Mine landform design using natural analogues

Ian Douglas Hollingsworth

Dip.App.Sci.Agr-Hawkesbury Agricultural College

Dip.Agr.Sci-University of New England

M.Rur.Sci.-University of New England

A thesis submitted in fulfilment of the requirements

for the degree of Doctor of Philosophy

Faculty of Agriculture, Food and Natural Resources

The University of Sydney

New South Wales

Australia

MMX

CERTIFICATE OF ORIGINALITY

The text of this thesis contains no material which has been accepted as part of the requirements for any other degree or diploma at any University of any material previously published or written, unless due reference to the material has been made.

Ian Douglas Hollingsworth

ABSTRACT

Current practice for landscape reconstruction following opencast mining relies on topographic reconstruction, adaptive land management and botanical characterisation. Environmental processes may be altered where reconstructed landforms have significant relief. Consequently, environmental outcomes in cases where there is large scale land forming are unpredictable. Moreover, landscape restoration lacks an integrated methodology, and while many mine closures have detailed ecosystem and biodiversity objectives based on natural analogue areas there has been no reliable way to design these objectives into mine landforms. The methods used in landscape restorations to describe reference conditions are based on generalised environmental factors using regional information and incorporating conceptual models. Such models lack the precision and accuracy required to understand and restore hillslope environmental pattern at mine sites.

However, methodological integration and statistical inference models underpinning the spatial inference methods in conservation and landscape ecology, and pedology may be applied to solve this problem. These inference models utilise digital terrain models as the core environmental data incorporating ecological theory to predict biodiversity and species distribution. Also, numerical mass balance models such as water and solute balance, which have been applied to understand environmental processes in landscapes, can be used to assess mine landform design. The objective of the work reported here was to investigate environmental variation, with sufficient accuracy and precision, in natural landscapes to design mature mine landforms and to demonstrate the capacity to predict ecological outcomes. This would extend current best practice - designing mine landforms with predictable hydrological and geotechnical outcomes needed to protect off-site environmental conditions – to the on-site environment after closure.

The specific aims of this thesis were to: (i) evaluate the predictability of ecosystems based on regional ecological mapping: (ii) develop and evaluate quantitative, site specific environmental mapping and natural analogue selection methodology; (iii) evaluate a trial final landform cover (reconstructed soil) using water balance, water chemistry monitoring; (iv) design and evaluate a conceptual mine landform through the assessment of environmental processes in natural analogue areas; and (v) make valid predictions of revegetation outcomes on the conceptual landform. In meeting these aims, links between ecological theory, landscape analysis and the current practice in mine landform design were identified.

The first phase of the thesis involved environmental investigations and surveys of extensive savanna environments on the Tiwi Islands (7320 km^{-2}) and similar environments in the vicinity of Ranger uranium mine (150 km^{-2}) in northern Australia. This first phase, reported in Chapter 3, investigated the reliability of conceptual landscape models used in regional ecological mapping in predicting ecological patterns in terms of vegetation and soil. The Tiwi Islands was selected

because of the relatively uniform parent material and its simplified climate. This allowed the study of physiographic control of soil and vegetation patterns. The results identified correlations between vegetation pattern and landform that were confounded by a subjective and complex land unit model of ecosystems. This investigation enabled the development methodological approach to analogue selection and ecological modelling at Ranger uranium mine – a site that will require a restoration approach so as to meet environmental closure objectives.

The second phase is the methodological development – involving an initial reconnaissance, is presented in Chapter 4. This phase was aimed at selecting natural analogue areas for mined land restoration. Environmental pattern recognition involving classification, ordination and network analysis was implemented based on methods of conservation ecology. This led to quantitative landscape model to identify natural analogue areas and design ecosystem surveys. This quantitative landscape model incorporated a grid survey of vegetation and soil variation into a nearby analogue landform that matched the area of mine disturbance. This analogue landform encapsulates the entire ecosystem types observed on rocky substrates in the broader reconnaissance survey. The natural analogue selection incorporated a combination of digital terrain analysis and k-means clustering of primary and secondary terrain variables to classify habitat variation on hillslopes. Landscapes with similar extent to the mine landscape were identified from numerical similarity measures (Bray-Curtis) of fine grained habitat variation and summarised using a dendrogram. The range in hillslope ecosystem types were described from stratified environmental surveys of vegetation and soils along environmental gradients in selected analogue landforms.

The results show that the mapped environmental factors in close correlation with water and sediment distribution were strongly associated with observed vegetation patterns in analogue areas at Ranger uranium mine. Environmental grain size and landform extent concepts were therefore introduced using landscape ecology theory to integrate different scales of environmental variation in a way that provides direct context with the area impacted by mining. Fine-grained environmental terrain attributes that describe runoff, erosion and sediment deposition were derived from a digital elevation model and classified using non-hierarchical multivariate methods to create a habitat class map. Patch analysis was used to aggregate this fine-grained environmental pattern into a grid that matched the scale of the mine landform. The objective was to identify landforms that were similar in extent to the reconstructed mine landscape. Ecosystem support depends on soil as well as geomorphic factors.

An investigation into critical environmental processes, water balance and solute balance, on a waste rock landform at Ranger uranium mine is presented in Chapter 5 to characterise waste rock soils and investigate cover design options that affect environmental support. This involved monitoring of water balance of a reconstructed soil cover on a waste rock landform for four years and the solute loads for two years. A one dimensional water balance model was parameterised and run based on 21 years of rainfall records so as to assess the long-term effects of varying cover thickness and surface compactness on cover performance. The results show that the quality of runoff and seepage water did not improve substantially after two years as large amount of dissolved metal loads persisted. Also, tree roots interacted with the subsoil drainage-limiting layer at one metre below the land surface in just over two years - and thus altering the hydraulic properties of the layer. Further, the results of water balance simulations indicate that increasing the depth to, and thickness of, the drainage-limiting layer would reduce drainage flux. Increasing layer thickness could also limit tree root penetration. It was also found that surface compaction was the most effective means of limiting deep drainage, which contained high concentrations of dissolved metals. However, surface compaction creates an ecological desert. Therefore long-term rehabilitation of the cover will be required to allow water to infiltrate for it to be available for ecosystems. A cover that can store and release sufficient water to support native savanna eucalypt woodland may need to be three to five metres deep, including a drainage limiting layer at depth so as to slow vertical water movement and comprise a well graded mix of hard rock and weathered rock to provide water storage and erosion resistance. The resulting waste rock soils would be similar, morphologically to the gradational, gravelly soils found in natural analogue areas.

The study then shifted from mined land back to a selected natural analogue landscape at Ranger mine in Chapter 6. The fine grained variation in terrain attributes is described to support a landform design that allowed for mine plan estimates of waste rock volumes and pit void volumes. A process of developing and evaluating the landform design was put forward, in the case of Ranger, that begins with key stakeholder consultation, followed by an independent scientific validation using published landform evolution and integrated, surface-groundwater water balance modelling. The natural analogue and draft final landforms were compared in terms of terrain attributes, landform evolution and eco-hydrological processes to identify where improvements could be required. The results of the independent design reviews are contained in confidential reports to Ranger mine and in conference proceedings that are referenced in Chapter 6. Independent validation will be a key element of an ecological landform design process and the application of published eco-hydrological and landform evolution models at the Ranger mine case study site are presented as an example of current best practice. Also, detailed assessment was made of environmental variation and soil and geomorphic range in the selected analogue landscape to support the landform design process with the mining department.

Ecological modelling of the distributions of framework species in the reconstructed landscape is proposed as an additional assessment tool in this thesis to validate an ecological landform design methodology. To this end, a detailed environmental survey is presented in Chapter 6 of the soils and vegetation in a selected natural analogue area of Ranger mine to identify common and abundant plant species and their distribution in a similar landscape context to the mined land. This work supported ecological modelling of species distributions in reconstructed and natural landscapes in the following chapter.

The results of species distribution models for reconstructed and natural landscapes at the Ranger mine site are reported in Chapter 7. The aim was to predict the distribution of common and abundant native woodland species across a landscape comprising a sculpted, post mining landform within a natural landscape. Species distribution models were developed from observations of species presence-absence at 102 sites in the grid survey of the natural analogue area that was reported in Chapter 6. Issues related to optimising predictor selection and the range of environmental support were investigated by introducing survey sites from the broad area reconnaissance survey reported in Chapter 4. Added to these are the published species abundance data from an independent regional biodiversity survey of rocky, well drained eucalypt woodlands, used as analogues of mined land. Plant species responses to continuous and discrete measures of environmental variation were then analysed using multivariate detrended correspondence analysis and canonical correspondence analysis to select independent variables and assess the relative merits of abundance versus presence absence observations of species. Then, generalised additive statistical methods were used to predict species distributions from primary and secondary terrain variables across the natural analogue area and a reconstructed post-mining landform. This analysis was completed with an assessment of the effect that survey support has on model formulation and accuracy. The scale of the mine landscape was found to provide important context for the stratified environmental surveys needed to support predictive modelling. Extending the geographic range of survey support did not improve model performance, while survey sites remote from the mine introduced some degree of spatial autocorrelation that could reduce the prediction accuracy of species distributions in the mine landscape. Further work is needed to address uncommon species or species with highly constrained environmental ranges and aspects of landform cover design and land management that affect woodland type and vigour.

The combined studies reported in this thesis show that the predictability of mine land restorations is dependent on the landscape models used to characterise the natural analogue areas. It is demonstrated that conceptual ecological models developed for regional land resources survey, commonly used to select natural analogue areas, are subjective, complex and unreliable predictors of vegetation and soil patterns in hillslope environments at particular sites. It was recognised that environmental patterns are subject to terrain and hillslope environmental variation across an extensive areas. The landform model for selecting natural analogues was refined by introducing grain size and ecological extent concepts, used to describe ecological scale in landscape ecology, to address these effects. These refined concepts were adapted to define environmental variation in the context of natural analogue selection for mining restoration, rather than home range habitat conditions for native animals as was their original purpose. It is demonstrated here that the grain size and extent of environmental variation in the natural landscape can be used to select natural analogue landforms, develop ecological design criteria and design field surveys that support the capacity to predict the distributions of common and abundant woodland species in a reconstructed landscape.

In conclusion, it is worth noting that an integrated ecological approach to landscape design can be applied to closure planning at mine sites where cultural and ecological objectives are critical to the success of the mine rehabilitation. Furthermore final landform trials could be used to support a restoration approach — providing an understanding of the interactions between critical physical and ecological processes in the soil layers and environmental processes at catchment scales. The accuracy of the inferences made is dependent on the understanding of hydrological processes in natural and constructed landforms. However, the natural analogue approach provides a clear landscape context for these trials. In a world where species extinction resulting from habitat loss is one of the most important global ecological issues, mine rehabilitation offers unique experimental opportunities to develop capability in ecosystem rehabilitation.

ACKNOWLEDGMENTS

I need to thank Associate Professor Inakwu Odeh for his supervision and advice and Drs John Ludwig and Elisabeth Bui for their suggestions and constructive supervision, which has greatly improved the quality of this work. I also express my gratitude to my wonderful wife Amanda and family for their tolerance and support and my parents Peter and Margaret Hollingsworth for their encouragement. I acknowledge EWL Sciences and ERA Ranger uranium mine, which supported the work financially. Dr Tony Milnes and Dr Laurie Corbett at EWL Sciences provided many valuable ecological insights.

The Tiwi Islands land unit survey presented in Chapter 3 was funded by the Northern Territory Government and supported in the field by Phil MacLeod, one of their senior land evaluation staff. The efforts of Kate Haddon at the Tiwi Land Council made this work possible. The ecosystem surveys presented in Chapter 4 and Chapter 6 could not have been done without the botanical skills of Anja Zimmermann and Ingrid Meek and encouraging support from the Gundjehmi Aboriginal Corporation, representing the traditional owners of the Ranger mine lease. The landscape evaluations presented in Chapter 6, namely landform evolution modelling and eco-hydrological modelling were performed independently by John Lowry at the Environmental Research Institute of the Supervising Scientist and James Croton from Water & Environmental Consultants. Their professionalism is gratefully acknowledged. I am also grateful to Geff Cramb, formerly with EWL Sciences, who worked on the field instrumentation, surveys and sampling and is a good friend.

Finally, I would like to thank my mother and my grandmother, who lived many years ago nearby the University of Sydney in Balmain.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF PLATES

PARTS OF THIS THESIS HAVE BEEN PUBLISHED OR SUBMITTED FOR PUBLICATION IN VARIOUS FORMS AS JOURNAL ARTICLES OR BOOK CHAPTERS

Chapter 3 – Published as a book chapter in

Hollingsworth, I., Odeh, I., Bui, E. and MacLeod, P., 2007. Rule Based Land Unit Mapping. In: P. Lagacherie, A.B. McBratney and M. Voltz (Editors), Digital Soil Mapping: an introductory perspective Chapter 29. Developments in Soil Science. Elsevier, Amsterdam, pp. 404-414.

Lines Composed a Few Miles above Tintern Abbey

Five years have past; five summers, with the length
Of five long winters! and again I hear
These waters, rolling from their mountain-springs
With a soft inland murmur. — Once again
Do I behold these steep and lofty cliffs,
That on a wild secluded scene impress
Thoughts of more deep seclusion; and connect
The landscape with the quiet of the sky.

William Wordsworth, July 13, 1798

Chapter 1 General introduction

1.1 Introduction and problem definition

Worldwide mining and quarrying move over 57 billion tonnes of rock and earth per year, a figure that rivals natural, geomorphic processes of earthmoving (Douglas and Lawson, 2000). Surface mining damages 2-11 times more land than with underground mining (Miao et al., 2000) leading to extensive areas of degraded landscape within and in the vicinity of the mine site. Typically a mine degraded landscape comprises: stripped areas (59%), open-pit mines (20%), tailings dams (13%), waste tips (5%) and land affected by mining subsidence (3%). The direct effects of mining activities can be an unsightly landscape, loss of cultivated land, forest and pasture land, and the overall loss of production. The indirect effects can be multiple, such as soil erosion, air and water pollution, toxicity, geo-environmental disasters, loss of biodiversity, and ultimately loss of economic wealth.

Historically mining and mineral processing have relied upon the capacity of the environment for dispersing and assimilating wastes. It is only since the latter part of the twentieth century that environmental problems associated with mining have been understood chiefly in terms of effects on the receiving environment outside of the mine site (Bridge, 2004). These effects have become more significant through a massive expansion of mineral production, driven by increased demands for raw materials and narrowing cost-price differentials (Mudd, 2007). These demands have led to exploitation of ever decreasing ore grades and larger scale opencast mining leading to ever increasing footprints on the landscape.

By 2004, in tropical areas with high levels of biodiversity, 75% of active mines and exploration areas overlapped with global areas of high conservation value (Figure 1-1) and areas of watershed stress (Warhurst, 2002). Nearly one third of all active mines and exploration sites were located within intact ecosystems of high conservation value (Miranda et al., 2004). This geographical shift in the location of mining since the late twentieth century has intensified long-standing concerns about the impact of mining on global biodiversity and critical ecosystems (Bridge, 2004).

