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Revegetation of mined land in the wet-dry tropics of northern Australia: A review

Report to the Alligator Rivers Region Technical Committee (ARRTC)

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This review was carried out by the Centre for Mined Land Rehabilitation (CMLR) at the University of Queensland under commission from the Australian Centre for Mining Environmental Research (ACMER) as part of a contract between ACMER and Supervising Scientist to develop and implement programs in technology transfer and strategic research addressing environmental issues facing the mining sector.

CENTRE FOR MINED LAND REHABILITATION

THE UNIVERSITY OF QUEENSLAND

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This review aims to assist the Alligator Rivers Region Technical Committee (ARRTC) in determining whether current practices and plans for revegetation at Ranger mine and elsewhere in the Alligator Rivers Region (ARR) are appropriate, and to establish research priorities in this region. The majority of information regarding revegetation in the wet-dry tropics (WDT) of northern Australia pertains to Ranger mine. However, there is a dearth of peer-reviewed published papers which is a significant problem that may limit effective communication and application of appropriate revegetation techniques on mines in the WDT.

The use of topsoil on hard rock mines in northern Australia is a contentious issue, with topsoil re-spreading being excluded from many rehabilitation programs. However, the experience of many WDT rehabilitation researchers indicates that use of topsoil containing indigenous microbes, valuable nutrients and organic matter increases the probability of achieving a successful, self-sustaining native ecosystem in the long term (eg Bell 1993, Hinz 1996, Tongway et al 1997). Because of the negative effects associated with stored topsoil use at hard rock mines, research is required on effective collection, handling and storage strategies for stored topsoil. Studies are also needed to determine the minimum amount of topsoil required for effective rehabilitation.

There is a paucity of literature on the long-term successional development of revegetated areas in the WDT. On disturbed sites in the WDT, the dominance by early successional species such as acacias has been found to retard successional development, with poor recruitment of eucalypts and other species to these systems (Setterfield et al 1993). Given the discrepancy between the time scales of many revegetation programs and subsequent lease relinquishment, and the time required to effect succession, accurate prediction of successional development of young rehabilitated areas is important. Research is also required to produce a body of organised, reliable theory and practice for industry on the selection, germination and establishment of a composite of species that will increase the likelihood of successional development toward a target ecosystem.

Perhaps the most important issue affecting the successional development of young rehabilitated areas towards self-sustaining native vegetation communities is fire. The existing literature pertaining to the role of fire in tropical systems focuses on mature systems rather than on young rehabilitated areas. Research is needed to establish the time required for the development of fire resistance in the various woody components of rehabilitated areas. There is also a need to quantify the frequency and timing of burning regimes that could reduce the risk of high intensity fire in younger rehabilitation and direct species composition/successional development of older rehabilitation.

Finally, there is a requirement, both during and upon completion of rehabilitation, for the redeveloped landform, soils and vegetation to be monitored and assessments made of how successful the rehabilitation process has been. This review examines five methodologies that have been used to assess the success of rehabilitation in the WDT. Success criteria based on a single or narrow set of parameters are likely to be inadequate. A study comparing the indicator value of the various monitoring methods would be valuable, with a possible outcome being the development of a ‘multi-discipline’ monitoring approach.

Gaps in the existing knowledge or practices that may limit the success of revegetation at minesites in the Alligator Rivers Region are identified. The most critical issues are identified broadly as: topsoil utilisation and management; fire; management/prediction of successional processes; establishment of symbiotic microorganisms; native seed collection, storage and germination; development of monitoring methodologies and acceptable success criteria; and technology transfer.

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List of abbreviations

eriss	Environmental Research Institute of the Supervising Scientist
oss	Office of the Supervising Scientist
NTDME	Northern Territory Department of Mines and Energy
CMLR	Centre for Mined Land Rehabilitation
ACMER	Australian Centre for Mining Environmental Research
ARRTC	Alligator Rivers Region Technical Committee
WRD	waste rock dump
WDT	wet-dry tropics
ARR	Alligator Rivers Region
TERC	Tropical Ecosystems Research Centre

1 Introduction

This review has been carried out by the Centre for Mined Land Rehabilitation (CMLR) at the University of Queensland under commission from the Australian Centre for Mining Environmental Research (ACMER). This forms part of a contract between ACMER and the Office of the Supervising Scientist (Environment Australia) to develop and implement programs in technology transfer and strategic research addressing environmental issues facing the mining sector. The review aims to provide a summation of the issues and information relating to revegetation in the wet-dry tropics (WDT), drawing on discussions with rehabilitation practitioners, regulators and researchers, visits to mine sites throughout the region and a review of both the published literature and unpublished company, consultants and government reports. In doing so, the review aims to assist the Alligator Rivers Region Technical Committee (ARRTC) in determining whether current practices and plans for revegetation at Ranger mine and elsewhere in the Alligator Rivers Region (ARR) are appropriate, and to facilitate the establishment of research priorities in this region.

The wet-dry tropics of northern Australia (figure 1) correspond to the northern-most part of the continent with a south-eastern extension along the western side of the Great Dividing Range (the SE extension of the WDT is not shown in figure 1).

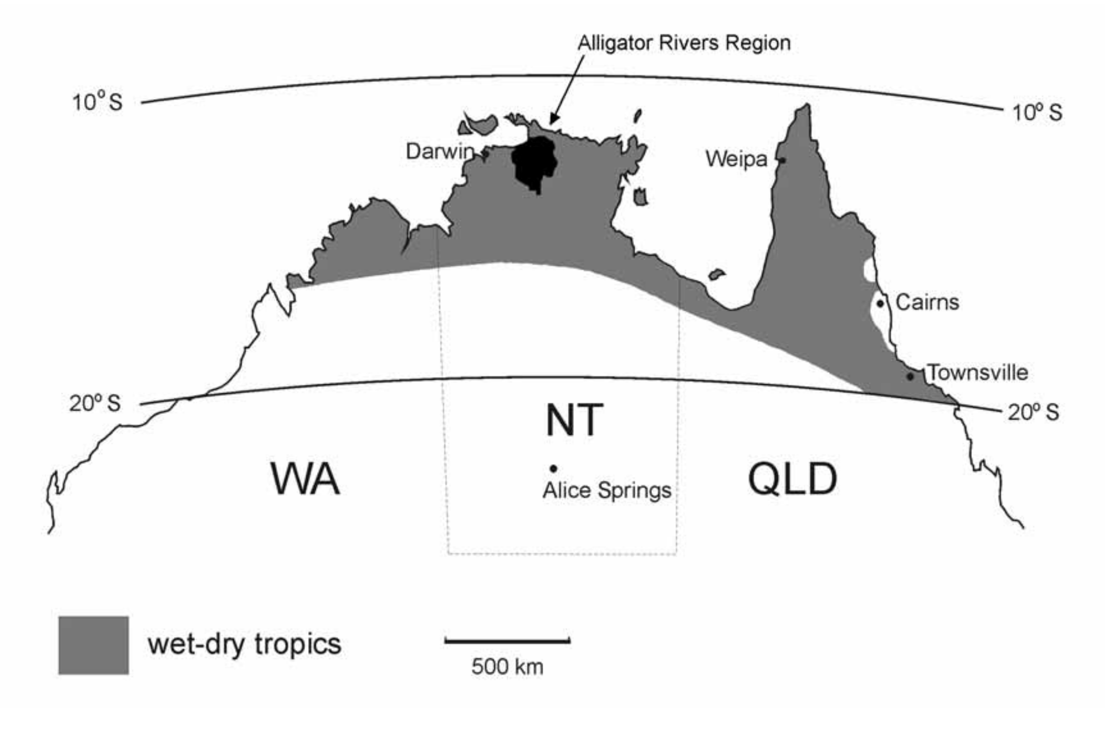

Figure 1 The Australian wet-dry tropics (rainfall band ~ 600–1800 mm)

The WDT differ markedly from other regions of Australia. Temperatures are high, rainfall occurs almost exclusively during a well-defined wet season, and fires are common (Ridpath 1985). Soil moisture and fire are very important determinants in the establishment, distribution and abundance of plants in the region and present significant challenges to the rehabilitation of mined land. Re-establishment of vegetation is often the most important component of rehabilitation. Successful revegetation implies that a suitable and stable substrate exists and that ecosystem function is being re-established. The Ranger rehabilitation plan requires the re-establishment of a stable ecosystem that supports vegetation communities with values comparable with those existing in the surrounding Kakadu National Park. Ultimately, the Ranger project area will be incorporated into the Park. The magnitude of this challenge is reflected in the amount of research relating to rehabilitation at Ranger that has been undertaken by the Office of the Supervising Scientist and the Environmental Research Institute of the Supervising Scientist, Energy Resources of Australia Ltd and a host of consultants and research organisations over the past 20 years. This effort is supported by a great deal of research and rehabilitation experience at other mines within the WDT of northern Australia. In particular, research at bauxite mines at Gove and Weipa has provided a wealth of knowledge. Experience can also be drawn from other hard rock mines in the WDT such as Nabarlek, Rum Jungle, Pine Creek, Groote Eylandt and Union Reefs.

2 Landform development and soil management

Post-mining landforms vary in slope, elevation and aspect from mine to mine. Bauxite mining at Gove and Weipa has resulted in basically flat landscapes (Nabalco Pty Ltd 1998), however, the slope of waste rock dump (WRD) batters at Ranger is currently typically 1:5. Ranger has stated its intention to use a final design of 1:3 for WRD batter slopes, 1:5 for laterite batter slopes and 1:8 for internal slopes (Ranger Minesite Technical Committee 1998). The Supervising Scientist's position is that no slope should be greater than 1:5, regardless of material (Loch 1998). Consultants were engaged to model the runoff and erosion from slopes of the three gradients and two proposed surface materials at Ranger mine using the WEPP (Water Erosion Prediction Program) computer model. The model was used to assess potential rates of erosion and off-slope sediment movement for a variety of slope angles, shapes and – most relevant to this review – vegetative cover. The study showed the most important factor influencing erosion, for all materials and slope gradients at Ranger mine, was vegetation (Loch 1998). The literature contains many examples of the large reduction in erosion which can be achieved through effective surface preparation and establishment of vegetative cover (eg Evans & Johnston 1997, Evans et al 1998). In an assessment of the effect of vegetation on the long-term stability of rehabilitation at Ranger mine using the SIBERIA landform evolution model, Willgoose (1995) predicted the erosion rate of a fully developed canopy and understorey to be about 5.8% of that of the unvegetated areas.

Given the scope of this review and the recent publication of the aforementioned landform/erosion studies, no attempt will be made to discuss issues of landform design. The following section will instead review the physical, chemical and microbial characteristics of WRD soils (with special reference to Ranger mine), the handling of topsoils, the use of fertilisers and surface preparation techniques in the context of the formation of a favourable root zone for the rapid establishment and maintenance of vegetation (Bell 1996).

2.1 Characterisation of waste rock dump soils (with special reference to Ranger mine)

2.1.1 Formation of soils on the Ranger waste rock dump

The WRD at Ranger mine consists of rocks that comprise carbonates, carbonaceous schists and mica, and quartz feldspar schist (Needham 1988). The waste rock is highly weatherable and breaks down into medium and fine gravel and clay-rich soil within a two or three year period (Riley & Gardiner 1994). Rate of soil formation on Ranger WRD is extremely rapid (Milnes 1989).

Fitzpatrick and Milnes (1988) present a generalised scheme of mine soil development on Ranger WRD. In that paper, the morphological, physical and chemical properties of soils are described, and several types of soils formed on the WRD over a five-year period are identified. These soils include lithosolic soils with minimal pedogenic development; soils with stony/gravelly lags, vesicular crusts and weak development of a B horizon; polysequel soils indicative of erosion events on the dumps; and pseudo-acid sulphate soils characteristic of waterlogged situations. The development of soil texture progressed from rocky and stony, to gravel, to clay.

2.1.2 Physical properties

Armstrong (1986) described the physical properties of Ranger WRD soils as:

- possessing a high degree of heterogeneity
- rapidly weathering
- becoming poorly structured upon weathering and exhibiting a high bulk density
- containing a high proportion of clay.

Evans et al (1998) noted a covering of coarse competent rock fragments on Ranger WRD surfaces. This may increase infiltration and decrease erosion (Agassi & Levy 1991). Riley et al (1993), however, concluded from rainfall simulation experiments that Ranger WRD soils have extremely low infiltration rates compared with natural soils. Fitzpatrick and Milnes (1988) drew similar conclusions regarding infiltration rates. Fitzpatrick et al (1989) compared disturbed mine soils with undisturbed natural soils and stockpiled topsoils at Ranger. Their study indicated that fine particles in most mine soils disperse on wetting, resulting in variation in soil texture leading to marked soil structural changes with time. The heterogeneous nature of mine soils, the loss of surface fine particles through dispersion and erosion, and low infiltration rates are physical characteristics likely to limit plant growth on WRD soils at Ranger.

2.1.3 Chemical properties

Fitzpatrick et al (1989) reported that the chemical, morphological and mineralogical properties of WRD soils have features in common with natural undisturbed soils in the surrounding landscape. WRD soils are distinguishable, however, in terms of their higher pH, lower organic carbon content, higher content of labile minerals and, consequently, their higher concentration of some plant nutrients. Based on soil and plant analyses and glasshouse experiments on seedling growth, Milnes (1989) concludes that minesoils at Ranger are more fertile than natural undisturbed soils and stockpiled topsoils. Analysis of the chemical properties of mine spoils and natural soils from Ranger, Nabarlek and Coronation Hill mines (Ashwath et al 1993) found that mine spoils differ from natural soils in that they showed higher pH, electrical conductivity (EC) (at Ranger only) and cation exchange capacity (CEC) and higher concentrations of magnesium (Mg) and sulphate (SO_4). They exhibited lower concentrations of Kjeldahl N and total organic carbon. They also found mine spoil at Ranger has higher total and plant-available phosphorus compared with other mine spoils and natural soils. Milnes (1988) found the pseudo-acid sulphate soils that form in the furrows of reshaped and ripped WRD soils have higher concentrations of soluble and exchangeable potassium, clay and silt content than other soil types.

Ranger mine soils have concentrations of Mg up to 100 times higher than adjacent natural soils (Ashwath et al 1993). The high concentrations of Mg present in the proterozoic rock can cause a severe Mg:Ca ratio imbalance (20:1 to 80:1) (Atkinson et al 1998). Atkinson et al (1998) demonstrated contrasting responses by native plants with changes in soil Mg:Ca ratios.

