thereof when necessary should be subjected to a rigorous peer-review process.

Finally, while there should be evidence that meets the highest scientific and engineering standards to evaluate success or failure, the general public should be constantly informed. The criteria for success and failure, the project scale and schedule, the costs and benefits, and other factors (including enhanced societal values) should be thoroughly and carefully explained to all citizens in a jargon-free, intelligible format. This should be an ongoing process. Once all citizens of the region understand and approve the process and have achieved the necessary level of environmental literacy about this particular program and have approved the objectives and goals, periodic reports should be issued to keep all informed. A citizens advisory committee that would be fully informed about the project at all times could then gain the public trust in such a way that communication with the general public is simplified. All of this will be somewhat expensive in time and money, but the problems caused by lack of information or misinformation will almost certainly be far more costly.

VI. PAYING FOR RESTORATION

Ultimately, of course, the general public always pays for everything either as part of the production cost or in taxes that pay for disposing of the waste products in various ways. Alternatively, if byproducts or wastes are not disposed of properly, one pays the societal costs when the ecosystems are restored. However, the cost of ecological restoration could be more equitably distributed in a variety of ways:

1. Bonding should be required that would pay for ecological restoration when land is surface mined, trees are harvested, grazing allocations are allotted that might result in overgrazing and ecological damage, or hazardous materials such as oil are being transported.
2. The cost of ecological restoration could be built into the extraction and production of a particular product. For example, a charge is made for each ton of phosphate rock mined in Florida near Bartow, which is specifically assigned to ecological restoration activities. Thus, only the producers and ultimately the users of extracted phosphorous pay for the ecological restoration. The same is true with surface mining and could be true for a number of other industrial activities as well.
3. Products with extensive packaging (such as cardboard boxes for breakfast cereals, detergents, disposable diapers, and the like) and even automobiles and household appliances could have a disposal charge added to the original price which would then be paid to the organization disposing of them in a manner that would not degrade the environment. This would reduce the cost of ecological restoration by reducing ecological damage.
4. For ecosystems already damaged (especially when the institution/person responsible for the damage can no longer be held liable because they have filed for bankruptcy, gone out of business, or died), an environmental trust fund[8] should be established so that ecological restoration can be carried out on "orphaned" damaged ecosystems. Since society permitted this damage, or at least was silent while it occurred, society should pay for the restoration that converts these damaged ecosystems into ones that provide more ecosystem services.

VII. HABITAT RESTORATION MAY RESULT IN BIOLOGICAL RESTORATION

In some cases, merely restoring the physical habitat may be adequate if no toxic residuals impair environmental quality and if nearby sources of colonizing species are adequate. A good example of this is hydrologist D. L. Rosgen's reconstruction of a section of the Rio Blanco River in southwestern Colorado.[11] The habitat characteristics to which Rosgen gave particular attention included: river width, depth, velocity, discharge, slope, energy slope, roughness, sediment load, sediment size, sinuosity, width/depth ratio, dominant particle size

of bed and bank materials, entrenchment of channel, confinement of channel, land form confinement of channel, land form features, soil errodability, and stability.[11] In short, Rosgen was restoring the natural hydrologic characteristics of the stream that existed before the U.S. Army Corps of Engineers modified the system at the request of some of the land owners of the area. Thus, the restoration was natural in the sense that it fit the hydrologic attributes of the system before disturbance. An unnatural habitat restoration would be one that might occur in other aquatic ecosystems but not be compatible with this particular hydrologic system. Unfortunately, these physical habitat restoration efforts are carried out without an understanding of the interactions of stream hydrology, fluvial geomorphology, and the indigenous organisms. In these circumstances, the most desirable outcome is that these structures will not endure for a long period of time. However, much more serious damage may result to the riverine environment by introducing undesirable compensatory adjustments of channels and banks.[12-14]

VIII. CONCLUSIONS

Mainstream science has accepted ecological restoration as a major new developing field. Professional meetings devote sessions to ecological restoration that are often standing room only and invariably well attended. News media attention to the release of the National Research Council book[8] was heartwarming, at least to a restoration ecologist. Public and institutional attitudes toward ecological restoration have been altered dramatically. Case histories are probably at least an order of magnitude more abundant than they were just a decade ago. However, long-term evidence of success of ecological restoration is now, and will remain, inadequate for many decades to come because episodic events such as hurricanes, drought, floods, and the like may alter the successional process markedly. In some cases, such as the restoration of wetlands, a flood might accelerate the recovery process, whereas a drought might effectively eliminate the progress made and discourage society from investing further funds in the restoration effort. Furthermore, in the United States, robust state, regional, or national restoration goals have not been established, for either acreage of terrestrial systems, lakes, or wetlands or for miles of streams within a particular time frame. Moreover, the illustrative goals of the National Research Council book[8] have received much attention, but this has not resulted to date in the formal adoption of goals either similar to or modified from those stated in the book. The Bureau of Land Management is presently engaged in a discussion of the degree to which grazing is damaging public lands and whether or not user fees should be raised and possibly grazing density reduced; however, again, no formal decision has been reached.

Probably the most important set of relationships is stated in the I = PAT equation in which environmental impact (I) equals population (P) X level of

affluence (A) or per capita pressure on resources X influence of technology (T).[6] There is a close relationship between all of these factors and the probable long-term success of ecological restoration.

One of the most important developments since *Rehabilitating Damaged Ecosystems*[3] was published is the developing interest in improving environmental literacy in the general population, decision makers, and the like in all disciplines. One important publication[15] is designed to improve environmental literacy. The importance of this Sigma Xi document is the fact that approximately 100 participants — half consisting of top administrators from major academic institutions and half faculty from the same institutions — reached an agreement on desirable goals and the general means of implementing these goals. Until environmental literacy is markedly improved in society as a whole, it seems unlikely that large-scale ecological restoration efforts will succeed and persist. Ecological values and human values must be co-mingled in order to reach a sustainable ecosystem repair that will persist once the repair has been achieved.[8]

Another major event affecting the larger context in which ecological restoration must occur is the Union of Concerned Scientists' *World Scientists' Warning to Humanity* signed by over 100 living Nobel laureates and by over 1600 members of the U.S. National Academy of Sciences and its counterparts in other countries. Additionally, the officers of the Royal Academy of London and the U.S. National Academy issued a somewhat similar statement (as noted earlier in this chapter) at approximately the same time. Both statements essentially indicate that concomitant human population growth and ecological destruction cannot continue without serious dangers to human society. Although the practice of ecological restoration has expanded enormously during the past decade, a number of factors could still seriously affect further development of the field.

1. With the economies of the United States and much of the world in mediocre to poor condition, society may well decide that ecological restoration should be postponed until the financial situation is better, just as maintenance of property and machinery may well suffer for the same reason. The educational system throughout most of the United States has suffered budget cuts and other reductions in resources, and it seems unlikely that substantial funds will be devoted to ecological restoration until both the educational system and the human medical care system are in better shape.
2. Legislation on ecological restoration, or at least its interpretation by the regulatory agencies, may become very prescriptive. The clear danger in this is that each ecosystem is unique and each restoration project will be experimental until the field has developed much further than at present.
3. Tenured faculty at most academic institutions has either strongly resisted interdisciplinary activities or accepted them with considerable trepidation because raises, tenure, and promotion are determined almost exclusively by personnel committees strongly oriented toward a particular discipline. Ecological restoration, on the other hand, requires the integration of a variety of information from a variety of disciplines, and, if students are not acquainted with this during the educational process, their performance after graduation on solving problems that require an interdisciplinary effort will not be exemplary.
4. Inevitably, some restoration projects will not live up to expectations or the originally projected schedule. There will then be the inevitable cry of mismanagement, which may be true in specific instances. However, because of a few failures, public confidence in ecological restoration may be severely impaired.
5. A stigma is still attached to applied ecology, and this may inhibit many scientists, particularly those at the beginning of their careers, from undertaking restoration projects because of the negative influence on career advancement of doing any applied work. Happily, this situation is changing, although it may still inhibit some persons who otherwise would be willing to engage in applied projects.
6. The number of people with significant experience in the field of ecological restoration is minuscule, while the number of areas needing restoration is enormous. Appropriate education and training at the Ph.D. level should take at least five years, and, therefore, there will be a significant lag time even if all major universities had Ph.D. programs in restoration ecology, which is not the case.

The enormous progress made in ecological restoration in the half decade between publication of these two books is reason for considerable optimism about the future of the field.

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CHAPTER 16

Restoration Ecology: A Synthetic Approach to Ecological Research

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While interest in ecological restoration has grown rapidly in recent years, especially during the decade since the passage of the Surface Mine and Reclamation Act of 1977, research in this area has generally been highly empirical in nature. In most cases, the objective has been to find techniques that are effective and economical and that can serve as the basis of prescriptions for reclamationists and restorationists working under various conditions. In other words, restoration and reclamation have generally been approached as problems that are basically *technical* in nature, the implication being that they represent a form of "applied" ecology — a task that may benefit from ecological concepts and knowledge, but that has little to contribute to it.

At the same time, it is generally recognized that the distinction between basic and applied research is not a sharp one. Information and ideas, questions and criticisms clearly do move in both directions, from the practitioner to the scientist as well as in the other direction, and work that may be carried out primarily for practical reasons often turns out to be a kind of proof-of-the-pudding test of basic ideas.

This chapter is a brief discussion of another book in which this idea is explored in some detail by some two dozen ecologists. The book was edited by myself, by Michael Gilpin of the University of California at San Diego, and by John Aber of the Forestry Department at the University of Wisconsin-Madison and was published by Cambridge University Press early in 1987. Its title is *Restoration Ecology: A Synthetic Approach to Ecological Research.*

"Restoration ecology" is a term of our own devising and perhaps calls for a bit of explanation. By this term we do not mean simply an ecologically sophisticated or environmentally sensitive approach to the task of ecological

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restoration. What we mean is ecological restoration carried out specifically as a way of raising basic questions and testing fundamental hypotheses about the communities and ecosystems being restored. We mean, in other words, restoration deliberately used as a technique for basic research. The idea is simply to draw attention to the fact that one of the most powerful ways of learning about a thing, a process or a system, and one of the most convincing ways to demonstrate that you understand it is first to take it apart, and then to put it back together and make it work. In this sense, then, restoration ecology is the second and complementary phase of experimental study: the synthesis that follows description and analysis. To make this point more clearly, and also to suggest an analogy with other disciplines such as chemistry and engineering, we have from time to time used the term "synthetic ecology".

This idea is clearly not a new one, and even in ecology it is one that probably tends to be taken for granted. At the same time, as far as we are aware, it has never been developed in detail or even very clearly articulated as a procedure, a pattern, or a paradigm for ecological research. This was what we have hoped to do in *Restoration Ecology*.

The book itself amounts to a discussion, from a wide variety of points of view, of the heuristic or intellectual value of ecological restoration or synthesis. (We construe these terms broadly to include attempts to assemble communities or construct ecosystems for purely experimental purposes. Indeed, one purpose of the discussion was to identify a continuum linking various synthetic techniques that are traditional in ecology with the full-scale ecological synthesis that takes place in the field in the familiar form of "restoration".)

The starting point for this discussion was the University of Wisconsin-Madison Arboretum, where attempts to restore a collection of ecological communities native to Wisconsin and the upper Midwest have been underway since 1934. This effort, which was apparently the first attempt anywhere to restore native communities systematically and on a large scale, was initially undertaken in order to establish a collection of communities that could be used for research along traditional lines. Thus, even though it was carried out for "scientific" purposes, this project was product- rather than process-oriented, as restoration projects have almost invariably been. At the same time scientists involved with the project soon recognized that the process of restoration itself provided numerous opportunities for research. Certain difficulties encountered in the early attempts to create a tallgrass prairie *without* fire, for example, had led to classic experiments on the role of fire in prairie communities. In addition, the restoration efforts eventually led to a number of other insights into the site preferences and dynamics of plant communities. What this suggested was that, though not undertaken in quite this spirit, the restoration efforts actually represented, or could be construed as, a form of manipulative research. This was a notion of considerable practical interest to the Arboretum since it was clear that the restored communities had limited value for research along traditional lines but were ideally suited for basic research using the "restoration ecology" approach.

Fortunately, however, we were not alone in our thinking along these lines. Just about the time we were consolidating our own thinking in the course of planning for celebration of the 50th anniversary of the Arboretum in 1984, several ecologists drew attention to the heuristic value of various forms of ecological management.

In particular, Anthony Bradshaw, one of the few academic ecologists in the English-speaking world to have taken a serious interest in restoration as an intellectual as well as a practical challenge, suggested that, far from being merely a form of "applied" ecology, restoration actually represented the "acid test" of ecological understanding. About the same time John Harper, reflecting on his own early experience in agricultural research, pointed out that ecology has learned some of its most important lessons from agriculture, and that this is true precisely because the constructing and managing of ecosystems, which is the distinctive business of agriculture, has led to valuable insights and tests of ideas that are inaccessible to descriptive or even analytical studies. What Harper proposed, in effect, is that agriculture provides a paradigm for ecological research, and that the way for ecology to become experimental is by working, as farmers do, by assembling working systems, beginning with single elements to produce increasingly complex — and realistic — combinations. This, of course, was precisely what we meant by "restoration ecology".

At the same time John Cairns was drawing attention to the heuristic value of restoration efforts in a variety of situations on this side of the Atlantic. All this encouraged us to organize a symposium on "restoration ecology", as one of several events marking the 50th anniversary of the Arboretum. The symposium was held in Madison in October 1984. It included 14 formal papers and was attended by more than 300 ecologists and managers from all over the U.S. and Canada. The papers read at that meeting became the basis for the book being summarized here, though a number of additional papers have since been added. Fortunately, all three ecologists mentioned above have contributed to this effort.

Our objective throughout this project was basically to test the idea that the synthesis, repair, or healing of systems might have as much heuristic value in ecology as it does in other disciplines, such as, for example, mechanics or medicine. As we proceeded it soon became clear that there were numerous reasons for believing this to be the case. Actual restoration of communities in the field, such as had been done at the Arboretum, is only one example of a much more general tradition of testing ecological ideas through the assembly and manipulation of communities, in the laboratory as well as in the field. In other words, what we were calling "restoration" or "synthetic" ecology was in fact a well-established approach to ecological research, even if it had not generally been recognized as such. Ecologists regularly put systems together as part of their attempts to test ideas about them, though this work rarely includes actual creation of replicas of naturally occurring communities in the field. By the same token, restorationists are continually assembling whole communities and ecosystems in the field, though rarely if ever for explicitly

experimental purposes. (More precisely, experiments in restoration are typically experiments of a purely technical or empirical nature, intended to find out what works, not why it works or how the community or ecosystem itself works.)

It was our intention to see how the two traditions might be combined or linked. The idea was to explore the idea that the tradition of synthetic research that ecologists had carried on for years, but which had usually been confined to the laboratory, might profitably be extended into the field and to work with more complex systems, modeled on those that occur naturally. In a sense we were testing an historical hypothesis having to do with the way ecological research is done. The hypothesis was that ecologists do in fact carry out various kinds of synthetic experiments, and that this suggests a pattern in ecological discovery, and also a paradigm that might be useful in planning ecological research as well as in thinking and writing about it, evaluating it, and even criticizing it.

Though this seemed a straightforward premise, we did encounter a few difficulties in our attempt to explore it. A basic problem was that it involved looking at a familiar subject from an unfamiliar direction. While recognizing the value of management work as an *occasion* for reaching new insights, both ecologists and ecological managers have tended to assume that, in the last analysis and for most practical purposes, information, ideas, questions, and answers to questions flow in the other direction — from the theorist and the academic to the practitioner. Our goal, in contrast, was to consider restoration and synthesis not as the beneficiary of more "fundamental" studies, but as a technique for research, a source of questions, including questions of the most basic kind, and finally as a test — perhaps even the most stringent test — of ecological ideas and, therefore, as a challenge that is not peripheral to ecology but central to it.

A second problem was that in order to do this it was necessary to approach the subject historically, which is unusual in the reporting of scientific results. A couple of generations ago, G. K. Chesterton remarked on "the modern innovation which has substituted journalism for history" (in *Charles Dickens: Last of the Great Men*). But if Chesterton thought journalism was nonhistorical, it would be interesting to know what he would think of a typical modern-day scientific paper. Science, of course, has gone far beyond journalism in the elimination of history from its day-to-day communications, and it has replaced the journalistic remnants of the story with something called "materials and methods", which, as any practicing scientist knows, is typically completely nonhistorical. (The interested reader will find further, and considerably more pungent, comments on this matter in Peter Medawar's essay "The Art of the Soluble".)

This tradition presented us with special problems in bringing together *Restoration Ecology,* since as the discussion and editing proceeded it gradually became clear that the whole project represented an attempt not just to figure out what ecologists know, but to get some idea of how they come to know it. Specifically, the idea was to tell the story of various research efforts in order

to test the historical hypothesis that a particular kind of activity — the synthetic and restorative — really has played a role in the shaping and testing of ecological ideas.

The results certainly suggest that this is the case. Detailed accounts are presented in the book. However, what may be of special interest here are several generalizations about the value of restoration as a technique for basic ecological research.

Most generally, it would seem that the greatest value of restoration as an ecological research technique is the power it has to draw attention to what is important in a system — in other words, to force the ecologist to an increasingly clear conception of the critical parameters governing the system with which he or she is dealing. This idea is developed in a series of straightforward examples in a chapter by Anthony Bradshaw dealing with attempts to restore vegetation on profoundly disturbed land. Bradshaw first points out that plants have certain very simple needs, such as the need for a suitable rooting medium, an adequate supply of water and nutrients, a lack of toxicity, and a means of reaching a site through some dispersal mechanism. He then describes in some detail how both descriptive and manipulative studies of changes in vegetation on disturbed sites provide opportunities for identifying those factors that are most significant in a given situation. Of special interest here is Bradshaw's discussion of the value of active restoration involving attempts to introduce plants and to accelerate and guide the development of a particular kind of plant community. Under these conditions, each act of disruption is likely to reveal a new set of factors crucial to the normal functioning of the system. Similarly, each manipulation of the system in an attempt to accelerate or guide its recovery is based on some kind of idea of these factors and therefore constitutes a test of that idea.

What emerges is an elegant corroboration of Harper's idea of agriculture, with its combination of analytic and synthetic manipulations of ecological systems, as a paradigm for experimental ecology. Thus, for example, at the level of ecosystem disruption and reconstruction represented by traditional farming methods on, say, prairie soils in the American Midwest, supplies of certain nutrients such as nitrogen soon become limiting, so that the attempt to manage the resulting artificial agro-ecosystems initially resulted in a preoccupation with macronutrients and a considerable amount of research related to their use by the community. As Harper points out, the resulting information is of direct relevance to ecology, and — which is of special interest here — it is information that emerged as a result of efforts to take ecosystems apart, to put them back together again in a simplified, ecologically stylized form, and then to make them *work,* reliably, predictably, even economically.

It is possible to discern a pattern here: the way to understand the system is first to describe it, then to disrupt it, and then to try to repair the disruption. Of course, different kinds or degrees of disruption elicit and bring into the foreground different aspects of the system. Thus, while the disruption represented by modern agriculture quickly results in failure of nutrient-supplying systems,

it does not draw attention quite so forcefully or immediately to problems involving the physical properties of the soil. These questions are encountered in their most unmistakable form when the soil itself is severely disturbed, or even destroyed, as in stripping for surface mining. Thus, research on reclamation has led to numerous insights into the interrelationships between plants and soils, some aspects of which are discussed by several contributors. Bradshaw, for example, discusses ways in which reclamation work has contributed to an understanding of the importance of soil structure and its relationship to rooting depth in plant communities, and Clark Ashby notes that forest restoration efforts have led to a growing awareness of the importance of macrochannels created by decayed roots and burrowing animals.