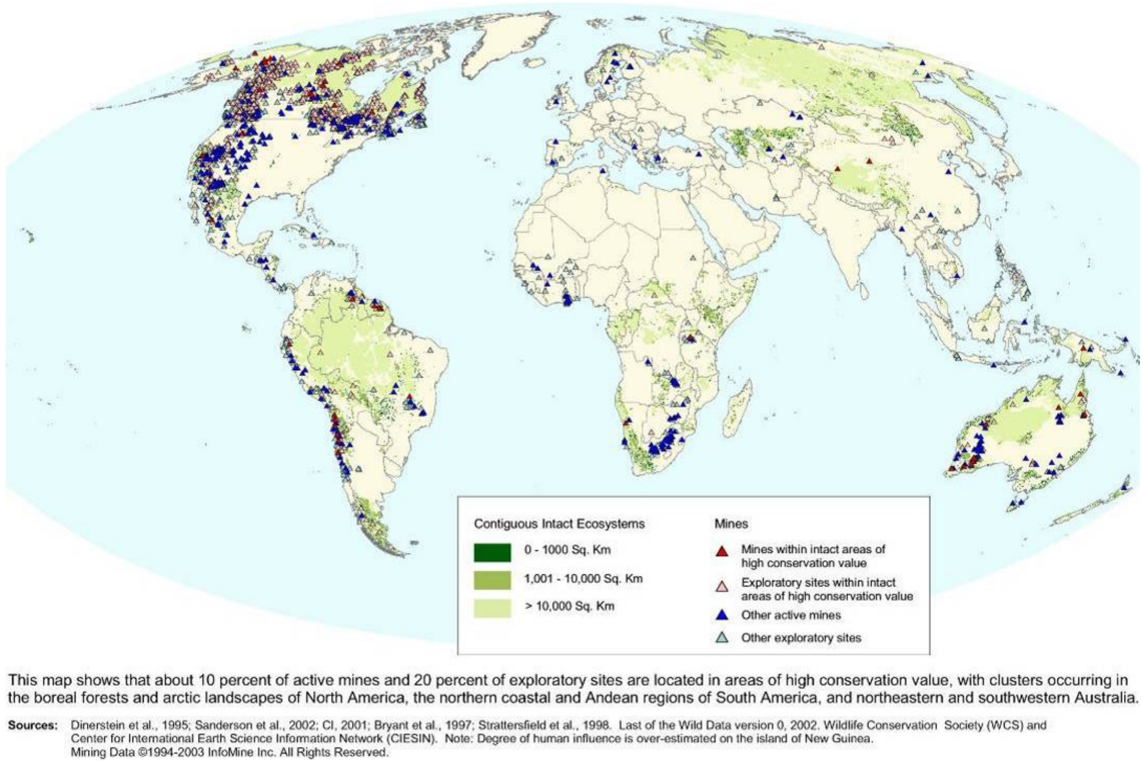


Figure 1-1 Active mining and exploration overlaid on areas of high conservation value

Because mines occupy a relatively small land area compared to other land uses like forestry or agriculture, the effects on the environment tend to be localized. Mining occupies considerably less than 1% of the world's terrestrial land surface. Estimates for the United States—a country with an extensive mining history—indicate that mineral extraction occupies only 0.25% of the land area (and only 0.025% for metal mining) in comparison with 3% for urban areas and 70% for agriculture (Hodges, 1995). In Australia, a principal raw materials supplier, coal, gold, bauxite, iron ore, base metal and mineral sand operations are spread across most of the nation's biogeographic zones. In spite of this wide distribution, the collective area disturbed by mining and mineral processing is less than 0.05% of the land surface and the mining sector makes the largest contribution to national exports (Bell, 2001).

Nriagu (1996) challenged this view of discrete, localized impacts by identifying that mining and minerals processing were the primary contributors of anthropogenic releases for most metals. Mining and minerals processing activities discharge pollutants into waterways and the atmosphere (Archer et al., 2005; Martinez-Sanchez et al., 2008; Moore and Luoma, 1990; Ripley et al., 1996). Nriagu (1996) concluded that industrial releases of heavy metals into our environment have overwhelmed the natural biogeochemical cycles of metals in many ecosystems. Consequently, direct regulation of contamination to the receiving environment in terms of water, soil contamination, radiological impact and loss of biodiversity has been the dominant approach to address environmental impacts associated with mining (Bridge, 2004).

However, the focus of environmental concern is shifting from off-site to on-site environmental impact. Mine restoration legislation in the USA provides for the reconstruction of the original topography (Toy and Chuse, 2005) and for restoring hydraulic and erosional stability to the mined landscapes. Instead of engineered slope designs for safe hydraulic management of water movement, this legislation prescribes that natural slope and catchment conformations are incorporated into the mined landscape. Requirements for landscape reconstruction and stabilisation after mining are increasing elsewhere — in South America (Griffith and Toy, 2001), Europe (Nicolau, 2003), Australia (Riley, 1995a) and China (Li, 2006). In Australia, international obligations, agreements and guidelines have guided the joint development of rehabilitation standards applied by the Commonwealth and States and Territories (Anon, 2006; ANZMEC, 2000). The approach to mine remediation is self–regulated by industry codes of conduct (Soloman et al., 2006). State and Territory legislation sets standards for closure planning, which increasingly refer to Commonwealth legislation (such as the Environment Protection and Biodiversity Conservation Act, 1999), and national water quality and contaminated land guidelines (ANZECC, 2002) to set acceptable levels of protection for the receiving environment once mining operations have ceased. Also, mine closure goals have shifted from restoring agricultural land capability to the introduction of indigenous plant species found in local native ecosystems (Grant and Koch, 2007).

The methodologies needed to provide acceptable outcomes have not been developed for native ecosystem restoration (Ehrenfeld and Toth, 1997). Where the closure objective is to restore analogous native ecosystems in the post-mining landscape, the link between the reconstructed topographies, analogue soil conditions and vegetation has not been addressed explicitly. As a consequence there is no assurance of ecological outcomes (Nicolau, 2003; Toy and Chuse, 2005). At the same time mining companies are becoming increasingly accountable for environmental costs after closure (Miller, 2006), due in part to declining community tolerance of poor outcomes (Soloman et al., 2006). Jurisdictions around the world (including Australia) have strengthened the requirement for financial assurance in recent years (Miller, 2006) and the mining industry has developed generic guidelines for mine closure planning and rehabilitation (Anon, 2008). The task is now to develop site specific methodology for restoration design at mine sites where the topographic is to be reconstructed.

Natural analogue or reference sites that represent relatively intact ecosystems are used to set objectives and develop strategies to restore land degraded by mining and other human activity (Palik et al., 2000; White and Walker, 1997). Using a landscape approach to select reference sites embraces spatial heterogeneity and identifies appropriate configurations of restored elements to facilitate recruitment of flora and fauna (Bell et al., 1997). However, methods for matching native plant species to land in ecosystem restoration tend to be based on general principles (Ehrenfeld, 2000; Holl et al., 2003) and the attention paid to local context in terms of geomorphology and edaphic factors at the landform design stage is rudimentary (Nicolau, 2003). While landform is an important factor driving biodiversity (Lawler and Edwards, 2002; Palik et al., 2000; Sklenicka and Lhota, 2002; Steiner and Kohler, 2003; Wardell-Johnson and Horwitz, 1996) and affecting the success of many restoration projects (Chapman and Underwood, 2000) there is no reliable methodology for ecological landscape design. Some assurance is needed that natural ecosystems, in context with the surrounding natural landscape, are being restored. Waste rock landforms designed for geotechnical stability and hydraulic performance according to engineering principles (Hancock, 2004) may lack context in natural landscapes. Also, ecosystems can be complex and demonstrating the same level of confidence in theoretical ecological models of landscape restoration as there is in hydraulic and erosion models is some way off (Nicolau, 2003).

In summary, government policy and community's views on post-mining land use have caused the minerals industry in Australia and overseas to shift closure objectives from agricultural land values to re-establishing native ecosystems. In this case landform re-construction is a critical first step in any restoration project. However, landform design and evaluation methods based on ecological as well as physical principles are required to provide assurance that key ecological values can be reinstated *ab initio* into the mine landforms. The challenge for landscape restoration following opencast mining is to provide both general guidance and more context sensitive landscape design methods that are relevant to particular ecosystem types found in specific situations.

1.2 Mining context

Typically, post-mining landforms comprise open pits and above grade waste rock landforms (Plate 1.1) that have been engineered (based on hydraulic design parameters) and constructed to a particular failure risk profile.

Plate 1-1 Rehabilitated waste stockpile and pit lakes at
Rum Jungle mine (Pidsley, 2002)

Where the objective is to restore natural topography and vegetation, pits are backfilled and waste rock stockpiles reshaped to resemble natural landscapes (Plate 1-2).

The cost of earth moving for the rehabilitation of opencast mining can be economically prohibitive. Earth moving operations for opencast mining involve much larger amounts of waste rock material per unit of ore compared to underground or strip mining techniques

(Figure 1-2). The amount, and distance that material needs to be moved, and the requirements to contain reactive and mineralised waste materials, are of primary consideration (Anon, 2008). Any additional requirements to restore ecological properties in the post mining landform need to be quantitative, cost effective, practical and able to be validated.

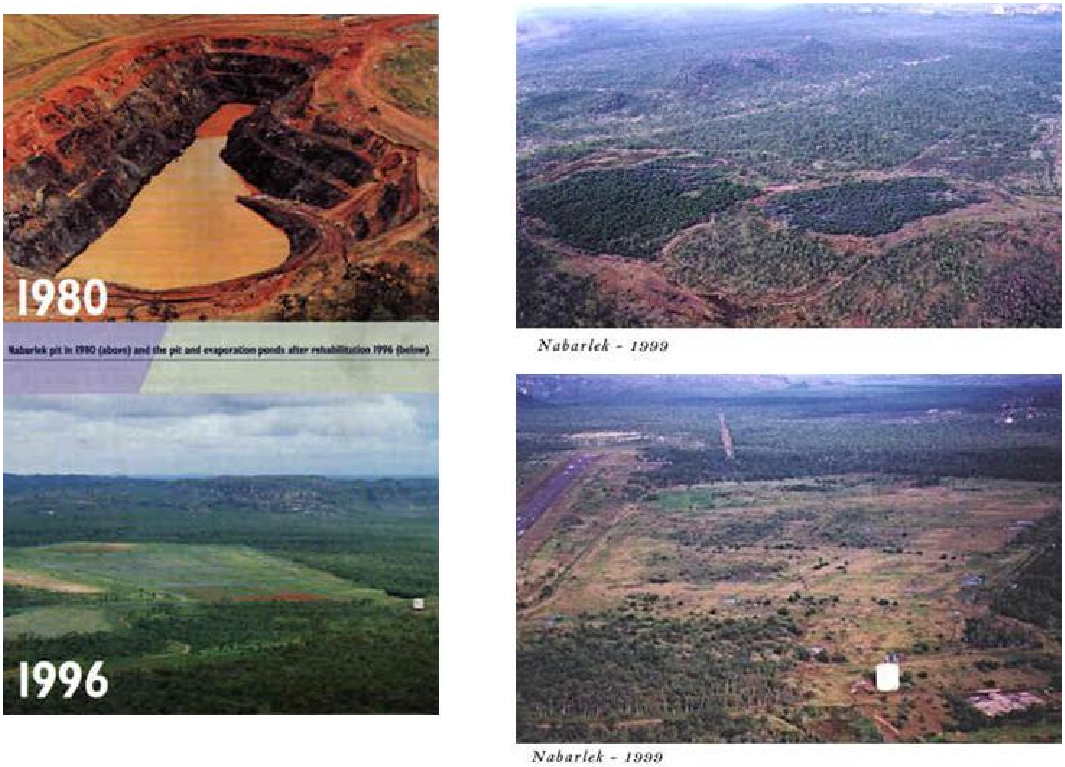


Plate 1-2 Images of Nabarlek mine before and after rehabilitation (Klessa, 2000)

Natural analogues have been used with limited success to design reconstructed natural landscapes following open cast mining (Ehrenfeld and Toth, 1997; Holl et al., 2003; Klessa, 2000; Nicolau, 2003). Deterministic modelling of ecosystem development as a function of topography, soils and management factors is an alternative to using natural analogues but its application is some way off (Austin, 2007). However, natural analogue selection in ecosystem restoration is a poorly defined process and failures may be attributed to inaccurate specification that may be addressed by realigning it with ecological theory (Ehrenfeld and Toth, 1997; Jim, 2001).

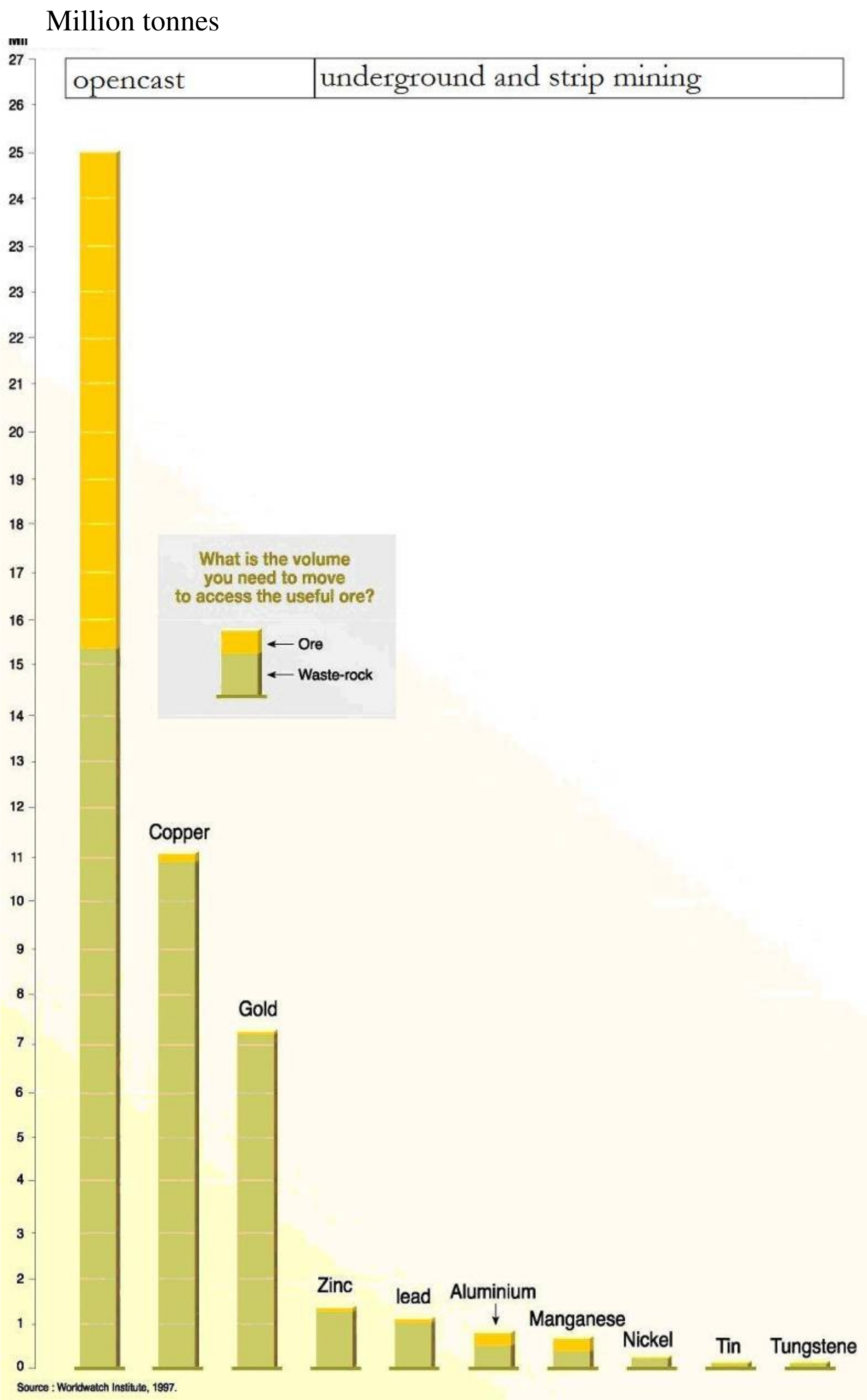


Figure 1-2 Industry waste rock and ore tonnages[1]

---

[1] Cartographer: Philippe Rekacewicz, Mining waste rock. (2004). In UNEP/GRID-Arendal Maps and Graphics Library. Retrieved 06:19, September 24, 2006 from http://maps.grida.no/go/graphic/mining_waste_rock.