They state that for normal growth of crop plants, a Mg:Ca ratio between 1:3 and 2:1 is acceptable. Milnes (1988) notes that the very high ratio of Mg to Ca in the soil solution is likely to affect nutrient availability for some plant species.

The higher pH of mine spoils could have some effect on the establishment of native vegetation given the natural acidity of local soils (Ashwath et al 1993). The salinity of Ranger mine spoil is due to high concentrations of magnesium sulphate ($MgSO_4$) (Ashwath et al 1993). This represents a potential problem for seed germination because magnesium salts have been found to be more inhibitory to seed germination than potassium or sodium salts (Malden 1995). The low total organic carbon and plant available nitrogen could also affect the establishment of native vegetation on WRD soils. Similarly, excessive concentrations of P in WRD soils could have an inhibitory effect on some native vegetation adapted to low P soils, particularly grevilleas (Ashwath et al 1993).

2.1.4 Microbiological properties

Based on short-term glasshouse experiments, Milnes (1989) concluded that Ranger mine soils had fewer symbiotic microorganisms (rhizobia and mycorrhizal fungi) than natural undisturbed soils. Brundrett et al (1992 a,b) found that disturbed sites at the Ranger mine had fewer spores and lower diversities of endomycorrhizal fungi compared with undisturbed natural sites. Where the mine sites were densely revegetated, the endomycorrhizal infectivity/spore numbers were greater than those found in climax natural vegetation. The diversity of endomycorrhizal fungi, however, was less than in natural sites. Reddell and Milnes (1992) found that:

> while rhizobia and mycorrhizal fungi were a ubiquitous component of the soil biota in all undisturbed woodland soils … they were absent or poorly represented in the stockpiled topsoils and some of the rudimentary soils formed in waste rock at the [Ranger] mine site.

The importance of soil microflora to the revegetation of disturbed land in the tropics is well documented in recent literature and will be reviewed further in section 2.3.2 (eg Malajczuk et al 1994, Gibson 1995, Menzies & Mulligan 1996, Reddell et al 1997).

2.2 Formation of a favourable root zone

2.2.1 Topsoil

Available or mineralisable nutrients, beneficial microorganisms, substrate quality, organic matter, structural integrity, seeds and other propagules are among the attributes of topsoil. In an assessment of indicators of ecosystem rehabilitation success (in which rehabilitated areas on 13 mines around Australia, including 4 sites in northern Australia, were examined), Tongway et al (1997) observed that, when used appropriately, rehabilitation with topsoil was invariably more successful than rehabilitation without topsoil. Grundy and Bell (1981) provide a comprehensive survey of the use of topsoil in minesite rehabilitation. They conclude that topsoil should be preserved wherever possible for use in revegetation work during the rehabilitation program. At Weipa, rehabilitation experience has shown that the successful establishment and growth of direct-seeded native species is strongly influenced by the nature of the replaced topsoil (Dahl & Foster 1989). Grigg et al (1999) also found a significantly higher soil microbial biomass where the A horizon had been accurately replaced at Weipa and concluded:

> accurate topsoil replacement promotes the development of healthy, productive communities with associated nutrient cycling capabilities, and is therefore vital in the long-term success of the reconstructed communities.

Schwenke (1999) discusses the importance of returning organic matter to mined areas through topsoil. He states that, in addition to conservation and cycling of nutrients, organic matter decline may affect seedling emergence and root penetration by changing aggregate stability, water retention and soil aeration. Hinz (1996) noted that the degree of success of the restoration of a self-sustaining forest community at Gove bauxite mine is dependent on good topsoil management.

The use of topsoil on hard rock mines in northern Australia, however, is a contentious issue. The shallow strip-mining process used to extract bauxite presents a very different set of rehabilitation challenges to hard rock mining. Rehabilitation of bauxite-mined areas is more analogous to borrow-pit rehabilitation (Setterfield et al 1993). The bauxite mining process strips a layer of topsoil equal to the final area to be rehabilitated, whereas on hard rock mines the area of the final landform is in fact much greater than the initial disturbed surface area.

Issues that have led to topsoil respreading being largely excluded from hard rock mining rehabilitation programs include: limited supply of suitable fresh topsoil, adverse effects of grasses out-competing sown native trees and shrubs for water and nutrients, fire fuel accumulation and the dominance of primary colonising acacias (T McGill, pers comm, NTDME) (see plate 1, p21). Respreading of topsoil would also increase the *initial* cost of a rehabilitation program. Rehabilitation experience of Fawcett (1995) at Pine Creek Gold Mine has shown topsoil to inhibit the successful establishment and survival of native tree species. He found native tree species seeded directly into oxide waste material produced more vigorous growth and suffered lower mortality which he attributed to reduced grass density and therefore greater water availability. He concluded that topsoil was found to be a source of annual *Sorghum* spp, aggressively colonising *Acacia* spp and numerous weed species and was not a significant source of native tree species, and that the final composition of revegetated areas was more controllable when topsoil was excluded (MNR Fawcett, pers comm, Acacia Resources Ltd).

In a revegetation trial conducted at Ranger mine, Reddell et al (1994) also reported superior vegetation establishment without the use of topsoil. In his review of revegetation research at Ranger, Batterham (1998) concludes that eucalypts have shown the ability to establish and grow without the use of topsoil and recommends that topsoil not be applied to Ranger WRD (see plate 2, p21). The exclusion of topsoil from rehabilitation areas on these mines is a practice endorsed by the Northern Territory Department of Mines and Energy (T McGill, pers comm).

The literature and experience of other WDT rehabilitation practitioners indicates that the indigenous microbes, valuable nutrients and the organic matter contained within the topsoil resource will increase the probability of achieving a successful, self-sustaining native ecosystem in the long term (eg Bell 1993, Hinz 1996, Tongway et al 1997). Proponents of the use of topsoil emphasise the importance of handling and management procedures (reviewed in section 2.2.1.1 and 2.2.1.2). Adverse effects associated with spreading stockpiled topsoil may be removed through effective management.

Tongway et al (1997) suggest that since many hard rock mines have minimal topsoil resources, research should establish the minimum amount of respread topsoil needed to successfully rehabilitate the site. Bell (1993) suggests that on WRD soil showing no major physical or chemical limitations to root growth – such as salinity, sodicity or acidity – a layer of topsoil as thin as 50 mm will help vegetation to establish.

2.2.1.1 Collection and handling

Literature pertaining to the collection, handling and respreading of topsoil emphasises the importance of careful management (eg Foster & Dahl 1990, Bell 1996, Grigg et al 1999). The degradation that topsoil undergoes when it is stockpiled for long periods of time is well documented (eg Anderson 1988, Hinz 1990b, Klessa & leGras 1994, Klessa et al 1995). Anderson (1988) documented losses of nitrogen through denitrification, decomposition of organic matter, structural breakdown and a decline in mycorrhizae propagules and other microfloral/faunal attributes. Klessa et al (1995) predicted that during the tropical wet, detrimental structural effects may result in anaerobiosis, which in turn will have a detrimental effect on the viability of seed and microbes. They also note the role of stored soil as a vector for the entrapment and accumulation of weed seed (see section 2.3.1.2).

A review of the effects of soil storage on soil properties at Nabarlek mine by Klessa and leGras (1994) found the availability of information on the storage and handling of topsoil in the tropics to be poor. More recently, however, there have been several detailed accounts of topsoil handling techniques at Weipa and Gove bauxite mines in northern Australia (Hinz 1996, Short et al 1996, Dahl & Mulligan 1996, Grigg et al 1999, Schwenke 1999). These accounts suggest rehabilitation success is most likely when the soil resource is stripped and replaced in the appropriate horizons, unmixed and as soon as possible. Weipa mine employs two scrapers ‘in tandem’ to strip topsoil and subsoil separately from an area to be mined, and to relay the material directly and accurately into a mined-out pit. Research on this ‘dual strip’ method indicates that soil organic matter is retained within the replaced soil profile relative to the unmined soil (Schwenke et al 1999). Grigg et al (1999) suggest that soils returned this way produced communities with a greater biomass which favoured the cycling of P. Nabalco Pty Ltd employ a similar dual stripping method at their Gove mine to that used at Weipa. Following vegetation clearing and burning, the soil is left fallow for a minimum of two years before stripping the topsoil to allow native propagules and soil microflora and fauna to build up (Hinz 1992). If direct emplacement of stripped topsoil is not possible, stockpiling is minimised to months rather than years (Hinz 1996).

It is not always practical to respread topsoil immediately after removal on hard rock mines and so stockpiling is often necessary. Both Klessa et al (1995) and Bell (1993) found an inverse relationship between microbiological activity, organic matter, some plant nutrients and viable native seed with depth in topsoil stockpiles. Klessa et al (1995) studied a topsoil stockpile at Nabarlek with a maximum depth of 6 m and noted an accumulation of available N below 120 cm depth and a corresponding increase in pH and EC. They suggested that ‘inverting’ the stockpile during re-emplacement may reduce the effect of surface accumulated weed seed and take advantage of available N lower in the profile. Bell (1993) recommended that if storage is required, the height of the stockpile should be minimised and the surface area maximised. He also recommended that surface and subsurface material be stockpiled separately.

Research is needed into practical techniques that adapt the broad principles of recent findings on effective topsoil handling at bauxite mines (ie accurate stripping and replacement) and apply them to hard rock mines. Optimal storage techniques to retain the desirable attributes of fresh topsoil should also be considered.

2.2.1.2 Soil seedbank

Topsoil can be a repository of unwanted seed. This can lead to replacement of sown native species by exotic or native grasses, exotic weeds and aggressive natives (ie *Acacia holosericea*), which have been shown to germinate readily in respread, stockpiled topsoil (Fawcett 1995). Klessa et al (1995) note the dearth of literature relating to seed banks and mine site revegetation. In their study of 16 year stockpiled soil at Nabarlek mine, Klessa et al (1995) found that half of all surface (0–5 cm) samples were infested with weeds and that there were no viable seeds below 20 cm depth. In a study of 3 year stockpiled soil on a Western Australian sand mine, Bellairs and Bell (1993) noted a ten-fold reduction in seed density and a 12–37% increase in the proportion of non-native species. They also observed a decline in the survival of perennial species.

Roberts (1994) described successional processes directing seed bank composition. Early successional species tend to produce large numbers of small seeds and form persistent seed banks. They tend to be annuals, have rapid growth rates, a dependency on labile pools of nutrients and dramatic growth response to fertiliser. At Nabalco's Gove mine, Hinz (1992) found some 100 native species, representing 77 genera, were present in fresh topsoil. Few tree species seeds were observed in the freshly returned topsoil, which was attributed in part to their biennial or triennial seed production. Luken (1990) argues that late successional species are generally not large seed producers. Additionally, small seeded plants such as eucalypts may not be able to emerge from depths greater than 20 mm while larger seeded species may fail to emerge from below 100–150 mm (Bell 1993). Williams (pers comm, CSIRO Tropical Ecosystems Research Centre) believes that, in some cases, seed harvesting rather than a limited or temporally variable seed production is a likely cause of a depleted woody species seedbank.

In an experiment aimed at quantifying the effect of grass competition on the establishment of native species at Weipa, Foster and Dahl (1990) detailed an inverse relationship whereby reducing native grass competition was found to dramatically increase native seedling survival. The amount of native grass emerging in freshly laid topsoil was dependent on soil moisture during stripping and the season of removal. They found that topsoil stripped 'late season', from mid-October to the start of the wet season in December, produced the best regeneration with the least competition by grasses. At that time, the soil was moist and retained its structure and biological activity and most grasses that had recently germinated were destroyed subsequently during the stripping operation.

Similar conclusions, on the importance of this seasonal variation and its implications for the effective use of topsoil in aiding the re-establishment of a self sustaining native community, were drawn by Roberts (1994). Luken (1990) believes an understanding of the seed availability at a site is important in determining which species could participate in the successional pathway. Williams and Lane (1999) for example found that the *Sorghum* seedbank can be eliminated by wet season burning prior to removal of topsoil (see section 4.1).

Therefore, research on native flowering and fruiting phenology such as Brennan's (1996a) description of the Ranger mine lease area and Hinz's (1990a) description of native communities in the Nabarlek area are vital to the effective management of topsoil.

2.2.2 Symbiotic microorganisms

Reddell and Milnes (1992) reported that mycorrhizal fungi and nitrogen-fixing rhizobia were ubiquitous components of the soil biota in undisturbed soils of Kakadu National Park, but were absent or found in low numbers in WRD and stockpiled soils. Given the widely accepted importance of re-establishing indigenous symbiotic microorganisms in mined soils in the tropics (eg Malajczuk et al 1994, Menzies & Mulligan 1996, Hinz 1997, Ragupathy et al 1997), it is appropriate that a review of conservation or re-introduction of symbiotic organisms be included in this report. Inoculation techniques are not discussed in detail in this review. However, a recent description outlining the inoculation of mycorrhizae and rhizobia in the tropics is provided by Doran and Turnbull (1997). For rhizobia, they recommend inoculating 2–3 cm tall seedlings with a suspension of surface sterilised and ground nodules

(10 g/1000 plants) collected from a healthy young tree of the required species. For mycorrhizae, they considered the advantages and disadvantages of soil inoculum, pot cultured inoculum, spore inoculum, mycelial inoculum and the use of mycorrhizal mother-tree seedlings to effect a symbiosis. In terms of simplicity, soil inoculum[1] and spore inoculum[2] may be the most efficient and cost-effective option for mine sites.

2.2.2.1 Mycorrhizae

The majority of plant species in natural woodland communities surrounding the Ranger lease area have been found to be mycorrhizal, with ectomycorrhizal[3] (ECM) associations occurring with most dominant tree species (Reddell & Milnes 1992). Consequently, the establishment and the nutrient acquisition strategies of those species are likely to be highly dependent on ECM fungi, which have been shown to be absent or poorly represented on Ranger WRD (Milnes 1989, Reddel et al 1994). The positive effect of mycorrhizae on plant growth through increased phosphorus availability is well documented (Harley & Smith 1983). Increased tolerance of saline conditions (Dodd & Thompson 1994), uptake of zinc (Reddell & Milnes 1992), protection against pathogens and enhanced water uptake (Malajcuk et al 1994) are some of the other potential benefits conferred by mycorrhizae.