Perhaps the greatest challenge, raising the largest number of questions, is restoration of natural communities on the most profoundly disturbed sites, since it is in doing this that the restoration ecologist encounters problems related to the widest range of ecological phenomena. Mike Miller, for example, who has done extensive work involving restoration of native vegetation on mine sites in the semiarid West, details how work of this kind has contributed to the understanding of mycorrhizae and the ways they interact with the vegetation in a community. He also describes a number of experiments involving the stockpiling and respreading of soil, some of which amounted to tests of various ideas about the nature and mechanism of succession in these communities. One idea, for example, was that succession beyond the early seral stages depends on mycorrhizae. This suggested that it might be possible to accelerate succession, or even to skip stages, simply by adding mycorrhizae, but when this was tried it produced negative results.

A recurring theme, running through many chapters in the book, is the value of restoration as a way of carrying the ecologist beyond what can be concluded or inferred from observation alone. Observations yield facts, but may provide little insight into the precise importance or significance of those facts. As early as the 1940s, for example, limnologists were aware of chemical reactions that might result in the release of phosphorus from lake-bottom sediments under certain conditions. Early models of lake eutrophication, based largely on experience with deep, oligotrophic lakes, represented lakes as simple "algae bowls", in which storage and release of critical nutrients were limited, implying that reductions in nutrient input ought to lead to corresponding decreases in productivity. According to Eugene Welch and Dennis Cooke, who contributed a chapter on lake restoration, it was not until managers started trying to restore eutrophic lakes that the weaknesses of this model were fully appreciated. Restoration efforts based on this model and involving reductions in phosphorus input were dramatically successful in certain lakes, but had little effect in others, leading directly to identification of large reserves of phosphorus in the sediments of some lakes and to a growing understanding of the factors influencing the movement of phosphorus in and out of these reservoirs.

Of particular interest here is the fact that some of the *chemistry* pertinent to the storage and release of phosphorus from lake sediments had been clearly

understood since the 1940s, but that the *ecological significance* of these reactions had not been appreciated until restoration efforts drew attention to them. In fact, this is typical of the way in which restoration has served to bring to the ecologist's attention factors that may have been overlooked in the course of work with undisturbed systems.

More generally, restoration may lead to more accurate assessments of the importance of factors the significance of which may have been under- or overestimated. The book offers a number of examples. Walter Adey's attempts to construct an artificial coral reef/lagoon microcosm in connection with his research at the Smithsonian Institution, for example, led directly to insights into the nutrient budgets and productivity of these complex systems. Similarly, by following changes in the distribution of plants on the artificial prairies at the Arboretum, Grant Cottam and colleagues have been able to show that these changes are correlated with soil moisture and pH, but not with several other soil properties, including texture and content of major mineral nutrients and organic matter. Similar observations might have been made on natural communities, but it is likely that the results would have been less revealing, since it is precisely the initial *misplacement* of plants in restoration efforts, together with the resulting tensions and unbalances and the response of the community to these, that make possible the kinds of distinctions suggested by the work Cottam describes.

In a similar way, the artificial forest communities at the Arboretum have also provided opportunities for studies of nutrient budgets that it might have been difficult or impossible to carry out in natural systems alone. Basically this reflects the difficulty of matching communities — and species — to sites. As a general rule, the restorationist attempts to make these matches as closely as possible, but inevitably slight mismatches occur, and it is the results of these, far more than the results of the most serendipitous successes, that provide insights into the ecology of the system being restored. Thus, the several artificial forests at the Arboretum, which have differed considerably in performance, have made it possible for John Aber and colleagues to carry out measurements of nitrogen cycling over a far wider range of conditions than is likely to be found in naturally occurring forests. The results have led Aber to suggest a refinement in the use of the term "mesic" as it is used to refer to the ecological condition of a site. Generally, this term is applied to sites that are regarded as having a generous supply of both water and nutrients. By measuring productivity over the unusually wide range of conditions provided by the restored communities at the Arboretum, however, Aber and colleagues have been able to show that the sugar maple is far more sensitive to changes in moisture than to changes in nitrogen availability, making it possible to discriminate between two factors that have usually been lumped together under a single term, and in this way coming a step closer to the isolation and identification of an important parameter influencing the development of these forests.

Restoration, in other words, provides a powerful tool for discrimination in the development of ecological concepts. Another example, not discussed in the

book, has to do with the definition of species. In an earlier publication ("After Description," in *The Plant Community as a Working Mechanism,* E. I. Newman, Ed., Blackwell Scientific, Oxford), John Harper pointed out that species defined in the classical manner, on the basis of morphological characteristics, may be highly inappropriate categories for ecological studies, in which subtle physiological factors may be of far greater importance than even fairly well-marked morphological characters. In fact, this is a matter of some concern to restorationists, who (like horticulturists) have long been aware that the behavior of individuals and populations of a species may differ across the range of the species in a way that may be highly significant ecologically. This experience clearly suggests the weakness of traditional taxonomic schemes for use in ecological work and also suggests ways by which more ecologically oriented systems might be worked out and refined through restoration projects involving the use of reciprocal transplants and creation of communities including species and ecotypes in various, perhaps novel, combinations. Indeed, the growing use of restoration as a strategy for dealing with environmental problems lends this issue some urgency, since restoration frequently involves bringing in species from considerable distances. Here the question of provenance and the definition and ecological significance of ecotypes becomes one of considerable importance from a purely practical and economic point of view as well as from the point of view of taxonomy and evolutionary theory. A chapter by Tom McNeilly in *Restoration Ecology* includes a discussion of some aspects of this issue.

Another aspect of the heuristic value of restoration that is discussed in some detail in the book is the value of restoration as a way of working out mechanisms underlying phenomena that may have been identified in the course of descriptive studies. An example is the characterization of the mechanisms that account for changes in a community during the process of succession. Most studies of succession have been descriptive studies in which attempts have been made to discern patterns of change and, insofar as possible, to infer the reason for it by establishing correlations between specific factors and the changes taking place. The problem, of course, is that correlations can be highly misleading, and this is especially true when one is working with natural or more or less undisturbed systems, where species and systems have long periods to form a more or less integrated whole. Under these conditions correlations may reflect quite obvious factors (e.g., water availability) or much less obvious factors, or even complex combinations of factors. John Aber's refinement of the term "mesic" suggests the value restoration has as a way of refining ideas that may seem perfectly adequate when tested by purely descriptive or even analytical studies.

In any event, it is clear that one way to test a guess about ecological systems based on observation, correlation, or analysis is to try to manipulate the system in such a way as to test the guess — and this frequently involves assembly, or partial assembly, of the system. Katherine Gross' work on mechanisms of

succession in old fields in southern Michigan provides an excellent example. Descriptive studies of these fields had revealed that several biennial species differed quite dramatically in the time at which they colonized these fields and also in the length of time they persist in them following colonization. These patterns of behavior were correlated with certain life-history traits such as seed size and number, the length of time the seeds remain viable in the soil, the growth form of the emerging seedlings, and so forth. In order to establish which of these factors really accounted for the behavior of the species, however, Gross undertook a series of experiments involving introduction of the plants into the community under conditions designed to test specific hypotheses accounting for their behavior. In this way she was able to show, for example, that the ability of large-seeded species such as *Daucus carota* to persist in increasingly dense vegetation longer than certain smaller-seeded species is a direct result of the large seed, which enables seedlings to become established in a vegetative cover where seedlings emerging from smaller seeds may fail.

In a similar way, to test the idea that clearings created by burrowing animals might play a key role in the persistence of certain species, Gross created artificial disturbances in the communities. In this way she was able to show that the species that colonize such disturbed areas vary depending on the season in which the disturbances are created.

This is obviously of considerable interest from a practical as well as from a theoretical point of view. Gross also points out, however, that the seasonal effects she observed only suggest, and may not necessarily indicate, the specific factors (i.e., the actual critical parameters, such as timing of changes in moisture supply) that are really responsible for the observed effect. This is an observation of great significance for restoration and reclamation research generally, since it draws attention to the fact that the results of purely empirical experiments are necessarily of limited applicability. What worked last spring on one site may very well not work next spring on another site, unless the restorationist actually understands the critical parameters that underly what is going on and is able to manipulate them, e.g., by irrigating at some critical moment. To identify these factors, however, it will be necessary to undertake additional synthetic experiments such as experimental irrigation of test plots, for example.

Another way of putting all this is to say that restoration provides the ecologist with a basis for distinguishing between what one really needs to know in order to understand a system and what one can afford to "black box" and, at least for the moment, ignore. Which factors, for example, determine the ability of plant species to coexist? To find out, Patricia Werner planted seeds of teasel *(Dipsacus sylvestris)* in old fields and then attempted to relate the success of the resulting plants to the species growing near them. What she found was that the results could be accounted for almost entirely in terms of the growth form or "architecture" of the various competing species. In short, these synthetic experiments, technically almost indistinguishable from certain

kinds of agricultural experiments, suggested that, so far as teasel is concerned, all that matters is the size and shape of the competition. This in turn points to the possibility of species substitutions in the creation of artificial ecological communities — a practice that may be of great importance in reconstructing communities from which a dominant species may have become extinct (e.g., American chestnut or perhaps American elm.)

A feature of this book that will be of special interest to those interested in the more philosophical aspects of ecology generally and restoration ecology in particular are several discussions in the introductory section that deal with the nature of restoration, its heuristic value and its relationship to the development of ecological theory. A key issue here is the suggestion that restoration represents the crucial, or "acid", test of ecological understanding. Plausible as this idea may seem on the surface, it is also evident that restoration or repair of a biological entity such as an ecological community may not be a stringent test of an idea simply because biological systems have a certain capacity for self-repair, so that it may be possible to "restore" a community simply by bringing elements together and letting nature take its course. Given enough time any system will recover, and this obviously proves nothing at all about anyone understanding the system.

What this means, however, is that the really critical test of understanding is not simply the ability to restore the system, but the ability to control its development and functioning. Restoration is only a particular form of control, even though it may be an especially dramatic one. By the same token, what is called for by way of an "acid test" of ecological understanding is not rote copying of natural systems (i.e., not restoration at all in the narrowest sense) but rather *imitation* of natural systems, the distinction being that copying implies reproducing the model in concrete form while imitation implies making a similar system that only *acts* the same as the model. Clearly it is this, and not copying, that demonstrates a grasp of the critical parameters governing the system. It is in this sense, then, that we use the word "restoration" in the term "restoration ecology".

Clearly a number of questions remain. Even granting that restoration has provided ecologists with opportunities for achieving significant insights, just what has the relationship between the restoration effort and the insight been? Has it been intentional? Has it even been necessary? So far as I am aware, restoration projects have rarely, if ever, been undertaken specifically as experiments to test basic hypotheses. Often, restoration efforts undertaken for purely practical reasons lead to new ideas about an ecosystem that can then be tested more systematically. An example is a new idea about the species composition of midwestern oak savannas developed by Steve Packard, a restorationist who works with The Nature Conservancy in Illinois. Packard developed these ideas, which differ considerably from earlier ideas about this ecosystem, as a result of his restoration efforts, and they are now being tested and explored by a number of researchers. This is an example of synthetic ecology proceeding

from practice in the field toward research in a more strictly controlled situation, but the process may also proceed in the opposite direction as it were, from the laboratory to the field. One of our ideas at the outset was that laboratory experiments involving the creation of experimental communities might be extended in this way to experiments in the field, but to what extent will this prove to be practical, intellectually fruitful, or heuristically economical? Granted that questions may be answered through restoration; still there may be easier, more economical ways to answer them. Just what will happen in the process? Will phenomena such as the role of competition in shaping a community, so elegantly revealed by Michael Gilpin's experiments involving mixtures of fruit fly species in bottles, simply become irrelevant in the complex community in the field?

All this remains to be seen. At the same time, we have at least emerged from this phase of our discussion with the sense that the discussion is worthwhile. Overall, a sense emerges that the gap between theory and practice may not be so great after all. Restoration in the sense we are using it here may indeed have a great deal to offer ecology. Anthony Bradshaw has suggested that it represents the acid test of the ecologist's understanding of his or her subject. Michael Rosenzweig adds to this observation that this is so precisely because the nonlinearity of response in many ecological systems makes it necessary not only to work with perturbed systems, but to work with *profoundly* perturbed systems, in which variables such as populations sizes have been pushed to extreme values. For ethical reasons, Rosenzweig suggests this be done not by perturbing systems artificially, but by beginning with sites disturbed by activities such as mining and then adding species experimentally.

Furthermore, the idea that restoration (or "control") of the ecosystem represents the acid test of ecology is one that obviously has important implications for the science. To the extent it is valid, for example, it suggests that, far from being a mere intellectual by-product of ecology, of marginal interest to the ecological theorist, restoration actually deserves to be regarded as the organizing principle for the science, providing a basis for deciding what is worth knowing and what is not.

This immediately raises yet another question: Is this principle comprehensive enough? Or might it be possible to "restore" (i.e., to control) a system in the way we have described without really understanding it? If an ecologist can finally create a given community anywhere, at any time, and under various conditions, specifying all the conditions that make this possible, is ecology then finished with that system? Or are there things ecologists properly want to know about their subject that are irrelevant to the challenge of restoration? Certainly there might be. Recently, for example, a colleague suggested that branching patterns might be an example. This is something that ecologists are interested in and that might seem irrelevant to restoration. Yet, as work like that of Pat Werner and Kay Gross clearly shows, it is not irrelevant. Still, this question awaits further discussion.

Finally, developing the idea of restoration ecology and taking advantage of it may well prove an excellent way to establish a closer, more fruitful relationship between ecological theory and ecological practice, since it suggests a common ground between theorist and practitioner. At bottom, this is the whole purpose of the discussion carried on in this book: to weaken the barrier between theory and practice by developing the idea that what the practitioner has done — perhaps for purely "practical" reasons — actually represents a test of the ideas on which the procedure is based and so, properly construed, is essentially an experiment.

Restoration, then, clearly provides ecology with a powerful technique. It may also provide it with a set of objective, definable goals and, hence, with a basis for self-criticism. Just as important, it provides ecology with a mission. If nothing else, restoration ecology does represent ecology taken seriously not simply as a science, but as one of the healing arts — in fact, the most comprehensive of the healing arts, the one that deals not with individual members of a single species, but with the whole. It is just this sort of ecology that Aldo Leopold had in mind when, in 1940, reflecting on the state of his own discipline of game management, he said that it was not merely a science because it was based on value and concern for particular things. ("The state of the profession," *Journal of Wildlife Management*, 4(3), 343.) It is also certainly what he had in mind when he looked forward to the development of what he called a science and art of land health.

What is of special interest here is the way restoration ecology suggests that the intellectual objectives and the ethical concerns of ecology not only overlap but may actually *coinicide*. What one needs to know about the system in order to restore it may also be just the thing that is most worth knowing from a purely intellectual point of view.

To the extent this is true, it is immensely encouraging since it suggests the possibility of a science wholly integrated with value. Perhaps it is just this integration, this coincidence of interest and value that characterizes the healing arts and distinguishes them from the "mere" sciences.

In any event the discussion is an interesting one. What we have heard so far encourages us to suspect that restoration will prove a healthy preoccupation for ecology, and especially for ecologists interested in the development of a discipline that has moved beyond description and diagnosis to take on the even more demanding tasks of prediction, prognosis, and cure.

NOTE

This chapter was previously published in the two volumes of *Rehabilitating Damaged Ecosystems* (CRC Press, 1988). Although the ideas, results, and main points still stand, the literature has not been extensively updated.

Karen Clarke

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AN ECONOMIC ASSESSMENT OF THE BENEFITS AND COSTS OF MINE SITE REHABILITATION

GARRY JOHNSTON
NEWCREST MINING LIMITED

INTRODUCTION

The mining industry utilises a fraction of the total Australian land mass (Graetz *et al.*, 1992) and spends large sums on the environment, most of it in rectifying land degradation. Over large areas across the country the industry is subject to fairly uniform rehabilitation standards, although the benefits and costs resulting from rehabilitation may display considerable regional variation.

Environmental issues can be addressed by political processes (e.g. lobbying), scientific evaluation or economic analysis. The first two processes have largely decided mine site rehabilitation requirements in Australia. Political processes alone are likely to produce unpredictable rehabilitation outcomes because they involve conflict resolution between rent seeking and rent defending behaviour from groups with vested interests (Sandmo, 1990). Science provides information about desirable rehabilitation outcomes but no information on how resources should be shared among competing uses. Resource economics can make a valuable contribution to the development of rational environmental policies for the mining industry by addressing how resources should be allocated.

Economic analysis has all but been ignored in the assessment of environmental regulations affecting mine sites. Two recent papers that have dealt with environmental mine site economics in Australia are:

- the Resource Assessment Commissions ambitious contingent valuation survey of the Kakadu Conservation Zone published in 1992. This concluded that Australians would be willing to spend $600 million per annum to forgo mining in the area while the adjacent national park some 300 times larger, was spending only $12 million per year on upkeep, a finding widely criticised; and

- the brief overview of mine site rehabilitation by Barnes *et al.*, (1992).

The evolution of environmental policy has inevitably brought economic issues to the fore; environmental regulation has necessarily involved costs (Cropper and Oates, 1992) - and the question of how extensive these remedial measures should be because of these costs remains a crucial and unanswered question.

Relating the use of the environment by mining to the supply and demand for environmental services highlights the crux of the interaction between mining and the environment; that is, that the resulting environmental disturbance is not the product of deliberate wrong doing but an unavoidable and integral part of socially useful activity (Industry Commission, 1991).

The setting of rehabilitation standards and regulations for mine sites presupposes that regulators can determine what the economically desirable levels of rehabilitation are. However the limited evidence available suggests that this may not have occurred for mine site rehabilitation and a misallocation of resources for rehabilitation is occurring.

In economic terms optimal allocation of finite resources occurs when the benefits to society are maximised. Optimal utilisation of a finite resource is not zero, but rather the allocation of the resource that results in maximising the benefits to society while minimising the costs. It follows therefore that complete restoration of a mine site will only be rarely justified as a socially desirable outcome (Coase, 1960; Randall, 1986; Kula, 1992).

This paper uses elements of welfare economics to discuss a framework for considering the optimal level of mine site rehabilitation. It advocates using a cost-benefit approach to maximise social benefits when establishing rehabilitation standards. The result is a unifying conceptual approach that can be used to consider mine site rehabilitation anywhere in Australia, for example mines located in populated coastal plains, national parks or the remote arid zones of Australia's interior.

There are limitations in the economic approach outlined; in particular the approach does not consider distributional issues, legal mandates or ethical considerations.

To remain competitive the mining industry must compete in world markets facing

costs that include the rapidly spiralling environmental costs imposed on the industry as the result of Government regulations. The paper begins by briefly considering the economic pressures facing the mining industry today.

COMPETITION AND THE MINING INDUSTRY

In attempting to stay competitive, the mining industry in Australia must measure its performance against other international producers.

Consider the case of copper. Figure 1.0 illustrates the trends in prices and consumption for copper that have occurred during the last century. Despite rising consumption for copper, real prices have been steady or declining. This has resulted in part because of changes in technology and increases in resource inventories. The implications are quite clear for copper producers. Either they manage their costs to remain internationally competitive or they perish! Similar arguments can be applied to most other raw materials.