1.3 General hypothesis

The broad objective of this thesis was to test the hypothesis that natural analogue landscapes can be used to develop practical ecological design and evaluation methodologies for restoring landscapes constructed from waste rock following opencast mining. The predictability of key species across mine and natural landscapes is used to test the hypothesis on two study areas in savanna woodland environments, namely the Tiwi Islands and Ranger uranium mine in the Top End of Northern Australia.

It was assumed that suitably stable analogue landforms exist and are amenable to developing methodologies based on quantitative techniques in landscape and conservation ecology. The method developed here is limited to extending topographic design for opencast mine landforms based on natural analogue landscapes (Toy and Chuse, 2005) to include explicit ecological design parameters and modelling methods. Ultimately, ecological outcomes may be predictable from deterministic modelling. While there has been recent development in modelling soil formation in hillslopes (Minasny et al., 2008), the capacity to reconstruct soil profiles to satisfactorily recreate natural edaphic conditions and to manage revegetation to achieve reliable outcomes is largely implied at this stage of simulation modelling.

Hobbs & McIntyre (2005) recommended using regional environmental and thematic mapping to select suitable analogue or reference sites for ecosystem reconstruction following mining. The regional environmental mapping of the Tiwi Islands was used to test the predictability of natural environmental variation as a function of a land unit classification and terrain. A broad range of low relief hill slope landforms (in context with mined landscapes) occur on the Tiwi Islands, characterised by low gradients in other environmental factors such as parent material and climate that affect species distribution.

The choice of the Ranger uranium mine to develop and evaluate ecological landform design methods was due to its proximity to a world heritage area (Kakadu National Park) and because its restoration has spurred developments in landform evaluation using landform evolution modelling (Willgoose and Riley, 1998).

1.3.1 *Aims*

The specific aims of this thesis were:

i. to evaluate the predictability of ecosystems as a function of regional ecosystem mapping based on land units, an information format commonly used to select analogue or reference sites for planning mine rehabilitation

ii. to develop and evaluate a quantitative environmental mapping and natural analogue selection method to support a landscape restoration approach using Ranger uranium mine as a case study site

iii. to evaluate a trial final landform cover (reconstructed soil) from water balance, water chemistry monitoring and water balance simulations to assess soil reconstruction options

iv. to propose an ecological landform design and evaluation process that is based on a consultative approach and an understanding of critical environmental processes in natural analogue areas

v. to make valid predictions of revegetation outcomes from a landform design.

This thesis constitutes several chapters. An overview of the literature on ecological methods and their application to restoring opencast mine sites is provided in Chapter 2. This presents the impetus to develop and implement the research needed to address the aims listed above.

The variation in soil and vegetation community related to landforms on the Tiwi Islands (7 320 km^2) is investigated in Chapter 3. The relatively uniform, climate and geology across the Tiwi Islands (Nott, 1994b) suggests that underlying environmental effects of landform variation on vegetation pattern are easier to understand. This chapter also describes how ecological models based on land units can be assessed using quantitative landscape analysis. From this analysis the limitations of using the land unit model for representing ecosystem variation are identified and an alternative quantitative approach, using environmental grain size and extent concepts of ecological scale from landscape ecology are introduced to target analogue landforms for guiding open cast mine restoration.

In Chapter 4, the first analogue stage of ecological design that involves analogue selection and ecosystem characterisation is presented. Terrain analysis and multivariate methods that were developed in conservation and landscape ecology were applied to select natural analogue landscapes and analyse environmental variation in a case study at the Ranger uranium mine. This analysis identified key soil and landscape variables affecting plant community variation and recommended where more detailed environmental surveys of selected analogue landforms is needed to support predictive modelling of species distribution. However, the level of environmental support in the near surface of waste rock landscape also needs to be understood to understand what the critical soil reconstruction issues are in the restoration design.

In Chapter 5, a mass balance approach is used to evaluate the environmental performance of a trial waste rock landform that incorporated a layered cover (soil zone) intended to isolate radioactive waste rock and support vegetation at Ranger mine. A one dimensional water balance model is used to assess the effect of varying layer thicknesses in the cover construction on deep drainage. The analysis gives a practical insight into the design and construction of a mine cover and the implications of cover design, or soil reconstruction, for ecosystem reconstruction and drainage water quality controls that affect off-site environmental impact.

The investigation moves back to the natural landscape in Chapter 6. A conceptual landform design at Ranger uranium mine is developed from the fine-grained terrain variation of terrain attributes in a selected natural analogue area (from Chapter 4) and evaluated with reference to the properties of a natural analogue landform. The evaluation incorporates:

- comparative geomorphic analysis with a natural analogue area
- independent one– and three–dimensional water balance assessment for ecosystem support and catchment water balance
- independent landform evolution modelling
- compliance with the expectations of the Mirrar Traditional Owners of the Ranger lease.

A second stage of ecological design (following on from Chapter 4) involving detailed soil and vegetation survey of a selected natural analogue area is presented in Chapter 6. The level of support required to develop reliable statistical models of the distributions of common and abundant framework species is presented. Application of species distribution modelling to open-cast mine landform design is the main methodological development in this thesis. The objective is to improve the reliability of ecosystem reconstruction by introducing an account of ecosystem outcomes at the mine landform design stage, in the same sense that erosion, slope stability and hydraulic performance are currently evaluated in the environmental design.

In Chapter 7, quantitative ecological modelling methods are applied to predict species distribution using survey support for the Ranger case study site. Survey design and ecological modelling parameters were assessed with the aim of adapting established methods from conservation ecology to ecosystem restoration at a mine site. The validation of species distribution models presented in this chapter is the test of the ecological landform design method. This assumes that other factors such as soil reconstruction and revegetation management can be addressed.

A discussion about ecosystem restoration of opencast mine sites based on natural analogues is provided in Chapter 8. It concludes by pointing out the inevitability of a shift in mine rehabilitation paradigm away from revegetation of landforms designed from environmental engineering principles to ecologically engineered landforms that support local context in biodiversity and cultural amenity. This subtle shift requires the development of quantitative ecological models with acceptable accuracy and reliability and comparative analysis of analogue and mine landscapes to validate landform design variables.

Chapter 2 Literature review – landscape design for ecosystem reconstruction, a synthesis

Restore vb. (tr.) 5. to reconstruct (an extinct animal, former landscape, etc.)

(Collins English Dictionary)

2.1 Overview

Landscape reconstruction is focussed on restoring physical landscape properties such as natural basin morphology (Toy and Chuse, 2005) and natural relief profiles (Hancock et al., 2003) using topographic design principles and three dimensional models developed from the natural landscape surrounding a mine (Hancock, 2004; Nicolau, 2003). Topographic design principles are aimed at restoring natural levels of erosion and sedimentation, ensuring effective containment of waste and managing off-site effects (East et al., 1994; Evans and Willgoose, 2000; Hancock et al., 2006). Revegetation planning follows topographic construction. However, links between the geomorphic, soil and vegetation factors that determine ecosystem pattern in the landscape are not made.

Currently, ecological outcomes from topographic reconstruction are unreliable for those situations where there is a high level of landscape disruption (Toy and Chuse, 2005). To extend topographic design to reconstructing particular ecosystems requires the inclusion of ecological theory. The outcome to predict the spatial distribution of ecosystems to a similar capacity as is evident in erosion and deposition modelling methods, and the ability to assess whether ecological outcomes are consistent with defined goals need to be demonstrated.

Nicolau (2003) considered the critical ecological issue to be integrating geomorphology with the soil and vegetation, so as to describe the formation of functional, self-sustaining ecosystems. This conclusion is repeated elsewhere (Brown, 2005; Duque et al., 1998; Ehrenfeld and Toth, 1997; Holl et al., 2003; Jim, 2001; Wang et al., 2001) and is considered to be a major factor limiting the success of rehabilitation programs in open cast mining (Johnson and Miyanishi, 2008).

There are some methodological and theoretical hurdles to this integrated approach to landscape restoration. Holl (2003) opined that landscape level restoration lacked a methodology. Other authors considered that landscape restoration works lacked a basis in ecology (Lipsey and Child, 2007; Temperton, 2007; van Diggelen et al., 2001). However, unless substrate conditions are extreme, ecosystems on mine sites function similarly (over time) to comparable ecosystems on adjacent unmined analogue sites (Huttl and Bradshaw, 2001).

Consequently, analogous natural landscapes have been used to guide ecosystem restoration strategies for mine landforms — by providing some capacity to identify long term outcomes and accelerate natural remediation processes (Bradshaw and Huttl, 2001; Wang et al., 2001). Although, inaccurate representation of water, nutrient, erosion and sediment distribution processes in mine and natural landscapes has lead to poorly defined or unrealistic ecological goals (Bell et al., 1997; Choi, 2007; Ehrenfeld, 2000; Ehrenfeld and Toth, 1997). The main focus has been on managing the revegetation process (Hobbs, 2007; Holl and Crone, 2004) rather than designing landforms to support biodiversity. Addressing this gap would conceivably improve mine rehabilitation from an ecosystem perspective and have broader application to restoring landscape degradation more generally.

The projected climate trends for the 21st century, which point to the prospects of future climate states that may produce different analogues to the present one and the disappearance of some extant climates (Williams et al., 2007) could compromise an analogue approach. It is likely that future 21st-century climates may promote formation of new species associations with concomitant increased risk of extinction for species with narrow geographic or climatic distributions resulting in disruption of existing communities. Consequently, reliable definitions of natural analogues may only apply to extensive habitat types and associations with relatively common species in many instances.

This review aims to provide some discourse on divergent ecological perspectives and explore an ecological design methodology based on natural analogues. Analogue selection is critical to this process and to developing reasonable design criteria. Methods to validate ecological designs are also reviewed. The themes covered in this review are outlined in Figure 2-1.

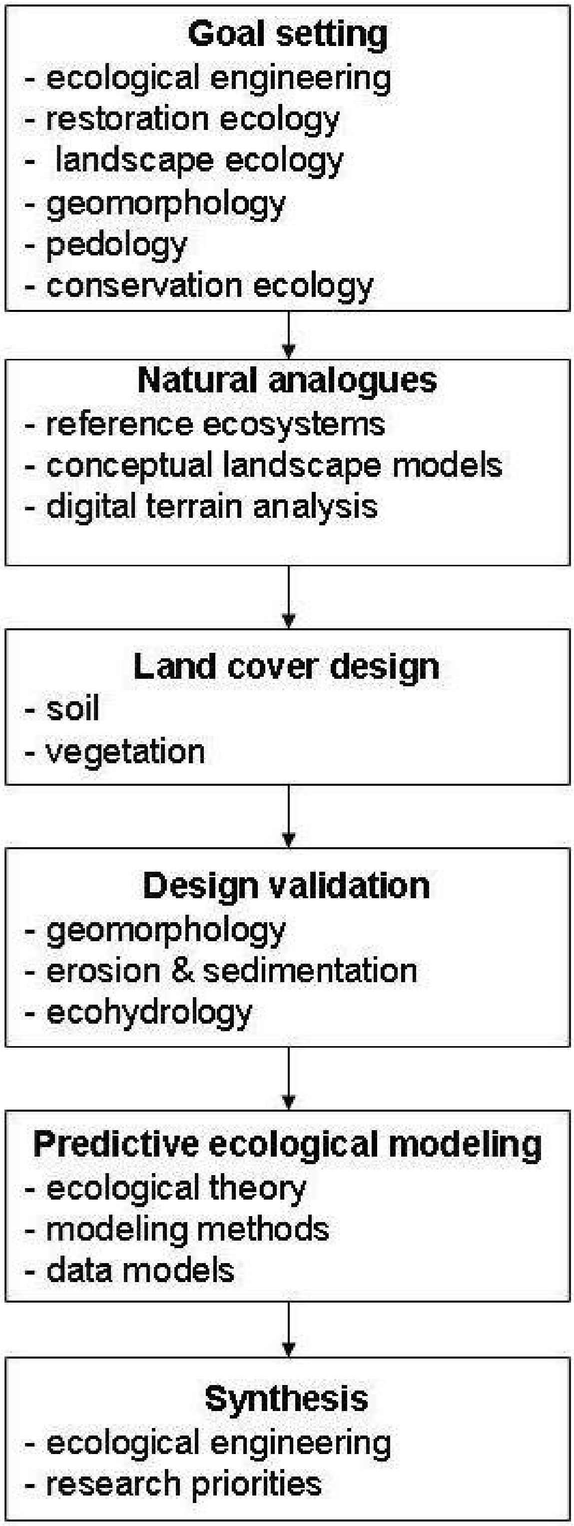


Figure 2-1 Conceptual framework and structure of this review

2.2 Goal definition

The main problem for ecological restoration may be goal definition, whether individual species, whole ecosystems or landscapes, or ecosystem functions are to be restored (Ehrenfeld, 2000). As a consequence, ecosystem reconstruction in open cast mine landform design lacks a clear methodology, and in situations where the community places a high value on indigenous ecosystems and the natural landscape, not much assurance can be given regarding the outcomes. Clear representation of the critical environmental processes that constrain developing ecosystems in the soil landscape continuum is best practice (Cairns, 1994). However, the range of technical perspectives involved in mine rehabilitation creates a diversity of goals and objectives that require synthesis to support an integrated ecological design approach.

Ecosystem perspectives in mine rehabilitation originated out of the Australian industry and have been used in other countries (Bell, 2001). These ecosystem perspectives are diverse, in contrast to well integrated physical process approaches to developing and evaluating mine landform designs (Nicolau, 2003). In the latter case, relief can either be designed with the idea of maximizing the viability of the ecosystem to be restored (on-site effects), or the relief is designed to export the smallest possible quantity of runoff and sediments to water bodies downstream to avoid any off-site effects and to conserve basic ecosystem services related to water quality (Ehrenfeld, 2000). This second approach is the most developed, both conceptually and methodologically. The first approach is the subject of the following review leading to research into an ecologically engineered approach to landscape restoration.

2.2.1 *Ecological engineering*

Ecological engineering operates on the principle of self-organising systems that perpetuate themselves (Allen et al., 2003). Ecologically-engineered systems seek to account for the environmental processes in appropriate natural landscapes (Odum and Odum, 2003) implying that the environmental factors causing diversity are known and are built *ab initio* into the design. Relief is a key parameter that sets the potential energy level of the landscape — driving biodiversity that is related to landscape heterogeneity, or patchiness, in mined landscapes where the original ecosystems have been totally destroyed (Hollingsworth and Spain, 2000; Sklenicka and Lhota, 2002) and also in natural landscapes (Hobbs and Cramer, 2003; Ludwig and Tongway, 2000; Ludwig et al., 1997).