Studies of the effectiveness of mycorrhizal inoculum in enhancing nutrient uptake by plants at Ranger mine have produced conflicting results. Both Gordon et al (1997) and Reddell et al (1997) found that inoculation of tubestock with ECM significantly enhanced establishment, growth and survival of eucalypt seedlings. By contrast, Lane (1996b) and Cramb et al (1997) reported either no significant benefit to seedling growth or negative growth effects compared with control plants following inoculation by mycorrhizae. Growth inhibition of inoculated plants, however, may be a result of host/strain (ECM) incompatibility (D Bowen, pers comm, CMLR).

At the Gove mine, both Reddell et al (1993) and Hinz (1997) noted the importance of mycorrhizae in the establishment and growth of native vegetation. Hinz (1997) believes the growth of the dominant woody species *Eucalyptus tetrodonta* is dependent on an effective association with mycorrhizal fungi (*Nothocastoreum cretaceum*). Reddell et al (1993) found that fungal root infection increased with age of rehabilitation and that ECM and macro-fungal fruiting bodies were most indicative of the development of rehabilitated areas.

Many studies have shown that disturbance caused by mining severely reduces diversity and propagule levels of both endo- and ectomycorrhizae (eg Jasper et al 1987 & Malajczuk et al 1994). Mycorrhizal population levels may be so depleted that the benefits may not be expressed in plant growth (Brundrett et al 1996, Ragupathy et al 1997). In a study aimed at assessing the feasibility of rehabilitating Ranger WRD without the use of topsoil, Reddell and Milnes (1992) described two processes of colonisation by mycorrhizae: the transport of the infective propagules to the site, and the subsequent infection of plant roots under the altered conditions of the new substrate. The dispersal of spores and other infective propagules of

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[1] The addition of fresh soil collected from an existing healthy stand of the desired species or community.

[2] Watering seedlings with a suspension of macerated fruiting bodies of ectomycorrhizae – or mushrooms, puffballs or truffles.

[3] ECM hyphae are external to the root and mainly form a sheath enclosing the developing root tip. Cortical infection is intercellular. They are *predominantly* associated with woody species and they demonstrate high host specificity. ECM infection results in changes to gross root morphology (Bowen 1990).

vesicular arbuscular (or endo-) mycorrhizal (VAM)[4] fungi relies on the activities of vectors such as vertebrate and invertebrate fauna, wind and water (Warner et al 1987).

Ranger mine does not presently inoculate tubestock or seed sown onto WRD areas or spread topsoil to re-establish mycorrhizae, despite the fact that little volunteer colonisation occurs on Ranger mine spoils (Ragupathy et al 1997). Rather, there is a reliance on dispersal of mycorrhizal propagules from the 'ecological islands' established about the mine and from surrounding native vegetation. Malajczuk et al (1994), however, indicate that the natural dispersal and re-establishment of ECM fungi on Ranger WRD occur at a very slow rate and that this may significantly impact on the rate of development and the resilience of the plant community. Therefore, future research emphasis should be placed on identifying the factors affecting establishment of viable mycorrhizal populations particularly on mines that neither respread topsoil nor undertake any inoculation.

2.2.2.2 Rhizobia

Batterham's (1998) review of revegetation research at Ranger mine concludes that *Rhizobium* treatment is of little benefit because acacias are the easiest species to establish on waste rock dumps. Many processes other than initial establishment, however, will govern long-term success of rehabilitated areas (Reddell & Milnes 1992). Nitrogen deficiency in mine soils can be amended via the establishment of nitrogen fixing legumes and root nodule bacteria (*Rhizobium*) (Gibson 1995). Legumes are only effective if an association is formed with the appropriate strain of rhizobia (Bell 1996). If local nitrogen fixing species are to be reintroduced to the mined area, their effective inoculation will be catalysed through the replacement of fresh surface soil or through the careful selection, collection and inoculation of the plants with the appropriate strains. The contribution of plant available nitrogen and of organic matter by acacias to a re-establishing ecosystem through mechanisms such as litter cycling may rely on symbiosis with rhizobia (Coleman et al 1983).

Ranger WRD soils have been shown to be lower in plant available N than surrounding undisturbed soils (Ashwath et al 1993) and initially devoid of symbiotic microorganisms (Reddell & Milnes 1992). In a glasshouse experiment on a young Ranger WRD soil amended with basal nutrients (excluding N), Reddell and Milnes (1992) found inoculation of *Acacia holosericea* seedlings with *Rhizobium* mitigated nitrogen deficiency. They recommended the introduction of viable populations of the appropriate microbial symbionts during the early (seedling) stage of vegetation establishment on Ranger mine WRD. Gibson's (1995) research based on a collection of 500 *Rhizobium* isolates, collected from 64 species in Kakadu National Park, emphasised the specificity of the host plant/*Rhizobium* strain relationship. He found that chance infection of legumes on the Ranger WRD have only a 20% likelihood of being fully effective with the host and recommended inoculation of tubestock with effective strains. Ashwath et al (1995) found the establishment of some legumes on Ranger WRD soils was dependent on inoculation with the appropriate strain of *Rhizobium.*

There has been a great deal of research relating to symbiotic microorganisms in post-mining environments, particularly Ranger mine WRD soils. This research, however, has focused on the effects of symbionts on vegetation establishment and growth, rather than on the parameters or processes that limit the recolonisation of mined environments by beneficial microorganisms.

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[4] Endomycorrhizae, or vesicular arbuscular mycorrhizae (VAM) hyphae infect roots intracellularly and are not host specific. Infection by VAM does not alter gross root morphology. Hosts are usually herbaceous or rainforest tree species (Bowen 1990).

2.2.3 Fertilisers

Most Australian native plant species have evolved under conditions of low fertility, particularly under low phosphorus levels (Bell 1993). While native vegetation must rely on symbiotic microorganisms and nutrient cycling mechanisms to meet nutritional requirements, sometimes fertiliser is required to facilitate the rapid establishment of vegetation under conditions of nutrient deficiency. The addition of high rates of fertiliser, however, has been demonstrated to inhibit the establishment and the effectiveness of symbiotic microorganisms (Bowen 1990, Doran & Turnbull 1997).

Bell (1996) suggests using a ‘starter dose’ of 10–30 kg/ha of N and 10–50 kg/ha P to provide for plant growth until natural nutrient acquisition processes are re-established. He also suggests that ‘little response of native species to the application of other elements would be expected where establishment directly into waste is being attempted’.

Fertiliser application rates vary dramatically from mine to mine in the WDT (see table 1) and application is often designed to achieve a balance between establishment of an exotic grass cover crop to prevent erosion and the need to minimise competition between the cover crop and establishing native trees and shrubs. Union Reefs gold mine uses no fertiliser in the establishment of native vegetation (F Lawton, pers comm) while Pine Creek gold mine adds 800 kg/ha (see table 1 for fertiliser compositions) (MNR Fawcett, pers comm, Fawcett Rehabilitation Services). Both mines broadcast a mix of native seed directly into oxide material without the use of topsoil. In a fertiliser rates experiment using *Eucalyptus bigalerita* at Pine Creek, Fawcett (1995) found improved growth rates with applications up to 1000 kg/ha. Conversely, a fertiliser trial at Rustler’s Roost mine demonstrated that a lower fertiliser rate (100 kg/ha compared with 400 kg/ha) produced the best revegetation result (Rustler’s Roost Mining Pty Ltd 1996). It is noteworthy that the criteria used to assess the success or otherwise of these treatments is unclear and effective documentation of results is often lacking.

Table 1 Fertiliser rates, composition and application methods used by various mines in the WDT

Mine	Fertiliser rate	Composition	Application method
Gove	200 kg/ha	Superphosphate	Applied with grass cover crop
Weipa	300 kg/ha, 500 kg/ha for stockpiled soil	Custom Mix 7.2 N:8.3 P:7.0 K:10.3 S: 6.9 Ca:1.2 Cu:0.8 Zn	Sown with seed by tractor mounted spreader or helicopter broadcast
Rustler’s Roost	100 kg/ha	Tropigro	
		Neutrogro Organic	
Nabarlek	100 kg/ha		
Pine Creek	800 kg/ha	Custom Mix 15N:8P:9K:11S + Trace elements	Two aerial applications of 400 kg/ha 6 weeks apart
Union Reefs	No fertiliser used		
Christmas Island	10 g/plant	Osmocote	Applied to base of seedling at planting
Ranger	15 g/plant or no fertiliser used	Slow release commercial	Applied at base of seedling at planting

Bell (1996) found *Eucalyptus* spp and *Melaleuca* spp respond markedly to applications of N and P, while other natives are less responsive. Different species have different nutrient requirements during the establishment phase and the fertiliser mix used may influence the final species composition on a site (Bell 1993). High applications of fertiliser could also affect the drought-hardiness of plants through an elevated shoot:root ratio (Bell 1996) and encourage the establishment of weeds (Alligator Rivers Region Technical Committee 1998a).

In his study of the long-term stability of revegetation at Weipa, Roberts (1994) writes 'fertilisers increase diversity and density in the short term, although the influence of fertilisers on long-term succession is not clear…'. At Gove, Richards et al (1992) re-analysed a twenty-year-old species/fertiliser trial investigating the effect of superphosphate applications up to 800 kg/ha on 15 exotic grass and legume species. Of the fifteen species originally sown in 1972, seven were absent from the site in 1992, three had very restricted occurrence, and five had spread extensively from the original sown sites. Similarly, analysis of field trials established at Rum Jungle in 1975 using a mix of exotic grass and legumes, a moderately high input of lime (20 t/ha), and high rate of NPK fertiliser (800 kg/ha) revealed that none of the original species remained after five years without further nutrient input (Richards et al 1996).

It is evident that care should be taken to establish fertiliser regimes that take into account the nutritional status of the growth media, the requirements of the species being reestablished, the potential effect on microbial populations and the potential impact on the long-term successional development of the system. It is difficult, however, to propose an optimal fertiliser regime based on the studies presented here because quantitative data on the effects of various fertiliser and application rates on species composition and vegetation development are often lacking.

2.2.4 Surface preparation

Creation of a suitable seedbed has always been a cornerstone of agricultural practice, so it is not surprising that the physical conditions of the seedbed are considered to be very important in the establishment of vegetation on mined areas (eg Foster & Dahl 1990, Bell 1996). At mine sites in the WDT, the final surface may be a mixture of rocks and fine weathered material (hard rock mines) or respread subsoil and topsoil layers over a compacted ironstone layer (bauxite mines). In order for seeds to germinate they require good contact with the soil to offset drought and attack by birds and insects (Chandrasekaran et al 1994). Alleviation of compaction in minesoils is usually achieved by deep ripping, which reduces physical impedance to plant roots and improves aeration and infiltration of water (Bell 1996). Tongway et al's (1997) assessment of rehabilitation success emphasised the importance of stable surface troughs and banks and their ability to improve soil stability, resist erosion, improve infiltration and promote nutrient cycling through the retention of leaf litter. Timely cultivation can also control competing vegetation (Doran & Turnbull 1997).

Moisture is often a limiting factor on disturbed mine sites as the water-holding capacity is often less than for a 'natural' soil (Davies et al 1992). Riley et al (1993) found ripping at 1–1.5 m spacing improved infiltration of water. However, the furrows in the substrate became choked with fine materials within the first season, and the silt fraction of weathered material formed a surface crust following cultivation operations (see plate 3, p22). Both Hinz (1990a) and Fawcett (1995) explain that rainfall between cultivation and sowing results in partial collapse of furrows and crusting on the surface of waste rock substrate. Foster and Dahl's (1990) experience at Weipa also emphasised the importance of timing on cultivation, and found that unless the soil was moist, cultivation resulted in soil structural decline. They achieved optimal seedling establishment by sowing seed immediately following cultivation with disc harrows[5] in the 3–4 week period after the first rains of the wet season but before the

[5] Disc harrowing resulted in 93%, 58% and 42% more seedlings than a 'no cultivation' control, scarifier tines and trailing chains.

drenching storms that follow. They also found that the exposure of the prepared seedbed to the wet-dry cycles of early wet season storms prior to sowing destroyed the favourable seedbed structure. Thornton and Dahl (1996) also believe that soil stripping, contouring, and ripping should be completed before the early season storms and that surface cultivation and sowing should follow immediately after the onset of the first consistent rains.

3 Vegetation establishment

The evolution of mine revegetation practices has seen the agronomic engineering approach of establishing ‘green’ landscapes with introduced grass and pasture species (Dahl & Mulligan 1996) replaced by the re-establishment of native flora to create a self-sustaining ecosystem. Early attempts to revegetate mined areas at Weipa and Rum Jungle relied on introduced grass and tree species and nitrogen-fixing shrubs (Dahl & Mulligan 1996, Richards et al 1996). Often driven by post-mining landuse agreements, there has also been a realisation that exotic ‘green covers’ require high levels of on-going management and that the establishment of self-sustaining communities of native flora can reduce or eliminate any need for further work or expenditure (Thornton & Dahl 1996). Thus, the re-establishment of functioning natural ecosystems has required an understanding by rehabilitation practitioners of the ecological processes driving the establishment, propagation and succession of the communities they wish to recreate. This section will review species/provenance selection, the use of exotic grass cover crops, seed technology, seeding and planting methods and the management of ecological succession in the context of native community establishment.

3.1 Species and provenance selection

The utilisation of provenance or local seed for mine revegetation takes advantage of the inherent attributes of fitness conferred on those species by natural selection (Van Leeuwen 1994). The assumption with locally collected seed is that those species are the most suited to local climatic, edaphic and ecological processes. The mined environment, however, may be hostile to some local species and the importance of selecting species suited to the ‘new’ local conditions must be recognised. Trials at Weipa, for example, showed that some locally occurring species were not suited to the mined environment, because the lowered topography results in wetter conditions because of its closer proximity to the shallow aquifer (Thornton & Dahl 1996). Also, the removal of the pisolitic bauxite layer meant species had to be selected that could penetrate the hard ironstone layer and grow in the altered substrate. Subsequently, successful establishment was achieved using local *Melaleuca* and *Lophostemon* species that are suited to seasonally waterlogged conditions and *Eucalyptus cullenii* that occurs locally on ironstone ridges. Of the 37 species sown in the most recent season at Pine Creek, 36 occur within a 5 to 6 km radius of the site (Fawcett, pers comm). If older rehabilitated areas are present in the locality, it may be prudent to take provenance selection a step further and harvest seed from plants that have proven their capacity for establishment, survival and propagation in the post-mine environment.