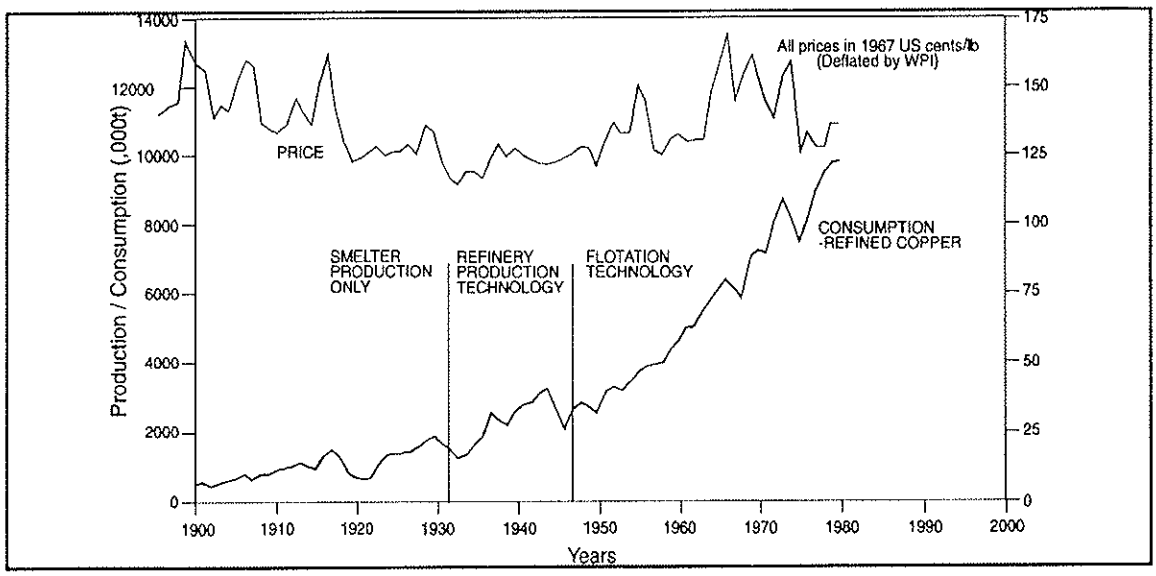

Figure 1.0
World Copper Prices and Consumption

Gould (1993) has summarised the economic implications of increasing international competitiveness (Figure 2.0).

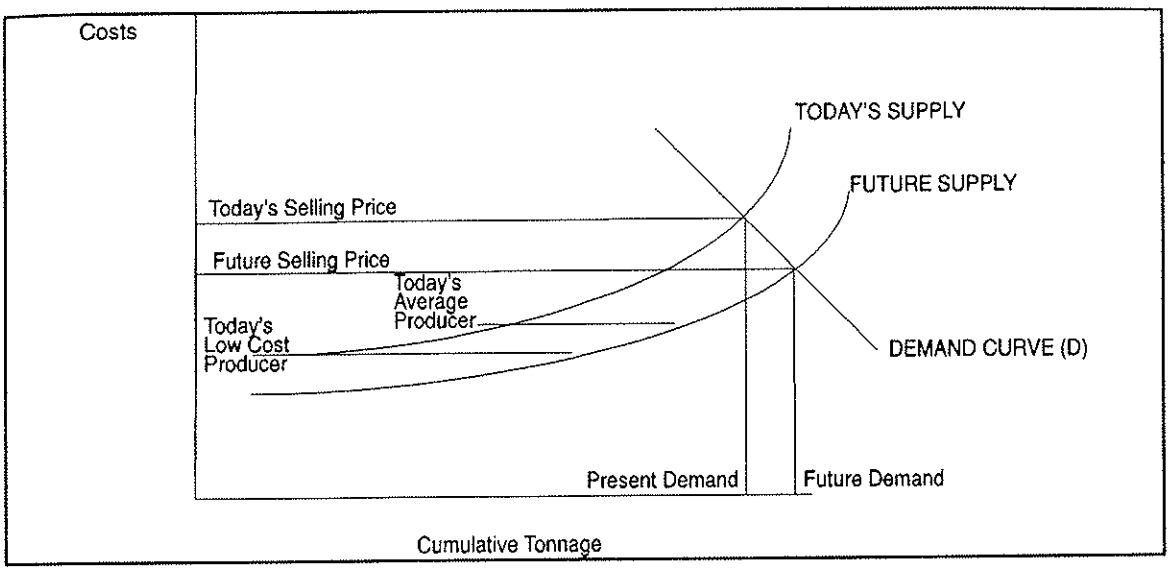

Figure 2.0
Downward Shift in Commodity Cost Curves
(adapted from Gould, 1993)

Given past trends, future supply curves will be below today's supply curve and producers will need to produce more for less. Today's average producer will be a marginal operator if productivity improvements do not keep pace. Similarly today's low cost producer could be tomorrow's average or marginal producer if productivity improvements do not take place.

What has this to do with rehabilitation of mine sites? *Put simply if environmental (and rehabilitation) costs are too high, marginal producers will no longer be competitive and will stop producing.* Furthermore, while rehabilitation of mine sites remains essentially a domestic policy matter, performance must be achieved against international competition.

ECONOMICS AND MINE SITE REHABILITATION

Before discussing the optimal level of mine site rehabilitation, it is first necessary to introduce the ideas of external costs and benefits and explain their relevance to mine site disturbance (or impacts). These terms refer to the fact that costs and benefits fall on others besides the producers and consumers directly involved.

The importance of externalities and property rights has become more widely recognised since Coase (1960) published his seminal paper. To overcome the problems presented by negative externalities (often called 'The Problem of Social Costs") governments have reacted by introducing prescriptive rehabilitation standards. The rate of environmental regulation has been escalating in recent years.

Rehabilitation Involves a Cost

Consider the case of a mine that creates off-site impacts. Examples are the contamination of shallow aquifers caused by metals leaching from waste rock dumps or decreased productivity for adjacent landholders because of dust. If the existence of the mine does not affect other individuals, then the marginal external damage cost imposed on others is zero. On the other hand, if the mine results in additional costs to others beyond the mine lease boundaries, such as the damage to a water aquifer used by adjacent land holders for potable and stock water supply, then these marginal external damage costs should be added to the private costs of mining. These additional costs of production are the external costs or off-site costs. The situation is best illustrated with the aid of Figure 3.0.

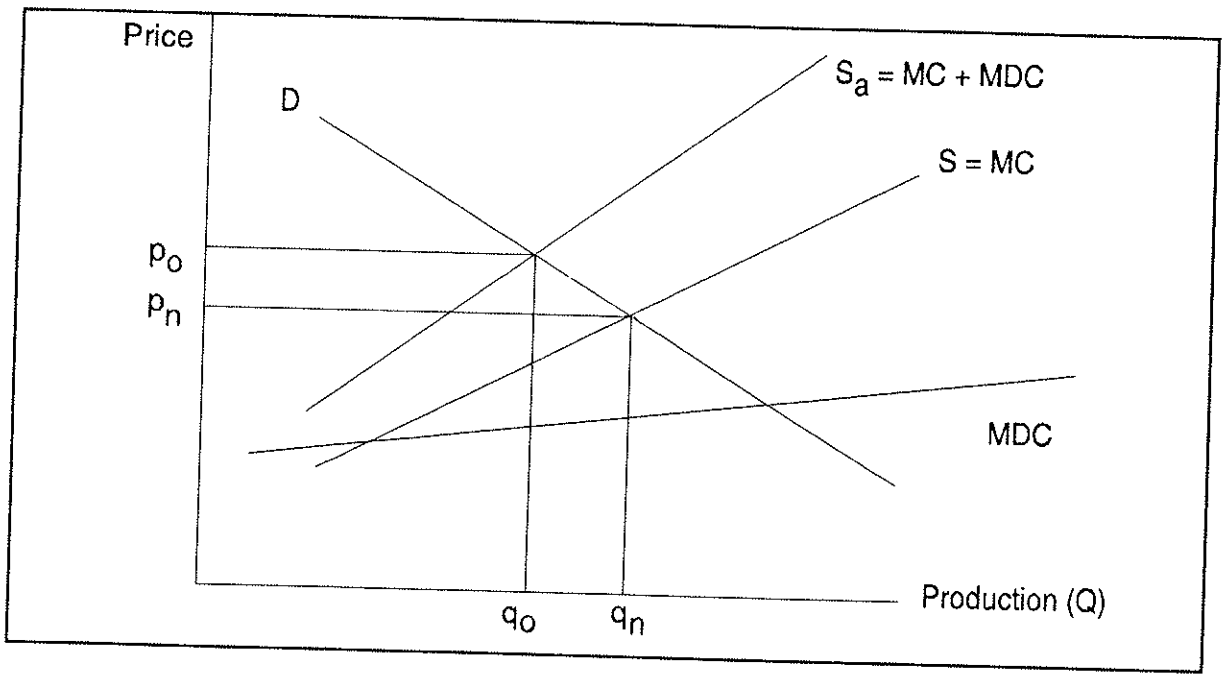

Figure 3.0
External Costs

The miner's supply curve, S is equivalent to the marginal private (production) costs of the miner. The supply curve S_a represents the marginal social cost curve and is the private cost curve S plus the marginal external damage cost imposed on others, MDC. D represents the demand for the commodity being mined and is equivalent to the marginal social benefits of mining.

If the price of the commodity is p_n then q_n tonnes are produced in the presence of the externality (the private optimum level of output). The socially optimum level of mine production is q_0 tonnes and is less than the private optimum, q_n. At the socially optimum level of production prices rise from p_n to p_o to cover the costs of controlling the pollution (the externality has been internalised).

In this illustration the effects caused by a decrease in supply (caused by rehabilitation costs) are lower output, a higher price and a lower return than would be the case without rehabilitation. Thus, the costs of rehabilitation are borne by both the miner who has reduced output and the consumer as higher prices.

The first point being made is that rehabilitation is not costless and that the costs are distributed in some way between the mining company and the community. The second point is that the presence of externalities suggests a potential resource allocation inefficiency.

Governments have reacted to control externalities by setting up regulations and standards and one way to view these is as a form of tax on the use of environmental inputs for mining (Kalt, 1983).

Defining Rehabilitation Benefits and Costs

As discussed above benefits and costs need to be carefully defined in economic terms. For example, there are costs of rehabilitation and there are benefits of rehabilitation. To avoid confusion benefits are normally measured relative to a specified reference state, typically a mining versus non-mining state. Care must be taken to avoid double counting the benefit or cost being valued.

Several classes of benefits might flow from environmental quality improvements brought about by rehabilitation. These include:

- recreation opportunities;
- increased productivity of the rehabilitated land;
- existence value (includes aesthetics); and
- reduction in damages (includes AMD control, erosion/silitation, adjacent land values, reduced water quality impacts).

Similarly costs for mine site rehabilitation can include:

- the costs of compliance with the rehabilitation standards (this includes the costs of the performance bonds if applicable and the costs of enforcement by governments);
- loss of habitat;
- lost economic opportunity costs (i.e. loss of income from alternative uses of capital); and
- loss of future productive capacity (i.e. sterilisation of in-ground resources see Carson, 1988).

THE APPROPRIATE LEVEL OF REHABILITATION - BENEFIT COST ANALYSIS

What rehabilitation standard is appropriate for an individual mine? The options vary from completely backfilling every pit and complete ecosystem restoration (assuming this is possible) to doing nothing. The best solution from an economic view point is to choose that option that maximises the net social benefits.

The usual way in which economists endeavor to measure these net benefits is by using benefit cost analysis. Benefit cost analysis is a technique used to identify, quantify, and weigh the advantages and disadvantages of public policies designed to increase society's general well being (Krupnick and Portney, 1991).

The quantification of benefits rests on the notion that anything has value if someone is willing to pay for it. Furthermore each individual is held to be the best judge of how policy affects them. The complete value to society (or the regional community) is measured by the sum of the individual valuations (see the discussion below).

Thus, benefit-cost analysis is not restricted to purely financial gains or losses but also to non-market goods such as the reduced well being caused by aesthetic degradation.

Costs are generally measured as the expenditure that private companies, governments, and individuals must make to comply with regulations. Analogous to benefits, however costs should also be measured by the amount of money required to compensate individuals for unfavourable effects associated with regulatory or other public policy; these could take the form of higher prices, reduced output or reduced employment etc.

Thus applied to mine site rehabilitation standards, benefit-cost analysis is about balancing the economic costs of environmental degradation against the net benefits generated from current and future uses of the resources (Randall *et al.*, 1978).

Building upon the conceptual formulation of the problem developed by Randall *et al.* (1978), let $NSB(t_0)$ denote the present value in time t_0 of the sum of the net benefits and environmental costs occurring in each time period t (t=t+1,...,T) because of mining. Thus:

$$NSB(t_0) = \Sigma_{t=0}^{T} PV \left[\Sigma_1^J \Sigma_1^K \left(B(t)_{jk} - C(t)_{jk} \right) \right]$$

where:

PV is the present value of future money and $B(t)_{jk}$ and $C(t)_{jk}$ are the benefits and costs in time t from resource using activity k (k=1,...,K) due to the change j (j=1,...,J) in resource quality which resulted from mining, respectively.

Thus all that is being done is that the change in the net benefits (the net benefit ma be negative) of each resource using activity (resulting from resource quality changes) is determined, reduced to present value and summed (Randall *et al.*, 1978). The economically optimum level of mine site rehabilitation occurs when the marginal benefit of rehabilitation just equals the marginal cost of rehabilitation (Sharpe *et al.*,1988). Thus the socially optimal level of rehabilitation occurs when:

$$MB = MC$$

Where:

MB is the marginal benefits; and

MC is the marginal costs.

Two issues associated with benefit-cost analysis are:

- aggregating of individual benefit and cost preferences. The larger the population that aggregation is applied over the more problematical the valuation is likely to be, particularly for non-market goods. Ways to overcome some of these problems include identifying groups who stand to benefit or lose and weighting their preferences. This is already done implicitly for many new mining projects where local community preferences are given prominence; and

- the use of discount rates to calculate present values (PV). Discount rates are used to consider the timing of when benefits and costs are received. The higher the discount rate the more future generations are discriminated against in favour of the present, investment is depressed and high extraction rates of natural resources are favoured. Alternatively lower discount rates are weighted more towards future generations, increased investment is favoured and therefore the demand for environmental services is increased. This creates something of a dilemma for environmentalists because on the one hand accelerated use is favoured and on the other extensive use of environmental resources is favoured (i.e. Kula, 1992; Randall, 1986; Pearce *et al.*, 1989). For rehabilitation the costs occur today but the benefits may occur in the future, therefore low social discount rates might be expected to apply *(ceteris paribus)*. The important point to understand is that it is through the use discount rates that intertemporal preferences are taken into account (i.e. time preferences for benefits and costs).

Rehabilitation Standards and Benefit Cost Analysis

If rehabilitation standards are too stringent and therefore costly to comply with, the total social costs of these standards may exceed their total benefits. Indeed society might experience a gain in welfare from relaxing these standards, if the marginal benefits are below the marginal costs at the level of the standard (Cropper and Oates, 1992).

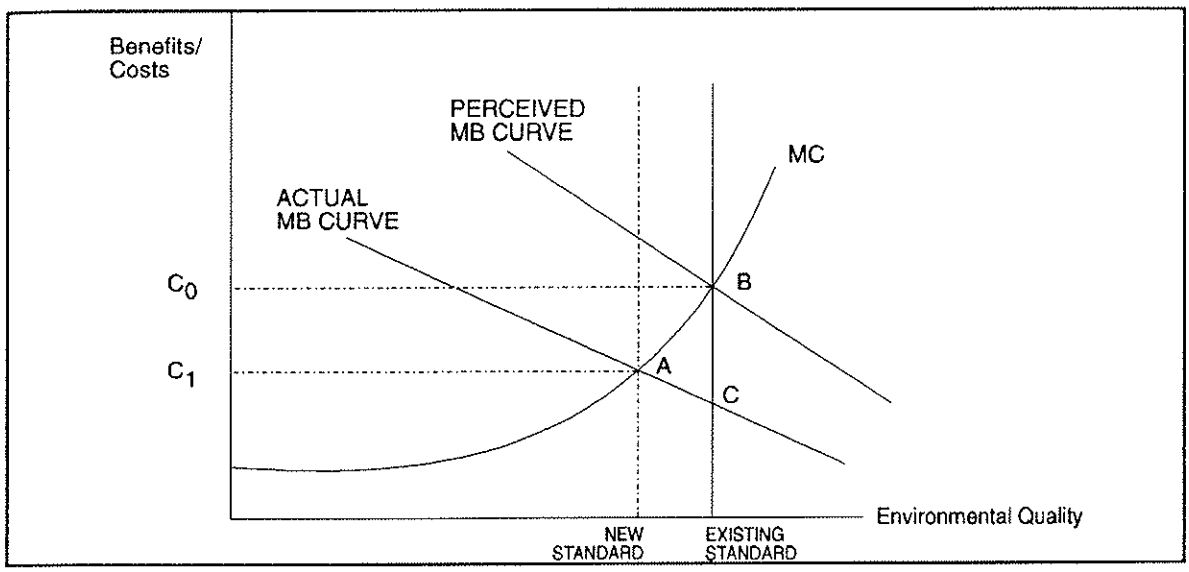

Figure 4.0
Welfare Gain From a Relaxation of Environmental Standards

The net welfare gain obtained from a reduction in marginal cost from C_0 to C_1 (Figure 4.0) is the area ABC (the total gain to society). In the alternative situation where costs are much less than benefits at the standard then a raising of the standard would result in a social welfare gain.

It is important therefore to understand where the benefit curve lies in relation to the cost curve and any standard imposed. There is mounting evidence that in many instances in the U.S.A. standards intercept the marginal cost curve in the steeper region and are too high e.g., the Clean Water Act, the Superfund Sites Program, the National Ambient Air Quality Standards Program (Krupnick and Portney, 1991; Cropper and Oates, 1992).

Valuing Non-Market Goods

Some of the benefit and costs listed above involve non-market goods (those that are not routinely traded in a market) which have no monetary value. In order to build these into a benefit-cost analysis a monetary value that society places on these goods must be estimated. There are several techniques available and a brief tabulation of

these occurs in Table 1.0.

These techniques do have problems and if they are poorly performed can provide unrealistic valuations (e.g., the Kakadu Conservation Zone Contingent Valuation). However perseverance and refinement of these techniques appears justified if they can be ultimately shown to be capable of producing realistic estimates. At present the setting of rehabilitation standards presupposes the regulatory body can decide what the economically desirable levels of rehabilitation are. There is very little evidence to support this contention and available information suggests that an over allocation of resources to rehabilitation may be occurring (as discussed below).

Table 1.0

Techniques for Estimating Non-Market Goods

(Pearce et al., 1989; Sinden, 1992)

Technique	Basis	Problem
1. Hedonic pricing	Adjacent land prices are used to derive a price for the good or commodity to be valued. Accuracy and reliability:	1. Land prices not readily available. 2. Market segmentation 3. Mitigating behaviour. Questionable, result can be out by an order of magnitude.
2. Travel Cost Method	Measures of willingness to pay are estimated from the amount payable and prepared to pay to travel to each location. Accuracy and reliability:	1. Data collection. 2. Single purpose visits. 3. Discriminates against different income levels. Requires strong assumptions about income effects and purpose of visits, long distance travel a problem.
3. Contingent Valuation Method (CVM)	Hypothetical market established and people asked how much they are willing to pay (WTP) or willing to accept (WTA) for the goods being valued. Accuracy and reliability:	1. Bias e.g., strategic, design, hypothetical, operational. 2. WTP vs WTA criteria lead to different valuations. 3. High level of skill needed and can be costly to perform. Provides a flexible framework for valuations. Accuracy depends upon how survey implemented and what is being valued. WTP vs WTA problems persist.

A Comparison of Costs

To illustrate the costs involved consider a mine located in the arid zone on crown land with the following characteristics:

- the area of disturbance is 250 ha and the land system is well represented elsewhere;
- groundwater quality is about 150,000 TDS (about four times the salinity of sea water) and there is no surface water; and
- the project is a fly-in/fly-out operation;

Furthermore in terms of cost benefit consideration discussed above the following observations also apply:

- the biological impacts in a regional context are almost unmeasurable;
- the groundwater resources in the region have limited value because of the high salinity;
- there is no alternative commercial use of the land (now or in the future);
- recreation to the area is non-existent because of the characteristics of the region and lack of infrastructure;
- dust and salinity impacts will be well controlled and largely limited within the lease boundaries; and
- aesthetic impacts will occur although the possibility of anyone seeing these is at best remote.

Using published unit rehabilitation rates (Chisholm and Dumsday, 1987; Barnes *et al.*, 1991) that are consistent with industry experience, then somewhere between $3,000-$12,000 per/ha will be spent to comply with the current rehabilitation standards. On this basis total direct costs for rehabilitation are likely to be of the order of $1-$3 million dollars (indirect costs are not considered). Table 2.0 summarises the internal and external costs for this example.

Table 2.0
Internal and External Costs for an Arid Zone Mine Example

I. Internal Incentive

	Cost ($millions)
Regulatory Rehabilitation Requirement	$1-$3

II. External Costs

Biological damage	unmeasurable
Water damage	unmeasurable
Land damage	?
Reduced recreation	unmeasurable
Reduced aesthetics	?