Consequently, with the increased focus on environmental objectives the trend in the design and construction of hill slope landforms in mine rehabilitation is moving away from platform-bank designs based on civil, environmental engineering principles of hydraulic and geotechnical stability to an ecologically engineering approach of restoring catchments characterised by analogous relief to a natural landscape (Nicolau, 2003). This is evident in the geomorphic approach to topographic reconstruction developed in the USA (Toy and Chuse, 2005). Previously, Willgoose (1998) suggested an analogue approach for designing natural two-dimensional basin forms from digital terrain analysis. Later, Hancock (2003) demonstrated that the concave slope profile designs based on natural slope properties could reduce, by half, the estimated sediment loss in comparison with linear slope designs. However, these ecological approaches to mine landform design measure success in terms of geotechnical and hydraulic performance criteria and, while some principles have been developed, the relationships between landform design and natural revegetation outcomes have not been demonstrated.

Toy & Chuse (2005) initiated a detailed topographic restoration approach that would support clearer specification of ecological goals. They proposed that drainage-basin area, weighted mean slope, and drainage density for the pre-disturbed or analogue landscape could closely be replicated in a reconstructed topography. They used digital terrain analysis and computer aided design techniques to create “steady state” landscape configurations typical of mature natural landforms. There is sufficient detail in this design process to begin defining specific target habitats and ecosystems in the natural environment, thus enabling the design envelope to be extended to ecosystem objectives.

Nicolau (2003) opined that the drainage-basin restoration concept will continue to underpin developments in the field. He recommended design evaluation based on erosion (Renard et al., 1997) or landscape evolution models (Hancock et al., 2008; Willgoose and Riley, 1998). However, Nicolau (2003) overlooked the use of eco-hydrological models to validate landscape designs (Beverly and Croton, 2002; Croton and Reed, 2007) and restoration strategies (Bonell and Molicova, 2003). Catchment reconstruction with a view to restoring natural patterns of water and sediment movement, including interactions between groundwater and surface water, is a fundamental design principle linked to restoring natural vegetation patterns. Feedback relationships between landforms, geomorphic processes and plant communities have been known for a considerable time (Wondzell et al., 1996).

Specification of target habitats and ecosystems in the landscape design would be a development of the drainage-basin area methodology whereby ecological outcomes as well as erosion rate and receiving water quality requirements could be addressed. This approach is akin to bio-geomorphic reconstruction of catchments (Hupp et al., 1995), with emphasis shifted to hill slope processes supporting woodland rather from fluvial processes and interactions between organisms and riparian habitat. To be successful, concepts and methods from the ecological literature need to be drawn into the landscape design process to predict ecological outcomes.

2.2.2 *Restoration ecology*

Rehabilitation strategies for mined land can be categorised as passive or active (Bradshaw, 2000). Passive strategies rely on natural colonisation and succession (Tischew and Kirmer, 2007). They are underpinned by careful management of natural propagules (Kirmer and Mahn, 2001; Koch et al., 1996) and planned intervention according to successional status (Grant, 2006; Jochimsen, 2001), strategic monitoring (Wassenaar et al., 2007), and ameliorating metal toxicity (Li, 2006) and fertility problems (Short et al., 2000). In Europe, time sequences extend back to the Middle Ages (Pelka-Gosciniak, 2006; Trpak et al., 2006). However, passive strategies have performed well, albeit slowly, in North America where industrial time sequences are more recent (Huttl and Weber, 2001; Pensa et al., 2004; Rayfield et al., 2005; Wiegleb and Felinks, 2001). Where natural propagules are not carefully managed acceptable results may not be achieved by passive methods in a reasonable time period.

An active, or interventionist approach relies on engineering, geochemical and agronomic intervention to produce natural landforms, to ameliorate metal toxicity and acidity, and to provide suitable root zone characteristics in mine waste materials (Burke, 2003; Huttl and Gerwin, 2003; Kramer et al., 2000; Maiti and Ghose, 2005). A key objective is to re-establish natural ecosystems by planting vegetation identified from analogue studies (Guo et al., 2007) or observations of pioneering (Guevara et al., 2005) and climax species (Pelka-Gosciniak, 2006) that self-establish on mine waste. However, despite its meticulous approach to mined land restoration, there may be problems with arrested succession and moreover, broad landscape scale impacts can detract from public perceptions of restoration success (Brown, 2005).

Current active rehabilitation practices as applied to strip mining (Grant 2006) involve a conceptual framework of "adaptive management" based on a state and transition model proposed by Hobbs and Norton (1996) and applied at a range of bauxite and sand mining sites (Cummings et al., 2005; Grant and Loneragan, 2001). Success is measured with respect to analogue or reference sites according to diversity and community structural measures (Araujo et al., 2006; Rayfield et al., 2005), resilience to environmental stressors (Grant, 2006) and indicators of functioning ecosystems (Andersen et al., 2004; Brennan et al., 2003; Davis et al., 2002; Ferreira and Aarde, 1997; Hoffmann and Andersen, 2003; Paschke et al., 2003). The natural complexity of local soil and geomorphic factors affecting ecosystem variation are considered distracting to clear communication (Hobbs, 1997; Hobbs and McIntyre, 2005). However, there is a need for developing links between spatial pattern, environmental process and landscape function in order to extend restoration methods to waste rock landscapes where the topography is reconstructed.

Hobbs & McIntyre (2005) identified the need for adaptive management to prevent rehabilitation from transitioning to an unacceptable state. An example is the acacia-grass fire succession in which aggressive acacias and annual sorghum seeds re-establish after hot fires burn a rehabilitated site where adaptive management is used to restore transition to stable native ecosystems based on eucalypts (Andersen et al., 2003). Restoration practices in an adaptive management approach produce ecological outcomes that are in transition between different states. A range of vegetation community types are used as analogues to define, in broad terms, the possible end results of restoration (Brown, 1994; Cummings et al., 2005; Filet, 1994; Gillson, 2004; Guo et al., 2007; Li, 2002; Meurk and Swaffield, 2000; Westoby et al., 1989). It is assumed that these transitions can be managed without reference to landscape design.

The assumption that local landscape processes can be ignored in open cast mine rehabilitation is difficult to support. Understanding the effects of landscape on vegetation pattern is a crucial issue for restoration practices (Lindenmayer et al., 2008). Landscape classification is recognised as fundamental to assessing rehabilitation success, for instance Aronson (1996b) defined vital landscape attributes to evaluate the results of ecological restoration or rehabilitation undertaken with a landscape perspective. However, landscape classification methods used to assess ecosystem restorations do not address the need to improve outcomes by creating landform conditions that would support particular ecosystems. A coupling of ecological theory and restoration practices has been recommended to improve the scientific benefits of restoration projects and restoration outcomes (Choi, 2007; Palmer et al., 1997). A landscape focussed ecological perspective on landscape reconstruction needs to be included in mined land rehabilitation.

2.2.3 *Landscape ecology*

The lack of consideration of spatial scale and structure in adaptive management methods contrasts with the literature in landscape ecology (Ben Wu and Archer, 2005; Ludwig et al., 2002; Ludwig and Tongway, 1995; Rempel and Kushneriuk, 2003; Tongway and Ludwig, 1997; Urban et al., 2002; Wu and Hobbs, 2002). Bell (1997) recommended incorporating spatial assessment methods embedded in landscape ecology to represent the scale and pattern of environmental processes causing ecosystem variation in restored landscapes. This development would associate the topographic, soil and vegetation properties of natural analogue areas to provide the basis for describing what Nicolau (2003) identified as ecosystem function in reconstructed landscapes.

The concept of patchiness in landscape ecology (spatial diversity in two dimensions) is a beneficial attribute of the natural landscape in terms of ecological resilience and species diversity (Lawler and Edwards, 2002; Ludwig et al., 2002; Ludwig et al., 2004; Tongway and Ludwig, 1997). Patch arrangement in the landscape as well as patch type can be linked to landscape function in terms of how water and nutrients are retained and used for plant production (Ben Wu and Archer, 2005; Ludwig et al., 1999). The geomorphic properties that distribute water and nutrients in the landscape exert obvious control over how vegetation is patterned (Bastin et al., 2002; Ludwig et al., 2002; Williams et al., 1996).

The concept of ecosystem function was developed in landscape ecology to describe resource use in rangeland ecosystems (Ludwig et al., 2004; Ludwig et al., 1997; Tongway and Ludwig, 1997). Ecosystem function varies from highly functional to very dysfunctional at hill slope scales in terms of nutrient and water retention. This is because environmental processes linked to topography at hill slope scales distribute, or make available, primary resources (such as water and nutrients) that affect ecosystem function. The concept envisages that toposequences in hillslope landforms are described in enough detail to reveal nutrient and water cycling patterns.

The environmental processes contributing to ecosystem function are generally linked to the spatial resolution used in data models and two-dimension geometric measures of landscape structure. Geometric measures of landscape organisation or patch metrics (McGarigal and Marks, 1995) lend themselves to quantitative landscape and habitat analysis. The patch metrics vary in their utility to describe aspects of landscape function (Appelberg and Svenson, 2001; Marsden et al., 2002; Pearson, 2002). For rangelands and savanna woodlands in northern Australia, the metrics that reflect the hill slope hydrology are robustly correlated with landscape function and the ability to retain material resources to support perennial vegetation (Bastin et al., 2002). Also, landscape metrics related to class area could be characterized by strong relationships with species dispersal (Neel et al., 2004).

Extending current landscape ecological methods using physically based patch metrics derived from terrain attributes to design natural landscapes appears to be a reasonable development. There is the potential to incorporate the patch metric derived from digital terrain analysis (Wilson and Gallant, 2000b) with functional analysis (Ludwig et al., 2003), an interdisciplinary development that would give a physical basis to landscape ecological analysis of natural and disturbed landscapes that would be relevant to topographic reconstruction. This development would link ecological pattern to geomorphology and address the perception that landscape ecological methods had limited application to ecosystem restoration in degraded landscapes (Wu and Hobbs, 2002).

2.2.4 *Geomorphology*

The spatial pattern of water infiltration, erosion and sedimentation processes that underpin ecosystem function can be described using a digital elevation model (DEM). Geomorphic properties of slope length, gradient and curvature have a measurable effect on water distribution, erosion and sediment delivery (East et al., 1994; Hancock et al., 2003; Loch, 1998). Digital terrain analysis using raster data at an appropriate resolution are ideal for analysing the fine pattern and scale of habitat variation in the hill slope environments (Busby, 1991; Moore et al., 1991). Software developed for this express purpose (Gallant and Wilson, 1996; Moore et al., 1993) has been applied to a range of ecological mapping and conservation tasks (Coops and Catling, 2002; Mackey et al., 2000; Moore et al., 1993).

Accurate representation of water movement in the landscape is a pre-requisite for realistic habitat assessment. The surface splining method implemented in the ANUDEM software package (Hutchinson and Gallant, 2000) generates a DEM that respects observed drainage pattern from contour, spot height and surface drainage data that has set the standard for environmental and topographic modelling. The appropriate grid resolution for DTM required for environmental modelling depends on landform morphology. On relatively flat alluvial landscapes sub-metre contours may be required (McKenzie and Austin, 1993). In an undulating to rolling terrain a grid resolution of approximately 50-metres is required (Gessler et al., 1995; McKenzie and Austin, 1993). As a rule of thumb, the grid resolution that can be derived is equivalent to 1 millimetre of the published map scale (Hutchinson and Gallant, 2000). For example a 50-metre grid would be a reasonable from 1: 50 000 topographic maps to standard specifications.

2.2.5 *Soil: pedological and edaphological considerations*

Jenny's model of soil formation (Jenny, 1941) provides the unifying ecological concept used in formulating soil landscape mapping models and in understanding factors affecting soil development,

$$S(s)=f(c,o,r,p,t,..) \qquad \text{(equation 2-1)}$$

,where S is the soil, s any soil property and the notations in parentheses on the right hand side are the climatic, biotic, topographic, parent material and time factors respectively. The model is impossible to solve in a general way because of interactions between factors (Phillips, 1989). A modified factor model has been proposed to map soil using digital data that incorporates geographic determinism (spatial component) from survey data (McBratney et al., 2003). The modification is based on a generalisation of Jenny's five factors, namely: (1) *s*: soil, other or previously measured attributes of the soil at a point; (2) *c*: climate, climatic properties of the environment at a point; (3) *o*: organisms, including land cover and natural vegetation; (4) *r*: topography, including terrain attributes and classes; (5) *p*: parent material, including lithology; (6) *a*: age, the time factor; (7) *n*: space, spatial or geographic position.

Relationships between soil and topography are easily quantifiable but there is evidence of quantitative relationships with the other soil forming factors (McBratney et al., 2003).

Functional relationships have been established using linear models (Webster and Oliver, 1990), generalised linear models (Lane, 2002), generalised additive models (Odeh, 1997), tree models (Bui et al., 1999; Lagacherie and Holmes, 1997), neural networks (Minasny and McBratney, 2002) and fuzzy systems (McBratney et al., 2002; Zhu, 1997). Classification and regression trees perform well and provide rules which are relatively easy to interpret and extend soil survey information (Bui and Moran, 2003).

A wide range of statistical and geostatistical techniques have been developed or adapted, on their own or in combination with classical models of soil formation, to map soil properties or soil classes as functions of secondary topographic variables (Wilson and Gallant, 2000a) derived from a DEM, along with other spatially dense environmental data such as imagery (McBratney et al., 2000). Digital terrain analysis has been particularly useful in soil property mapping (Bui et al., 1999; Bui and Moran, 2000; Bui and Moran, 2001; Gessler et al., 1995; McBratney, 2004; McKenzie and Ryan, 1999; Odeh et al., 1991; Odeh et al., 1994; Odeh et al., 1995), which traditionally relies on a hill slope model to describe soil variation in many situations. Quantitative pedological models (based on terrain analysis) are matched by those used in conservation ecology where the focus has been on selecting reserves to conserve core habitats and maintain species diversity over regional extents.

2.2.6 *Conservation ecology*

Topographic variables derived from DEMs are used in almost every published example of predictive mapping of terrestrial vegetation at the regional scale (Guisan and Zimmermann, 2000). While this may be a gross simplification for rare and endangered species (Williams et al., 2007) it justifiable for common species with extensive habitats. Ecological models based on environmental correlation with topography could be used to predict the realized (ecological) niche; assuming that the vegetation is in equilibrium with the current environment in altered landscapes or environments.

Ecological theory holds that plant communities tend to change continuously along environment gradients (Austin and Gaywood, 1994) and the relationships between species and environment tend to be either linear or unimodal depending on the environmental range over which samples were taken (Guisan and Zimmermann, 2000). Skewed unimodal distributions with implications for the type of analysis used are also common (Austin, 2002b). The association of a particular species with the physical environment is defined as its habitat. The literature on prediction of biodiversity and species distribution in the landscape supports regional reserve selection strategies (Austin, 2007; Guisan et al., 2006; Guisan and Thuiller, 2005; Margules and Austin, 1991; McKenzie et al., 1989) for regional habitat conservation rather than the specific habitat reconstruction task in mine landscape design.

Habitat patterns in landscapes can be complex because organisms respond to their surroundings at multiple spatial scales, along multiple environmental gradients and different organisms respond differently to the same environment. Fischer (2004) conceptualized landscapes as overlaid species-specific habitat contour maps. Habitat contour modelling requires corroboration from field investigations stratified on the basis of ecological gradients other than human-defined patches and patch boundaries. This approach is similar to that being used in ecological modelling to predict individual species distributions and less commonly plant community distribution (Ferrier and Guisan, 2006) and is different from patch based habitat models that rely on land units or other thematic landscape classifications.

The habitat contour approach bridges the gap perceived between pattern and process in landscape ecology (Hobbs, 1997). The data models that support this approach are stratified according to gradients in ecological or environmental variables, which also links to quantitative soil survey methods (McKenzie and Ryan, 1999). These data models have a lot to recommend them over broad regional and unstructured patch based survey designs to support an ecological design methodology for topographic reconstruction.