3.2 Seed mixes and sowing rates

Woody species selected for revegetation of mines in the WDT are predominantly of the genus *Eucalyptus*, *Corymbia*, *Acacia*, *Grevillea*, *Melaleuca*, *Brachychiton*, *Terminalia*, *Cochlospermum* and *Erythrophleum* with the addition at some sites of species of *Callitris*, *Allocasuarina*, *Leptospermum*, *Hakea*, *Pandanus*, *Livistona* and *Calytrix*. On mines where the agreed end land-use requires the re-establishment of a native vegetation community, seed mixes typically reflect the species composition of adjacent target communities and evolve through observation of previously revegetated areas.[6] Seeding rates are determined by seed size, viability, ease of collection, substrate quality and the desired end-point.

At the Pine Creek Gold mine, the aim is to create a self-sustaining native community reflecting the species composition of surrounding woodlands. In 1991/92, a seed mix of 17 local species sown at a rate of 1.8 kg/ha produced an acacia dominated community. In 1995/96, a eucalypt-dominated mix comprised of 39 local species was sown at 2.67 kg/ha (Fawcett 1995) and in 1998/99, acacias where excluded completely from the seed mix as they were deemed to be self-establishing in sufficient numbers (Fawcett, pers comm). While there is a lack of detailed data for these trials, Fawcett (1995) believes the evolution of the seed mix has resulted in the establishment of eucalypt-dominated communities reflecting the density and diversity of surrounding communities.

A seed mix trial at Ranger resulted in Cramb et al (1997) recommending that *Acacia* spp be limited to less than 16% by weight and that the ‘standard’ sclerophyll mix be applied at the rate of 7.5 kg/ha. On freshly topsoiled areas at Weipa, a seed mix comprising approximately 35–40 native species is sown at 1.2–1.5 kg/ha, based on seed viability tests each year (Dahl & Mulligan 1996). This seed mix comprises local species and species from other regions in Australia selected for their potential to survive on the mined substrate. At Gove a mixture of about 20 native species at a ratio of roughly 5 *Eucalyptus*: 2 *Acacia*: 1 *Brachychiton*: 1 *Livistona*: 1 *Grevillea*: 1 *Alphitonia* is broadcast at a rate of 400–600 g/ha (Hinz 1992).

Unfortunately, data on the species composition of areas established using these various seeding rates and mixes are unavailable or non-existent, so it is difficult to draw meaningful conclusions about the relative success or failure of these methods in establishing the desired vegetation communities. This highlights the importance of the need to monitor (see section 5) and, secondly, to record and report the outcomes of such trials (see recommendation 6.8) to enable the synthesis of this information into a well-organised, reliable body of information as an industry reference.

3.3 Cover crops

Cover crops are used on many mines in the WDT for short-term erosion control, maintaining mycorrhizal populations in stockpiled soils and to build up soil organic matter.

Generally, a cover crop is achieved using introduced pasture species such as rhodes grass (*Chloris gayana*), couch grass (*Cynodon dactylon*) or *Stylosanthes* sp and high rates of fertiliser (Gray 1994). A cover crop species is selected based upon: ability to bind the soil surface; capacity for rapid germination and growth; ability to survive the dry season; a low growth habit to minimise the inhibition and growth of native species (Hinz 1990a). Experience at Gove indicates that *Chloris gayana* ‘fades out’ of the system after approximately five years due to competition by recolonising native flora and reduced nutrient availability after cessation of fertiliser application (Hinz 1981). Cover crops are no longer incorporated into seed mixes at Pine Creek after Fawcett (1995) found that they out-competed tree seedlings for moisture and resulted in an increased fire fuel load. He also found that, where couch grass had established from the topsoil seedbank, native seedling mortality was 100%.

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[6] Foster (1985) calculated the seeding rate for native species, based on previous season’s revegetation as: seeding rate (g/ha)= density (stems/ha)/[germinants per gram x field success rate].

An assessment of the success of ERA's revegetation of the Jabiru East area (Lane 1996a) found the dense cover of introduced grasses and legumes, which comprised the original seed mix, suppressed the establishment of the preferred native woody and herbaceous species. Ashwath et al (1994a) contended that the introduction of non-native species conflicts with the rehabilitation goals at Ranger mine and expressed concern at the potential of exotic cover crop to invade the adjoining Kakadu National Park.

3.4 Native grasses

Problems associated with grass competition are not limited to exotic grass species. In their assessment of rehabilitation success at Gove mine, Reddell et al (1993) demonstrated an inverse relationship between native grass cover and leaf area index and basal area of trees. Similarly, several researchers at Weipa have demonstrated the competition effect of native grass on seedling establishment (Morton 1983, Foster & Dahl 1990). The competition and the accumulation of fire fuel by native *Sorghum* and speargrass, particularly following the repreading of stockpiled topsoil, is also well documented on hard rock mines in the tropics (eg Fawcett 1995, McGill, pers comm). An investigation of the potential of native grasses in revegetation programs in the WDT (Ashwath et al 1994a), however, outlines a number advantages of using native rather than exotic grasses. Table 2 outlines the advantages and disadvantages of using native grasses in revegetation.

Table 2 Advantages/disadvantages of using native grasses in revegetation

Advantages	Disadvantages (Gray 1994)
The production of large amounts of seed ensuring subsequent self-sowing Ashwath et al (1994)	The poor availability of native grass seed on a commercial scale
Resilience to fire Ashwath et al (1994)	A poor understanding of their ecological characteristics
Adaptation to local conditions Ashwath et al (1994)	Slower growth rates than exotic species
Minimal competition with other native species (Cowie & Finlayson 1986a)	The difficulty of seed collection
The ability to establish without high applications of fertiliser or irrigation (Armstrong 1986)	A lack of understanding of the germination requirements

Ashwath et al (1994a) recommended matching native grass species to mine site conditions through careful selection from the 184 species of native grasses that occur in the Alligator Rivers Region (Brennan 1996b). They suggest, for example, short-term erosion control of steep slopes could be achieved with *Setaria apiculata*, a rapid germinating and growing annual. If longer-term stability is required, *Aristida* sp and *Ectrosia leporina*, which are initially slow growing but produce perennial tussocks and extensive root systems, could be used. *Ectrosia leporina* may have further potential on Ranger WRD in that it has been shown to be tolerant of high Mg:Ca ratios in the soil (Gray 1994).

Ashwath et al (1994a) also suggest native grass species for low lying areas *(Eragrostis* sp, *Ectrosia leporina*, *Echinochloa* sp and *Eriachne burkittii*), well drained areas (*Schizachyrium fragile* and *Eriachne ciliata*) and compacted areas (*Aristida* sp) based on their natural occurance, growth habit and root system. Field trials investigating the performance of 10 native species and rhodes grass conducted in Ranger mine spoil demonstrated that without amendment with gypsum or fertiliser, native grasses can produce similar or greater ground cover than rhodes grass (Ashwath et al 1994a, Gray 1994). Similar results were found by Lane (1997, unpublished data, cited in Batterham 1998) who found high germination rates, long seed viability, large ground cover establishment and no fertiliser requirement by native grasses (*Eriachne schultziana* and *Ectrosia leporina*) trialed at Ranger.

The disadvantages of using native grasses as cover crops (table 2) appear, largely, to be a lack of understanding of their ecological characteristics and germination requirements and poor commercial availability of seed. Research should therefore aim to overcome these limitations.

3.5 Seed technology

Hinz (1990b) nominated seed quality as the most important factor in successful revegetation. Seed quality in turn is a function of proper seed collection, processing and storage techniques (Hinz 1990b, Roberts 1994). While this review provides a brief overview of seed technology applicable to species in the WDT, Langkamp (1987) or more recently Doran and Turnbull (1997) provide comprehensive coverage of this topic.

3.5.1 Timing of collection

Central to the collection of large quantities of viable seed is an understanding of flowering and fruiting phenology of target species. Harvesting the fruit too early may yield immature and possibly unviable seed (McLaughlin 1993), while if collection is late, seed may be shed and lost on the ground (Hinz 1990b). Both Hinz (1990b) and Brennan (1996a) noted the temporal and spatial variation in regularity and intensity of fruit production in the ARR, highlighting the importance of careful observation and timing for seed collection. Brennan (1996a) provides flowering and fruiting periods for 288 tree, shrub, grass and herbaceous species that occur in the ARR. Hinz (1990a) documented the flowering and fruiting times for 56 species of tree and shrub occurring in undisturbed communities adjacent to the Nabarlek uranium mine.

Brennan (1996a) found rainfall and fire to be the most important influences on timing of seed production. Flowering and fruiting of herbaceous species in the ARR was largely confined to wet season months (January to April) with March recording the greatest activity. While 136 herbaceous species were observed flowering or fruiting between February and April, only 7 were observed after April. The flowering of 15 herbaceous species during mid dry season months (June to August) appeared to be stimulated by fire. The two peak fruiting times for tree and shrub species of September and again in February/March generally occurred 2–3 months after flowering times. Several *Eucalyptus* spp, *Gardenia megasperma* and *Planchonella pohlmaniana* displayed an extended fruit to flower maturation (>3 months, up to 9 months). Chandrasekaran et al (1994) also found that most native ground cover species in the WDT mature during the second half of the wet season while most trees and shrubs mature towards the end of the dry season. Williams et al (1999) and Brennan (1996a) showed that phenological events generally occurred at similar times for species throughout the region.

The current information relating to the phenology of species currently used in revegetation programs in the WDT appears to be comprehensive and consistent.

3.5.2 Collection and handling

Methods of seed collection vary according to growth habit of the target species and the ingenuity of the collector. Collecting seed from grasses, herbs or shrubs is generally easier than from trees. Techniques to collect seed from trees (summarised by McLaughlin 1993) include:

- collecting seed or fruit from the forest floor – this technique is commonly used with larger seeded species which shed their mature seed;
- shaking the tree and collecting the fallen seed or fruit on a tarpaulin;
- sawing or breaking whole branches that contain a large quantity of seed;

- shooting very tall seed-laden branches with a rifle;
- climbing to the crown to collect seed.

Chandrasekaran et al (1994) point out that less careful mechanised methods can be utilised for collecting seed for routine rehabilitation, as opposed to collecting for research purposes. Mechanised collecting methods may be restricted by the nature of occurrence of a species within a community (eg homogeneous stands in accessible terrain lend themselves to mechanised collection). Lawn mowers, commercial vacuum harvesters and agricultural harvesters are sometimes used to collect seed on a large scale. Ashwath et al (1994a) discussed the difficulty associated with collecting native grass seed on a large scale. The staggered timing of seed maturity for some species and the fact that native grasses often occur in mixed stands hinders the collection of large quantities of their seed.

McLaughlin (1993) outlined the difficulty of collecting seed from species in the ARR which mature over a staggered period, or produce small quantities of seed, or have a large proportion of the seedlot empty due to the development of an unfertilised fruit (see table 3). The Environment Protection Agency (EPA) (1995) recommends the establishment of a seed orchard for species that are rare, produce limited seed, or are difficult to collect.

Table 3 Species from the Alligator Rivers Region described by McLaughlin (1993) for which the collection of large numbers of seed is often difficult or impossible[1], the seedlot is often empty[2] (fruits developed without fertilisation of the ovaries) or have unknown seed storage requirements[3]

Acacia conspera[1]	*Acacia helicophylla*[1]
Acacia limbata[1]	*Acacia multisiliqua*[1]
Acacia wickhamii[1]	*Brachychiton diversifolius*[3]
Brachychiton megaphyllus[3]	*Canarium australianum*[1,2,3]
Cathormion umbellatum[1,2]	*Cochlospermum fraseri*[3]
Erythrophleum chlorostachys[1]	*Grevillea pteridifolia*[1,3]
Owenia vernicosa[3]	*Pavetta brownii*[2]
Petalostigma pubescens[3]	*Planchonella arnhemica*[2,3]
Terminalia platyphylla[2]	*Terminalia sp*[3]
Terminalia pterocarya[2]	*Xanthostemon paradoxus*[2]

Invariably there is a requirement to extract seed from the collected fruit. Dry indehiscent fruits are sometimes stored and sown whole, while fleshy indehiscent fruits often require processing to remove the seeds if they are to be stored (McLaughlin 1993). The fleshy material may be removed by soaking in water and/or the use of abrasion. The release of seeds from dry dehiscent fruits may be stimulated by drying (sun or <40^{o}C oven). The separation of chaff (portions of the fruit and other collected debris) from the seed is usually done by sieving or winnowing, although this is often impossible for Myrtaceaous species. Methods for processing native grass seed are poorly developed but often adapt agricultural threshing techniques (Ashwath et al 1994a).

3.5.3 Seed storage

The temporal variability of seed production means that collected seed is rarely planted immediately, and invariably requires storage. The loss of seed viability following storage is common and potentially costly (Chandrasekaran et al 1994). The general consensus regarding the storage of tropical orthodox seed is that the seeds should be fumigated with carbon dioxide for 24 hours prior to storage and the storage temperature and relative humidity should be kept low (<10^{o}C and approximately 50% respectively) (Foster 1985, Hinz 1990b, McLaughlin 1993). Storage requirements for several recalcitrant species (eg *Persoonia, Terminalia*, *Buchanania, Ficus* (Hinz 1990a)) – often included in revegetation programs for their bush food value – are presently unknown (see table 3). These species, several of which fruit immediately prior to the wet season (Brennan 1996a), may be immediately nursery-propagated and planted at a later date. Kabay and Lewis (1987) recommend storing recalcitrant species in airtight containers with silica gel at 1 to 4^{o}C. Schaefer et al (1989), as quoted in Chandrasekaran et al (1994), identified the storage of recalcitrant species at ultra low moisture content (3–5%) as a high priority research area.

3.5.4 Seed treatment

The success of revegetation programs adopting a direct seeding approach depends on the seeds germinating quickly, evenly and in as large numbers as possible (McLaughlin 1993). Both dormant and non-dormant seeds may display increased vigour, evenness and/or capacity for germination after treatment prior to sowing. Dormancy may increase the time that sown seeds are exposed to predators and adverse weather conditions. For non-dormant seeds, immersion in water at room temperature results in a rapid uptake of water that may otherwise not occur, resulting in increased vigour and evenness of germination.

Doran (1986) states that the most suitable treatment is one that allows the nurseryman to apply the method practically to a large quantity of seed. For example, in the study of the germination characteristics of 27 tree, shrub and palm species from the Alligator Rivers Region, McLaughlin (1993) found that immersion in boiling water for 60 seconds or manual nicking of the hard seed coat of *Acacia* species consistently increased germination, but that nicking large quantities of seed was not practical. Hinz (1990a), however, believed that pre-treatment of seed was not required to achieve successful revegetation at Nabarlek mine, and that the advantages conferred to a species by natural mechanisms should be left alone. He used the example of acacia seed, which if treated would germinate shortly after sowing and be susceptible to periods of dry desiccating weather that may occur between early wet season storms. Similarly, the revegetation program at Gove bauxite mine does not attempt any pre-treatment of seed but prefers to rely on natural stimuli to germinate seed (D Hinz, pers comm, consulting ecologist to Nabalco Pty Ltd).