If optimal rehabilitation is being achieved MC = MB and $1-$3 million = Costs of {land damage + reduced aesthetics}

Since we have defined that there will be limited physical or biological impact or loss of alternative productive capacity of the land disturbed most of this cost must reflect the regulator's interpretation of the aesthetic or existence benefits the community would place on the disturbed area. ***Is this a rational use of limited resources and an accurate reflection of community values?***

If the community actually valued the aesthetic effects of the benefits of rehabilitation at considerably less and/or the lost economic opportunity costs were high (money could derive more benefit by drilling for new deposits or opening or developing new mines) then there is an over-allocation of resources and the situation in Figure 4.0 is occurring. *How representative is the example given?* For a large number of mines located in arid zones the example given is fairly typical.

To provide an economy wide perspective and comparison with amounts spent on other land degradation problems, Tables 3.0 and 4.0 have been prepared.

Table 3.0
Sources and Extent of Land Degradation in Western Australia

(Adapted from Western Australian Government, 1992)

Land Use	Area Affected (% of state)	Notes
1. Mining [1]	<1500 km^2 (0.6%)	No account of Rehabilitated land Made.
2. Arid and Tropical Rangeland[2]	308,471 km^2 (12.3%)	Veg. classified as degraded from poor to very poor.
Ratio Degraded Rangeland/Mining	200:1	

Thus for every hectare of disturbance created by mining, two hundred have been degraded from other forms of land use in the arid and tropical rangelands of Western Australia.

Table 4.0
Land Degradation Expenditure

Land Degradation other than that caused by Mining		
Annual Cost	**Description**	**Source**
Federal Allocation + WA State Allocation $3.7m + $9.6m = $13.3m (1990-91)	Used for rectification, education.	Dep. of Primary Ind. & Energy 1991 Dep. of Primary Ind. & Energy 1992
Unit rate $0.43/ha	Calculated on arid zone and rangeland degradation only	
Mine Site Environmental and Rehabilitation Costs		
Total $611m (1990-91) WA $469m (1990-91)	Includes all environmental work	(ABS Cat. No. 8402-1991)
Unit rate $3,000/ ha (can be up to $12,000)		

Shown in Table 4.0 are the levels of resources that have been allocated to various forms of land degradation in Western Australia

[1] The Perth Metropolitan area is about 1300 km^2.

[2] Majority of Government controlled land.

for the period 1990-91. The intensity of disturbance from mining activities is clearly much higher than that from other forms of land use, however the area disturbed is a fraction of that disturbed by other activities. Do the benefits to the community from mine site rehabilitation to current standards justify the costs?

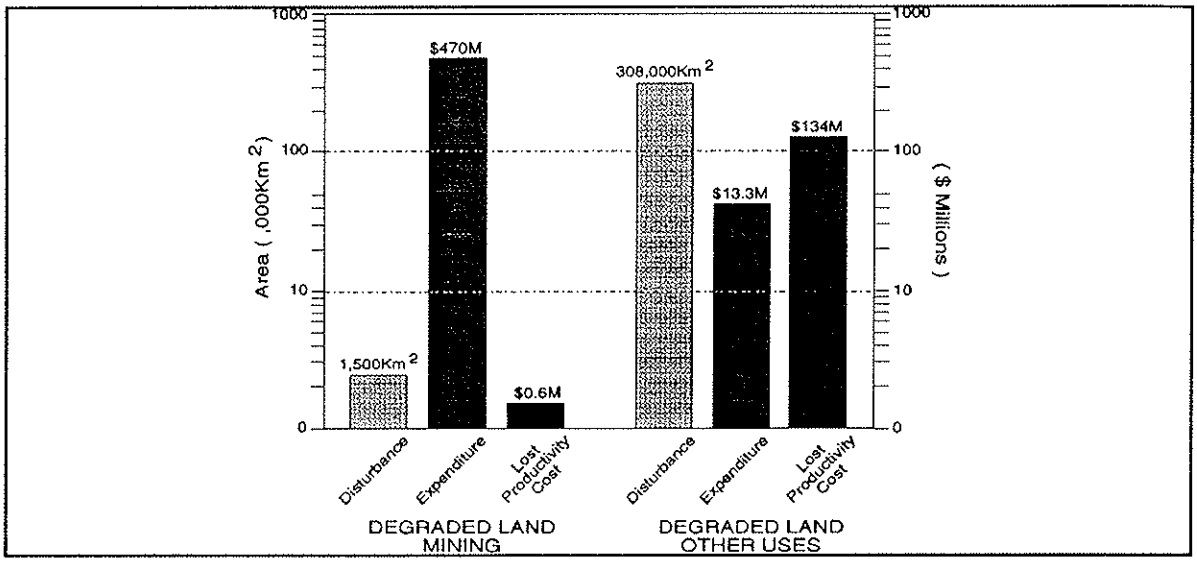

Figure 5.0
Land Disturbance, Productivity Costs and Expenditure on Land Degradation in W.A.[3,4,5] (1990-91)

One figure for the economic costs for lost production due to land degradation in Western Australia is $134 million, the wheat-belt and south-west have been included in this cost estimate (1991 dollars, Robertson, 1984: cited in Chisholm and Dumsday, 1987). Assuming productivity losses from the much smaller areas disturbed by mining (some 200 times less) were the same, the mining industry is still spending three and half times more than is being forgone in productivity from the total degraded area from other land uses (Figure 5.0). ***These comparisons suggest that an over allocation of resources for mine site rehabilitation is occurring in W.A. with the costs disproportional to the expected benefits.***

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[3] Rangeland & arid areas used for other areas

[4] No private expenditure included in other uses

[5] Productivity cost for mining based upon unit rate for state arid and rangeland areas

CONCLUSION

This brief overview has highlighted how mine site rehabilitation can be considered from an economic perspective. Mine site rehabilitation occurs because it is believed that the benefits of rehabilitation to society outweigh the costs. Governments have reacted by regulating to ensure mining companies internalise costs.

In recognising this general economic principle it needs to be understood that if using the criterion of maximising social welfare, then rehabilitation standards must be set at levels that equate the benefits both present and future from rehabilitation with the costs that are involved. Thus often it may be in the interests of both the miner and society for rehabilitation standards to be significantly less than or different to what they currently are. Indirect evidence suggests that at present there may well be a significant misallocation of resources to rehabilitation that if redirected into other areas would result in a significant welfare gain (this doesn't mean that the savings should be used to cross subsidise other land users who degrade land because this would offer no incentive for them to consider the full costs of their actions).

An important related notion that derives from social benefit-cost analysis is that existence of unrehabilitated or partially rehabilitated old mine sites does not in itself constitute a compelling case for additional rehabilitation. Previous mine site disturbance is a *sunk cost* to society and whether further rehabilitation is warranted can only be decided based on the present value benefits and costs, not on what has occurred in the past (e.g. consider the $20 million in 1980 dollars spent on rehabilitating the old Rum Jungle mine site).

In arid zones current blanket government regulation or guidelines for angle of repose of dumps and specification for tailings dam covers and decommissioning are not appropriate. As long as off site impacts are minimised then permanent man made landforms, away from population centres even if aesthetically unappealing or unsightly to some may be an acceptable end-point for rehabilitation. Particularly when it is considered that the benefits derived from mining far outweigh any current or future benefits that might occur from existing or alternative land uses.

Thus the tendency to gold plate rehabilitation should be avoided and a site specific rehabilitation program developed that is commensurate with surrounding

characteristics and benefits and costs associated with rehabilitation in the area.

The conceptual approach outlined of using a benefit-cost approach helps explain why rehabilitation in a Jarrah forest or coastal sand dune should be different from a mine in a remote arid zone. In the former cases the community values the benefits from present and future uses of the forest or coastal landform highly and therefore rehabilitation should reflect this using the criteria of maximising social benefits. In the latter case it is not unreasonable to expect that the community will value the benefits of rehabilitation at considerably less and therefore the rehabilitation performed should reflect this.

Extended further it is not unreasonable to expect that people from different cultural backgrounds will have different values and therefore place a different monetary value on the benefits and costs of a project than those that would occur for a similar project in Australia. Thus rehabilitation standards particularly in developing countries are likely to be quite different to those that apply in Australia.

It is worth reflecting that the logic that allows open pits to remain at the end of mining because the benefits derived from backfilling are far out weighed by the costs involved, is not applied when it comes to other mine site rehabilitation.

How will future generations assess the benefits from our rehabilitation efforts and the capital stock we bequeathed to them?

ACKNOWLEDGMENTS

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Defining the Limits of Restoration: The Need for Realistic Goals

Joan G. Ehrenfeld[1]

Abstract

The search for a universal statement of goals for ecological restoration continues to generate discussion and controversy. I discuss the diverse roots of restoration ecology, and show how the complex lineages within the field have led to diverse, and divergent, sets of goals. I then review the three major themes that currently are used to develop statements of goals: restoration of species, restoration of whole ecosystems or landscapes, and the restoration of ecosystem services, and point out both the advantages and the limitations and problems associated with each category. Finally, I suggest that restoration ecology would be better served by recognizing that the diversity of conditions requiring restoration demands much flexibility in goal setting, and that restorationists should seek to develop guidelines for defining the sets of conditions under which different kinds of goals are appropriate. I further suggest that goals would be more easily and more appropriately set if restorationists would set forth at the outset the true scope and limitations of what is possible in a given project.

Key words: goal-setting, wetlands, conservation biology, ecosystem management, ecosystem services, landscape management.

Introduction

The specification of goals for restoration projects is frequently described as the most important component of a project, because it sets expectations, drives the detailed plans for actions, and determines the kind and extent of post-project monitoring. Not surprisingly then, the nature of restoration goals is the subject of frequent comment (Aronson & Le Floc'h 1996; Box 1996; Hobbs & Norton 1996; Kershner 1997; and many others). Although the statement of restoration goals can take many forms, it remains unclear if there is an optimal way of specifying goals, and if so, what form this statement should take. In particular, debate within the communities of both restorationists and conservationists has been concerned with the appropriate level of organization at which goals should be specified (Aronson & Le Floc'h 1996; Goldstein 1999). The level of organization (species, communities, ecosystems, watersheds, or landscapes) in turn reflects the ecological processes that practitioners may perceive as critical to the restoration effort (Allen et al. 1997; and associated papers therein).

Like conservation efforts, restoration can be oriented around particular species, can address community composition, or may be centered on whole ecosystems or landscapes (Risser 1995; Falk et al. 1996; Kershner 1997). Goals may also be stated in terms of ecosystem services. The recent attempt to place a dollar value on such services (Costanza et al. 1997) has accentuated the interest in using services as a basis for goal-setting. The debate as to which of these levels is the appropriate, optimal, or best focus of restoration and conservation goals has excited strongly worded statements (Goldstein 1999; Risser 1999; Walker 1999) addressed to both conservationists and restorationists. Here, I review the relative merits and pitfalls associated with specifying restoration goals based on species, ecosystem functions, and ecosystem services, and offer some suggestions.

Multiple Goals from Multiple Origins

The goals set for restoration projects are highly variable, in part, because restoration ecology has a complex, heterogeneous lineage. Four main themes can be discerned (Fig. 1). Each reflects a largely separate course of development of the theory and practice of restoration. One line is derived from conservation biology and is centered on the restoration of individual species. Species-centered goals for restoration are reflected in a variety of recent publications (Falk et al. 1996). Conservation biologists have also emphasized the recognition and preservation of rare or endangered communities, and restoration goals centered on the reestablishment of such assemblages can be considered a facet of the conservation-derived root of restoration. Efforts by state Natural Heritage programs to identify and restore rare communities exemplify this theme. Practitioners of this form of restoration emphasize the need to duplicate natural conditions as a standard of restoration success. Currently, the restoration of communities—particular associations of organisms—is a primary focus of many restoration efforts.

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Restoration Ecology: Repairing the Earth's Ecosystems in the New Millennium

R. J. Hobbs[1]
J. A. Harris[2]

Abstract

The extent of human-induced change and damage to Earth's ecosystems renders ecosystem repair an essential part of our future survival strategy, and this demands that restoration ecology provide effective conceptual and practical tools for this task. We argue that restoration ecology has to be an integral component of land management in today's world, and to be broadly applicable, has to have a clearly articulated conceptual basis. This needs to recognize that most ecosystems are dynamic and hence restoration goals cannot be based on static attributes. Setting clear and achievable goals is essential, and these should focus on the desired characteristics for the system in the future, rather than in relation to what these were in the past. Goal setting requires that there is a clear understanding of the restoration options available (and the relative costs of different options). The concept of restoration thresholds suggests that options are determined by the current state of the system in relation to biotic and abiotic thresholds. A further important task is the development of effective and easily measured success criteria. Many parameters could be considered for inclusion in restoration success criteria, but these are often ambiguous or hard to measure. Success criteria need to relate clearly back to specific restoration goals. If restoration ecology is to be successfully practiced as part of humanity's response to continued ecosystem change and degradation, restoration ecologists

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practice and policy. Restoration ecology is likely to be one of the most important fields of the coming century.

Key words: ecosystem repair, conceptual framework, dynamic ecosystems, restoration goals, thresholds, success criteria.

Introduction

The start of the new millennium is a useful time to reflect and take stock of where we are and where we think we should be going. The latter part of the last millennium saw unprecedented changes in all aspects of human existence on Earth, not the least of which were the increasing numbers of humans on the planet and increasing impacts of humanity on Earth and all its ecosystems. In the twilight of the second millennium we switched to a new relationship with our planet, one in which humanity dominates all other living things, sequesters the majority of the products of photosynthesis and most of the available freshwater, and increasing proportions of Earth's fish stocks for its own use (Vitousek et al. 1997). In addition, humanity is collectively changing the composition of the atmosphere and transforming the Earth's ecosystems at an unprecedented rate, and in the process causing widespread damage to the life-support systems upon which we, and every other living thing, depend.

The new millennium is thus something of a nexus for humanity, at which we need to decide whether we wish to proceed with this huge transformation of our planet, and in so doing, put our continued existence at increasing risk. Or whether we want to seek alternatives in which we aim to protect the resources, both living and abiotic, that we have left, and set about repairing some of the damage we have inflicted in the past. It is our hope that we have the collective wisdom to choose the latter course, and it is in this context that we consider the rapidly developing field of restoration ecology. If we are to persist on our planet, repair of Earth's ecosystems and the services they provide will be an essential component of our survival strategy. How well placed is the science of restoration ecology to meet this challenge? Does it have a sufficiently well-developed conceptual or theoretical base to be applied broadly? Does it have a suitable arsenal of strategies and tactics to tackle the often intractable problems it encounters? Moreover, does it have sufficient pathways into policy and practice to enable it to be applied effectively and quickly?

In this paper we examine these questions, and provide what we hope will provide pointers for the way ahead.

Landscape Restoration: Moving from Generalities to Methodologies

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KAREN D. HOLL, ELIZABETH E. CRONE, AND CHERYL B. SCHULTZ

Large-scale, landscape-level restoration actions are widely implemented but receive little attention from academic ecologists. We review the methods used to assess the role of these processes in past studies, and suggest ways to use past and ongoing restoration activities to increase our understanding of large-scale processes and improve restoration projects. To make better use of past restoration, we recommend the use of a number of alternative analytical approaches that have become widely applied in conservation biology and wildlife management but have yet to be adopted in restoration ecology.

Keywords: Bayesian statistics, information-theoretic, large-scale, reclamation

Ecological restoration is among the most expensive and extensive conservation actions worldwide. For example, between 1986 and 1997, the US Army Corps of Engineers required that 17,500 hectares of wetlands be restored as compensatory mitigation for destroyed wetlands (NRC 2001). The price tags on some high-profile restoration projects, such as the Kissimmee River in Florida and the San Francisco Bay delta, range to billions of dollars. At the Golden Gate National Recreation Area (GGNRA) in San Francisco, more than 200,000 volunteer hours are spent annually on restoration (Theresa Kreidler, volunteer manager, GGNRA, personal communication, 2002), and smaller-scale volunteer restoration efforts are increasingly common worldwide.

Despite their vast spatial and economic extent, restoration efforts have typically focused narrowly on how to restore a given piece of land or waterway, and these efforts have not received much attention from academic ecologists. It is widely recognized that the long-term success of these efforts depends on the landscape matrix in which the projects are embedded (Hansson et al. 1995, Bell et al. 1997, Hobbs 2002). Nonetheless, this recognition has rarely translated into specific recommendations that are put to use on the ground. In part, this gap between general theory and application reflects the difficulty of investigating landscape-level patterns and processes, as landscape-level processes take place at spatial and temporal scales that are not amenable to traditional methodologies of experimental design and inferential statistics. Underutilization of large-scale restoration projects as experiments to inform ecological theory also results from other reasons, such as poor documentation of restoration protocols and lack of monitoring after restoration projects are in place.

We review ways in which large-scale, landscape-level patterns and processes have been incorporated and tested in past restoration studies, discussing the strengths and weaknesses of the different approaches. We suggest ways to use past and ongoing restoration activities to test hypotheses about the importance of landscape processes for ecosystem dynamics in general and restoration success in particular. To make better use of past restoration, we recommend the use of a number of alternative analytical approaches that have become widely applied in conservation biology and wildlife management but have yet to be adopted in restoration ecology.

What do we mean by landscape restoration?

As an initial step, we systematically reviewed the existing literature for landscape restoration studies. We documented methods that have been used to incorporate landscape processes in restoration planning, and we analyzed the results of past large-scale restoration when results were reported in the ecological literature (box 1). Over the past 20 years, the terms *landscape* and *restoration* have become increasingly common in the literature (figure 1) and have been defined in different ways. We use the inclusive definition of ecological restoration from the Society for Ecological Restoration

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What is Good Ecological Restoration?

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Abstract: *The rapid rise of ecological restoration is forcing consideration of what good restoration entails. Defining an end point for restoration is as much an ethical matter as a technical one, but scientifically trained restorationists have largely ignored the former issue. I argue that good restoration requires an expanded view that includes historical, social, cultural, political, aesthetic, and moral aspects. This expanded definition is necessary at a practical level to guide practitioners in the pursuit of excellence and at a conceptual level to prevent restoration from being swamped by technological activities and projects that veer away from ecological fidelity. Ecological fidelity is based on three principles: structural/compositional replication, functional success, and durability. These principles produce effective restoration, which is a necessary but not a sufficient condition of good restoration. An examination of characteristic problems that emanate from technological practices—reverse adaptation, an attention to product at the expense of process, and the separation of actions from consequences—leads directly to an expanded, inclusive framework for restoration. The results of an inclusive restoration process set up conditions necessary for restoration to achieve both ecological fidelity and harmonious human relationships within ecosystems.*

¿Que es una Buena Restauración Ecológica?

Resumen: *El rápido desarrollo de la restauración ecológica lleva a considererar que implica una buena restauración. La definición de una meta para la restauración es un asunto tanto ético como técnico, pero los restauradores con formación científica han ignorado este último. En este trabajo argumento que una buena restauración requiere de una amplia visión que incluye aspectos históricos, sociales, culturales, políticos, estéticos y morales. Esta definición expandida es necesaria en un nivel práctico para guiar a profesionistas en la búsqueda de la excelencia y en un nivel conceptual para prevenir que la restauración se empantane por actividades y proyectos tecnológicos que se desvían de la fidelidad ecológica. La fidelidad ecológica se basa en tres principios: replicación estructural/composicional, éxito funcional y durabilidad. Estos principios producen restauración efectiva, que es una condición necesaria, más no suficiente, para una buena restauración. Un examen de los problemas característicos que emanan de las prácticas tecnológicas—adaptación en reversa, la atención al producto a expensas del proceso y la separación de acciones de las consecuencias—conduce directamente a un marco de referencia expandido para la restauración. Ello resulta en un conjunto inclusive de condiciones necesarias para que la restauración logre tanto la fidelidad ecológica como relaciones humanas armónicas con los ecosistemas.*

> *What we require, then, is neither disparagement nor celebration of place-sense but an account of those specific conditions under which it significantly furthers what Relph calls environmental humility, an awakened place-awareness that is also mindful of its limitations and respectful that place molds us as well as vice versa.*
>
> —*Lawrence Buell,* The Environmental Imagination

Paper submitted June 19, 1995; revised manuscript accepted May 21, 1996.