2.3 Natural landform analogues for landscape restoration

2.3.1 *Reference ecosystems*

Approximating and understanding ecological variation with respect to reference conditions requires a clear context with the landscape that is being restored (White and Walker, 1997). Soils and geomorphology exert a high degree of control over ecosystem identity (Palik et al., 2000), although local management regimes can control ecosystem quality or condition (Holl and Crone, 2004). Consequently, to set regional landscape restoration objectives reference sites are chosen that represent ranges in disturbance within different plant community types (Shinneman et al., 2008) and terrain types (Gibbons et al., 2008; Palik et al., 2000). This produces complex and demanding survey designs that extend beyond the scope of the rehabilitation objectives for particular mine sites. Selecting the reference ecosystem types that represent relatively pristine habitats of target species, or desirable, restoration outcomes implies that the spatial scale and extent of environmental processes to be restored (critical to the selection and description of these reference ecosystems) is understood.

Finding a close match with the various dimensions of ecological variation can be difficult (White and Walker, 1997). Stratified sampling and multivariate analytical methods are required to ensure that survey and analysis programs assess the environmental range (Gibbons et al., 2008; Rempel and Kushneriuk, 2003). Statistical methods can then be used to incorporate topographic variables in a GIS framework that predicts vegetation type and condition by interpolation (Gibbons et al., 2008; Palik et al., 2000). Where there is regionally extensive survey support, extrapolation from areas of remnant vegetation can be used to predict pre-clearing vegetation patterns (Accad and Neil, 2006) and guide landscape restoration efforts. However, extrapolated prediction assumes (often incorrectly) that all of the environmental gradients are understood.

In summary, regional conservation strategies have driven quantitative methods for selecting and characterising reference ecosystems. This has resulted in strategies that focus on characterising ecosystem variation and condition by key species (often with high conservation value) and a regional landscape context to restore natural vegetation patterns in cleared or degraded landscapes. However, in mined land restoration, the landscape model is not regional but local and refers to the mine landscape. Also, the key species may be those that are critical to ecosystem function rather than rare and threatened species with high conservation values.

2.3.2 *The land unit ecological model*

In Australia, thematic soil mapping at the regional scale was the mainstay of environmental analysis up until the emergence of satellite data and DEMs. Thematic maps were, and are used to plan development (Atega, 1992; Kiesel et al., 1985; Wood and Wells, 1979), to design and stratify environment for sampling (Wood and Wells, 1979) and to identify natural analogues for mined land restoration (Brennan, 2005; East et al., 1994). However, the resolution and validity of regional thematic environmental mapping and the conceptual models on which they are based may limit their predictive capacity and accuracy.

The land unit concept was developed to map regional soil and land resources as unique classes of soil, landform and vegetation (Christian and Stewart, 1964). Land units are derived from air photo interpretation and ground truth data collected using free survey methods with sampling biased to higher land capability values and more accessible areas. The definition of land units emphasises detailed description rather than quantitative measurement or a sampling strategy that would support accurate prediction (McKenzie and Austin, 1993). Consequently, sampling and classification strategies in land unit mapping tend to confirm and extend concepts that are rarely tested. Land unit, patch-based landscape classification, is routinely used to select natural analogue, or reference sites, to plan restoration in mined land rehabilitation (Aronson and LeFloch, 1996a; Brennan, 2005; Riley, 1995a).

Land unit mapping has been fundamental to identifying natural analogues for landscape restoration (Aronson and LeFloch, 1996b; Zonneveld, 1989). For example, land units were used to develop design criteria for landscape restoration at the Nabarlek (Riley, 1995a) and Ranger uranium mines (Riley and Rich, 1998), both in East Arnhem Land in the Top End of Australia. The critical issues at these sites were ensuring long-term containment of radioactive waste and restoring natural woodland environment to the satisfaction of the traditional aboriginal owners of the land (RUEI, 1977).

The work influenced mine landform design generally (Nicolau, 2003; Toy and Chuse, 2005) and promoted the use of quantitative landform evolution models to assess failure risks for natural slope profiles that were constructed in place of engineered platform-bench slope profiles (Hancock, 2004). The Nabarlek site has since been rehabilitated and while containment criteria have been met (Hancock et al., 2006), the revegetation did not meet objectives for restoring native eucalypt woodland environments (Klessa, 2000). The unsatisfactory revegetation may arise from an imprecise definition of natural analogues and the inaccurate assessment of the landform design factors that determine long term ecological outcomes. Continuous, rather than patch based models of environmental variation in landscapes offer an alternative to land unit mapping for selecting analogues and analysing the factors that affect ecosystem variation in the landscape.

2.3.3 *Continuous environmental variation*

The concept of environmental variation along a continuum is central to predicting species distribution along environmental gradients (Austin, 2002a). Catenary concepts of continuous landform, parent material and soil pattern along toposequences in hill slope environments have been used to describe ecosystem diversity at hill slope scales (Ben Wu and Archer, 2005; Hobbs and Cramer, 2003; Ludwig et al., 1999) as an alternative to patch based land unit classification of landscapes. The catenary concept of continuous environmental variation is consistent with the habitat contour model proposed by Fischer (2004) and toposequence studies used in landscape function analysis (Ludwig et al., 2004). Landscape models representing gradient and scale dependent phenomena may be the key to better identification of landscape processes associated with ecological complexity.

Wu & Hobbs (2002) argued that landscape ecologists who focus on discrete patterns have developed only a limited capacity to understand ecological processes. Similarly, studies on direct rehabilitation processes have had only limited success in their applications to real landscapes. The choice of landscape model is critical to linking pattern and process in landscape ecology (Fischer et al., 2004). Quantitative landscape models are based on continuous variation in environmental variables that account for environmental processes at a hill slope scale (Ben Wu and Archer, 2005; Wilson and Gallant, 2000b). Such models could be used to link pattern and process description in natural (Fischer and Lindenmayer, 2007) and reconstructed landscapes. This would provide enhanced knowledge of environmental variation in natural landscapes, which could then be applied to designing mine landscapes.

2.3.4 *Digital mapping methods*

The availability of digital elevation models and continuous environmental variables from remote sensing has provided continuous environmental surfaces that have promoted the development of quantitative spatial prediction methods in pedology (McBratney et al., 2003), conservation biology (Austin, 2002b; Guisan and Zimmermann, 2000) and landscape ecology (Ludwig and Tongway, 1995; McGarigal and Marks, 1995). Digital mapping methods for spatial inference of soil properties have progressed to the extent that environmental factor models based on continuous variables are replacing thematic, patch based classifications such as land units (Lagacherie, 2008) and are underpinning mechanistic modelling of soil formation (Minasny et al., 2008).

Peneplanated landscapes represent a challenge to selecting analogues using digital terrain analysis to guide landscape restoration following mining (Russell-Smith, 1995; Wardell-Johnson and Horwitz, 1996). Wardell-Johnson and Horwitz (1996) concluded that mining produced subtle changes in water distribution that are related to landscape, which could have significant effects on biodiversity values in the temperate, high rainfall zone jarrah woodlands of south Western Australia. Accurate representation of water movement in the landscape from a DEM is a pre-requisite for using terrain analysis techniques in habitat assessment and the basis for advances in the biogeographical description of landscapes (Mackey et al., 2008). Accuracy in the terrain information needed to model water movement realistically over a disturbed mine site has hitherto been difficult to demonstrate (Nichols et al., 1985). Consequently, while land management factors may be well defined in a given restoration approach (Grant, 2006), the local spatial environmental context incorporating hillslope landforms tends to be overlooked (Hobbs and McIntyre, 2005). Also, the restoration of natural soil processes, which is important for ecosystem reconstruction (Bradshaw, 2004; Nicolau, 2003) is often overlooked.

2.4 Landform cover design and soil reconstruction

The surface cover, or reconstructed soil, is primarily intended as a physical or drainage barrier (Gatzweiler et al., 2001; Leoni et al., 2004; O'Kane et al., 2003; O'Kane and Wels, 2003; Williams et al., 2003a) where the aim is to isolate potential contaminants such as metals (Carlsson and Buchel, 2005) and radioactive materials (Bollhöfer et al., 2008; Lottermoser and Ashley, 2006) and to minimise erosion and sedimentation (Johnson, 2002; Simon-Coinason et al., 2003). Although, the engineering and ecological disciplines that dominate mine rehabilitation tend to overlook soil science (Callaham et al., 2008; Heneghan et al., 2008), restoring environmental processes is also an important factor that determines rehabilitation success (Koch and Hobbs, 2007; Roux, 2002). Restoring soil quality amounts to restoring the capacity to function within ecosystem boundaries, to sustain biological productivity, maintain environmental quality and promote plant and animal health (Doran and Parkin, 1994).

Soil conditions in rehabilitated landscapes can be critical to ecological outcomes (Croton and Ainsworth, 2007; Parrotta and Knowles, 2001; Szota et al., 2007). Consequently, management interventions such as deep ripping to ameliorate soil compaction and allow roots to exploit the subsoil water store may be critical for restoring ecological processes following mining in seasonally dry environments (Szota et al., 2007). Restoring landscapes so that vegetation has access to water is a key factor in restoring ecosystem function.

Mine soils are heterogeneous and characterised by the lack of clear links between mine soil chemistry and morphology and analogous natural soils (Huttl and Bradshaw, 2001). However, environmental processes such as nutrient dynamics (Vitousek and Farrington, 1997) moderated by the accumulation of organic carbon are soil factors that have been linked to ecosystem development in mine soils (Koch and Hobbs, 2007; Smith et al., 1997; Walker et al., 1996). Soil nutrient status (Bradshaw, 1997; Bradshaw, 2004; Huttl and Weber, 2001; Li, 2006; Short et al., 2000) and water relations (Jim, 2001) are particularly important for mine rehabilitation. Sustainable soil nutrition and moisture conditions require the establishment of organic carbon pools as well as vital soil fauna and flora for the successful rehabilitation of mine soils (Huttl and Gerwin, 2005; Koch and Hobbs, 2007). Altered soil properties and landscape positions can also affect the distribution of revegetation types (Wiegleb and Felinks, 2001).

The most limiting factor often determines environmental outcomes (Koerselman and Meuleman, 1996). Where waste material properties are different to the native soils then demonstrating the long-term success of ecosystem restoration should rely on quantifying ecosystem support processes such as water and nutrient cycling rather than endemic species occurrence or soil properties post-rehabilitation (Knoche et al., 2002). Although there are complex issues of measurement scale, accuracy and uncertainty to be addressed (Beven, 1997), deterministic, mass balance models of land cover and landform design interactions with vegetation, runoff and groundwater can be developed (Croton and Bari, 2001; Croton and Reed, 2007).

Species selection for revegetation is an issue that is usually based on the perception that they are endemic (Roux, 2002; Sarrailh, 2002). Selecting endemic species may not account for edaphic changes (Paschke et al., 2003) or common and abundant local species (Brown, 2005). The success of the reconstructed soil zone to support canopy leaf areas that are similar to natural woodland vegetation are critical to restoring landscape hydrological processes (Coops et al., 2004; Hutley et al., 2000; Leuning et al., 2005). Therefore the design should focus on restoring natural levels of plant available water in the reconstructed soil required to restore similar canopy structures, evapotranspiration rates and natural hydrological processes.

2.5 Validation of ecological design

2.5.1 *Geomorphic reconstruction*

Relief is a key ecological design parameter to consider in reconstructing the landscape (Nicolau, 2003). The landscape relief defines the potential energy and determines a range of flux conditions for sediment, surface and groundwater flow (Ehrenfeld and Toth, 1997). Starting conditions in the natural landscape are thought to control soil development (Phillips, 2001) and fine perturbations in initial relief could introduce significant variability in soil formation (Minasny and McBratney, 2001). Consequently, predicting the dynamics of soil (Minasny et al., 2008) and ecosystem (Austin, 2007) development presents some problems.

Landform and the type and variety of available waste material provide the framework for restoring the ecosystems (Brown, 2005; Miao et al., 2000; Nicolau, 2003; Toy and Black, 2000), while erosion and sedimentation processes affect how mine landscapes evolve. Mine landform reconstruction that conforms to pre-existing catchment boundaries and drainage densities has been recommended (Toy and Chuse, 2005). Designing hill slopes with curvature can also restore natural patterns of water and sediment distribution (East and Cull, 1988; Hancock et al., 2003). Geomorphic processes in natural analogue areas will need to match with those in the mine landscape, indicating that catchment geometry is also a key design parameter.

2.5.2 *Erosion and sedimentation*

Soil erosion reduces the viability of vegetation on slopes (Holl, 2002; Ludwig et al., 2002). Loch (1997) reported that, in environments which suffer intensive water erosive precipitation, plant growth and rainfall erosion risk are closely linked. In another study by Nicolau and Asenio (2000) they opined that environments where water deficit is the main limiting factor for plants it is frequently associated with intense water erosion of the soil surface. As a consequence there is a decrease in water availability caused by a decrease in soil depth (diminishing water storage capacity) and by crusting of the surface (reducing infiltration capacity) of soils with low plant cover. Geomorphology influences the stability and diversity of restored ecosystems because it is the key to the supply of water to plants and it determines the local intensity of soil erosion, which can hold back ecological succession (Tischew and Kirmer, 2007).

Erosion and sedimentation patterns in landscapes can be assessed using empirical (Renard et al., 1997; Wilson and Gallant, 1996) or deterministic models (Hancock et al., 2008). Transport limited erosion processes are readily modelled and the choice of flow routing algorithm can affect the results. The Terrain Analysis Package for the Environmental Sciences, TAPES-G, (Wilson and Gallant, 1996) has a range of flow routing options including demon (Costa-Cabral and Burges, 1994) and FRho8 (Moore et al., 1993) that perform well on a range of surfaces. Simpler flow routing methods such as d8 (O'Callaghan and Mark, 1984) are more generally used but are inappropriate to model flow dispersion on convex surfaces or flat areas and do not perform well on artificial landscapes.

The weakest point in applying erosion models to ecological design is assigning a value of maximum tolerable erosion. Alternatively, ecological models that assume non-linear relationships between vegetation and erosion and which incorporate feedback systems between them could be developed (Chartier and Rostagno, 2006). Furthermore, erosion rates in the mined landform could be expected to conform to similar natural analogue landforms (Evans, 2000; Lowry et al., 2006; Moliere et al., 2002), while sediment delivery rates would need to comply with preserving the receiving environment (Hancock et al., 2006).

2.5.3 *Eco-hydrology*

Maintaining plant available water store in the soil zone at pre-mining levels is important for restoring pre-mining vegetation type and structure. This is particularly so in seasonally dry environments where water supply and drainage constrain plant growth (Bowman et al., 1993; Williams et al., 1996). Hill slope curvature also affects the way that water and sediment are redistributed to support ecosystem function (Ludwig et al., 2002; Ludwig et al., 1999) that affects the plant community distribution in the natural landscape (Ben Wu and Archer, 2005). However, mining may cause hydrological perturbations within catchments that are difficult to rectify. Groundwater-dependant ecosystems are particularly vulnerable to altered water balance conditions (Eamus and Froend, 2006; Johnson and Miyanishi, 2008). Additionally, where the hydrological and hydrogeochemical changes are severe, revegetation alone may not be effective and engineered systems may be needed (Hatton et al., 2003).