Treatments to overcome seed dormancy can be divided into those that overcome seed coat imposed dormancy and those that overcome embryo-imposed dormancy. The latter of these two dormancy mechanisms is, however, uncommon in tropical tree species seed (McLaughlin 1993). Treatments to overcome coat-imposed dormancy include water treatments (soaking in cool or hot water), manual or mechanical scarification of the seed coat, chemical scarification of the seed coat, dry heat and smoke treatments (see table 4). No work to date has been published on the effect of smoke on the germination of species in the WDT. A project is currently underway, however, to assess the potential of this method to enhance the poor germination of some native species (including; *Xanthostemon paradoxus, Brachychiton paradoxus, Terminalia latipes*) used in revegetation at Ranger mine (Alligator Rivers Region Technical Committee 1998b).

McLaughlin (1993) described several species in the ARR for which little is known of their germination requirements (see table 5). This list is derived from the 27 species commonly used in revegetation studied by McLaughlin and is by no means exhaustive.

Table 4 Seed treatment requirements for some genera commonly used in revegetation programs in the wet-dry tropics

Eucalyptus	Generally no pretreatment required (Doran 1997)
Acacia	Immersion in boiling water for 60 seconds, 1:10 seed:water ratio (McLaughlin 1993, Doran 1997)
Grevillea	McLaughlin (1993) lists *G. pteridifolia* as very difficult to germinate. Doran (1997) found the germination of some grevilleas (including *G. pteridifolia*) was stimulated by a 24 h soak in cold water. Lawton (pers comm) reports improved germination after soaking in 5% KNO_3.
Melalaeuca	Generally no pretreatment required (Doran 1997)
Brachychiton	Generally no pretreatment required (McLaughlin 1993, Doran 1997)
Terminalia	Difficult to germinate. More research required on the manual scarification of the thick woody seed coat (McLaughlin 1993)
Cochlospermum	Manual nicking or >30 minutes soak in H_2SO_4 improved germination of *C. fraseri* (McLaughlin 1993)
Erythrophleum	*E. chlorostachys* germinates readily without pretreatment (McLaughlin 1993)
Callitris	Generally no pretreatment required (Doran 1997)
Casuarina	Generally no pretreatment required (Doran 1997)
Leptospermum	Generally no pretreatment required (Turnbull & Doran 1987)
Hakea	Lawton (pers comm) reports improved germination after soaking in 5% KNO_3
Pandanus	Generally no pretreatment required (Fox et al 1987)
Livistona	Slow germinating palm species (up to 6 months). McLaughlin (1993) recommends soaking or leaching the fruit in warm (30°C) water with research required on soaking times and the effect of wetting/drying cycles
Calytrix	Generally no pretreatment required (Turnbull & Doran 1987)

Table 5 Selected species from the Alligator Rivers Region which require further studies in regard to their germination requirements (McLaughlin 1993)

Canarium australianum	*Cochlospermum fraseri*
Grevillea pteridifolia	*Livistona humilis*
Owenia vernicosa	*Petalostigma pubescens*
Planchonella arnhemica	*Terminalia latipes*
Terminalia platyphylla	

Ashwath et al (1994a) found that of the 30 species of native grass from the ARR that were tested, 14 species failed to germinate and 5 species had low (<10%) germination. They concluded that germination characteristics of native grasses are poorly understood and further study is required to fully exploit them in mine site revegetation. The same applies to many woody and herbaceous species of WDT savannas (Ashwath et al 1994b).

Methods for collecting, handling and germinating seed of species to be used in revegetation in the WDT appear to be ad hoc rather than scientific and consistent. Development of research and the synthesis with current successful methods of collecting and handling native seed in the WDT into an industry manual would be very useful. Such a manual should incorporate phenological data, collection, storage and germination characteristics.

3.6 Seeding methods

3.6.1 Tubestock vs direct seed

The majority of revegetation programs on mines in the WDT rely on broadcasting seed directly onto areas to be revegetated. The discussion of the relative advantages of direct seeding or planting nursery grown tubestock is well represented in the proceedings of a direct seeding conference organised by Greening Australia (1990). Kabay and Lewis (1987) state that direct seeding results in a more spatially ‘natural’ vegetation community, natural selection of ‘fit’ seed falling in favourable niches, a greater genetic diversity and is generally highly cost effective.

Rehabilitation experience at Nabarlek mine indicates that the performance of direct seeded revegetation is superior to that of tubestock planted areas (Hinz 1990b). In addition to the cost savings and the ability to cover large areas in a short space of time, direct seeding provided a greater ability to control diversity and density in a given area and the plants grown from seed were generally more healthy and robust. Revegetation at Pine Creek mine utilises direct seeding methods exclusively, due to both economics and field observations that indicated directly seeded plants displayed superior growth rates compared with tubestock (Fawcett 1995) (see plate 5, p23). Both Fawcett (1995) and Queensland Mines Ltd (1990) reported a ‘stagnation’ of tubestock shoot growth after planting. At Gove bauxite mine, Hinz (1992) found direct seeded plants developed a stronger root system than tubestock and the plants adapted to local conditions more rapidly and subsequently proved more sustainable. Mulligan (pers comm, CMLR) proposes that if a sustainable community is to be established, seed must be able to germinate in the post mining media and that the establishment of a tube-grown plant may not result in its self propagation.

In contrast, Reddell and Hopkins (1995) found that tubestock revegetation trials at Ranger mine had consistently outperformed direct seeded trials. A study on the effectiveness of mycorrhizal fungi on seedling survival and growth rates by Gordon et al (1995) reported survival rates of *Eucalyptus miniata* tubestock on Ranger WRD to be 95–100% regardless of treatment. It should be noted, however, that this trial was irrigated.

At Ranger, a system of ‘ecological islands’ were trialed in 1989 and 1992 whereby small, high-input areas (topsoiled and tubestock planted) were dotted throughout areas to be rehabilitated by less expensive direct seeded areas. To date establishment in the inter-island areas is poor and there is little evidence of natural recruitment from the ‘islands’ (see plate 4, p22). The persistence and development of such islands, however, is encouraging

In terms of the rehabilitation goal of establishing self-sustaining vegetation communities reflecting the surrounding unmined environment, the weight of evidence suggests direct seeding appears to be superior to tubestock. Contrary evidence is restricted to studies on Ranger mine, where the trials of ‘ecological islands’ suggest that although establishment may be superior, subsequent propagation is restricted. Research should examine the specific limitations to germination and establishment of seed in Ranger WRD soils.

3.6.2 Timing of seeding or planting

Soil moisture status has been described as the single major determinant of the distribution and abundance of vegetation in the WDT with the greatest potential for variability (Ridpath 1985). The duration, intensity and the onset of the wet season show appreciable variability (Taylor & Tulloch 1985). Dry periods of 2 to 3 weeks frequently occur early in the wet season and will dramatically reduce survival of previously sown seed. Therefore sowing of seed or tubestock in the WDT should ideally coincide with the onset of *consistent* rainfall (Foster & Dahl 1990, Fawcett 1995, Thornton & Dahl 1996). In his study of the germination characteristics of 27 species of the Alligator Rivers Region, McLaughlin (1993) found germination conditions were most favourable in early to mid wet season.

Similarly, it was recommended by Nisbet (1997) that planting of nursery-grown tubestock at Ranger mine should commence in early January. Ranger has also trialed the use of irrigation to assist vegetation establishment on the WRD with mixed results. Cramb et al (1997) found irrigation increased the survival rate of seedlings on the Ranger WRD seven-fold. Lane (1996b), however, indicated the establishment of native vegetation on Ranger WRD was not improved by dry season irrigation. Non-acacia/eucalyptus species demonstrated greater survival under a rain-fed regime (Lane 1996b) and a combination of dry season irrigation and fire exclusion produced a change in community structure favouring acacias (Lane 1996c).

Taylor and Tulloch (1985), in their study of rainfall in the WDT over a 113 year period concluded that in order to achieve predictable responses in rain-dependent activities, management in the region must be opportunistic and scheduled according to biological, rather than calendar events.

3.6.3 Method of seeding or planting

Seeding is generally carried out with a ground broadcaster, aerially or by hand. When using a ground broadcaster, care needs to be taken to preserve soil structure and to prevent the initiation of gully erosion. Fawcett (1995) found hand seeding to be cheaper, and as efficient as mechanical ground broadcasting and less damaging to the integrity of the seedbed. Hinz (1990b) also found hand broadcasting to be the most effective method for small areas.

At Weipa, seed is mostly broadcast by tractor-mounted spreader with large-seeded species planted by hand (Schwenke 1999). Areas deemed unsuccessful through subsequent monitoring (approximately 2 months after seeding) receive supplementary hand seeding or in some cases, advanced nursery-grown plants. Seed is broadcast onto large and inaccessible areas at Ranger mine using a hydroseed canon (Batterham 1998). Though more expensive, seed may also be broadcast on large areas by plane. In a report on slope research and erosion to Ranger, Loch (1998) recommends that remedial work be carried out to establish vegetation on areas that have not achieved adequate cover.

Batterham (1998) recommended that tubestock should be a minimum of 3 months old before planting and that the seedlings be hardened prior to planting by decreasing water supply and increasing exposure to sunlight. Chandrasekaran et al (1994) stressed the importance of the following when transplanting native seedlings: loosening circled roots; trimming excessive roots; placing the seedling at an appropriate depth (to prevent collar rot); compacting soil around the planted seedling; providing protective irrigation immediately after transplanting and reducing leaf area to minimise transpirational loss during the initial establishment phase. A useful summary of nursery practices and planting techniques for tropical Australia can be found in Doran (1997).

3.7 Succession

Minesite rehabilitation has been described as managing a succession from the post mining landscape to (in some cases) a self-sustaining ecosystem (Werner & Wigston 1989). Rehabilitation seeks to accelerate succession such that a relatively stable ecosystem is achieved over a short time scale. Tongway et al’s (1997) analysis of ecosystem success on 13 mine sites across Australia, including 4 sites in the WDT, notes that:

> mines which took pains to establish the desired final species mix early on were conspicuously more successful than those which relied on some hopeful expectation of favourable succession.

Plate 1 Seven year old topsoil experiment at Ranger mine showing domination by grass and acacias, March 1999 (photograph Matt Corbett)

Plate 2 Grevilleas and eucalypts growing in minesoil in Ranger mine soil, March 1999 (photograph Matt Corbett)

Plate 3 Ground preparation on Ranger mine three months post-planting, March 1999 (photograph Matt Corbett)

Plate 4 Poor vegetation establishment on areas adjacent to an ‘ecological island’ established 1992 at Ranger mine, March 1999 (photograph Matt Corbett)

Plate 5 Hand seeded area on waste rock at Pine Creek mine after two wet seasons, March 1999 (photograph Matt Corbett)

Plate 6 ‘The cricket pitch’ at Woodcutter’s mine representing 50 year-old natural regeneration and succession, March 1999 (photograph Matt Corbett)

Plate 7 Woodcutter's mine WRD sown with an exotic grass cover crop in 1997 (photographed March 1999, Matt Corbett)

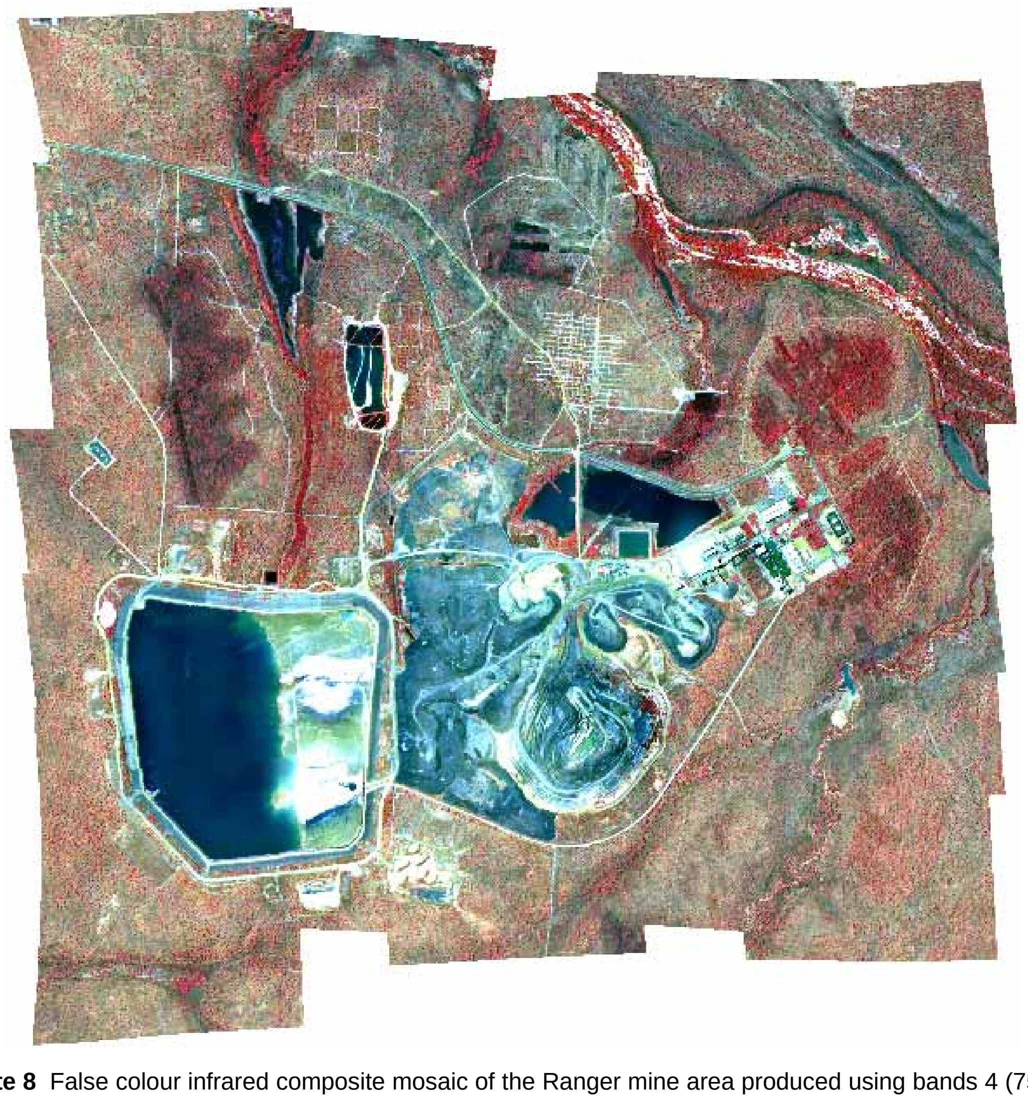

Plate 8 False colour infrared composite mosaic of the Ranger mine area produced using bands 4 (758–782 nm), 3 (638–662 nm) and 2 (538–562 nm) from DMSV images acquired on 28/9/95. Defines different patterns of vegetation condition and distribution at a spatial resolution that shows single trees (from Ong & Hick 1998)

Both Bell (1996) and Menzies and Mulligan (1996) agree that an initial attempt should be made to establish native understorey, ground cover and grass species in addition to canopy species, the long-term densities of which will be driven by succession.