Introduction

The rapid rise of ecological restoration practice is forcing consideration of what good restoration entails. Most evaluative work has focused on technical performance criteria (e.g., Briggs et al. 1994), such as structural replication and composition. Other factors that have been introduced relate to ecological functions of direct interest to people (e.g., wetland filtration) and efficiency considerations (e.g., least investment of materials for greatest

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Enhancing the links between restoration ecology and ecological restoration

By Ian Lunt

For the past two years, I have taught a third-year undergraduate subject on issues in vegetation management, as part of an applied science degree. In both years I asked the students what topics they would most like to see added to the curriculum in the future. Both times, the most popular choice—by a wide margin—was bush regeneration and restoration.

At first, I was surprised by the strength of the vote for restoration and bush regeneration. But on reflection, the motivations seemed clear enough. While my perceptions of priority issues in vegetation conservation were flavoured by *past* experience and current circumstances, the students' choices were more strongly driven by their perceptions of where jobs would lie in the *future*.

In most Australian states, the historical phase of national park expansionism is drawing to a close. Consequently, the traditional market for on-ground employment in national parks agencies is likely to remain static at best. By contrast, job openings are burgeoning in fields such as bush regeneration (especially in urban areas), and catchment protection and land rehabilitation in regional and rural areas. Mooted policy options for large-scale revegetation works to help manage salinity are likely to further enhance employment prospects in revegetation.

In response to my students' requests, I began collating teaching materials on environmental restoration and rehabilitation. Naively, I endeavoured to find case studies which illustrated clear linkages between ecological theories and concepts, applied research results, and reports of inspirational on-ground activities (at both large and small spatial scales).

Not surprisingly perhaps, I found the restoration literature to be highly fragmented and disjointed. At one end of the research–practice spectrum, it is not hard to find inspirational accounts of on-ground works, especially from community and industry-based organizations (such as the Australian Association of Bush Regenerators and the Indigenous Flora and Fauna Association in Victoria) and in the seminal USA journal *Ecological Restoration* (formerly *Restoration and Management Notes*). At the other end of the spectrum, there are many excellent compilations of conceptual and philosophical works on restoration (e.g. Jordan *et al.* 1987; Throop 2000). Both fields are healthy and vibrant. But many conceptual approaches (especially the philosophical) appear to offer little of value to practitioners, and many practical accounts of challenges and successes are characterized by minimal conceptual content.

Surprisingly, I found it difficult to find relevant applied studies in the published literature to bridge the gap between the conceptual and experiential literatures. Such studies obviously exist, since the mainstream ecological journals contain lots of applied research, much of which is on enhanced management of natural ecosystems. The extensive literature on rangeland rehabilitation and fire ecology, for example, has greatly enhanced our ability to manage and rehabilitate these ecological systems. However, while most applied studies acknowledge (at least tacitly) the relevant conceptual literature (and vice versa), far fewer applied studies have demonstrable links to on-ground restoration activities (or vice versa). This schism might be expected in a young discipline, but the dichotomy is far wider than I have encountered in other fields of environmental management.

Alternatively, in more reflective moments, I wonder if this paucity of applied restoration research simply highlights the fact that, as a nation, we have not yet recognized the need to temper our ongoing ecological destructiveness, let alone seriously considered ways to redress past damages.

A common perspective in conservation biology is to ask retrospectively, 'How did we get in this hole?', whereas restoration ecologists tend to look forward and ask, 'How do we get out of it?'.

Restoration is one of many approaches to environmental conservation. In a recent article, Truman Young (2000) compared the key concerns of conservation biology and restoration ecology, based on a review of published literature. The two fields differ in a number of ways. Conservation biology studies were

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more zoological, descriptive, theoretical, and focused on small organizational scales (e.g. populations and genetics), while restoration ecology studies were more botanical, experimental and based on larger scales (populations, communities and ecosystems). More importantly, the two fields have a different temporal focus. A common perspective in conservation biology is to ask retrospectively, 'How did we get in this hole?', whereas restoration ecologists tend to look forward and ask, 'How do we get out of it?'. Answers to the first question do not necessarily help to address the second one. Clearly, both disciplines have complementary roles to play, but the comparative youth of restoration ecology restricts our ability to deal with many environmental issues.

These ruminations have focused on restoration ecology, but a number of equally important questions can be asked of ecological restoration and management. Currently, it is fashionable to justify many on-ground activities under a mystical veil of 'adaptive management'. Unfortunately this mantra is unlikely to be helpful, since many examples of 'adaptive management' seem to be based on little more than 'start it up, stuff it up and start it up again', which is exactly what *true* adaptive management (as proposed by Holling 1978; Parma *et al.* 1998) was intended to overcome. In good adaptive management, comprehensive plans are first developed using all available knowledge, and learning opportunities are built into all implementation activities, so that *all* activities provide lessons to guide future plans and activities. Learning is not an afterthought. In a similar vein, the anti-intellectualism of Australia's National Heritage Trust scheme for funding environmental rehabilitation ('Never mention the dreaded R-word, Research, and forget about monitoring and experimentation') can only have led to lessons being lost from this huge investment in on-ground works.

How can we best learn from the many rehabilitation and restoration projects that are now being conducted? How can practitioners provide feedback to influence conceptual frameworks about ecosystem functioning? Clearly, we need to maximize as many opportunities as possible to integrate learning opportunities (i.e. research) into on-ground works (and vice versa). Similarly, mechanisms need to be fostered to minimize the fragmentation which currently characterizes the restoration literature.

In this context, this new journal, *Ecological Management and Restoration*, provides a unique opportunity to fill the gaps in the young field of restoration ecology in Australasia by incorporating a wide range of voices in ecosystem management, from conceptual, descriptive, experimental and experiential backgrounds. Hopefully in future years, it will be a simple task to scan back-copies of the Journal, to find case studies which integrate all levels of ecological knowledge and experience, including conceptual ecologists (who might ask, 'How did this activity influence our conceptual models?' and vice versa), applied researchers ('Which activities gave the best ecological outcomes?'), and practitioners ('What did we achieve, and how can others learn from our experience?').

Truman Young proposed that 'the long-term future of conservation biology is restoration ecology' (Young 2000 p. 77). The motivations of new conservation graduates leave little doubt that restoration ecology will play an increasingly important role in our efforts to heal the damage which we continue to inflict on this planet. Journals such as *Ecological Management & Restoration* provide the intellectual infrastructure to support these inspirational endeavours.

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Ecological management and restoration: Assessment, setting goals and measuring success

By Richard J. Hobbs

The journal *Ecological Management & Restoration* is now in its fourth volume and has found a niche as a bridge between the science and practice of ecology in Australasia and beyond. The range and quality of material in the first three volumes has been remarkable and is a clear indication of the health and vigour of field. The application of ecology to the management and restoration of Australia's ecosystems is not new, of course, but recognition has been growing of the importance of ensuring that a clear link exists between science and practice.

This special issue brings together a series of papers that focus on the science of assessment, setting goals and measuring success. Setting goals for management and restoration is perhaps one of the most important steps in designing and implementing a project or program, and yet it is often either overlooked entirely or not done very well. There is a tendency to jump straight to the 'doing' part of a project without clearly articulating the reasons why things are being done and what the outcome should be. Ensuring that goals are both explicit enough to be meaningful and realistic enough to be achievable is a key to the development of successful projects. For most ecological management and restoration projects, goals also have to be decided inclusively so that they are agreed to by everyone with interest in the outcomes of the project.

We also need to emphasize the importance of interpreting aspirations or broad aims in terms of specific goals that can be used as 'performance targets'. Many people express aims and set objectives but forget to set goals that are measurable. Objectives are usually 'action focused' and so only measure outputs, not ecological outcomes. For instance, number of trees planted is often taken as a performance measure in revegetation projects, but this does not give any indication of the success of the planting — either in simple terms of the survival of planted material or in more meaningful measures such as the extent to which useful habitat is recreated. Goals or performance targets hence need to be related to ecological outcomes and be measurable in terms such as increases in health indicators (e.g. increasing similarity of species or structure with reference community) or decreases in indicators of degradation (e.g. active erosion, salinity extent or impact, non-native plant cover).

Setting clear goals is becoming increasingly important now that 'biodiversity targets' are being discussed, and priorities for action are being identified. Clearly there is a need for targets and priorities, but there has not been much discussion of the appropriate level for targets to be set (i.e. national, regional, local) or for priorities to be decided (i.e. national agencies, state agencies, regional groups, local groups). Clearly national and state priorities are important for deciding where public investment in management and conservation should go; but those priorities may not mesh well with community aspirations. If, for instance, priorities are set for investment in measures to protect or restore assets at risk from salinity, what happens to the areas that are not designated priorities? Priority areas are likely to account for only a small percentage of the total, meaning that large areas will not be a priority. But maybe the local community in a non-priority area thinks otherwise and sees its local surroundings as a priority! We have not been very good at debating this issue and reaching some sort of agreement on the way ahead. Maybe it's time that we did!

Of course, setting goals depends on a prior assessment of the current condition of the area or ecosystem being considered. What is its current state? What factors are causing it to degrade or preventing it from recovering? While the capture of data on aerial extent of conservation reserves, native vegetation and the like has become much more efficient and effective with remote sensing and GIS technologies, the assessment of condition within particular areas is still problematic. Can we find ways to assess current condition both at a local level and more broadly so that it can be incorporated into goal and priority setting? Several papers in this issue look at this question.

Measuring success of management and restoration depends on finding adequate approaches to the two topics already discussed — we can only measure success if we have some idea of what success looks like (i.e. we know what we are aiming for) *and* we have ways of assessing condition and detecting change in condition. There is any number of ecological variables that can be monitored, and many monitoring schemes contain great lists of all sorts of parameters to be measured, often at great expense of time, effort and money. Therefore, it's hardly surprising that many monitoring schemes are not maintained. It's often also not clear that the variables being monitored actually give

any useful information relevant to the goals of the project. The choice of parameters to be monitored thus has to go hand in hand with the setting of goals. And if the monitoring is to be done by managers or community members, there is little point in proposing complicated schemes which involve expert identifications or analyses. Finally, measuring success has to also include the possibility of measuring failure — there is little point in monitoring the progress of a project unless there is the opportunity for mid-course corrections or even complete changes in direction.

This, of course, means that adaptive management is essential — but while adaptive management is constantly advocated, it is seldom applied effectively, and will not be unless managers have the option of admitting that the current management or restoration practice is failing to deliver on the agreed goals. While we can intellectualize about all management being an experiment, the reality is that there are often adverse consequences if the experiment goes wrong, and the agency or individual involved gets the blame. We need a more open recognition of the likelihood that what we do now may not be the right thing and that we thus have to constantly think about modifying our practices. After all, what we thought was right 20 years ago has, in many cases, proved to be wrong. If we are to make progress with the important management and restoration tasks ahead of us, we need to openly admit the limits of our current understanding, take advantage of every situation to learn more about how our ecosystems work, and also learn from our mistakes. Our mistakes will only continue to be a problem if we keep repeating them.

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When is management 'restoration' and restoration, 'management'?

What is the meaning of our journal's title, 'Ecological Management & Restoration'? What is the distinction between 'management' on one hand and 'restoration' on the other?

From the journal's point of view there is no clear distinction between ecological management and ecological restoration because the two terms overlap. By 'management' we mean 'improved', ecologically sustainable management of natural ecosystems; whatever degree of utilization by humans. As this type of improved management often needs to involve the repair of component parts already damaged, 'restoration' might be assumed to be covered by it. For many, however, the term 'restoration' needs to be more explicitly stated, given the apparent need to couple the idea of management more tightly to more recent philosophies of repair, rehabilitation and even reconciliation between humankind and the rest of nature.

While debates about definitions are, at worst, unhelpful and pedantic; at best, they can provide deeper insight into ideas that can otherwise remain buried. Critics of the over-use of the term 'restoration' have some good points to make: excessive emphasis on repair and rehabilitation runs the risk of deflecting attention and resources away from protecting healthy ecosystems from damage; nostalgic attachment to a single historic state can waste critical opportunities for conserving a variety of possible states; and excessive preoccupation with redressing past wrongs can blind us to the need for creative, lateral and future-directed solutions.

Yet here is where already-existing definitions can help. A number of workers have developed analyses and definitions of restoration in the context of ecological potential, degree of degradation and societal expectations (Cairns 1989, 1991, 1993; Bradshaw 1992; Wali 1992; Aronson *et al.* 1993; Brown & Lugo 1994; Hobbs and Norton 1996). Most see ecological restoration as a process of repairing the composition, structure, function and dynamics of pre-existing indigenous ecosystems. But most would agree that restoration potential exists as a *continuum* over time and space, dictated by degradation degree and practicality. Certainly, most would see restoration as distinct from 'rehabilitation' or 'reclamation' in that the latter activities do not necessarily lead to the repair of pre-existing indigenous ecosystems. However, where the intention is an incremental improvement over time and partial treatments do not prejudice future restoration potential, many would argue that the difference between restoration and rehabilitation is less distinct and less important.

Our journal title, therefore, makes no clear distinction between improved management and restoration. Both are needed. A broader view of restoration would incorporate not only projects which fully conserve natural areas and attempt full restoration where these are damaged but would also incorporate projects which incrementally rehabilitate, by best-practice management decisions, already-converted landscapes. Converted landscapes managed in this way would never be 'restored' in the strict sense and would continue to be primarily managed for human use. Incrementally applying 'restoration' strategies over the decades, however, may allow these sites to contribute to the maintenance and reintegration of some ecosystem function at a higher ecological level. Such an outlook allows the journal to walk the same tightrope our whole society is walking — in an attempt to develop solutions that may ameliorate destruction at the same time as actively pursuing 'restoration' of our preferred futures.

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THE HILL RIVER PROJECT AND THE PROPOSED CONSERVATION RESERVE AT LESUEUR

A report to the Environmental Protection Authority from the Department of Conservation and Land Management

Edited by

Andrew A Burbidge and Stephen van Leeuwen

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Repair versus despair

Hope and reality in ecological management and restoration

By Richard J. Hobbs

It is with great pleasure that I introduce this new journal. It has been developed jointly by the Ecological Society of Australia and the Land and Water Resources Research and Development Corporation, with Blackwell Science Asia as a joint venture partner. The overall purpose of the Journal is to improve long-term management and restoration of ecosystems by providing an effective link between the findings of scientific research and the needs and actions of on-ground managers.

The new Journal appears at a time when the need for effective solutions to important environmental and social issues has never been greater. Globally, the human population has recently passed six billion, the reality of human-induced climate change is widely recognized, and human activities are changing the earth's ecosystems, both terrestrial and aquatic, at a pace unprecedented in history. Locally, this translates into problems of land degradation, urban congestion, loss of production, loss of species, transformation of ecosystems, and changes to the social and economic fabric of entire regions.

It is often tempting to view these changes in a despondent way. How, in the face of the huge and potentially irreversible changes we are forcing on the earth's ecosystems, can we hope to sustain food production, the quality of human life and the variety of non-human life? Although the problems have been known for some time, the extent of the remedial actions being taken is frequently woefully inadequate. While this is sometimes attributed to lack of knowledge or the unavailability of practical and affordable solutions, often we do have sufficient knowledge to make reasonable assessments of what needs to be done. Instead, the block to action stems from an inability or unwillingness of individuals, institutions and society as a whole to change from current behaviours. If we take salinity as an example, we have known for some time that large proportions of the Western Australian wheatbelt are under threat from rising watertables and salinization, and we have broadly recognized the array of treatments necessary to treat, or at least slow down, the process. Part of the solution is broad-scale replacement of perennial vegetation in the landscape, probably to a level of 20% or more in most areas. While the Landcare movement and local community action has greatly increased the numbers of farmers participating in revegetation and other remedial actions, the results of these actions still sum to only approximately one per cent of the farming area, or at least an order of magnitude less than is required (Hobbs & Saunders 1999). Similar analyses could be conducted on most other environmental issues.

In the face of this, despondency and despair are understandable responses. We certainly have to recognize the enormous reality of the problems facing us. We also probably have to recognize that some of the problems we've created are not amenable to simple solutions — and some may not be amenable to any sort of solution that lies within a reasonable budget and can be applied in a reasonable timeframe. For instance, groundwater rise in some areas of Western Australia may be happening at such a rate that there are no effective treatments that can reverse the situation. In such cases, we need to move from the 'fix-it' mindset which always hopes for and seeks a workable solution, and instead figure out what to do for these 'terminally ill' landscapes and the people that depend on them for a living.

Alongside these sombre thoughts, however, should be more positive and hopeful thoughts about what can be done and how things can change for the better. While we seem to be a long way from implementing effective solutions to many problems, the seeds of change are already apparent. Many people are managing ecosystems more effectively and the potential for ecosystem repair is immense. True, we often need more detailed information on particular situations, and we could still develop much more efficient and effective methods and tools. However, we definitely need to learn as we go, rather than waiting for the final answer or ultimate piece of data — after all, the answer keeps changing, and we'll always need more data!

We should also be hopeful that the necessary changes in individual and societal attitudes and behaviour are possible. One need only think of the radical social and political changes that have occurred this century to see the scope for change — for instance, improved rights and conditions for women across the world, the integration of black people in the United States and the fall of the Berlin Wall to name but a few. Of course, much more remains to be done and changes often bring new problems to the surface, but the point is that things can change, and often rapidly.

A key element in change is hope. I referred to this in a recent President's report in the Ecological Society of Australia Bulletin, but I think it bears repeating here. In their recent book, Peter Newman and Jeffrey Kenworthy suggest that a key element of solving environmental problems is 'practicing hope rather than despair' (Newman & Kenworthy 1999). They suggest that 'hope is not blind optimism; it recognizes the depth of the problem and refuses to

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accept defeat. It is not a feeling — it is a choice', and they quote from a person called Jim Wallis, who works with the urban poor in Washington D.C., who discusses hope thus:

> *Hope is not just a feeling or a mood, but the very dynamic of history. It is the energy of transformation and is the door from one reality to another. What seems impossible looking towards it (e.g. the abolition of slavery in the U.S.) was inevitable with hindsight.*
>
> *Between the impossible and the inevitable, between the impossible and possible is a door, and that door is hope. The possibility of the transformation of history lies at that door. On one side of the door of hope is nonsense. On the other side of the door is the best news ever heard. Hope not believed is nonsense. Hope that is believed is transformation.*
>
> *Victories and transformation always seem impossible to begin with. They only become possible by stepping through the door of hope. To walk through the door of hope you have to first see it and believe that there is something on the other side. It is never easy. It is always hard! This is particularly so for the first few to walk through. Others then find it easier to follow. And that's how historical changes take place.*
>
> *Hope is believing in spite of the others, and watching the others change.*
> [Wallis 1994, cited in Newman and Kenworthy (1999)]

If we are to have hope, I think we need three main ingredients; namely vision, strategy and tactics. The dictionary definition of vision is 'statesmanlike foresight', and implies the ability to see a goal towards which efforts should be directed. Strategy and tactics are concepts used in warfare, sport and other endeavours. Broadly stated, strategy in warfare is the planning, coordination, and general direction of military operations to meet overall political and military objectives. Tactics implement strategy by short-term decisions on the movement of troops and employment of weapons on the field of battle. As Carl von Clausewitz put it, 'Tactics is the art of using troops in battle; strategy is the art of using battles to win the war.'

The broad vision thus answers the question 'Where do we want to go?'. The strategy answers the question 'How do we want to get there?' and tactics provide the answer to 'What actions do we need to take?'. I suggest that all three elements are essential for success, and part of the failure to date can be attributed to the lack of one of these elements. I and colleagues have argued elsewhere that a lack of vision has hampered our attempts to develop strategies for effective management in our agricultural landscapes (Saunders 1996; Hobbs & Saunders 1999). Similarly, strategies are useless unless they are backed up with adequately resourced and planned tactical measures.