Deep tillage can be used to remove root barriers and compacted layers in truncated soil profiles in mined areas (Kew et al., 2007). Tillage can increase the depth of tree water use (Szota et al., 2007) and thus help restore catchment hydrological balance (Croton and Reed, 2007). Water supply and drainage are key environmental factors in landscape or ecosystem restoration.

There exist spatially distributed water balance models that address plant water use in detail and better represent the mined and natural landscape hydrology in three dimensions (Croton and Reed, 2007); these models have been used to validate ecosystem restoration strategies (Beverly and Croton, 2002). This methodology could be used to augment recommended landform validation using erosion and landscape evolution models (Nicolau, 2003). Assuming that natural catchment processes can be restored, the distribution of key species across the mine landscape needs to be a predictable function of landform in context with natural analogue areas.

2.6 Predictive ecological modelling

Statistical methods that use environmental correlation to predict target soil or ecological variables rely on an accurate description of environmental space (Guisan et al., 2002; McBratney et al., 2000). Conceptual landscape models using unstructured surveys may include theoretical assumptions that compromise reliable prediction (Austin, 2002b; Guisan and Harrell, 2000; Margules et al., 1987). Consequently, statistical models that require the environmental range to be adequately sampled (Austin and Gaywood, 1994) have largely replaced conceptual models that produce environmental patterns such as land units (Aronson and LeFloch, 1996a; Zonneveld, 1989) from thematic information and unstructured survey observations. The reliability of statistical methods is explicit during model selection and calibration.

Ecosystems are highly multidimensional. Species diversity and community structure can vary in response to multiple environmental gradients (exogenous) as well as proximal (endogenous) factors, which can compromise statistical inference methods (Wagner and Fortin, 2005). To further complicate this, different environmental variables may act over different ranges (White and Walker, 1997). Consequently, while detailed representation of environmental surfaces underpins quantitative methods in ecological modelling, a level of regionalisation in environmental domains (to a landscape or bioregion as well as proximal variation) is a factor in designing surveys to elucidate environmental and identify habitat pattern (Wu and Levin, 1997; Wu and Hobbs, 2002).

Species distribution models can be categorised into two types: (i) static or equilibrium models and (ii) dynamic models. The relationship between vegetation distribution and environmental variables modelled with sampled observations is assumed to extend throughout the study area (Franklin, 1995). Static vegetation models (SVM) are based on simplifying assumptions such as vegetation distribution being in temporary (or pseudo-) equilibrium with the environment. Static models do not directly consider dynamic ecological processes such as competition, predation and disturbance, all of which can affect the spatial arrangement. Although SVMs developed for individual species are more robust than those developed for plant communities and can account for dynamic processes such as climate change (Ferrier and Guisan, 2006). Individual species SVMs can be combined to describe the pattern of plant community variation (Austin, 2007).

Alternatively, dynamic vegetation models (DVM) seek to predict vegetation distribution based on dynamic processes such as climate, eco-disturbances, and soil. These processes can be far more complex and difficult to validate in space and time (Austin, 2007; Ferrier and Guisan, 2006; Guisan and Thuiller, 2005). Transition between phases or central tendencies (such as grassland and woodland) mediated by disturbance above a threshold by fire (Gillson, 2004) or erosion (Chartier and Rostagno, 2006) can represent temporal perturbations in the ecosystems. The state and transition models applied by Grant (2006) to assess and manage eucalypt woodland revegetation at mine sites with respect to revegetation endpoint criteria in the natural landscape are an example. Generally, in DVMs the complexities mean that the temporal and spatial components are not explicitly addressed.

Ecological modelling to validate design for biodiversity in a mine landform requires an explicit and quantitative understanding of the effects of spatial distribution of environmental factors on revegetation endpoints. Consequently, a static or “equilibrium” modelling approach, based on data from stable analogue ecosystems is preferable to dynamic modelling at this point in time. The next sub-section describes the methods involved with SVM within a spatially distributed environmental framework.

2.6.1 *Static vegetation distribution modelling methods*

To use static vegetation models of species distribution with environment in mined landscape restoration the working assumption is that stable natural ecosystems exist that are analogous to those that will develop in the reconstructed mine landscape and that dynamic ecological processes, particularly during ecosystem establishment, can be managed. The validity of these assumptions varies between sites and comparative analysis of the environmental factors across a broad range of restoration sites could explain variable outcomes (Aronson and LeFloch, 1996b; Holl et al., 2003). However, methodological papers in the restoration ecology literature report that identifying the ecological endpoints, in terms of the dominant overstorey native species in local ecosystems, are the key to successful ecosystem reconstruction (Grant, 2006; Holl, 2002; Holl and Crone, 2004). While environmental and physiological constraints are known by conservation ecologists to limit species distribution in hillslope landscapes along environmental gradients (Austin, 1987).

Most SVMs are empirical and the species relationships with environmental gradients are fitted statistically (Guisan and Thuiller, 2005). Austin & Heyligers (1989) categorized ecological gradients into three types, namely *resource*, *direct*, and *indirect* gradients. Resource gradients are used to quantify matter and energy fluxes that are consumed by plants or animals (nutrients, water, and light for plants, food, and water for animals). Direct gradients are environmental parameters of physiological importance, which are not consumed (temperature, pH). Indirect gradients have no direct physiological relevance for a species' performance (slope, aspect, elevation, topographic position, habitat type, geology).

Indirect gradients of environmental variables usually represent the combined effects of different resource and direct gradients in a simple way. The environmental gradient classification may not be exclusive, for example water can be a resource gradient under conditions of low availability and an indirect gradient when it is sufficiently abundant to cause anaerobic conditions due to waterlogging. Consequently, indirect variables may be the best predictors for modelling at small spatial scales and in complex topography, but the species response models based on them often fail to extrapolate to landscapes beyond the geographic range of survey support (Austin et al., 2006). However, indirect environmental gradients are most easily measured in the field, can be derived from digital elevation models and often correlate well with observed species patterns (Guisan et al., 1999).

Where indirect environmental gradients are measured, which are effective through correlation with the direct and resource gradients, modelling results are particularly sensitive to variable selection. Adequate sampling of the environmental range and selecting highly correlated and independent environmental variables are important to ensure accurate predictions of species distribution from indirect variables. Quantitative models should be fit with the least number of parameters (Guisan et al., 2002; McBratney et al., 2003) so as to avoid collinearity between environmental predictors – a situation that invalidates regression models. Restricted sampling reduces the combinations of environmental variables under which a model is calibrated and consequently reduces the reliability of predictions that are made (Thuiller et al., 2004). In this situation, a broad range of environmental parameters need to be reviewed to select the best. Independent predictors of species response and multivariate analysis are suited to high-dimensional problems.

Hybrid gradient analysis and indirect gradient analysis refer to the two general classes of multivariate gradient analysis and the way that environmental variables are handled can affect results. Hybrid gradient methods, based on constrained ordination such as CCA (Canonical Correspondence Analysis) have been popular for modelling species response to environment (Leps and Smilauer, 2003). A weakness of the multivariate analysis methods such as CCA that purport to directly relate plant community variation to environmental gradients is that they ignore community structure that is unrelated to the environmental variables. In contrast, in multivariate analysis methods that assess gradients indirectly an ordination is performed on just the community data, and then secondarily relates the ordination to the environmental variables, allowing an expression of pure community gradients, followed by an independent assessment of the importance of the measured environmental variables. CCA combines a multivariate ordination of species occurrence data with a regression constrained to maximise the linear correlation between the species ordination axes and selected environmental variables. The assumed linearity of response is often invalid (Bio et al., 1998) and the principle components analysis (PCA) technique used to generate the axes produces methodological artefacts by compressing either end of the range in highly dimensional data.

Indirect gradient analysis on the other hand involves initial ordination and then rotation in ordination space to maximise correlation with independent environmental variables to focus on significant environmental variables. Indirect gradient techniques are particularly effective when nonmetric multidimensional scaling (NMS) methods are used for ordination. NMS avoids assumptions of linear relationships among variables and performs well in highly dimensional data sets (Minchin, 1987a). Such unconstrained ordination often results in lower correlation between key environmental variables and ordination scores but provides a better representation of the overall community structure (McCune and Grace, 2002; Minchin, 1987b).

Multivariate ordination regression (direct gradient methods) such as CCA have value as exploratory overview of plant community trends with environmental variation and to identify and exclude collinear environmental predictors (Guisan and Harrell, 2000). Although, multivariate methods of indirect gradient analysis such as NMS are less prone to misinterpreting highly dimensional data and where there are non-linear community trends with environment are used as a precursor to quantitative prediction by univariate statistical methods (Urban et al., 2002). Indirect gradient analysis using non-metric multidimensional scaling ordinations methods avoid distortions in species space produced by PCA in hybrid gradient methods (Minchin, 1987a).

CCA also assumes that unimodal or bell shaped curves describe species distribution in response to the underlying environmental gradients as specified by the ordination axes. Species interactions (competition) may change the response shape even if the fundamental response was symmetrical (Minchin, 1987a). Also, the relationships between species response and environmental factors in CCA are described by straight lines, whereas in reality these relationships are more often more complex (Bio et al., 1998). Generalised linear models GLMs, more especially the generalised additive models (GAMs), are an improvement over multivariate, hybrid gradient analysis using CCA for species specific modelling in most situations (Bio et al., 1998) where interspecies competition for resources plays a significant part in determining asymmetrical habitat responses (Austin, 2007; Heikkinen and Makipaa, 2010). GAMs provide probabilistic predictions and are more theoretically valid in most situations than CCA (Austin, 2002b).

The choice of environmental parameters needs to be considered. SVMs fitted using GAM techniques and based on indirect environmental parameters (such as topographic variables) of environmental gradients perform as well as SVMs that use direct measures of environment, such as radiation and temperature (Austin et al., 2006). Classification and regression tree techniques (CART) are another approach that has been used for predicting species distributions (Accad and Neil, 2006) and to detect species interactions (Franklin et al., 2000). Although CART functions are discontinuous when a continuous function may be more appropriate (Austin, 1987; Austin and Smith, 1989). The best practice may be to include species interaction using CART combined with autocorrelation in GAM models (Austin, 2002b). Lehmann et al. (2003) implemented this in a statistical software package called Generalised Regression Analysis Species Prediction (GRASP) implemented in SPLUS 2000.

2.6.2 *Ecological data models*

The scale and purpose of a given project determine the data model to be adopted (Austin, 2007). Two important aspects of scale are extent and resolution (Guisan and Thuiller, 2005). While extent refers to the area over which a study is carried out, resolution is the size of the sampling unit at which the data are recorded or interpolated. Data availability may severely restrict the purposes for which the data can be used or place caveats on the usefulness of the results for the intended purpose. If the purpose is to investigate the environmentally realized niche of a species then the extent of the study should range beyond the observed environmental limits of the species (Austin et al., 1990). If this is not the case, then the species responses are truncated and the actual shape of the species distribution function cannot be determined.

The spatial resolution of a sampling unit governs what variables can be measured and what processes can be considered to operate in determining species distribution and abundance. For instance, Leathwick & Austin (2001) found that differences in measurement plot size between 0.4 and 0.1 hectares limited the generalisations that could be made about inter-species competition in New Zealand. Species competition may occur at a different scale to edaphic variation and redistribution of water in the landscape. Also, Ben Wu and Archer (2005) found that topographic effects on runoff and runon at hill slope scales overrode finer scale effects of soil texture on vegetation pattern in woodland savanna ecosystems. Consequently, digital terrain data may describe vegetation pattern more accurately than detailed site based observations of soil properties.

2.6.3 *Biotic response data*

Biotic response data can be used to assess whether mine landforms will support similar levels of biodiversity that exist in natural landscapes. There are three types of biotic data usually considered in spatial prediction: (i) abundance in various forms; (ii) presence–absence data; and (iii) presence-only data. The development of GLM and GAM statistical methods has enabled the incorporation of nominal data on abundance and binary presence–absence data (Guisan and Zimmermann, 2000). In applying the response data coded as abundance or presence–absence the strength of species-environment relationships can be affected (Cushman and McGarigal, 2004b). However, for species distribution modelling, there is no disadvantage in using species presence–absence data (Guisan et al., 2006). However, abundance data are essential for describing species dynamics (Austin, 2007).

2.6.4 *Environmental predictors*

In many cases the main prerequisite for survey design and prediction using environmental correlations is the DEM (Guisan and Zimmermann, 2000). Biophysical data required for spatially explicit predictive ecological models are derived from DEMs using digital terrain analysis to produce spatially distributed high resolution environmental metrics (Wilson and Gallant 1996). The value of landscape metrics in terms of surrogacy to water balance, radiation balance, erosion and deposition depends on using a resolution appropriate to describe the variation in hill slope environments and natural drainage geometry (Gallant and Wilson, 1996). Also, surrogacy limits the extent of the environmental conditions for which metrics can be used to areas where the links to environmental process are understood.

Modelling biotic response using environmental predictors has advantages over purely spatial modelling methods based on geographic correlation (Guisan and Zimmermann, 2000). Geographic correlation (autocorrelation) may violate assumptions of non-stationarity in the data that are implied in regression modelling (Guisan et al., 2002) and incorporate process knowledge that is outside of the static ecological model. As well, autocorrelation implies that the ecological model could be better defined (Austin, 2007). It is therefore important to design sampling strategy that minimise spatial autocorrelation (Guisan and Zimmermann, 2000). The context of landscape design requires predicting a biotic response from the environmental factors that are believed to be the causal, driving forces for their distribution and abundance.

2.6.5 *Autocorrelation*

Autocorrelation, the cornerstone of spatially explicit predictive modelling, implies that places close to one another in time or space tend to have similar values, whereas ones that are farther apart differ more on average. Guisan (2006) and Guisan & Thuiller (2005) reviewed modelling approaches developed to take into account environmental interactions and spatial autocorrelation. Autocorrelated variation can account for biological processes associated with species dispersal ability, historical disturbance or physical dispersal barriers if they are not accounted for in presence–absence measures of species distribution. Quantifying the spatial autocorrelation at the scale of interest has been used to minimize and estimate errors in models that rely on environmental correlation (Bishop and McBratney, 2001; Dormann, 2007; Maggini et al., 2006; Zhang and Gove, 2005; Zhang et al., 2005).

Miller & Franklin (2002) used indicator kriging with presence–absence data to assess spatial autocorrelation in CART and GAM models of species distribution and found that the spatial parameter was usually highly significant and related to the way the model had been specified. Spatial autocorrelation in ecological data can inflate Type I errors (where the null hypothesis is falsely rejected) in statistical analyses (Lichstein et al., 2002). These errors may be due to poorly specified models (Austin, 2007), missing covariates (Barry and Elith, 2006) or spatially biased sampling strategies (Hawkins et al., 2007).

Miller et al. (2007) reviewed methods of incorporating spatial dependence in vegetation models and identified that geostatistical methods such as indicator kriging were the most appropriate for species presence–absence data at landscape scales. Earlier, Pfeffer (2003) used universal kriging to interpolate detrended correspondence analysis (DCA) scores in ordinations of alpine vegetation where the trend was a linear function of topographic variables. However, incorporating autocorrelation without considering the processes that underlie species distributions could lead to less than adequate ecological models (Austin, 2002b). The extent to which vegetation is in equilibrium with environment is at the root of the issue. Consequently, autocorrelation will be a critical factor in modelling ecosystem restoration following disturbance, and in identifying where dynamics that need further investigation occur in natural analogue vegetation types.