Structural or competitive dominance in a community can prevent the ascendancy of later seral species (Connell & Slatyer 1977). In some instances, perennial grass or legume monocultures have resisted successional change (Luken 1990). On disturbed sites in the WDT, the dominance of early seral species such as acacias has been observed to effectively stagnate successional development, with recruitment of eucalypts and other species to these systems being poor (Setterfield et al 1993). Hinz (1990a), however, found recruitment or introduction of eucalypts and other genera was more successful after the establishment of pioneering acacias.

There is little literature on the long-term successional development of previously mined areas in the WDT, with the exception of studies by Reddell et al (1993) at Gove bauxite mine and Roberts (1994) at Weipa bauxite mine. At Weipa, Roberts (1994) found rehabilitated areas dominated by acacias and expressed concern at their potential to inhibit succession. At Gove, three distinct phases were described with regard to the development of vegetation communities over a 16-year unburnt period (Reddell et al 1993): the initial 3–4 years post rehabilitation, when biomass at the site was dominated grasses; the subsequent 4–10 years in which acacias are the dominant biomass component; and the period between 10–16 years, which saw a reduction in the growth rates of acacias and signs of senescence. During this third period, eucalypts developed an emerging canopy. Reddell et al (1993), however, were unable to predict the development of the lower vegetation strata or the impact of fire on rehabilitated areas.

In 'Directing ecological succession' Luken (1990) outlines several key parameters that managers may use to manipulate successional processes:

- nutrient availability (fertiliser regimes/symbiotic microorganisms/litter cycling – with nitrogen the most significant nutrient)
- physical removal of unfavourable species
- fire
- water availability (irrigation)
- propagule availability (seeding/topsoiling).

An interesting site representing natural regeneration and succession is located at Woodcutter's mine (Normandy Mining Ltd). A native woodland with well-developed canopy, understorey and ground cover has re-established on a World War II cricket pitch (presumed to have been a 100 m radius clearing) abandoned without any active regeneration. Though this area may provide a useful research site for successional development studies, it differs from a disturbed mine area in that it is small, flat, would have a relatively undisturbed soil profile and has had over 50 years to redevelop (see plates 6 & 7, p23–24.)

There is often a discrepancy between the time scale of revegetation programs and/or lease relinquishment of many mines and the time required to effect succession. Research, therefore, should aim to develop methods to enable accurate prediction of the successional development of young revegetated communities. This requirement was illustrated at the Nabarlek mine where current revegetation (<4 years old) does not reflect the adjacent unmined woodland. Prendergast et al 1999 have questioned whether given the current structure and composition of the revegetated landscape, succession to a woodland will take place in the absence of further inputs.

4 Disturbance regimes

The post mining land use agreement that a self-sustaining vegetation community be established requires that rehabilitated sites be able to withstand disturbance. This is particularly critical upon relinquishment of a lease, where there may be little further management input to the re-established system. It is also vital throughout the establishment phase to be able to predict and manage an area's response to disturbance. It is unlikely that any area at Ranger mine will be subjected to grazing pressure and thus for the purpose of this review, consideration of disturbance will be restricted to fire and weed invasion.

4.1 Fire

The importance of fire in the formation of northern Australian biota and its historic and contemporary use as a land management tool is well documented (eg Andersen et al 1998, Bowman 1998, McKaige et al 1997, Russell-Smith et al 1997). The existing literature pertaining to the role of fire in tropical systems, however, focuses on mature systems rather than young rehabilitated areas. McGill (pers comm) believes that fire is the single most important issue affecting the success of revegetation efforts on mine sites in WDT.

Anthropocentric burning regimes account for the vast majority of dry season fires in the WDT (Williams et al 1998). These burning patterns – which typically see tropical savannas burnt at frequencies of between one and three years – are often at odds with attempts to restore natural ecosystems (Tongway et al 1997). The paucity of research examining the relationship between fire and rehabilitated lands in this region (Hinz 1996) is therefore conspicuous. The exceptions are Williams and Lane's (1999) investigation of wet season burning as a fuel management tool as Ranger mine, Hinz's (1996) research on the effects of fire on the rehabilitation at Gove mine and Hooper's (1985) discussion of the use of fire as a management strategy in regenerated mined lands in northern Australia.

Issues associated with fire and rehabilitated areas in the WDT can be summarised as:

- age class, or height of the dominant stratum that can survive fire
- burning regimes (protection from fire, frequency and timing of burning)
- maintenance of species and stratum diversity
- fuel loads (*Sorghum* sp)
- nutrient loss (litter cycling)
- fire management in lands adjacent to rehabilitation areas.

During the vegetation establishment phase, protection from fire is crucial. Juvenile woody species may be killed by fire early in revegetation, constituting a significant setback to the process (Williams & Lane 1999). Prescription management burns may be used to reduce fuel loads and thus the risk of fire or the potential fire intensity.

As the dominant vegetation at ground level in WDT savannas, grasses constitute the majority of fire fuel (Williams & Lane 1999). Native annual *Sorghum* species grow up to 2 metres in the wet season, brown off rapidly and burn fiercely unless conditions are very moist. At Ranger mine, wet season burning trials were carried out in an attempt to reduce the fuel load associated with the *Sorghum* (Williams & Lane 1999). They found that the optimal time to burn is mid-wet season (late December) when the greatest reduction in the density, cover and seedbank of *Sorghum* that will have germinated but not seeded is achieved. Additionally, during this time, the landscape can carry fire at spacial scales appropriate for managing mine sites (square kilometres). This regime is supported by Mott and Andrew (1985) who found no persistent seed store of *Sorghum* following early wet season germination. Williams and Lane (1999), however, found that the cover and seedbank of *Sorghum* had returned to pre-fire levels by the second dry season post-fire and recommended that wet season burns would need to be carried out every second year if reduced fuel levels are to be maintained.

The frequency with which prescription burns are applied is important. Tongway et al (1997) found analogue sites at Gove, Weipa, Pine Creek and Ranger mines that had been subjected to frequent burning (1–3 years) and showed an absence of ground layer vegetation, a narrow range of fire-tolerant species and restricted age class diversity. They observed that rehabilitation plots at Gove mine that had been protected from fire since 1974 'had a remarkably different species composition at all levels and an open, friable soil fabric with extremely active visible decomposition processes mediated by micro-arthropods and fungi'. Hooper (1985) states that protection of rehabilitation areas from fire results in the development of an understorey, the removal of grass cover through shading and the build up of litter. Hinz (1996) concluded that fire in young rehabilitation areas hinders soil development and nutrient cycling processes and may produce an unstable monoculture. The threat to rehabilitated areas, he also argues, is not fire as such, but its frequency and timing. While Williams et al (1999) also found that long-term absence of fire from WDT savannas appears to increase structural and compositional complexity, they found that under this regime, any unplanned fire would be likely to be of high intensity and would result in high mortality of tree stems. In relation to mature WDT savanna, Williams et al (1999) note that 'a balance needs to be struck between the possible or perceived deleterious effects of a frequent-fire regime and the risks of intense fires if a regime of low-frequency fires is prescribed'.

Both Roberts (1994) and Foster (1985) provide a valuable review of the relationship between fire and the litter cycling processes at Weipa mine. Because the decomposition rate of litter declines after mining due to reduced soil microorganism populations, litter may accumulate in rehabilitation areas at greater rates than in native forests. Heavy fuel loads created by the build up of litter increase the potential for destructive hot fires. For this reason, fire is excluded from revegetation at Weipa for at least ten years to enable trees to reach a height at which they can survive fire, build up invertebrate and fungal populations and re-establish litter cycling processes.

The fate of nutrients during fires in tropical savannas was described by Cook (1994). He found that most nutrients were redistributed locally through the transfer of ash and that permanent loss are more likely for those nutrients transferred in the non-particulate or gaseous form, in this case of Nitrogen. Cook (1994) estimated rates of biological fixation of N to be insufficient to replace persistent losses and expressed concern at such losses from a system already low in this nutrient. He recommended that to prevent the depletion of N reserves, the frequency of fires should be decreased.

Eucalyptus is generally considered the most fire resistant genus in the world (McArthur 1968). Although this resistance has been demonstrated to be age or size dependent (Williams et al 1999, Cowie & Finlayson 1986a), the height class of the woody component of rehabilitated systems that may resist fire has yet to be determined. The complex interaction between acacias and fire is discussed by Cowie and Finlayson (1986a). While many acacias occurring in revegetation in the WDT are susceptible to fire (eg *A. holosericea*), they will regenerate readily from soil stored seed. Frequent fire, however, would exhaust this seed supply.

To summarise the effect of fire on fauna, Corbett (1998) indicated that 'early-dry season' fire has less impact on fauna than 'late-dry season' or 'no fire' regimes. Williams et al (1998) found the mean intensity of early dry season fires was almost one quarter that of late dry season fires but also state that the season of burning is not necessarily a precise predictor of fire intensity. This highlights the care that must be taken if controlled burns are to be applied to rehabilitation areas. While it is beyond the scope of this review to examine fire behaviour and burn management, Cheney and Sullivan (1997) provide an excellent description in ‘Fuel, weather and behaviour of grass fires’.

An understanding of the influence of fire on the species and structural composition and succession of mature WDT savannas is developing (eg Williams et al 1999, Andersen et al 1998). However, Reddell et al (1993) describe fire as ‘the most critical unresolved factor that could have a major impact on the development of the vegetation communities in [the] rehabilitated areas’.

The majority of mining operations in the WDT are under an agreement to return the land to self-sustaining natural systems. These systems will be challenged with fire, be it in their infancy or as they begin to approach systems reflecting the surrounding communities. It is crucial to the sustainability of these re-establishing systems that fire and its intrinsic relationship with other aspects of rehabilitation operations be understood and managed and that burning regimes on mine sites are developed in the context of the surrounding land management. Therefore it is important to establish the age class of the woody component of the rehabilitated landscape that may be fire resistant and the optimal frequency and intensity of a managed fire regime.

4.2 Weeds

The Environment Protection Agency (1995) states that weed infestations on rehabilitated areas can be very difficult to control and emphasis should be placed on prevention rather than cure. If alien plants persist or invade revegetation areas, then the objectives of rehabilitation may not have been met (Cowie & Finlayson 1986a). The requirement that Ranger mine be incorporated into the adjacent Kakadu National Park after completion of rehabilitation demands strict weed management. The total number of alien plants within the ARR is low (71, or 5.3% of all vascular plants) and so the potential for weed invasion may be lower than for other regions but the consequences of weed infestation would be greater (Cowie & Finlayson 1986b). In this context, Prendergast et al (1999) expressed concern at the high proportion of weeds (>35%) in the herbaceous flora at the Nabarlek mine.

Exotic species may be introduced as contaminants in commercial seed mixes, in imported soil and materials, on vehicles, via natural dispersal mechanisms or be deliberately introduced. Mechanical disturbance of vegetation and soil, elevated nutrient levels following fertiliser application, altered water and fire regimes are all factors which may increase the susceptibility of revegetation areas to weed invasion (Cowie & Finlayson 1986a). Storrs’s (1996) weed management strategy for Kakadu National Park provides a comprehensive and practical examination of the issue and could be easily adapted to a mine situation. He subscribes to the prevailing successional theory that weeds are a symptom of poor land management and that emphasis should be placed on prevention of invasion. His strategy is built around 7 key principles:

- prevention
- identifying ecosystems prone to invasion
- surveillance and early intervention
- minimising an area’s susceptibility to invasion
- managing existing weeds

- researching existing and potential weed problems
- undertaking regular reviews.

Implicit in these principles is their dependence on other aspects of land management. In the case of a mine site, all reconstructed ecosystems will be initially prone to weed invasion. Prevention and minimisation of an area's susceptibility to invasion will rely on careful planning to minimise disturbance to an area and the rapid establishment of a stable, vegetated landform. Surveillance and intervention require an effective monitoring regime. Managing existing weeds requires an understanding of current eradication methodologies, successional and fire theory, and the ecology of invasive species. Cowie and Werner (1987) outline some general weed control measures which should be observed:

- Always work from areas of low infestation toward areas of high infestation
- Minimise disturbance
- Allow regeneration of desirable species to determine the rate of weed removal
- Time weed removal to precede seeding
- Remove flowers or seed heads where possible.

Lists of alien plant species occurring in the ARR and their attributes are presented in Storrs (1996) and Cowie and Finlayson (1986a).

5 Monitoring and indicators of rehabilitation success

There is a requirement, both during and upon completion of rehabilitation, for the redeveloped landform, soils and vegetation to be monitored and assessments made of how successful the rehabilitation process has been and is likely to be. Success criteria enable mining companies, communities or landowners and regulating authorities to determine whether post-mining land use agreement conditions are being or have been met. These criteria are poorly developed (Bell 1996). Unlike the monitoring of freshwater for example, there have been no quantitative industry standards developed to assess revegetation quality. The development of acceptable standards is handicapped by non-specific, ambiguous and often unrealistic goals of final land-use agreements (ie what is to be achieved?) and completion criteria (ie when will it be achieved?). Ranger mine has an agreement to return the mined area to a state reflecting the surrounding Kakadu National Park, whereas at Weipa the aim is not necessarily to mirror the unmined environment but rather to ensure the establishment of a self-sustaining native community. At Nabarlek and Gove mines, traditional land uses such as hunting and gathering are central to the revegetation effort (Hinz 1990a, Hinz 1996). Underpinning all of these rehabilitation attempts, however, is the requirement for the rehabilitated system to be stable and self-sustaining. Chapin et al (1996) define a sustainable ecosystem as one that maintains its characteristic diversity of major functional groups, productivity, soil fertility and rates of biogeochemical cycling over the normal cycle of disturbance events.