This new journal will, we hope, provide contributions in all three of the above areas; that is, developing visions of what we want our ecosystems and landscape to be like, presenting strategies for achieving the goals which arise from these visions, and discussing tactical measures for the management and repair of systems. It seems that in today's world we can't count on our political leaders to provide visions for the future to which we can all agree. Thus, we all need to be in the business of helping society to develop its own set of visions and making sure that we have adequate strategies & tactics available to meet future challenges as well as today's challenges. This is something to which everyone can contribute, including scientists, practitioners, policy makers, economists and many others. If we can collectively concentrate on repair rather than despair, then we will have hope for the future. If this journal helps to achieve this, then it will have been successful.

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Restoration ecology: A two-edged sword?

By David Goldney

As a small boy my father taught me to wet my finger in my mouth and then to hold it above my head to determine the direction of the wind. But it was not enough to have this additional datum if I did not know the cardinal directions of where I was then standing. As ecologists and land-managers we need to be able to discern the winds of change that are creating new ways of perceiving people-dominated landscapes across the world, their management and restoration - a paradigm shift that is essential for understanding the principles of landscape restoration and management.

Determining pattern in the midst of so much landscape noise is difficult but not impossible. Ecologists are moving from autecology and community ecology to reclaim the understanding of landscapes in ecosystems terms and the recurring patterns within those landscapes. We are moving from responding to issues in the landscape (e.g. salinity, erosion, species loss) to system understanding and restoration and management based on that understanding. We are moving from local, often point-based environmental management plans and actions, to catchment or bio-regional planning and long-term strategic outcomes. We are moving from 'greens' lying down in front of bulldozers to 'greens' proactively and creatively working with landholders.

In summary, we are moving out of an Age of Degradation and into an Age of Restoration (Goldney & Bauer 1998); but do we have the relevant knowledge and interdisciplinary capacity to respond to the challenge?

Learning to read landscapes – and teaching others. The 'landscape ecology' paradigm is not well understood by either ecologists or land managers and that continues to be a challenge since we ecologists have been found wanting at a most critical moment in our history. We are learning to read landscapes, to value the long-term anecdotal experience of landholders (often disregarded by researchers), to revisit the systems thinking of elementary courses in ecology and to integrate essential interdisciplinary information such as hydrology, geomorphology, spatial statistics, sociology and policy formulation!

Conversely, while we are learning to be more inclusive of other disciplines, it is also necessary for us to assist those working in other disciplines to adequately incorporate ecological information. It seems counterproductive to me, for example, for resource agencies to develop a strategy for salinity or erosion abatement - or for a landholder to develop a farming plan - if these do not simultaneously address incontrovertibly linked biodiversity issues (system functioning and species and community conservation). Also appearing are myriads of species recovery plans, often very useful documents, but few if any being exemplars of holistic landscape-catchment planning where a range of values and desired outcomes are optimized. There are very few competent accounts as to how landscapes do function (e.g. Ludwig *et al.* 1997) and no exemplars of community owned and driven future scenarios of how Australian landscapes should look and effectively function in the future.

We need interactions with other stakeholders for two reasons: to expand our own understanding of the more complex picture that is emerging; and to disseminate biodiversity information further than our traditional target audiences.

The two-edged sword for ecologists and land-managers is:

1. If we do not understand the landscape systems in which we work and research, our contributions to resource management committees and strategic planning endeavours may prove counterproductive in landscape terms and result in adding further to landscape 'noise'. At best there will be additions to local knowledge without the ability to enable technological transfer to occur to other disciplines or landscapes.
2. We need to ensure that there are tertiary courses (graduate and postgraduate level) that meet the needs of the Age of Restoration. While there are undergraduate units in landscape ecology, there is not yet to my knowledge a fully fledged Restoration Ecology or Landscape Ecology course in Australia, let alone one that also embraces needed aspects of agricultural science, sociology and policy formation whose graduates can view the landscape in a holistic manner. The wave of environmental, ecological and agricultural science graduates that came into the workplace from the 1970s to the present have made a stirling contribution to resource management. But we now know that our needs are far more complex. Will we as ecologists rise to the challenge?

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Adaptive management: a synthesis of current understanding and effective application

By E. Sabine G. Schreiber, Andrew R. Bearlin, Simon J. Nicol and Charles R. Todd

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Summary Adaptive management (AM) remains a commonly cited, yet frequently misunderstood, management approach. The aim of AM is to improve environmental management through 'learning by doing' and understand the impact of incomplete knowledge, but AM more commonly consists of ad hoc changes in managing environmental resources in the absence of adequate planning and monitoring. Here, we trace and review the development of AM, the central roles of consultation, collaboration and of monitoring, and of quantitative models and simulations. We identify a series of formalized, structured steps included in one AM cycle and review how current AM programs build upon such cycles. We conclude that the best AM outcomes require rigorous and formalized approaches to planning, collaboration, modelling and evaluation. Finally, simulating potential outcomes of an AM cycle in the presence of existing uncertainty can help to identify management strategies that are most likely to succeed in relation to clearly articulated goals.

Key words *adaptive environmental assessment and management, modelling, uncertainty.*

Introduction

The development of adaptive management (AM) reflects growing concerns over the past 30 years about the effects of people on ecosystems (Holling 1978; McDonnell & Pickett 1993). Basic empirical ecological research or the development of general ecological theory alone was thought unlikely to improve the way that natural resources were managed (Holling 1978; Walters 1986), and investigations into how natural resources respond to management were deemed necessary. AM is an approach that includes scientific methodologies in the design, implementation and evaluation of managment strategies. The other essential components of AM build on methods from a wide range of disciplines, including the natural and social sciences and recognize the importance of institutional and social structures to management and policy decisions.

Ecological research can contribute to the understanding of environmental effects of management by suggesting plausible ecological mechanisms. For example, the management of algal blooms in some Amer-

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logical knowledge was used to manipulate trophic structure in order to reduce risks of algal blooms (Carpenter *et al.* 1999). Nevertheless, connections between ecological understanding and environmental management are rarely made explicit, and questions of particular management relevance, such as whether human exploitation of natural resources is essentially different from natural disturbances, remain unanswered (Fairweather 1998).

Understanding what adaptive management is and how it can be applied requires more than a mere definition of words. The terms 'adaptive environmental assessment and management' (AEAM) and 'adaptive management' (AM) are used interchangeably throughout the literature. Here we do not enter the debate on differences between these terms, as we do not believe that a re-definition of these terms will assist in identifying the benefits and common misunderstandings of this management approach. Instead, we review the development of AM in order to synthesize our current understanding and improve future application of this approach.

This synthesis will concentrate on the two primary reasons for the implementation of AM: to improve environmental management, and to understand the impact of incomplete knowledge.

The origin of adaptive management

The first synthesis of AM was a response to the burgeoning realization in the 1970s that natural resources were limited and their continued use demanded prudent management (Holling 1978). The integration of environmental, social and economic issues in the development of environmental policy and management strategies was identified as critical, as well as the iterative cycle in which managerial experiences gained in the application of environmental policy should contribute to further policy development (Holling 1978). Examples of AM were not restricted to natural resource management, but also included management issues dominated by economic and social issues, for example the development of an Austrian mountain village (Holling 1978).

The importance of design, consultation and incorporation of existing knowledge and uncertainties was stressed by Walters's (1986) separation of 'adaptive environmental assessment' (AEA) from management applications. In particular, Walters (1986)

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TURNER REVIEW No. 9
Ecosystem services: an ecophysiological examination

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Abstract. This review aims to discuss ecosystem services, provide illustrative case studies at catchment and local scales and present future research needs. This review discusses the following:

(1) Ecosystem services (ES) are those goods and services that are provided by or are attributes of ecosystems that benefit humans. Examples of ES include the timber derived from a forest, the prevention of soil and coastal erosion by vegetation and the amelioration of dryland salinity through prevention of rises in the water table by trees. The provision of ES globally is in decline because of a lack of awareness of the total economic value of ES in the public, policy and political fora.

(2) Providing a scientific understanding of the relationships among ecosystem structure, function and provision of ES, plus determining actual economic value of ES, are the central challenges to environmental scientists (including triple-bottom-line economists).

(3) Some ES are widely dispersed throughout many different ecosystems. Carbon accumulation in trees and the contribution of biodiversity to ES provision are two examples of highly dispersed attributes common to many ecosystems. In contrast, other ES are best considered within the context of a single defined ecosystem (although they may occur in other ecosystems too). Mangroves as 'nursery' sites for juvenile fish is one example.

(4) Examples of catchment-scale and local-scale provision of ES are discussed, along with future research issues for the nexus between ES and environmental sciences.

1.0. Introduction

Student numbers in Business and IT degrees are increasing as students pursue degrees with obvious economic links. In contrast, student numbers in science, including biology and ecology, are falling, partly because of the lack of perceived economic value of these topics. Ironic it is, then, to see that ecosystem services have been valued at between US$16–54 trillion per year (with an average of $33 trillion per year), more than the global gross national product (Costanza *et al.* 1997). It is doubly unfortunate that the study of ecology and ecophysiology is not perceived as contributing directly to national economies and policy development. Doubly unfortunate because not only is the study and funding of botanical research suffering, but also because the lack of understanding of the economic value of ecosystem services to economic health is causing long-term impoverishment of both the economy and ecology of developed and developing countries alike.

We present this review in four stages (Sections 2–5). We begin by (i) defining ecosystem services and (ii) considering issues pertinent to valuing ecosystem services in the marketplace. In doing so we aim to establish the importance of ecosystem services as a valuable way of thinking and to establish a common currency among policymakers, managers, researchers and the public as to the value of natural systems. To highlight the broad applicability of this way of thinking, we provide case studies that cover a range of different ecosystem types and consider ecosystem services at both (iii) catchment (across several ecosystems) and (iv) individual ecosystem scales. These will provide examples of ecosystem services and illustrate how basic science in ecology and ecophysiology has illuminated our understanding of ecosystem services.

Our catchment-scale (terrestrial) case studies have a distinctly Australian focus, whereas our studies of particular ecosystems, which are mostly aquatic, have a more international flavour. We have taken this approach because there have been far fewer studies of aquatic ecosystem services than terrestrial ecosystem services in Australia and we argue that aquatic ecosystems are more universal across

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Table 9: Total Land Area Disturbed

Annual Environmental Reporting System

TOTAL AREAS REPORT

YEAR 2002
Results for whole State

Activity	2002 Annual (ha)			Cumulative Total to 31/12/2002		
	Disturbed by mining operations	Preliminary Rehabilitation land forming	Revegetation	Disturbed by mining operations	Preliminary Rehabilitation land forming	Revegetation
BOREFIELDS AND PIPELINES	64	162	5	1749	411	79
CAMP SITE	14	55	16	1358	391	302
EXPLORATION	122	50	28	4912	1498	830
MINESITE INFRASTRUCTURE	5406	683	479	50628	5218	3873
PIT	1525	1304	908	34900	8642	5981
TAILINGS/EVAP DAMS	8768	232	149	33256	2350	1791
WASTE DUMPS/HEAP LEACH	1215	877	612	35380	16745	10803
TOTAL	17114	3363	2197	162183	35255	23659

Environmental Management in the Mineral Industries

Major Achievement for 2003/04

Environmental Assessment of Onshore Mining Projects

A review was finalised of the Memorandum of Understanding between the Department and the Environmental Protection Authority for the environmental assessment of onshore mining projects in Western Australia. The revised MOU is ready for signing in the next reporting year.

Legislation

The Department played a key role in working with industry stakeholders and the Department of Environment in the development of new land clearing regulations.

The Department has developed a draft administrative agreement with the Department of Environment for the management of land clearing permits for mining projects. The agreement will clarify the roles of the two agreements and ensure the industry has a clear understanding of the process.

An administration agreement that deals with the issuing of tenements and mining approval in water reserves was signed between the Waters and Rivers Commission and the Department.

Regulatory Services

The Department:

- Approved 345 Notices of Intent (NOIs)
- Conducted 113 Annual Environmental Reviews
- Carried out 383 general inspections and audits of mining and exploration operations
- Processed 1010 Ground Disturbing Approval Applications for exploration activities
- Dealt with 15 complaints of an environmental nature, ranging from unauthorised mining to environmental incidents
- Initiated one work direction.

At the end of June, the Department held over 3 400 Unconditional Performance Bonds with a total value of $379 million, to cover the cost of post-operational remediation and rehabilitation should the operators fail to meet their commitments and conditions of approval. Based on these figures, an average bond of $2 289 is held for each hectare of disturbance on sites covered by the *Mining Act 1978*. (See Table 7 below)

The Department received payment of around $1.8 million in environmental bond monies in respect to the clean up of tailings and other disturbances on tenements surrendered at various mine sites around the State. Approximately $235 000 of the $2.25 million held in trust by the Department was accessed during the year to progress the remediation and rehabilitation of seven identified areas.

Demolition of the old mine infrastructure at Wittenoom was completed and all materials buried on site. There are ongoing concerns in regard to access to the area as large dumps of asbestos tailings still remain. The Department is providing technical advice and assistance to the Wittenoom Strategy Committee on the development of a long-term strategy to manage the tailings.

Stakeholder Liaison

A number of meetings were held with the Chamber of Minerals and Energy endeavouring to define the process for achieving successful and sustainable mine closure.

The Department participated in the Minerals Industry Forum, commenting on the framework for sustainable mine management. This was an opportunity for the regulator to contribute to the development of this framework, which if accepted will replace the current Minerals Council of Australia (MCA) environmental code of practice.

Through these meetings it is hoped to translate the expectations of the community regulator and industry into practical outcomes. This work is ongoing and includes participation in the development of the MCA Framework for Sustainable Development.

Exploration and Promotion of Exploration

The Department worked closely with a number of interested mining companies, and the Chamber of Minerals and Energy, to develop a new process of Exploration Environmental Management Plans, whereby companies holding numerous tenements under the same licence can obtain up-front approval to cover their full exploration project for up to five years. This alternative process will allow a more strategic approach to environmental management issues with demonstrated long-term benefits to industry, and the Department, particularly in the allocation of resources. Relevant guidelines are expected for release to industry by December 2004.

Table 7: Total Land Area Disturbed

Year 2003

Results for Whole State

	2003 Annual (ha)			Cumulative Total (ha) to 31/12/2003		
Activity	**Disturbed by mining operations**	**Preliminary rehabilitation land forming**	**Revegetation**	**Disturbed by mining operations**	**Preliminary rehabilitation land forming**	**Revegetation**
Borefields and pipelines	11	4	6	1 932	415	85
Camp site	18	6	4	1 376	397	306
Exploration	61	28	7	4 989	1 526	837
Mine site infrastructure	506	209	191	51 295	5 290	4 062
Pit	766	314	110	35 820	8 844	6 155
Tailings/evaporation pond	309	345	307	33 734	2 779	2 133
Waste dumps/ heap leach	829	827	896	36 439	17 931	11 720
Total	2 500	1 733	1 521	165 585	37 182	25 298

ha seem small

1,500 ha x $5,000

$ 7.25 m

Rehabilitation Expenditure

As at November 2003, 38 companies were signatories to the *2000 Australian Minerals Industry Code for Environmental Management*.

These represent over 92 per cent of annual production in the Australian minerals industry, and apply the Code at numerous sites, both domestically and overseas.

The annual expense for rehabilitation in 2002/03 was $344 million.

The figures presented in this survey cover only part of the industry's total environmental expenditure.

In 2002/03, the industry provided $344 million for expenditure on rehabilitation. The accumulated balance of the provision for rehabilitation expenditure rose to $2,155 million at the end of 2002/03. The strong rise in the balance over the past few years is consistent with a continued focus on environmental rehabilitation by the minerals industry. The balance is provided for the purpose of rehabilitation and can be expected to be drawn down in future years.

Expenditure on rehabilitation is projected to decrease next year, although some respondents are conservative in their forecasts in this area. In addition, a decrease would reflect, to some extent, the increasingly targeted, cost effective and efficient rehabilitation methods developed by respondents, as well as the 'maturing' of some provisions to reflect the full anticipated future costs of rehabilitation.

It should also be noted that the figures presented in this survey cover only part of the industry's total environmental expenditure (see note at the end of this section).

The future of the minerals industry is inseparable from the global pursuit of sustainable development. The idea of meeting the needs of this generation without compromising the ability of future generations to meet their own needs has become a key concept in the drive to maintain and enhance industry's 'social licence to operate'.

The Australian minerals industry is well progressed in responsible environmental management. The role of the *Australian Minerals Industry Code for Environmental Management* in providing a framework for continual improvement is becoming better understood and accepted by industry and its stakeholders. A common criticism of the Code, however, has been that it does not go far enough in addressing broader social issues such as the rights of local and Indigenous communities.

The Australian and global Mining, Minerals and Sustainable Development (MMSD) reports contained a strong reference to the role and value of a collective industry commitment to a sustainable development code or protocol. In response to this the International Council on Mining and Metals (ICMM) has developed a Principles-based Sustainable Development Framework. Modelled on the Australian Environmental Code, and developed with significant input from the Minerals Council, the ICMM Framework forms the basis of the industry's international commitment to meeting our social, environmental and economic stewardship responsibilities, and to maximising the industry's contribution to sustainable development.

In response to these international developments, the Australian minerals industry has commenced the development of an Operational Framework for Sustainable Development Implementation. The initiative is designed to assist implementation of the ICMM Sustainable Development Framework at the site level while also providing a framework for sustainable development implementation for those companies who are not currently engaged with the ICMM process. In doing so the SD Code directly aligns with the Principles of the ICMM Framework, thereby ensuring that the Code enhances rather than duplicates the ICMM Sustainable Development Framework and its value to industry.

The Operational Framework for Sustainable Development Implementation will be developed using a two-tiered process of stakeholder engagement throughout 2004.

The idea of being granted a 'social licence to operate' by taking into consideration the community response to resource development is an increasingly important determinant of successful project development. In response to this many leading mining companies are going beyond their traditional roles and investing in community development to demonstrate their long-term commitment to capacity building in local communities, and to genuine processes of engagement and negotiation regarding issues such as access to lands.

In light of this, the Minerals Council's sustainable development activities during 2003 focussed heavily on building industry capacity across the social and environmental dimensions of sustainable development.

Key areas of focus included the development of a comprehensive resource for member companies to assist in Agreement making with Indigenous communities; an identification of key challenges to the attraction and retention of a skilled and diverse workforce; and the continuation of a range of initiatives to strengthen the industry's leading role in public reporting and continually improve its stakeholder engagement.

The Minerals Council also showcased the industry's performance across the key social and environmental areas at the 2003 Sustainable Development Conference, *Value Through Sustainable Development* (incorporating the 28th Annual Environmental Workshop). Attracting approximately 400 delegates from across Australia as well as key mineral producing regions internationally, the Conference successfully facilitated the transfer of industry knowledge in three key areas, namely management systems, eco-efficiency and community relations.

The Australian minerals industry undertakes considerable research and development and professional training in areas related specifically to improving environmental performance. Research and training is undertaken directly by companies and through sponsorship and support of research and training institutions such as the Australian Centre for Mining Environmental Research and AMIRA International.