2.7 Synthesis

Ecosystem restoration criteria and revegetation plans still follow the landform design process rather than contribute to it, which reflects the lack of ecological specification in the current design paradigm. The current ecological design principles are framed in two dimensional plan and profile properties of landscapes. A capacity to quantify ecological outcomes as functions of landform design is needed before revegetation specifications can also be considered in the landform design process along with erosion and hydraulic performance. A clear understanding of the salient environmental processes in the soil landscape continuum will underpin the solution to landform design problems. Ecological engineering methods that need to be developed to restore habitats in landscapes that are reconstructed from waste rock will be based on extending plant community models from natural landscapes to mine landforms.

The critical issue for designing biodiversity into opencast mine landforms successfully is understanding control of species distribution and diversity in analogue natural landscapes. Initially, this requires selecting natural analogues that cover the expected range of environmental outcomes in the mine landform. Secondly, ecological outcomes in the post mining landscape need to be predictable functions of quantitative parameters that can be built into mine landscapes.

Without understanding how to re-create physical and ecological processes there is limited capacity for ecosystem restoration and no capacity for design validation. In such a situation the focus of reclamation may miss several key processes, such as biogeochemistry, the loss of, or dramatic shallowing of soils, and linkages among ecosystems. However, conceptual tools for structuring and evaluating landscape restorations can be developed from environmental processes and properties of mine landform analogues in the natural landscape. Analysis of the extent and environmental grain size attributes of ecological scale in the natural and the mine landform to resolve hill slope environmental variation is needed. Also, the results need to be qualified by assumptions about the effects of predicted changes in climate on current habitats and ecosystem distributions.

2.7.1 *Research priorities*

An ecological design capacity with predictable outcomes needs to be developed to restore mined landscapes after opencast mining. To this end, ecological engineering methodologies based on topographic reconstruction could be extended to address restoration of endemic natural ecosystems by: (i) including quantitative methods to select natural environmental ranges as analogues to the restored landscape; and (ii) providing predictive capacity at the landform design stage to quantify species and biodiversity outcomes so that these aspects of the landform design can be evaluated. There is a disjunct between landscape ecology and ecosystem restoration that needs to be addressed to implement ecological methodologies in landscape design.

The landform model that is used is also a critical issue for communication between ecological and engineering disciplines and for extending ecological principles into mine landform engineering. Using quantitative ecological and ecohydrological models to bridge these gaps could potentially improve mine rehabilitation outcomes and test and improve the theoretical understanding of disturbed ecosystems. Therefore, the investigations that follow in this thesis are concerned with: (i) testing land unit models and establishing predictive relationships between landforms in hill slope environments and ecosystem variation; (ii) developing a quantitative approach to selecting natural analogues to represent the environmental range for ecological design of mine landforms; (iii) assessing a constructed mine landform trial in terms of the water balance support for ecosystems; (iv) validating environmental performance of a draft landform design; and (v) predicting vegetation species distributions across a post mining landscape terrain model. This final investigation is the capability that is required to demonstrate ecological principals are incorporated in landscape reconstruction.

Chapter 3 Rule-based land unit mapping of the Tiwi Islands[2]

3.1 Introduction

3.1.1 *Background*

The working hypothesis for designing ecosystem restorations and predicting revegetation outcomes is that regional land and ecological surveys using conceptual ecological models can be used with adaptive ecosystem management to identify targets and predict revegetation outcomes (Hobbs and McIntyre, 2005; Lindenmayer et al., 2008; White and Walker, 1997). The Tiwi Islands mapping project was an opportunity to test how well land unit classes predict vegetation distribution in situations where landform was the main factor affecting environmental pattern — such as occurs in many land forming operations at mine sites. Hence, the inclusion of the Tiwi Islands survey in a thesis that investigates using analogous natural landforms to design for biodiversity in mine landforms.

Land units are an ecological concept that was developed for qualitative mapping of regional land resources (Hewitt et al., 2008). Land unit mapping represents soil, landform and habitat diversity in a landscape where the native vegetation is largely intact. The map product of this process is widely used for development, conservation planning and identifying environmental analogues for landscape restoration (Aronson and LeFloch, 1996b; Brennan, 2005; Riley, 1995a; White and Walker, 1997; Zonneveld, 1989). Land units, as described in the Northern Territory land resources survey, represent unique sub-class combinations of geomorphic unit, lithology, vegetation and soil classification which occur as a repetitive pattern within broader land systems (Christian and Stewart, 1964).

Land units are an attractive model for ecological survey and analysis because of their holistic conceptualisation of landscape (Zonneveld, 1989) and because the information is widely available from government agencies (Hobbs and McIntyre, 2005). Land units model soil variation according to the environmental factor model of soil formation (Jenny, 1941) that was discussed in Section 2.2.5 and are a complex classification of landforms, delineated from stereographic air photo interpretation of physiography, vegetation pattern

[2] This chapter was accepted for publication as: Hollingsworth, I., Odeh, I., Bui, E. and MacLeod, P., 2007. Rule Based Land Unit Mapping. In: P. Lagacherie, A.B. McBratney and M. Voltz (Editors), Digital Soil Mapping: an Introductory perspective Chapter 29. Developments in Soil Science. Elsevier, Amsterdam, pp. 404-414.

and land surface type. Uniform units are described following overlay of geological and air photo mapping and field survey based typically on 1:100 000 scale topographic maps. The complexity of the unit concept and the subjectivity of sub-class definitions in different surveys may lead to disparity in the quality and types of unit being used.

Digital Soil Mapping (DSM) methods have been used to assess and improve the quality and extent of thematic land resource mapping (Lagacherie, 2008). Guisan & Zimmermann (2000) described a broad range of applications using this technique, with Digital Elevation Models (DEMs) and a range of terrain variables derived from them as the core environmental data, for quantitative modelling of ecosystem variation. The effectiveness of environmental mapping using digital terrain attributes is sensitive to the scale of the base mapping and the resolution of the DEM that can be derived (Gallant and Hutchinson, 1997). Studies of the sensitivity of mapping reliability to DEM resolution in southern Australia (Gessler et al., 1995) found that 30 to 50 metre grid resolution was effective for representing hillslope variation in undulating to rolling landscapes. As a rule of thumb, a 50 metre grid can be derived from 1:50 000 topographic base data (Gallant and Hutchinson, 1997). Lower relief landscapes may require higher resolution, and therefore more accurate base data (McKenzie et al., 2000). Different environmental processes operate at different scales in landscapes (Ben Wu and Archer, 2005). Consequently, improved vegetation or soil mapping performance doesn't necessarily follow from higher resolution data. However, a grid resolution that is some fraction of the hillslope length is needed to resolve the processes that affect vegetation and soil properties.

A most relevant DSM technique is Decision Tree Analysis (DTA), which can generate rule based mapping, analogous to expert systems used by traditional land resource surveyors, from continuous and discreet environmental predictors. DTA has been applied extensively to map vegetation properties (Franklin et al., 2000), to predict habitat distribution(Guisan and Zimmermann, 2000; Sesnie et al., 2008), to extract soil mapping rules from extensive geology and DEM-derived attributes (Bui et al., 1999) and to produce continuous soil property maps from disparate soil survey data (Bui and Henderson, 2003; Henderson et al., 2005). However, DTA is prone to error when extrapolated to unfamiliar landscapes (Gahegan, 2000).

Technical developments with remote sensing have replaced thematic environmental information with continuous environmental layers in the definition of land units. For instance, the thematic geological layer in the land unit delineation can be replaced with continuous data surfaces from multispectral remote sensing — using band ratios that reflect differences in surface soil mineralogy (Laffan and Lees, 2004) — or where high resolution airborne radiometric data is available the Uranium, Potassium and Thorium bands can be correlated with surface soil mineralogy with less interference from vegetation cover (Wilford, 1992; Wilford, 2006).

The main aim of the work reported here was to apply a land unit model to an extensive area of savanna woodland, the Tiwi Islands (7 320 km^2), with relatively little variation in parent material and climate to identify physiographic effects on ecological pattern. The Islands are located in tropics at 12^0 S, 130^0 E (Figure 3-1). Land unit maps had been created over some parts in the 1970's to assess land capability for forestry and agriculture (Olsen, 1980; Van Cuylenburg and Dunlop, 1973; Wells and Cuylenburg, 1978; Wells et al., 1978). Since this time, digital 1:50 000 topographic mapping (capable of deriving a 50 metre DEM with ± 2 m vertical accuracy) over the Tiwi Islands has become available.

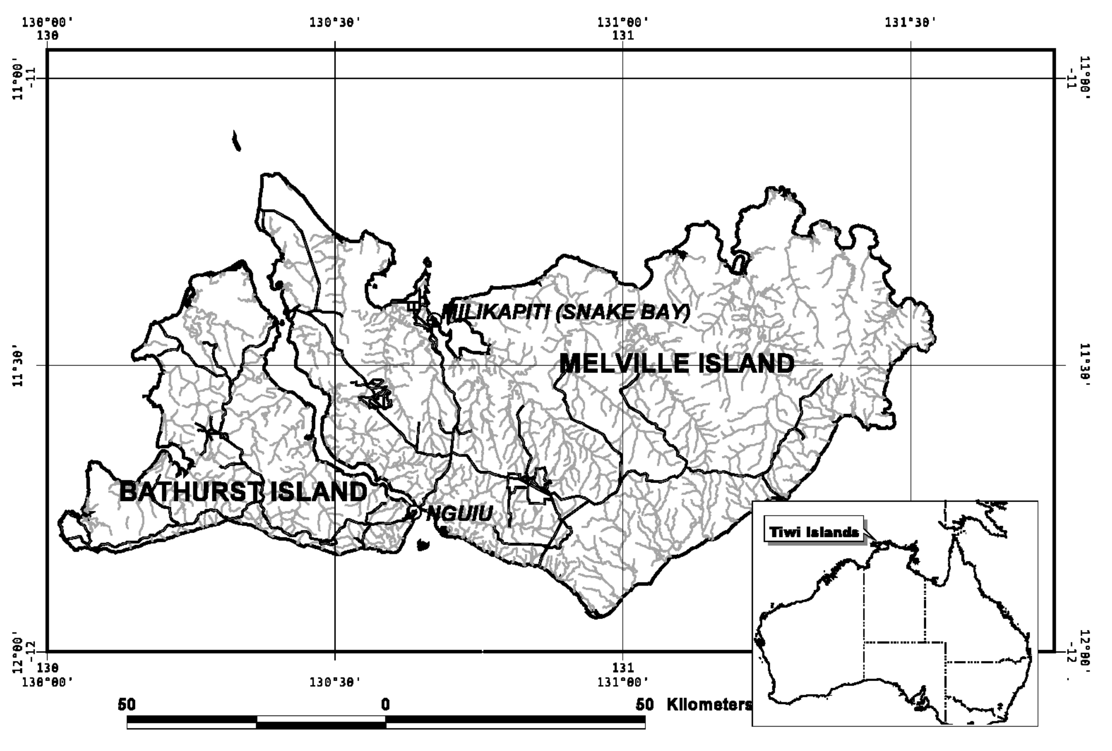


Figure 3-1 Location of the Tiwi Islands in Australia

An application of digital soil mapping (DSM) is presented that substitutes a DEM and terrain analysis for airphoto interpretation and uses statistical inference in place of expert judgement to map a land unit legend compiled from historical surveys across the Tiwi Islands — an extensive landscape with variable relief and relatively uniform geology and climate. There were gaps and inconsistencies in the existing land unit maps that needed to be filled or corrected. To do this, DTM methods were used to correlate land unit classes with environmental data surfaces. The predictability of land units and their component vegetation and soil types from environmental data is used to assess the land unit concept as a tool for identifying natural analogues in mined land restorations.

3.2 Materials and methods

The work reported here makes a continuous and consistent assessment of land resources of the Tiwi Islands, comprising Bathurst and Melville Islands, which was subsequently used for land capability assessment and strategic development planning. The DTA mapping method made use of legacy soil survey data, digital topographic mapping and Landsat TM data. Because climate and lithology are relatively uniform across Tiwi Islands (Nott, 1994a) the sampling design focussed on physiographic factors that are known to control soils and land unit diversity.

3.2.1 *Environment*

The Tiwi Islands have an equatorial savanna climate that is dominated by northwest monsoon and the southeast trade weather patterns. The islands experience seasonal drought during the southeast trade season between May and September. The surface hydrology exerts considerable control over habitat diversity in this climate because water supply is a limiting factor for plant growth (Eamus, 2003).

The geology consists of Quaternary alluvium, flat bedded, Tertiary Van Diemen Sandstone and the underlying Cretaceous Mookinu mudstone and Wangarlu Mudstone members of the Bathurst Island Formation. The Tertiary Van Diemen Sandstone is fine- to medium-grained quartzose sandstone of fluvial and partly littoral origin. The sandstone covers most of Melville and Bathurst Islands and varies in thickness to a maximum of 80 meters and dips gently to the northwest. The origin of the clastic sediments is the sandstone in the highlands, located to the south of the Tiwi islands during the Tertiary Age (Nott, 1994a).

Dissected plateau remnants 150 metres high form the current highlands in the centre of the islands, which are fringed by coastal mangroves and cheniers. Soils on sandstone are typically Oxisols (Ustoxs) with deep sandy to sandy loam surface horizons. Soils originating from mudstone are typically Ultisols (Ustults) with strongly acid subsoils overlain with a layer of pisolithic gravel.

3.2.2 *Definition of land units*

Historical soil and vegetation survey reports of the Tiwi Islands were correlated to produce a legend of 27 land units describing 10 soil family classes and 26 plant community types (Brocklehurst, 1998; Olsen, 1980; Wells and Cuylenburg, 1978; Wells et al., 1978) an excerpt of which is produced in Table 3-1. Soils were classified according to the soil family system used in the original land unit survey reports of the Tiwi Islands. Also, soil orders from the USDA Soil Taxonomy are used to facilitate communication of general soil properties.

3.2.3 *Digital elevation modelling*

A 50-metre grid digital elevation model (DEM) was generated using the ANUDEM program, which fitted a spline to a 5-metre contoured elevation surface (1:50 000 digital topographic mapping) interspersed with spot heights and tied to the 1:50 000 drainage network and coastline data provided by the Defence Imagery and Geospatial Organisation. Positional accuracy of the raw data was consistent with 1:500 000 topographic mapping standards (±0.5 m horizontal accuracy and ±2 m vertical accuracy. The raw drainage data was pre-processed to identify flow direction and discontinuous drainage was edited to an integrated network to ensure that the statistically fitted surface respected the natural drainage system (Hutchinson, 1997).

3.2.4 *Digital terrain analysis*

Slope, steady state wetness index and a sediment–erosion index were derived using the TAPESG program to characterise the spatial distribution of a steady state soil wetness index and erosion–deposition processes (Wilson and Gallant, 2000b).