For the purpose of this review, methods to monitor the development and success of rehabilitation areas are divided into five categeories, each of which will be discussed separately in this section.

- Quantitative ecological assessment (composition)[7]
- Ecosystem functional analyses (function)

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[7] Broad classifications of these methodologies based on ecosystem attributes proposed by Hobbs and Norton (1996) are in brackets.

- Remote sensing (structure/pattern)
- Faunal recolonisation (function/composition)
- Other indices of ecosystem recovery

Comparison of rehabilitation areas with ‘natural’ or analogue sites forms the foundation of all these modes of assessment and is the point of some contention. An initial discussion is therefore appropriate on the selection and use of analogue or reference sites.

Tongway et al (1997) suggest that the use of reference sites relies on the assumption that such sites are either static or represent the optimal state of the ecosystem. Difficulty arises in knowing where a reference system lies within its successional development at any point in time. Hobbs and Norton (1996) note the inadequacy in measuring success against a system which may be poorly understandood in terms of its composition, structure, function and dynamics. They consider that measuring restoration success against one particular ‘natural’ state constrains restoration efforts and promotes the setting of unattainable goals. When defining rehabilitation goals it should be recognised that even under natural conditions, alternative states are possible for any one location.

Aronson et al (1995) on the other hand stress the importance of some standard for comparison and evaluation, even if the ‘ecosystem of reference’ is arbitrary and imperfect. They acknowledge that there will be no ideal system or state to be used as a blueprint but see no alternative to a rehabilitation project having a reference by which to evaluate results. Clewell and Rieger (1997) contend that while comparison with a single reference ecosystem is inadequate, discrepancies caused by successional differences between rehabilitation sites and reference sites, as well as variation within reference sites, can be minimised by utilising multiple reference sites. They argue that descriptions of reference ecosystems must encompass the developmental stages of those systems. Norton (1991) states that if reference ecosystems are to successfully guide rehabilitation, they should be based on similar landforms, soil, biotic and climatic conditions and their range of potential conditions should be recognised.

5.1 Quantitative ecological assessment

Typically, assessment of rehabilitation success has involved a comparison of rehabilitation areas with target communities based on traditional quantitative ecological assessment, often of soils and vegetation. Species abundance, species diversity, structural diversity, plant growth, community ordination and soil analyses allow a comparison of communities. This approach usually involves the establishment of permanent grids, transects or quadrats to collect statistically valid data that can be used to compare the *composition* of paired communities. Temporal changes in composition are usually measured also. Should a rehabilitated area not reflect the composition of the reference area, this approach should provide a reasonably precise diagnosis on which to base remedial work. The assessment of the relative success of sown species also facilitates the comparison between, and refinement of, rehabilitation techniques (Foster 1985, Thornton & Dahl 1996).

Clewell and Rieger (1997) believe that in developing models for successful rehabilitation, a thorough account of species composition is a crucial aspect of the description of ecosystems and their developmental stages. Similarly, Palmer et al (1997) conclude that ‘practical restoration efforts should rely heavily on what is known from theoretical and empirical research on how communities develop and are structured over time’. Quantitative measurement of parameters is often more labour intensive, but is invariably more robust and less ambiguous than an indicator value of those parameters. Traditional, quantitative ecological assessments, however, often assume edaphic and vegetational parameters to provide reliable indices of ecosystem health.

5.2 Ecosystem functional analysis

Many contend that the measurement of rehabilitation success should focus on community *function* rather than species composition (Hobbs & Norton 1996, Palmer et al 1997, Tongway et al 1997) or that the role of individual species within the function of the community be targeted (Palmer et al 1997). With the emerging disciplines of landscape and restoration ecology, Ecosystem functional analyses (EFA) (Tongway et al 1997) has developed as a method to assess the degree of success of rehabilitation efforts. Originally devised for application to rangelands and largely based on the assessment of various ecosystem *functions*, EFA is being developed with the mining industry to determine the proximity to conditions that meet completion criteria. Several mines in the WDT (including Ranger, Gove, Pine Creek, Union Reefs and Brocks Creek) have used or anticipate using EFA to assess their rehabilitation efforts.

Tongway et al (1997) identify indicators of rehabilitation success on lands affected by mining. The EFA procedure incorporates three components to compare indicator values for rehabilitation sites with analogue sites:

1 Landscape Functional Analyses (LFA)

2 Vegetation development

3 Habitat complexity

LFA, which forms the foundation of the methodology, is a rapid field procedure that appraises a site in terms of the distribution, regulation and utilisation of vital resources such as water and nutrients. The method is characterised by an assessment of resistance to erosion (resource loss), water infiltration or storage (resource retention) and nutrient status of the soil (resource utilisation). These functions are interpreted by the spatial arrangement of 'source' zones, or areas of erosion of soil and organic matter by water, and 'sink' zones, where those resources are absorbed or deposited. This component also makes an assessment of the soil surface condition class (SSCC) based on 10 indicators (see table 5). Indices of stability, infiltration and nutrient cycling are derived by amalgamating several of these SSCC indicators.

Vegetation development is assessed using measurements of floral species composition, species similarity to the analogue site, presence of 'framework' species[8] and development of target species, in spatial relation to the source/sink zones. Structural characteristics, particularly of 'framework' species are also measured. An index of habitat complexity is also determined by estimating canopy cover, shrub cover, ground cover, free water availability and the amount of litter, logs and rocks.

All of these indices are determined through the allocation of qualitative scores, which effectively produces a scorecard that enables a comparison of areas. Perhaps most appealing to rehabilitation practitioners is the ease and cost effectiveness with which the assessment can be made (approx. 2 hr/site). The qualitative nature of the method may, however, produce problems of consistency when applied by different assessors.

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8 'Framework' species, such as those that provide shade or shelter, survive fire or facilitate nutrient cycling, have a higher indicator value

Table 6 Soil surface indicators and their process-based interpretations (from Tongway et al 1997)

Indicator	**Interpretation**
1. Soil Cover	Assess vulnerability to rainsplash erosion [1]
2. Basal cover of perennial plants	Assesses contribution of below-ground organs to nutrient
3. Litter cover, origin and degree of decomposition	Assesses the availability of surface organic matter for decomposition and nutrient cycling [1,2,3]
4. Cryptogram cover	An indicator of surface stability, resistance to erosion and nutrient availability [1,3]
5. Crust brokenness	Assesses loose crusted material available for wind ablation or water erosion [1]
6. Erosion features	Assesses the nature and severity of the current soil erosion features [1]
7. Deposited materials	Recognises mobile soil deposits [1]
8. Microtopography	Assess surface roughness for water infiltration, flow disruption and seed lodgment [2,3]
9. Surface resistance to erosion	Assesses likelihood of soil detachment and mobilization mechanical disturbance [1,2]
10. Slake test	Assesses soil stability/dispersiveness[1]

1 – used to derive *stability* index;

2 – used to derive *infiltration* index;

3 – used to derive *nutrient cycling* index

The application of a system largely designed to assess rangeland condition (LFA) (Tongway & Hindley 1995) may also pose some limitations in a mine situation. For example, Tongway et al (1997) found that the infiltration index at some sites was higher than the associated analogue site due to the 'competent preparation of troughs and banks'. This highlights the potential for rehabilitation techniques to be tailored to enhance indicator values rather than ecological values. A modified soil surface, through cultivation for example, may not persist. Similarly, litter material may accumulate more rapidly in rehabilitation areas than in native communities due to the initial absence of decomposing microorganisms (Foster 1985). This could result in an inaccurate indication of the nutrient cycling function of the system. However, as Tongway et al (1997) point out, the indicator system is not designed to replace other detailed investigations but to provide a tool to determine the *trajectory* of the rehabilitated area towards the desired end point. They contend that the method will determine the *proximity* of a system to conditions that satisfy the completion criteria (see figure 2). EFA is currently undergoing modification to improve its applicability to mine sites (A Lane, pers comm, ERA Environmental Services Pty Ltd).

When used in conjunction with other more quantitative assessment approaches, this system could provide a simple, rapid and cost effective diagnostic tool for the rehabilitation practitioner. It would be premature, however, to rely on this method as a means to achieve a desirable rehabilitation 'score' to facilitate relinquishment of a mining lease. It would be a valuable exercise to subject the method to some vigorous validation against a suite of quantitative approaches that encompass the functions examined by EFA, to assess its accuracy and efficiency as an indicator of rehabilitation success.

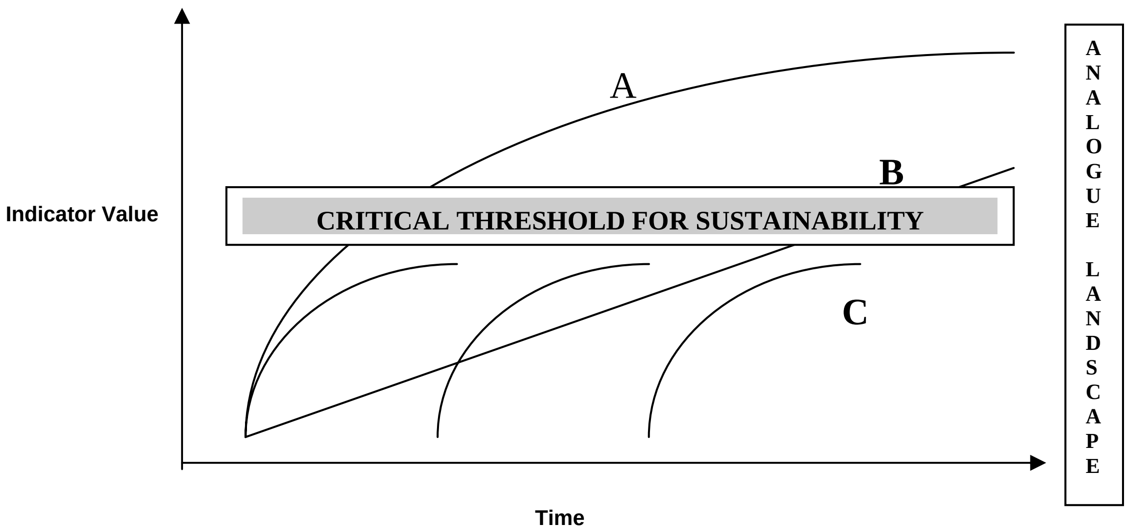

Figure 2 Three rehabilitation trajectories (from Tongway et al 1997). Curve A represents a desirable trajectory, characterised by a steep initial response followed by a steady increase over time. Curve B represents a system that is progressing towards a condition reflecting the analogue landscape but remains vulnerable to disturbance events. Curve C represents a system that has low resilience and requires remedial work to achieve sustainability.

5.3 Remote sensing

Remote sensing is the science of obtaining information about an object or area through the analysis of data acquired by a device that is not in direct contact with the subject under investigation. The devices or sensors used to obtain these data are referred to as active or passive: passive if they measure reflected or emitted solar radiation; active if they generate, radiate and measure that radiation returning from interaction with the atmosphere and features on the earth's surface.

There are four main elements of remotely sensed data to be considered when attempting to define features on the surface (Strahler et al 1986):

- Spectral resolution refers to the number and width of specific wavelength intervals in the electromagnetic spectrum to which a sensor is sensitive eg blue, green, red, near-infrared (NIR), mid-infrared (MIR), etc.
- Spatial resolution determines the smallest feature able to be detected by the sensor.
- Radiometric resolution defines the sensitivity of a detector to differences in signal strength as it records emitted or radiant flux from the target of interest.
- Temporal resolution refers to the time between successive image acquisitions of an area, and the time of day that an image is acquired.

All the above elements need to be considered when mapping and monitoring different vegetation types using remote-sensing techniques. Frameworks have been developed to provide an objective approach to selecting remotely sensed data sets for specific environmental monitoring problems, including monitoring of rehabilitated environments (Phinn et al 1999).

Remote sensing facilitates the comparison of whole areas of rehabilitation with native reference areas and reducing errors associated with under-sampling the variable post-mining environment. Importantly, remotely sensed data integrated with geographic information systems (GIS) and global positioning systems (GPS) introduce a spatial perspective. These data may be used to complement existing field point monitoring and allow extrapolation and comparison of rehabilitated and reference areas on site. This may also be extended to a local and regional context to provide a clearer picture of landscape function.

Remote sensing literature relating specifically to mine rehabilitation, however, is limited in Australia (eg Phinn & Hill 1992, Hick et al 1994, Warren & Hick 1996, Ong & Hick 1998, Jasper et al 1998). Recently, research emphasis has been placed on the application of airborne multi/hyperspectral data. Although most existing satellite systems provide the most cost-effective data, they were perceived to be too coarse, both spatially and spectrally, for monitoring mine rehabilitation (Warren & Hick 1996, Hick & Ong in press).

Separation of different vegetation types using remote sensing techniques requires measurements of the different spectral signatures characterising those types. Ground based spectral characterisation may be carried out during vegetation survey work and can provide a valuable database to facilitate the interpretation of new data sets collected over time.

Using airborne digital multi-spectral video (DMSV) data calibrated with ground spectral measurement of vegetation types at Weipa bauxite mine, Warren and Hick (1996) were able to map vegetation type and density. Analyses of the ground based spectral measurements demonstrated good separation between bare soil, litter cover and living vegetation. It also discriminated between the foliage of eucalypts, acacias, grevilleas and various grasses. Warren and Hick concluded that multi-spectral data could be used to characterise rehabilitated areas in relation to native reference sites and that maps distinguishing between major canopy species (acacias and eucalypts) could be produced. They noted, however, that the large amount of pre-classification computation required did not make the program feasible on an annual basis and DMSV data would most likely be acquired every 2 to 3 years. They also noted the fact that the trialed altitude and level of resolution could not produce useful information about plant density, cover and floristics in regeneration younger than three years.

In a recent study at Ranger mine, Ong and Hick (1998) also demonstrated that the different field spectral signatures of six tree species common to the woodlands surrounding the Ranger mine could be used to differentiate between eucalypts, melaleucas, acacias and grasses (see figure 3). They also found that airborne DMSV provided high precision monitoring of vegetation at a spatial resolution that shows individual trees (see plate 8, p24). A number of commercial airborne multispectral and hyperspectral imaging systems are available. Data from these systems can be calibrated to enable quantitative, ‘change over time’ comparisons to be made. However, aerial photography may be a more valuable tool if only qualitative comparisons are sought (Ong & Hick 1998).