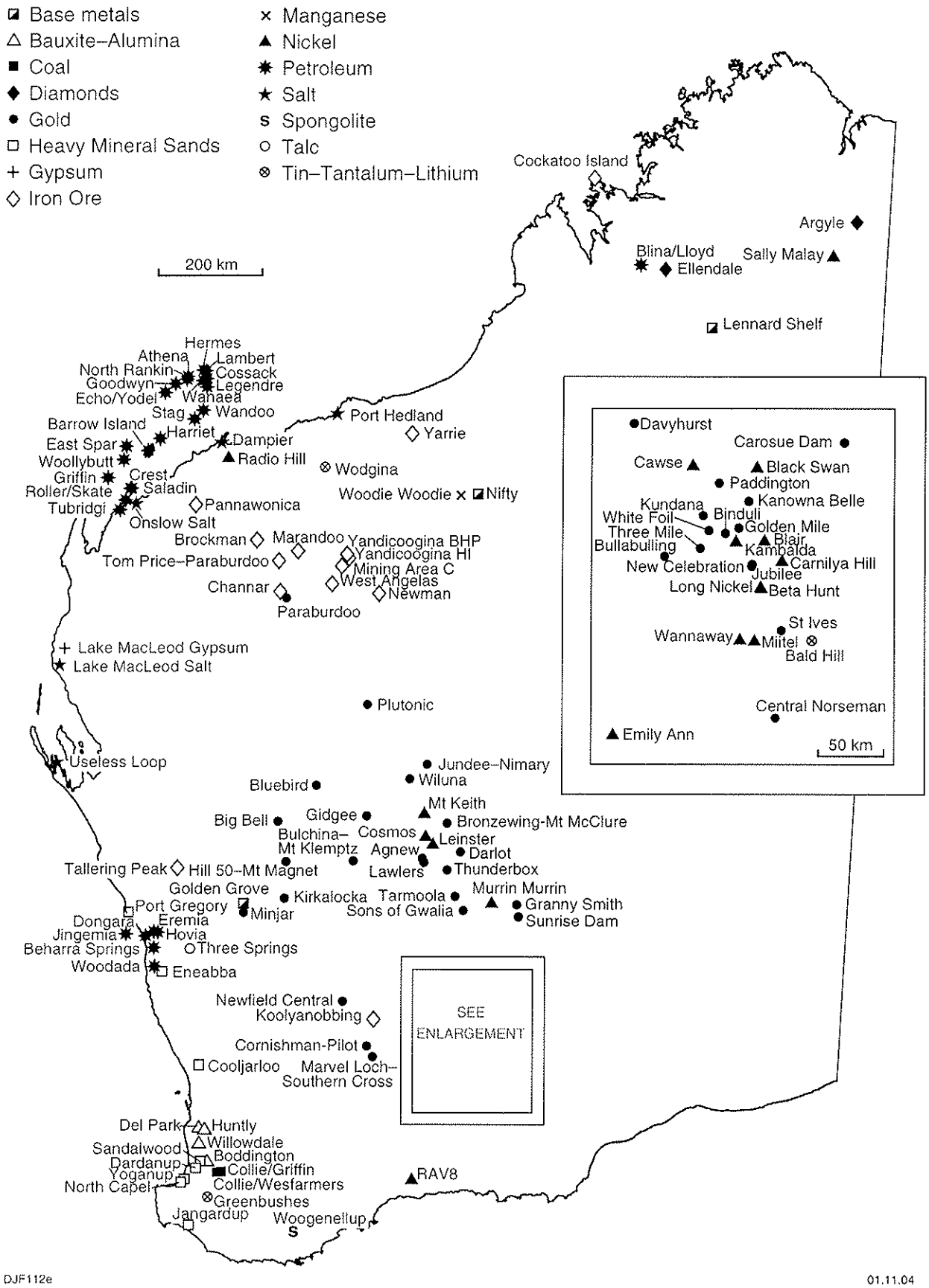
Note: The figures presented in this survey cover only part of the industry's total environmental expenditure. In addition to minesite rehabilitation, substantial environmental expenditures are incurred in research, pollution monitoring and control, clean up and in capital expenditures designed to minimise the environmental impact of mining and minerals processing plant and equipment. In providing figures, respondents may also distinguish between the amount spent and the amount charged to the statement of financial performance. Under the 'full liability method' of accounting for rehabilitation, a provision is made for the total present value of the future cost of repairing past damage and other related shut down costs as soon as the commitment is incurred, and the amount capitalised under this method is amortised over the life of the mine.

Rehabilitation Expenditure	2003/04 Forecast $ million	2002/03 $ million	2001/02 $ million	2000/01 $ million
Annual Provision [a]	154	344	229	185
Accumulated Balance of Provision [b]		2,155	1,838	1,619

Note: (a) Annual provision for rehabilitation represents the amount charged to the statement of financial performance during the period, which increases the total rehabilitation provision. Actual payments made will be made directly from the total rehabilitation provision, and will not necessary equal the amount charged to the statement of financial performance.

(b) The change in the mix of respondents means the accumulated balance of provision figure at the beginning of 2002/03 differs from the 2001/02 end of year figure.

MAJOR MINERAL AND PETROLEUM PROJECTS IN WESTERN AUSTRALIA



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restionaceous /ˌrɛstiəˈneɪʃəs/, *adj.* of or pertaining to rush-like plants belonging to the family Restionaceae.

restitute /ˈrɛstətjut/, *v.t., v.i.* to make restitution (of).

restitutio in integrum /rɛstəˌtjutsioʊ ɪn ˈɪntəgrəm/, *n.* the rescinding of a contract when the parties can be put back into their original positions, entailing a giving back and taking back on each side. [L: restoration to the original position]

restitution /rɛstəˈtjuʃən/, *n.* **1.** reparation made by giving an equivalent or compensation for loss, damage, or injury caused; indemnification. **2.** the restoration of property or rights previously taken away, conveyed, or surrendered. **3.** restoration to the former or original state or position. **4.** *physics.* the return of an elastic material to its original form when released from strain. [ME *restitucion,* from L *restitūtio* a restoring]

restive /ˈrɛstɪv/, *adj.* **1.** restless; uneasy; impatient of control, restraint, or delay, as persons. **2.** →**refractory.** **3.** refusing to go forward, as a horse. [REST[2] (def. 3) + -IVE; replacing ME *restif* stationary, balking, from OF *restif* inert] – **restively,** *adv.* – **restiveness,** *n.*

restless /ˈrɛstləs/, *adj.* **1.** characterised by or showing inability to remain at rest: *a restless mood.* **2.** unquiet or uneasy, as a person, the mind, heart, etc. **3.** never at rest, motionless, or still; never ceasing. **4.** without rest; without restful sleep: *a restless night.* **5.** characterised by unceasing activity; averse to quiet or inaction, as persons. – **restlessly,** *adv.* – **restlessness,** *n.*

restless flycatcher /– ˈflaɪkætʃə/, *n.* a black and white flycatcher, *Myiagra inquieta,* which hovers a metre or so above the ground when feeding, and which has a call resembling the sound a grindstone makes; scissors grinder.

rest mass /ˈrɛst mæs/, *n.* the mass of a body which is at rest relative to the observer.

restock /ˌriˈstɒk/, *v.t., v.i.* to stock again; replenish.

restoration /rɛstəˈreɪʃən/, *n.* **1.** the act of restoring; renewal, revival, or re-establishment. **2.** the state or fact of being restored. **3.** a bringing back to a former, original, normal, or unimpaired condition. **4.** restitution of something taken away or lost. **5.** something which is restored. **6.** a representation or reconstruction of an ancient building, extinct animal, or the like, showing it in its original state. **7.** a putting back into a former position, dignity, etc. **8. the Restoration, a.** the re-establishment of the monarchy in England with the return of Charles II in 1660. **b.** the period of the reign of Charles II (1660-85), sometimes extended to include the reign of James II (1685-88). –*adj.* **9.** (*cap.*) denoting, pertaining to, or produced during the period of the Restoration, esp. a form of drama: *Restoration comedy.*

restorative /rəˈstɒrətɪv/, *adj.* **1.** serving to restore; pertaining to restoration. **2.** capable of renewing health or strength. –*n.* **3.** a restorative agent. **4.** a means of restoring a person to consciousness.

restore /rəˈstɔ/, *v.t.,* **-stored, -storing. 1.** to bring back into existence, use, or the like; re-establish: *to restore order.* **2.** to bring back to a former, original, or normal condition, as a building, statue, or painting. **3.** bring back to a state of health, soundness, or vigour. **4.** to put back to a former place, or to a former position, rank, etc. **5.** to give back; make return or restitution of (anything taken away or lost). **6.** to reproduce, reconstruct, or represent (an ancient building, extinct animal, etc.) in the original state. [ME, from OF *restorer,* from L *restaurāre* restore, repair] – **restorer,** *n.*

restrain /rəˈstreɪn/, *v.t.* **1.** to hold back from action; keep in check or under control; keep down; repress. **2.** to deprive of liberty, as a person. [ME *restreyn(en),* OF *restrei(g)n-,* stem of *restreindre,* from L *restringere*] – **restrainable,** *adj.* – **restrainedly,** *adv.*

restrainer /rəˈstreɪnə/, *n.* **1.** one who or that which restrains. **2.** a chemical, as potassium bromide, added to a photographic developer to retard its action.

restraining order /rəˈstreɪnɪŋ ɔdə/, *n.* an injunction made by a court, as one to prevent a husband from visiting his wife during divorce proceedings.

restraint /rəˈstreɪnt/, *n.* **1.** restraining action or influence: *freedom from restraint.* **2.** a means of restraining. **3.** the act of restraining, or holding back, controlling, or checking. **4.** →**child restraint. 5.** the state or fact of being restrained; deprivation of liberty; confinement. **6.** constraint or reserve in feelings. [ME *restraynte,* from OF *restraint(e),* n. use of pp. of *restraindre* restrain]

restraint of trade, *n.* the restriction of business activity and freedom to compete.

restrict /rəˈstrɪkt/, *v.t.* to confine or keep within limits, as of space, action, choice, quantity, etc. [L *restrictus,* pp., restrained, restricted] – **restricted,** *adj.* – **restrictedly,** *adv.*

restricted /rəˈstrɪktəd/, *adj.* **1.** *Mil.* limited to persons authorised to have access, as to information, etc. **2.** limited; confined. **3.** admitting only members of certain groups, as a restricted club. **4.** of or pertaining to a horse which is eligible to compete in certain races by complying with the conditions of that race.

restricted area /– ˈɛəriə/, *n.* **1.** an area in which traffic is not permitted to exceed a certain speed. **2.** any area from which certain groups or races of people are excluded.

restricted exhibition /– ɛksəˈbɪʃən/, *n.* of or pertaining to a film considered suitable for adult viewing only. *Abbrev.:* R.

restricted race /– ˈreɪs/, *n.* a horserace in which entry is restricted to horses who have not previously entered in other specified races or won specified amounts of prize money.

restricted title /– ˈtaɪtl/, *n.* the statement at the beginning of a parliamentary bill of its scope, which delimits the debate and the amendments which can be moved.

restriction /rəˈstrɪkʃən/, *n.* **1.** something that restricts; a restrictive condition or regulation; a limitation. **2.** the act of restricting. **3.** the state of being restricted.

restrictive /rəˈstrɪktɪv/, *adj.* **1.** tending or serving to restrict. **2.** of the nature of a restriction. **3.** expressing or implying restriction or limitation of application, as terms, expressions, etc. – **restrictively,** *adv.*

restrictive clause /– ˈklɔz/, *n.* a relative clause, usu. not set off by commas, which identifies the person or object named by the antecedent (opposed to *descriptive clause*).

restrictive covenant /– ˈkʌvənənt/, *n.* a negative covenant entered into by the purchaser of land, as a promise to the vendor not to build a factory on it.

restrictive practice /– ˈpræktəs/, *n.* **1.** a practice on the part of the members of an association such as a trade union, tending to limit the freedom of choice of their coworkers or employers. **2.** →**restrictive trade practice.**

restrictive trade practice, *n.* an agreement between trading companies which is contrary to the public interest, as resale price maintenance, exclusive dealing, price discrimination, etc.

rest room /ˈrɛst rum/, *n.* **1.** a room set aside for people to rest in, as in an office, factory, etc. **2.** a lavatory, or similar room having washing facilities, etc.

restump /ˌriˈstʌmp/, *v.t.* to replace the stump foundations of (a house).

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reprovision, *v.t.*
republish, *v.t.*
repunctuate, *v.t.,* **-ated, -ating.**
repunish, *v.t.*
repurification, *n.*
repurify, *v.t.,* **-fied, -fying.**
repursue, *v.t.,* **-sued, -suing.**
requalify, *v.,* **-fied, -fying.**
requestion, *v.t.*
requicken, *v.*
requote, *v.t.,* **-quoted, -quoting.**
re-radiate, *v.,* **-ated, -ating.**
re-rate, *v.t.,* **-rated, -rating.**
re-read, *v.t.,* **-read, -reading.**
re-record, *v.t.*
re-reel, *v.*
re-register, *v.*
re-registration, *n.*
re-regulate, *v.t.,* **-lated, -lating.**
re-root, *v.t.*
re-route, *v.t.,* **-routed, -routeing.**
resaddle, *v.,* **-dled, -dling.**
resalt, *v.t.*
reschedule, *v.t.,* **-uled, -uling.**
rescrutinise, *v.t.,* **-nised, -nising.**
reseal, *v.t.*
resecure, *v.t.,* **-cured, -curing.**
reseed, *v.*
resegregate, *v.,* **-gated, -gating.**
resegregation, *n.*
reseize, *v.t.,* **-seized, -seizing.**
reseizure, *n.*

Effects on Woodland Restoration in Western Australia.
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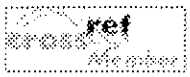

a definite principle; marked by steadiness or uniformity of action, procedure, or occurrence 1594. **b.** Recurring or repeated at fixed times or uniform intervals 1756. **c.** Habitually or customarily used, received, observed, etc.; habitual, constant 1797. **4.** Pursuing a definite course, or observing some uniform principle, of action or conduct; adhering to rule; now *esp.* observing fixed times for, or never failing in, the performance of certain actions or duties 1602. **b.** Orderly, well-ordered, well-behaved, steady 1705. **c.** Acting at the proper intervals 1783. **5.** Conformable to some accepted or adopted rule or standard; recognized as formally correct 1647. **b.** *Gram.* Of parts of speech, esp. verbs: Following some usual and uniform mode of inflection or conjugation 1611. **6.** Properly constituted; normal 1638. **b.** Of persons: Properly qualified or trained 1712. **c.** *colloq.* Thorough, complete, absolute, perfect 1821. **7.** *Mil.* Of forces or troops: Properly and permanently organized; constituting the standing army 1706.

2. I cannot, however, tell you that her features are r. 1716. A r. and appropriate nomenclature 1815. Small r. teeth 1863. **b.** R. Figures are those where the Angles and Lines or Superficies are equal 1679. *R. Curves*, are such Curves as the Perimeters of the Conick Sections, which are always curved after the same R. Geometrical Manner 1704. **3.** He supposes the philosopher to proceed by r. steps, until he arrives at the idea of good JOWETT. **b.** How r. his meals, how sound he sleeps! COWPER. Her r. pulses SHELLEY. **c.** It's past my r. time for going to bed DICKENS. **4.** The Herr Doctor was a r. man, and always appeared at his window at the same hour 1883. **b.** He grew first r., and then pious JOHNSON. **5.** Making acquaintances..without r. introductions 1831. **6. b.** A r. doctor JOHNSON. **c.** On Wednesday we had a r. flood, and it has been raining..ever since 1846. **7.** His Majesty's r. forces 1756.

B. *adv.* Regularly, steadily; thoroughly 1710.

C. *sb.* **1.** *Eccl.* A member of a religious order observing a RULE; one of the regular clergy 1563. **2.** A soldier belonging to the standing army; a member of the regular forces. Usu. *pl.* 1756. Hence **Re·gular-ly** *adv.*, **-ness.**

Regularity (regiŭlæ·rĭti). 1603. [f. prec. + -ITY. Cf. (O)Fr. *régularité.*] The state or character of being regular.

Regularize (re·giŭlărəiz), *v.* 1780. [f. REGULAR + -IZE, after Fr. *régulariser* (XVIII); see -IZE.] *trans.* To make regular. Hence **Re:gulariza·tion.**

†**Re·gulate**, *ppl. a.* 1577. [– late L. *regulatus*, pa. pple. of *regulare*; see next, -ATE².] Regulated; regular –1644.

Regulate (re·giŭle¹t), *v.* 1630. [– *regulat-*, pa. ppl. stem of late L. *regulare*, f. L. *regula* RULE *sb.*; see -ATE³.] **1.** *trans.* To control, govern, or direct by rule or regulations; to subject to guidance or restrictions; to adapt to circumstances or surroundings. †**b.** To bring or reduce (a person or body of persons) to order –1839. †**c.** To correct by control –1682. **2.** To adjust, in respect of time, quantity, etc., with reference to some standard or purpose; *esp.* to adjust (a clock, etc.) so that the working may be accurate 1662.

1. Can freedom be regulated without being..in some part destroyed? 1792. **c.** To r. the Errors of the Mind 1680. **2.** Clocks ought to be regulated by the mean solar time 1812.

Regulation (regiŭlē¹·ʃən). 1672. [f. prec.; see -ATION.] **1.** The act of regulating, or the state of being regulated. Also, an instance of this. **2.** A rule prescribed for the management of some matter, or the regulating of conduct; a governing precept or direction; a standing rule 1715. **3.** *attrib.* That is prescribed by, or in accordance with, a regulation or regulations; hence, regular, ordinary 1836.

1. The advancement and r. of manufactures and commerce BLACKSTONE. **2.** It's against regulations for me to call at night DICKENS. **3.** He can't afford more than his r. chargers THACKERAY.

Regulative (re·giŭle¹tiv), *a.* 1599. [f. REGULATE *v.* + -IVE.] Tending to regulate. Chiefly *Philos.*

Logic is not useless; it has a r., not a creative virtue 1874. Hence **Re·gulatively** *adv.*

Regulator (re·giŭle¹tə). 1655. [f. REGULATE *v.* + -OR 2.] **1.** One who regulates. **b.** *Eng. Hist.* A member of a commission appointed in 1687 to investigate and revise the constitution of various boroughs, for the purpose of influencing the election of members of parliament 1688. **c.** *U.S.* A member of one of the bands formed at various times in wild parts of the country, with the professed object of supplying the want of the regular administration of justice 1767. **2.** *techn.* A device for controlling machinery in motion, or for regulating the passage of air, electricity, gas, steam, water, etc. 1702. **b.** A device for adjusting the balance of a clock or watch, in order to regulate its speed 1696. **3.** A clock or watch keeping accurate time, by which other timepieces may be regulated 1758. **4.** Something which regulates; a regulating principle or power 1766.

1. c. The lynchers, or 'regulators' as they are often called, soon find that their foes organize also 1847.

Reguline (re·giŭlən), *a.* 1669. [f. REGULUS + -INE¹.] *Chem.* Of, or pertaining to, of the nature of, regulus.

Re·gulize, *v. rare.* 1778. [f. as prec. + -IZE.] *trans.* To reduce to regulus.

‖**Regulus** (re·giŭlŏs). *Pl.* **-li** (-ləi). 1559. [L., dim. of *rex*, *reg-* king; see -ULE.] **1.** *Astron.* A bright star (α Leonis) in the constellation Leo, called also *Cor Leonis.* **2.** *Chem.* †**a.** The metallic form of antimony, so called by early chemists, app. on account of its ready combination with gold. **b.** The purer or metallic part of a mineral, which sinks to the bottom of a crucible or furnace and is thus separated from the remaining matter. **c.** A product of the smelting of various ores, as copper, lead, and silver, consisting of metal in a still impure state. 1594. **3.** A petty king or ruler 1682. **4.** The golden-crested (and fire-crested) wren 1824.

Regurgitant (rĭgə·ɹdʒĭtănt), *ppl. a.* 1866. [See next and -ANT.] *Path.* Regurgitating; characterized by regurgitation.

Regurgitate (rĭgə·ɹdʒĭte¹t), *v.* 1653. [– *regurgitat-*, pa. ppl. stem of med.L. *regurgitare*, f. *re-* RE- + late L. *gurgitare.*] **1.** *intr.* Of fluids, air, or gases: To gush, rush, or pour back (again). **b.** *transf.* of the containing vessel. BOYLE. **2.** *trans.* To pour or cast out again from a receptacle, *esp.* from the stomach 1753.

2. *absol.* The Whale that swallowed Jonah found him hard meat, and..was forced to r. 1657.

Regurgitation (rĭgəɹdʒĭtē¹·ʃən). 1601. [– med.L. *regurgitatio*, *-on-*, f. as prec.; see -ION.] The act of pouring or gushing back; the fact of reissuing or being ejected again from a receptacle. Chiefly *Med.* with ref. either to the blood or to food.

Rehabilitate (rĭhăbi·lite¹t), *v.* 1580. [– *rehabilitat-*, pa. ppl. stem of med.L. *rehabilitare*; see RE-, HABILITATE *v.*] **1.** *trans.* To restore by formal act or declaration (a person degraded or attainted) to former privileges, rank, and possessions; to reestablish (a person's good name or memory) by authoritative pronouncement. **b.** To reestablish the character or reputation of (a person or thing) 1847. **2.** To restore to a previous condition; to set up again in proper condition 1845.

1. The king alone can r. an officer noted, condemned, and degraded; or a gentleman who has derogated from his rank 1727. **2.** The unwearied Lord Lieutenant..has been rehabilitating Courts of Justice in Dublin CARLYLE. Hence **Re:habilita·tion**, the action of rehabilitating, or state of being rehabilitated.

Rehash (rĭhæ·ʃ), *sb.* 1849. [f. RE- 5 a + HASH *sb.* 2.] A mere restatement in different words of opinions previously expressed; something served up afresh under a different form or name.

Reha·sh, *v.* 1822. [RE- 5 a.] *trans.* To restate (old ideas or opinions) in new language. All they did was to r. the old..arguments 1884.

Rehearsal (rĭhə·ɹsăl). late ME. [f. next + -AL¹.] **1.** The act of rehearsing; a recounting or recital; a repetition of words or statements; recitation. **2.** The practising of a play or musical composition preparatory to performing it in public; a private meeting of actors or performers held for this purpose 1579.

1. Many..made it a pretext for r. of old grievances 1842. **2.** The second part of the *Beggar's Opera*..was almost ready for r. GAY. *Dress r.*, a final r. in full costume. *In r.*, in process of being rehearsed.

Rehearse (rĭhə·ɹs), *v.* [ME. *reherce*, *-erse* – AFr. *rehearser*, OFr. *reherc(i)er*, perh. f. *re-* RE- + *hercer* harrow (see HEARSE, HERSE).] **1.** *trans.* To recite or repeat aloud in a formal manner; to say over, or read aloud, from beginning to end. **b.** To repeat, say over again (something previously said or heard) ME. †**c.** To say, utter, speak –1567. **2.** To give an account of; to relate, narrate, recount, describe at length. Now *rare.* ME. **3.** To recount in order; to name or mention one after another; to enumerate, reckon up, †number. late ME. †**b.** To mention, make mention of; to cite, quote –1562. **4.** To go through or practise (a play, scene, part, etc.) in private, in preparation for a more formal or public performance 1579. **b.** To exercise, train, or make proficient by rehearsal 1768. **5.** To perform, practise, as in rehearsing (*rare*) 1700. **6.** *intr.* To recite; to engage in rehearsal 1693.

1. Words learned by rote a parrot may r. COWPER. **b.** The critic brings thee praise, which all r. 1822. **2.** First of all we shall r...The Nativity of our Lord LONGF. **3.** I will r. the captains and their fleets COWPER. **4.** Sit downe..and r. your parts SHAKS. **6.** We got together, in order to r. GOLDSM. Hence **Rehea·rser.**

Rei, assumed sing. of REIS¹.

‖**Reich** (raiχ). 1924. [G., kingdom, empire, state.] The German state or commonwealth.

Reif (rīf). Chiefly *Sc.* [OE. *rēaf* = OFris. *rāf*, OS. *rōf*, OHG. *roub* :– Gmc. **raub-*; see REAVE *v.*¹] †**1.** Spoil, plunder –1557. **2.** The act or practice of robbery; spoliation. *Obs.* exc. *arch.* ME.

Reify (rī·ifəi), *v.* 1854. [f. L. *res*, *re-* thing + -FY.] *trans.* To convert mentally into a thing; to materialize. Hence **Reifica·tion.**

Reign (rē¹n), *sb.* ME. [– OFr. *reigne* (also mod.) *règne*, in OFr. always 'kingdom' – L. *regnum*, rel. to *rex*, *reg-* king.] **1.** Royal power or rule; kingdom, sovereignty; also *transf.* power or rule comparable to that of a king. Now *rare.* **b.** *transf.* Influence, dominion, sway (of something immaterial). late ME. †**2.** A kingdom or realm; a territory ruled over by a king –1725. †**b.** The kingdom of heaven or of God –1594. **c.** *poet.* A place or sphere under the rule of some specified person or thing, or having a specified character. Now *rare.* late ME. †**d.** = KINGDOM 5. (*rare*) –1781. **3.** The period of a sovereign's rule ME.

1. Under the r. of Queen Victoria THACKERAY. **b.** The owlet Night resumes her r. SHELLEY. **2. c.** A shout that..Frighted the R. of Chaos and old Night MILT. **d.** The vegetable and the mineral reigns THOMSON. **3.** After a r. of seventy years, he died 1841. Phr. *R. of Terror*: see TERROR.

Reign (rē¹n), *v.* ME. [– OFr. *reignier* (mod. *régner*) – L. *regnare*, f. *regnum* (see prec.).] **1.** *intr.* To hold or exercise the sovereign power or authority in a state; to rule or govern as king or queen. Also *transf.* or *fig.* **2.** Of persons: To exercise authority of any kind; to hold sway; to rule ME. **3.** Of things: To have power, sway, or predominance; to prevail or be prevalent ME.

1. During the time Edward the third did raigne SHAKS. Who reigned'st in thy heaven, yet felt'st our hell 1633. While..Reigns in pomp the perfect moon 1871. **2.** Yet he who reigns within himself, and rules Passions, Desires, and Fears, is more a King MILT. **3.** Lord, Lord: to see what folly raignes in vs SHAKS. In thy heart eternal winter reigns POPE. Famine has long reigned CARLYLE. Hence †**Rei·gner**, a ruler.

‖**Reim** (rīm). *S. Afr.* 1865. [– Du. *riem.*] A strip of ox-hide, a thong, strap.

Reimburse (rī¸imbə·ɹs), *v.* 1611. [f. RE- 5 a + IMBURSE, after Fr. *rembourser.*] **1.** *trans.* To repay or make up to a person (a sum expended). **2.** To repay, recompense (a person) 1637. **3.** With double object 1624.

1. The tardy sale of so voluminous a work could not have reimbursed the cost HALLAM. **2.** I will see you fully and thankfully reimbursed for what charges shall attend the same PEPYS. **3.** To r. him the costs of his trial 1841. Hence **Reimbu·rsable** *a.* that is to be reimbursed, repayable. **Reimbu·rsement**, the act of reimbursing, repayment. **Reimbu·rser.**

Reim-kennar (rəi·m¸ke:naɹ), *pseudo-arch.* 1821. [app. formed by Scott on G. *reim* rhyme

Acts 3:21). late ME. **5.** †**a.** Reposition, replacement (*rare*) –1658. **b.** Tendency to return to a previous position by virtue of elasticity or resilience 1656.

1. Euer the Frenche Ambassadours promised restitucion of euery thyng, but none was restored 1548. Phr. *To make r.*; They had wronged her . . ; therefore they ought to make her r. 1720. **2.** David passes sentence . . that there should be a fourfold r. made 1729. **3.** After the R. of King Charles the Second DE FOE. Phr. *R. in blood*, readmission to the privileges of birth and rank of one under sentence of corruption of blood (see CORRUPTION 2), or of his heirs. **4.** The R. of all Things to their first State of Perfection 1781.

Restive (re·stiv), *a.* 1599. [Later form (by assim. to -IVE) of RESTIFF – OFr. *restif* (mod. *rétif*) :– Rom. **restivus* 'inclined to remain stationary', f. *restare* REST *v.*[2] See RESTY *a.*[1]] **†1.** Inclined to rest or remain still; inactive, inert –1833. **†2.** Persistent, obstinate, settled or fixed *in* an opinion or course of action –1826. **3.** Of horses: Refusing to go forward; stubbornly standing still; obstinately moving backwards or to the side when being driven or ridden; hence, intractable, refractory 1656. **b.** *transf.* Of persons or things 1687. **4.** Of actions: Characterized by unwillingness or resistance to control 1806.

1. What great imployment with stirring and mettald spirits, what perpetuall quiet with heavie and r. bodies 1599. **2.** Every one being r. in his opinion, there can nothing . . be concluded 1660. **3.** The beasts . . became r. and went back MACAULAY. **b.** He turned r. at the least attempt at coercion 1863. **4.** The outward man yielded a reluctant and r. compliance SCOTT. Hence **Re·stive-ly** *adv.*, **-ness.**

Restless (re·stlés), *a.* OE. [f. REST *sb.*[1] + -LESS.] **1.** Deprived of rest; finding no rest; *esp.* uneasy in mind or spirit. **b.** Marked by unrest; affording no rest 1605. **2.** Constantly stirring or active, or desirous to be so; averse to being quiet or settled 1475. **†b.** Const. *to* (with inf.) or *of*: Impatient –1725. **c.** *spec.* in names of animals, as *r. cavy, thrush*, etc. 1771. **3.** Of conditions: Unceasing, continuous. late ME. **b.** Of things: Never ceasing or pausing 1596. **4.** quasi-*adv.* Restlessly. late ME.

1. R. he passed the remnants of the night DRYDEN. **b.** R. was the chair; the back erect Distress'd the weary loins COWPER. **2.** Cities, humming with a r. crowd COWPER. All the reason . . For so much rambling, was, a r. mind CRABBE. **3.** A world of restlesse Cares SHAKS. **b.** That Godde blind, that stands vpon the rolling restlesse Stone SHAKS. Hence **Re·stless-ly** *adv.*, **-ness.**

Restorable (rĭstō·răb'l), *a.* 1611. [f. RESTORE *v.* + -ABLE.] That can be restored or brought back to a former condition.

Restoral (rĭstō·răl). 1611. [f. RESTORE *v.* + -AL[1] 2.] Restoration, restitution.

Restoration (restŏrē·ʃən). 1660. [Later f. RESTAURATION, after RESTORE *v.*] **1.** The action of restoring to a former state or position; the fact of being restored or reinstated. **b.** *Theol.* = RESTITUTION 4. 1781. **2.** *Hist.* **a.** The re-establishment of the monarchy in England with the return of Charles II in 1660; also, the period marked by this event 1718. **b.** The reinstatement, in 1814, of the Bourbons in the sovereignty of France 1839. **3.** The action of restoring a person to health or consciousness; recovery of physical strength 1605. **4.** The action or process of restoring something to an unimpaired or perfect condition 1801. **b.** *Arch.* The process of carrying out alterations and repairs with the idea of restoring a building to something like its original form; a general renovation 1824. **c.** A representation of the original form of a ruined building, extinct animal, etc. 1836. **5.** The action of restoring something to one who has been previously deprived of it 1788.

1. The happy R. of his Majesty to his People and Kingdoms 1660. That period which has been distinguished as the r. of letters 1841. **4.** The r. of disfigured and decayed works of art 1835. The passages which defy r. 1874. **b.** Under the name of 'r.' the ruin of the noblest architecture . . is constant throughout Europe RUSKIN. **c.** Fig. 81 represents a r. of this extinct elephant 1878. **5.** The r. of estates that his predecessors had alienated 1877. Hence **Restora·tioner,** = RESTORATIONIST.

Restora·tionism. 1834. [f. prec. + -ISM.] The doctrine that all men will ultimately be restored to a state of happiness in the future life. So **Restora·tionist,** a believer in r.

Restorative (rĭstō·rătiv, *-ǫ·rătiv*), *a.* and *sb.* late ME. [var. of †*restaurative* (XIV–XVII) – OFr. *restauratif, -ive*; cf. RESTAURATION and see -IVE. Cf. med.L. *restaurativus* (in medical use, XIII).] **A.** *adj.* Pertaining to restoration (of strength or health); capable of restoring or renewing.

To try if there was any r. quality in the more genial air of that climate 1807.

B. *sb.* A food, cordial, or medicine, which has the effect of restoring health or strength. late ME. **b.** A means of restoring one to consciousness 1852. Hence **Resto·ratively** *adv.*

†Resto·re, *sb.* 1450. [f. next.] Restoration –1646.

Restore (rĭstō·ɹ), *v.* ME. [– OFr. *restorer* (mod. *restaurer*) :– L. *restaurare.*] **1.** *trans.* To give back, to make return or restitution of (anything previously taken away or lost). **2.** To make amends for; to compensate, make good (loss or damage). Now *rare* or *Obs.* ME. **b.** To set right, repair (decay, etc.). *rare.* 1567. **3.** To build up again; to re-erect or reconstruct. Now *spec.* to repair and alter (a building) so as to bring it as nearly as possible to its original form ME. **b.** To bring back to the original state; to improve, repair, or retouch (a thing) so as to bring it back to its original condition 1679. **c.** To reproduce or represent (an extinct animal, etc.) in its original form 1771. **4. a.** To replace (mankind) in a state of grace; to free from the effects of sin ME. **b.** To reinstate or replace (a person) in a former office, dignity, or estate 1450. **c.** To bring (a person or part of the body) back to a healthy or vigorous state. late ME. **d.** To bring back to mental calm. Now *rare.* 1582. **5.** To renew; to set up or bring into existence again; to re-establish, bring back into use, etc. ME. **b.** To replace or insert (words or letters which are missing or illegible in a text) 1855. **6.** To bring back (a person or thing) *to* a previous, original, or normal condition ME. **b.** To grant to or obtain for (a person) reinstatement *to* former rank, office, or possessions 1533. **c.** To take or put back *into*, to convey or hand back *to*, a place 1450.

1. Your helthe shall be restored to yow CAXTON. **2.** Time may r. some losses FULLER. **3.** At Winchester, where they are restoring the cathedral 1820. **4. a.** R. thou them that be penitent *Bk. Com. Prayer.* **b.** *To r. in blood*: see BLOOD *sb.* III. 5. **c.** The quiet place, the pure air . . will r. you in a few days DICKENS. **5.** It was with great difficulty that the . . man in the cocked hat restored order 1820. **6.** The application of faradic electricity quickly restored the patient to consciousness 1882. **b.** The innocent were restored to their rank and fortunes GIBBON. Hence **Resto·rement,** the act of restoring; restoration, restitution. **Resto·rer,** one who restores.

Restrain (rĭstrē·n), *v.* [ME. *restreyne, restrayne* – OFr. *restrei(g)n-, -ai(g)ne-*, pr. stem of *restreindre, restraindre* :– L. *restringere* bind fast, confine, f. *re-* RE- + *stringere* draw tight.] **1.** *trans.* To check, hold back, or prevent (a person or thing) *from* some course of action. **b.** Without const. To keep (a person) in check or under control. Freq. *refl.* late ME. **c.** To place under arrest or in confinement; to deprive of personal liberty or freedom of action; also, to shut in by material barriers 1494. **d.** To deprive (a person) *of* liberty by restraint 1530. **2.** To check, to put a check or stop upon, to repress, keep down (a desire, feeling, activity, physical agent, force, etc.) ME. **3.** To restrict, limit, confine ME. **†4.** To withhold, keep back, *from* a person –1594. **†5.** To forbid or prohibit (a person) *to* do something; to keep back *from* something desired –1791. **†6. a.** To draw tightly (*rare*) –1596. **b.** To compel or constrain (*rare*) –1655. **7.** *intr.* To refrain (*from* something; †also with infin.). Now *rare.* 1594.

1. This faculty tends to r. men from doing mischief to each other 1729. **b.** If I want skill or force to r. the Beast that I ride upon COWLEY. **2.** I could hardly r. my feelings 1839. The necessity of restraining population MILL. **6. a.** *Tam. Shr.* III. ii. 59. **b.** By antient custome no Vestal Virgin or Flamen of Jupiter was restrained to swear FULLER. Hence **Restrai·nable** *a.* capable of being restrained. **Restrai·nedly** *adv.* with restraint. **Restrai·ner. Restrai·ningly** *adv.* **†Restrai·nment,** restraint –1688.

Restraint (rĭstrē·nt), *sb.* late ME. [– (O)Fr. *restreinte*, fem. pa. pple. of *restreindre*; see prec.] **1.** The action of restraining or checking a thing, operation, etc.; an instance of this, a stoppage. **2.** A means of restraining or checking persons from a course of action, or of keeping them under control; any force or influence which has a restraining effect; an instance of restraining or of being restrained. late ME. **b.** Without article. Restraining action or influence, as applied to persons 1567. **c.** The state or condition of being restrained; *esp.* abridgement of liberty, confinement 1547. **3.** †**a.** A prohibition –1594. **b.** An embargo. Usu. *r. of princes.* 1475. **4.** Constraint; reserve 1601. **†5.** Restriction or limitation –1746.

1. A bill for the r. of the Press 1863. *Without r.*, freely, copiously. **2.** All Government is a R. upon Liberty 1672. **b.** R. she will not brook MILT. **c.** 'Tis not R. or Liberty That makes Men prisoners or free BUTLER. **4.** She . . did angle for mee, Madding my eagernesse with her r. SHAKS. **5.** This r. of Easter to a certaine number of dayes HOOKER.

Restrict (rĭstri·kt), *v.* 1535. [– *restrict-*, pa. ppl. stem of L. *restringere* RESTRAIN.] *trans.* To confine (some person or thing) *to* or *within* certain limits; to limit or bound. **b.** To restrain by prohibition 1835.

The power of preaching was restricted by the issue of licences only to the friends of the Primate 1874. **b.** The act . . which restricted the Bank from making payments in gold 1835. Hence **Restri·ctedly** *adv.*

Restriction (rĭstri·kʃən). late ME. [– (O)Fr. *restriction* or L. *restrictio, -on-*, f. as prec.; see -ION.] **1.** A limitation imposed upon a person or thing; a condition or regulation of this nature. **b.** The action or fact of limiting or restricting 1629. **2.** *Logic.* Limitation or qualification of a term 1551. **3.** Constriction (*rare*) 1758.

1. The restrictions under which our first parents were laid 1772. **b.** Yet this must be understood with some r. BLACKSTONE. **3.** Severe r. of the waist 1871. Hence **Restri·ctionary** *a.* imposing restrictions.

Restri·ctionist. 1849. [f. prec. + -IST.] One who advocates the restriction of some practice, institution, etc., such as the liquor-trade.

Restrictive (rĭstri·ktiv), *a.* and *sb.* late ME. [– (O)Fr. *restrictif, -ive* or med.L. *restrictivus*, f. as RESTRICT *v.*; see -IVE.] **A.** *adj.* **†1.** = RESTRINGENT *a.* 1. –1727. **2.** Of terms, expressions, etc.: Implying, conveying, or expressing restriction or limitation 1579. **3.** Restricting; limitative *of* the power or scope of some thing or person 1652.

1. This Plaister being r. 1607. **2.** In order to restrain the devise . . it was necessary to shew r. words 1827. **3.** The r. negative power of conscience CLARENDON.

B. *sb.* A term or expression having the force of, or implying, a restriction or qualification 1671.

The indeterminate character of the restrictives, *alone* and *only* BENTHAM. Hence **Restri·ctive-ly** *adv.*, **-ness** (*rare*).

Restringe (rĭstri·nd3), *v.* 1597. [– L. *restringere* bind fast, confine, f. *re-* RE- + *stringere* draw tight.] **†1.** *trans.* To affect (a person) with costiveness; to have an astringent effect upon (a part of the body) –1758. **2.** To confine, limit, restrict. Now *rare.* 1604.

2. Of Passions . . some . . dilate, and some compresse and r. the heart 1604.

†Restri·ngent, *a.* and *sb.* 1578. [– L. *restringens, -ent-*, pr. pple. of *restringere*; see prec., -ENT.] **A.** *adj.* **1.** Having astringent or binding properties; *esp.* tending to restrain the action of the bowels –1799. **b.** Of outward applications: Styptic –1834. **2.** Constipated, costive (*rare*) –1635. **B.** *sb.* **1.** A word which has a limitative or restricting force –1671. **2.** A medicine or application which possesses astringent or styptic properties –1792. So **†Restri·ngency,** the quality or property of being r.

Resty (re·sti), *a.*[1] *Obs.* exc. *dial.* 1515. [var. of RESTIFF, RESTIVE; cf. *hasty, tardy.*] **1.** = RESTIVE *a.* 3. **†2.** Disinclined for action