A number of studies show much promise in the application of remotely sensed data to quantifying ecosystem biophysical parameters. Skidmore et al (1997) points out that hyperspectral data have been shown to have been successfully related to a number of soil parameters, such as soil moisture, organic matter, iron oxide and particle size. Mapping material directly is usually confounded by the complexity of various environments. This would suggest that in a simplified environment (eg minesite with sparse vegetation), with further work on the spectral properties of different material, these types of data would be extremely useful in the monitoring of erosion and other indicators of ecosystem stability/function.

Airborne radiometrics is another remote sensing method that has demonstrated potential in its implementation at Nabarlek. Significantly, uranium is present in most mineralisation throughout the Pine Creek Geosyncline. Thus, airborne radiometrics has potential as an indicator of erosion from rehabilitation areas in the ARR (B Noller, pers comm, NRCET).

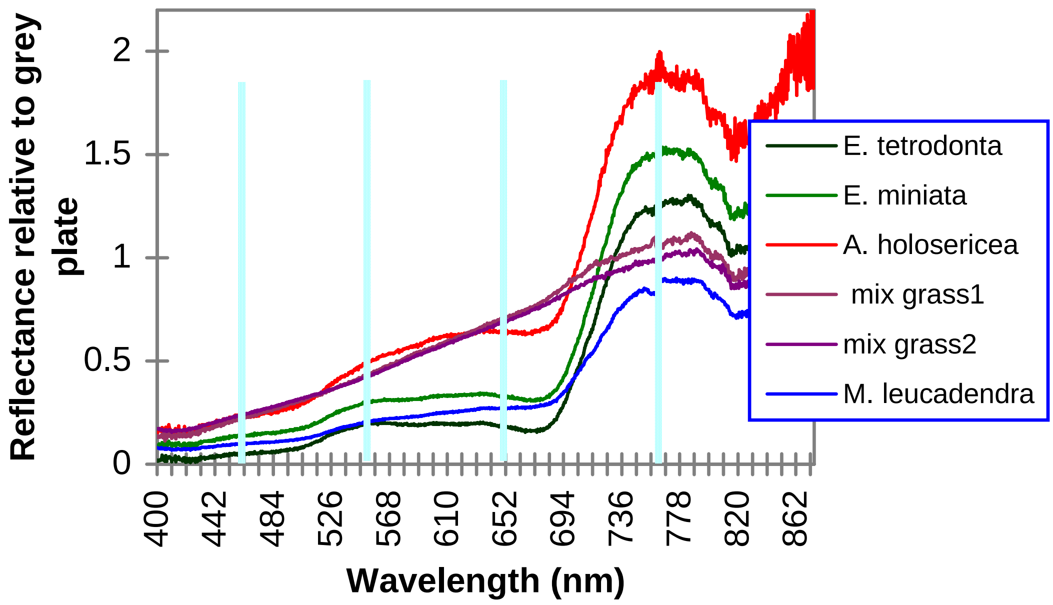

Figure 3 Visible–Near Infrared spectra of some of the common vegetation found in tropical woodlands measured with an Ocean Optics field spectrometer (from Ong & Hick 1998)

The limited number of studies examining the ability of remotely sensed data to quantify ecosystem recovery post-mining reflects the infancy of the technology. However, to date its development has been hampered by:

- limited understanding of the spectral characteristics of various components of vegetation communities and their indicator value of performance (Jasper et al 1998):
- the number of processing steps required prior to the extraction of useful information (Warren & Hick 1996).

Remote sensing represents a powerful tool that, if a number of types and scales of data are integrated, has the potential to provide a cost-effective and accurate method of monitoring over time. Modern hyperspectral sensors (such as HyMap™ and the compact airborne spectrographic imager CASI) with high spatial and spectral resolutions, while as yet untrialled on rehabilitation sites, have been recognised for their potential and are currently the subject of a significant amount of research. Measurable components such as: species associations, horizontal and vertical structure (succession), foliar chemistry (vegetation health), cellulose (litter and senescence), organic matter (soil disturbance, microbial activity) and soil chemistry (sulfitic/dispersive) allow for many ecological variables and indicators to be derived from these data (Hick & Ong, in press).

A methodology incorporating different types of remotely sensed data is the ideal approach (Stanford, pers comm, Biophysical Remote Sensing Group, University of Queensland). This includes data measured in the optical range encompassing hyperspectral (including handheld spectrometry), multispectral, airborne and satellite; aerial photography and also active systems such as laser and synthetic aperture radar (SAR).

Research should aim to streamline the processing of remotely sensed data to facilitate quantitative assessments of indicators of ecosystem recovery and stability. The adaptation and application of current remote sensing technology (often adopted from mineral exploration) to mine rehabilitation monitoring, should be a priority, particularly integration of remotely sensed data sets with field-based sampling.

5.4 Faunal recolonisation

Traditionally, rehabilitation and monitoring approaches have focused on vegetation communities, on the basis that they form the primary component of faunal habitat. Palmer et al (1997), however, questioned whether restoration of habitat is sufficient to reestablish species and function and assert that many untested assumptions are made concerning the relationship between habitat structure and restoration ecology. Conversely, the presence of a particular faunal species or assemblage may indicate that the ecosystem, or a component within that system, is functioning as it should. This assertion is the basis of a considerable amount of minesite research in the WDT (Reeders 1985, Majer 1990, Corbett 1997, Andersen et al 1998) and formed the focus of a recent conference devoted to faunal habitat reconstruction after mining (Asher & Bell 1997)

The use of invertebrates as bioindicators is well accepted in the study of aquatic ecosystems and has been the subject of considerable research on mine sites. Majer and Brown (1997) detail their value as bioindicators based on the involvement that invertebrates have on various ecosystem functions. Invertebrates perform functional roles in pedogenesis, organic matter decomposition, mycorrhizal activity, herbivory, pollination and propagule dispersal and thus are considered essential to the progress of rehabilitation.

Andersen et al (1998) described the degree to which ants provide an indication of the status of mine rehabilitation at Ranger. Andersen (1990) nominated ants as logical indicators because they:

- are ubiquitous and abundant
- interact with a wide range of ecosystem attributes and are functionally important at all trophic levels
- are readily sampled and processed
- are highly sensitive to environmental variation
- exhibit rapid response to environmental change.

Andersen (1990) contended that classification of species into functional groups, whose relative abundances vary predictably with climate, vegetation type and level of disturbance, facilitates data collection and interpretation. He found, for example, that changes in the relative abundance of a group of opportunistic ant species correlated with level of disturbance. Andersen et al (1998) also found a high correlation between ant and plant species richness across 39 sites at Ranger representing unmined areas and various stages of rehabilitation. In an assessment of indicators of ecosystem recovery at Gove, Reddell et al (1993) found termite and earthworm activity increased in rehabilitation areas with time.

Corbett (1997) suggests that the use of single indicator species is inappropriate due to large spatial and temporal variations in species richness and abundance. The ‘whole-ecosystem approach’ compares disturbed areas and reference areas for number and composition of trophic groups.[9] The approach compares the similarity between sites of the number of trophic groups present and species richness within them. Varying levels of similarity allow insights

9 Trophic groups are defined by life form and/or function. They comprise the food chains of ecosystems and form the basic unit of the whole-ecosystem approach to monitoring.

into ecological processes. For example, similar number and types of trophic groups with disparity between the species richness within them, may indicate a rehabilitated system is developing toward an analogue community. Absence of entire trophic groups would indicate a poorly developed system.

5.5 Other indicators of ecosystem recovery

In an attempt to define indicators of ecosystem recovery at Gove mine, Reddell et al (1993) examined a wide range of ecosystem attributes encompassing: floristic and vegetation attributes, and soil biological, chemical, physical, hydrological and structural characteristics. They examined a wide range of rehabilitation age classes (1 to 16 years since rehabilitation), as well as native undisturbed areas, and drew the following conclusions with respect to useful indicators of ecosystem recovery:

- Soil microbial activity increased with age of rehabilitation. Older (>10 years) rehabilitated sites displayed similar rates to those that occur in native sites.
- Fruiting bodies of macrofungi reflect changes in substrate and development of nutrient cycling processes and was strongly indicative of the stage of development of the plant communities and their disturbance history.
- Colonisation of roots by mycorrhizal fungi increased with age of rehabilitation and progressed toward the patterns found in native sites. This trend was most apparent with ectomycorrhizae which reflect the increasing dominance of woody species.
- Concentration of total carbon (representing organic matter) increased systematically with age and approached values reflecting those of native forest soil.
- The production of seed by species in rehabilitated areas, 10 years and older, is an important indicator that these communities can be self-sustaining.
- Substantial litter accumulation occurred between 5 and 10 years after rehabilitation.

These conclusions indicate that nutrient cycling processes may provide a useful indication of ecosystem recovery. This is supported by studies of post mining organic matter dynamics such as those at Weipa by Schwenke (1999) and Grigg et al (1999).

6 Recommendations for future research

6.1 Topsoil

Topsoil use on hard rock mines in the WDT is a contentious issue. The exclusion of indigenous microbes, valuable nutrients and the organic matter contained within the topsoil resource may reduce the probability of achieving a successful, self-sustaining native ecosystem. The availability of fresh topsoil in hard-rock open-cut mining and a reduction in quality resulting from poor handling and storage techniques pose limitations to its effective use.

To reinforce the recommendation by Tongway et al (1997):

> ***research should focus on the amount of topsoil that could confer positive benefits upon rehabilitation. There is also a need for research examining practical techniques that can adapt the broad principles of recent detailed research examining effective topsoil handling on bauxite mines (ie accurate stripping and replacement) to hard rock mines. Such an adaptation should include the development of effective collection, handling***

and storage techniques that maximise the retention of the desirable attributes of fresh topsoil and preclude the negative effects associated with topsoil use on hard-rock mines.

6.2 Fire

The existing research examining the role of fire in tropical systems focuses on mature systems rather than young rehabilitated areas. In their assessment of ecosystem recovery at Gove, Reddell et al (1993) describe fire as 'the most critical unresolved factor that could have a major impact on the development of the vegetation communities in the rehabilitated areas'**.**

Investigation into the role of fire in the establishment and development of revegetation should be a priority. Research emphasis should be placed on determining the time required for the development of fire resistance in the various woody components of rehabilitated areas. The use of fire as a management tool, in particularly the frequency and timing of burning regimes which could reduce the risk of high intensity fire in younger rehabilitation and which could direct species composition/successional development of older rehabilitation, also needs evaluation.

While fire exclusion will allow the development of a compositionally and structurally more complex community, such a regime is a gamble against the increased risk of a major setback to the redeveloping system. In this context, burning regimes on mine sites should be developed in the context of surrounding land-use and management and eventual lease relinquishment.

6.3 Succession

There is not much literature on the long-term successional development of rehabilitated areas in WDT. Tongway et al (1997) found mines that attempted to establish the desired final species mix early on were conspicuously more successful than those that relied on some hopeful expectation of favourable succession. Often, there is a discrepancy between the time scale of revegetation programs and lease relinquishment of many mines and the time required to effect succession.

Research is required to produce a body of organised, reliable theory and practice as an industry reference on the selection, germination and establishment of a composite of species that will increase the likelihood of succession toward a target community. Future research should also examine succession in a rehabilitation context, particularly developing methods to accurately predict the successional development of young revegetated communities. Such research should examine the dynamics of acacias and eucalypts in rehabilitation.

6.4 Symbiotic microorganisms

Establishment and nutrient acquisition strategies of species used in revegetation at Ranger are likely to be highly dependent on mycorrhizal fungi, which have been shown to be absent or poorly represented on Ranger WRD. Indications are that the natural dispersal and re-establishment of ectomycorrhizal fungi on Ranger WRD occur at a very slow rate and that the rate of development and the resilience of the plant community may be affected. Similarly, inoculation with appropriate strains of rhizobia may be necessary to establish some legumes on Ranger WRD.

Research should focus on identifying the factors affecting the establishment of viable mycorrhizae and Rhizobium populations on WRD soils, particularly where no respreading of topsoil or inoculation takes place.

6.5 Cover crops

Non-local species have the capacity to become invasive and out-compete native species for water, nutrients and light. The introduction of non-native species is also in conflict with the rehabilitation goals at Ranger mine.

Research should therefore continue to examine the use of native grasses for erosion control, particularly the use of **Ectrosia leporina** ***and*** **Eriachne schultziana*****, which have been shown to be particularly suitable for erosion control at Ranger. Emphasis should be placed on understanding their ecological and germination characteristics and developing methods to collect and store seed.***

6.6 Native seed collection, storage and germination

Methods for collecting, handling and germinating seed for native species to be used in revegetation in the WDT appear to be ad hoc rather than scientific and consistent.

The development of reliable information and the synthesis of existing information relating to the collection and handling and germination of native seed for the WDT into a body of reliable information would be a valuable exercise. Further research is required in regard to the germination requirements of many herbaceous and woody species in the WDT. Research should also examine the specific limitations to germination and establishment of seed in Ranger WRD soils.

6.7 Monitoring and success criteria

Ecosystem Function Analysis (EFA) is being proposed and gaining acceptance as an industry standard in the region.

It would be a timely and valuable exercise to subject the method to some vigorous validation against a suite of quantitative approaches that encompass the functions examined by EFA, to assess its accuracy and efficiency as an indicator of rehabilitation success.

Success criteria based on a single or narrow set of parameters are likely to be inadequate.

Therefore, a study comparing the indicator value of the various monitoring methods reviewed in this paper would also be valuable, with a possible outcome of the development of a 'multi-discipline' monitoring approach.

The potential for the application of remote sensing, particularly of airborne multi/hyperspectral data, to large-scale mine rehabilitation monitoring is yet to be realised.

Research should aim to streamline the processing of remotely sensed data to facilitate quantitative assessments of indicators of ecosystem recovery and stability. The adaptation and application of current remote sensing technology (often adopted from mineral exploration) to mine rehabilitation monitoring, should be a priority. A focus should be on the integration of remotely sensed data sets with field-based sampling.

6.8 Technology transfer

The vast majority of written information available to this review was in the form of unpublished reports written by or for mining companies. The dearth of peer-reviewed published papers (about 25% of material reviewed) is conspicuous. This is a significant problem which may limit effective communication and application of appropriate revegetation techniques in the WDT.

> ***Rehabilitation practitioners, research organisations, consultants and regulatory authorities should therefore be encouraged to publish material in relevant, peer-reviewed journals to facilitate the transfer of effective methodologies both within the mining and the wider resource management communities.***

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