Table 3-1 Land unit, vegetation and soil classes

Mapped Land Unit	Land unit code	Landform element	Mapped Vegetation Unit	Described Vegetation community	Soil family
(a): 85	Gec084	foot slopes	(a): Eucalypt forest	1a: E. miniata, E. tetrodonta & E. nesophila open-forest with Chrysopogon fallax grassland understorey	K11 Koolpinyah: deep, gravelly, imperfectly drained, yellow sandy loam over sandy clay loam
	Gaq085	hill crests	(a): Eucalypt forest	1a: E. miniata, E. tetrodonta & E. nesophila open-forest	K9 Hotham: deep, gravelly, well drained, red, sandy loam over sandy clay
	Gfc085	slopes	(a): Eucalypt forest	1a: E. miniata, E. tetrodonta & E. nesophila open-forest	K9 Hotham: deep, gravelly, well drained, red, sandy loam over sandy clay
	Udf085	slopes	(a): Eucalypt forest	1a: E. miniata, E. tetrodonta & E. nesophila open-forest	K9 Hotham: deep, gravelly, well drained, red, sandy loam over sandy clay
(b): 86	Gaq086	summit surfaces	(a): Eucalypt forest	1b: E. miniata & E. tetrodonta open forest/woodland	K8 Berrimah: very deep, well drained, red, sandy loam over acidic, sandy clay
	Laq086	summit surfaces	(a): Eucalypt forest	1b: E. miniata & E. tetrodonta open forest/woodland	K7 Berrimah: deep, slightly or non-gravelly, well drained, red, sandy loam over sandy clay loam
(c): 88	Lfc087	plains	(a): Eucalypt forest	1b: E. miniata & E. tetrodonta open forest/woodland	K8 Berrimah: very deep, well drained, red, sandy loam over acidic, sandy clay
	Gec088	fan	(a): Eucalypt forest	1b: E. miniata & E. tetrodonta open forest/woodland	T6 Cockatoo: deep, non-gravelly, well drained, red sandy
	Uec088	fan	(a): Eucalypt forest	1b: E. miniata & E. tetrodonta open forest/woodland	T6 Cockatoo: deep, non-gravelly, well drained, red sandy soils

The steady state wetness index (ω) is used as a surrogate for sub-surface flow in a humid environment (Troch et al., 1993) and can be defined as:

$$\omega = \ln\left(\frac{A_s}{\tan\beta}\right) \qquad \text{(equation 3-1)}$$

, where A_s is the specific catchment area (catchment area draining across a unit width of contour; m^2/m), and *b* is the slope angle (in degrees). This index is similar to the specific catchment area, or upslope area per width of contour used widely in soil property mapping from digital terrain data (Wilson and Gallant, 2000b). The wetness index was divided into six quantiles to represent the distribution of runon and runoff elements in the landscape (Figure 3-2). The visual assessment of the distribution of wetness index quantiles in the landscape indicated a close association between surface drainage and hill slope topography.

Figure 3-2 Static soil wetness index derived from 50 m DEM and classified into six quantiles

The erosion-deposition index (*ΔT*$_c$), a dimensionless sediment transport capacity, was also computed using TAPESG as a non-linear function of specific discharge and slope, expressed as:

$$\Delta T_{cj} = \phi\left[A_{sj-}^{m}\left(\sin\beta_{j-}\right)^{n} - A_{sj}^{m}\left(\sin\beta_{j}\right)^{n}\right] \qquad \text{(equation 3-2)}$$

,where β is the slope (in degrees) and A_s is the specific catchment area, or the drainage area per unit width orthogonal to a flow line (m^2/m). *ΔT*$_c$ represents the change in sediment transport capacity across a grid cell, and can be used as a measure of the erosion or deposition potential in each grid cell (Wilson and Gallant, 2000a). Uniform rainfall excess conditions were assumed.

A clay index calculated as a ratio of LANDSAT TM bands 5 and 7 was used to assess the variation in lithology (Laffan and Lees, 2004). Vegetation cover was assessed from the vegetation layer in the 1:50 000 digital topographic map. The mapped classes were: medium density woodland, scattered woodland, saline coastal flats and marine swamps, pine plantation, dense vegetation, mangrove, inland water, inter-tidal flat foreshore, bare areas, lakes, dunes and cheniers.

3.2.5 *Survey design*

The stratified survey design of 129 sites included 4 replicate observations in each of 6 elements of 5 landscape patterns. Variation in landform element and landform pattern (Speight, 2009) was pre-mapped using classification of the static soil wetness index at fine (50-metre grid) and coarse levels (100 hectare) of resolution. To represent landform pattern, a network of regular 100 hectare hexagons was overlaid on the wetness class grid and the areas of each wetness index class were cross tabulated for each hexagon. Five landform classes representing coarse (100 hectare resolution) landform pattern variation were created using an agglomerative, *k-means* classification, ALOC, in the PATN software program (Belbin, 1987). The landscape pattern classification is depicted in Figure 3-3 and the survey site coverage in Figure 3-4.

Prior to field work, replicate survey site locations were selected in elements identified with the six wetness index classes (landform elements) in each landscape pattern (five classes). Access considerations, element size and overall coverage of the islands also influenced where survey sites were placed.

3.2.6 *Survey analysis*

DTA models were fitted using the See5 program (http://www.rulequest.com/). DTA models were fitted to predict land units, and their component soil family and vegetation classes from latitude, longitude, elevation, slope, wetness index, wetness class, landform class, erosion–deposition index, vegetation cover, and clay index. To restrict the size of the tree that was generated, at least four observations were required for each DTA leaf (final node). DTA models were developed on a training data set (129 sites in the recent survey) and tested on historical survey data (108 sites) that had been collected in the south east of Bathurst Island and the western end of Melville Island in the 1970’s to support forestry development projects. The quality of the resulting maps was assessed from the frequency of correct and incorrect allocations to classes.

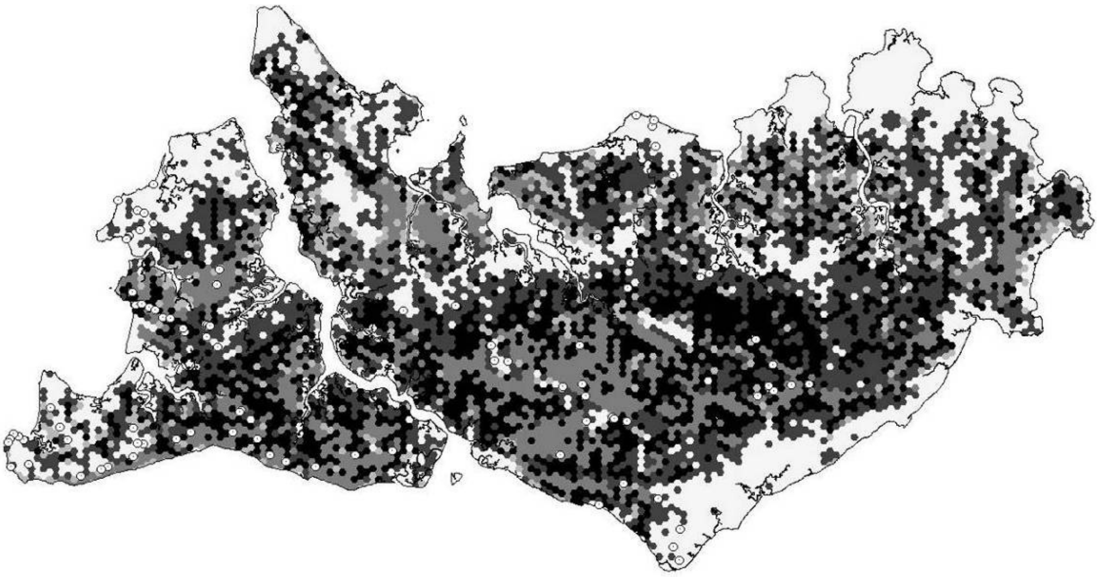


Figure 3-3 Five landform pattern classes over the Tiwi Islands indicated by shaded 100 hectare hexagons

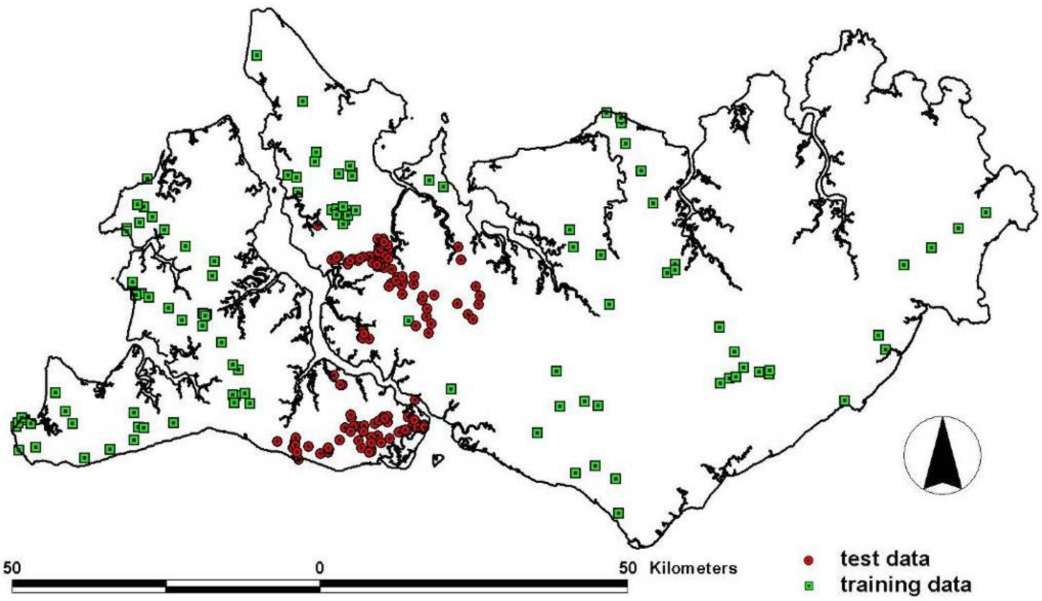


Figure 3-4 Survey site location showing training and test sites

3.3 Results

A selection of the resulting land unit classes, vegetation classes and soil classes was presented in Table 3-1 along with the more detailed classifications that were extracted from historical survey legends. The 9 aggregated classes, extracted from 27 land unit classes obtained from historical survey reports, exhibited a reasonable mapping result (Tables 3-2 to 3-4). A simplified vegetation classification (7 classes instead of 26) was used to achieve generalization in this case.

The tabulated results for training and test data sets for selecting land unit classes (Table 3-2), soil classes (Table 3-3) and vegetation classes (Table 3-4) all showed considerable increase in prediction errors between the training data and the test data. This is because the test data selected from historical surveys were less accurately located (± 100 metres compared with ± 10 metres) and were concentrated in a part of the islands not well represented in the training data set, thus adding to errors in prediction. For example, in the test survey area, land unit 109 comprised undulating landscapes with underlying cretaceous clay sediments, which were misclassified as the more extensive land unit 88, a similar landscape but underlain with unconsolidated sandstone and sand colluvium. These misclassifications were manually corrected in the map so as to agree with site observations.

Deep Ustoxs that occur on flat to undulating terrain occur in land units 85, 86, 88 and 90 showed surface horizon texture variation from sandy (Kiluppa) to sandy loam (Berrimah) and gravel content variation from non-gravelly (Berrimah) to gravelly (Hotham). The Hotham, Berrimah and Kiluppa soils mapped relatively consistently with most of the miss-classifications arising from confusion between each of the land units listed above.

The soil mapping result is depicted in Plate 3-1. The red and orange colours indicate extensive areas of Ustoxs. The soil prediction model could not distinguish between gravelly and non-gravelly phases of the soils (described as Hotham and Berrimah soil families in the previous land unit mapping). The grey colours represent Ustults and the blue colours Aquic soils. The mapping model did not distinguish Ustox formed in residual sandstone from Ustults formed in cretaceous clay sediments in a reliable way. These inaccuracies in the mapping were ascribed to shortcomings in the ability to discern lithological boundaries using the clay index imagery or regional geological mapping. Woodland vegetation with a thick understorey and grass cover would have obscured soil spectral signature where land had not been recently burnt.

Table 3-2 Land unit classification tree evaluation

Training data

Total	**Errors**	**(a)**	**(b)**	**(c)**	**(d)**	**(e)**	**(f)**	**(g)**	**(h)**	**(i)**	**<- Land unit classified as**
33	3	23	1	2			5		2		(a): 85
11	2	2	7		1		1				(b): 86
25	1	1		22			1		1		(c): 88
5	1	1			4						(d): 90
0	0										(e): 95
18	1				1		17				(f): 105
3	0							3			(g): 112
23	8		2	1	1		2	2	15		(h): 109
2	0									2	(i): 114
120	**16**										**Error rate = 13%**

Test data

Total	**Errors**	**(a)**	**(b)**	**(c)**	**(d)**	**(e)**	**(f)**	**(g)**	**(h)**	**(i)**	**<- Land unit classified as**
15	9	6		3	3		1	1	1		(a): 85
8	7	2	1		2		1	2			(b): 86
35	25	6		10	8		4	5	2		(c): 88
0	0										(d): 90
12	12	1		1	1		2	2	5		(e): 95
23	21	6	1	8	1		2	3	2		(f): 105
15	10	1					1	5	8		(g): 112
0	0										(h): 109
0	0										(i): 114
108	**84**										**Error rate = 77%**

The error rate for vegetation type prediction was less (36%, Table 3-4) that land unit (77%, Table 3-2) or soil type (73%, Table 3-3). Most of the generalisation in the land unit legend that was compiled from historical surveys (27 land units to 9) was achieved by aggregating the component vegetation classification from 26 vegetation types to 7 plant community types. There is as a result less detailed specification included in the vegetation classification than for either the soil or land units. The plant community types that were used in the analysis conformed broadly to the vegetation layer in the 1:50 000 digital topographic data.

Table 3-3 Soil classification tree evaluation

Training data

Total	**Errors**	**(a)**	**(b)**	**(c)**	**(d)**	**(e)**	**(f)**	**(g)**	**(h)**	**(i)**	**(j)**	**<- Soil family classified as**
34	11	22	9	1	1					1		(a): Hotham
44	9	4	35	1	4							(b): Berrimah
9	1		1	8								(c): Mirrikau
9	3		1		6		2					(d): Ramil
2	2	2										(e): Irgil
6	3		2				3			1		(f): Kiluppa
3	0							3				(g): Wangitti
2	2							1		1		(h): Rinnamatta
10	5	1	2	1				1		5		(i): Marrakai
1	1									1		(j): Koolpinyah
120	**37**	**32**	**50**	**13**	**13**	**0**	**5**	**7**	**0**	**9**	**0**	**Error rate = 31%**

Test data

Total	**Errors**	**(a)**	**(b)**	**(c)**	**(d)**	**(e)**	**(f)**	**(g)**	**(h)**	**(i)**	**(j)**	**<- Soil family classified as**
11	7	3	3	4						1		(a): Hotham
28	10	7	14	1	2			2		2		(b): Berrimah
13	12	4	1	1				0		7		(c): Mirrikau
14	13	5	3	1	1			1		3		(d): Ramil
2	2	1	1					0				(e): Irgil
14	13	6	4	1	1		1	0		1		(f): Kiluppa
1	0							1				(g): Wangitti
1	2	1						0				(h): Rinnamatta
13	9	6	2					1		4		(i): Marrakai
11	11	6	4					1				(j): Koolpinyah
108	**79**	**13**	**32**	**8**	**4**	**0**	**1**	**6**	**0**	**4**	**0**	**Error rate = 73%**

3.4 Discussion and conclusions

Land unit surveys have a tendency to over–specify land unit classes in relation to the field survey data sets, even in low relief landscapes with fairly monotonous lithology such as the Tiwi Islands. This has also been identified as a map quality issue in Australian land resources mapping (McKenzie and Austin, 1993). Consequently, the expectations of end users of land unit mapping need to be attuned to the need for further field survey work to meet specific information requirements.

The traditional land resources survey method produces a knowledge base that is expressed in soil map legends and can be formalized in terms of sets of mapping rules based on distributed environmental attributes (Bui, 2004). DTA produces sets of rules that are analogous to traditional survey methods. However, with this digital soil mapping technique it is possible to upgrade the mapping, or knowledge base, in a way that is consistent with current soil and land survey practice as more field survey data become available. In the Tiwi Islands mapping, additional surveys were environmentally stratified according to landform pattern and landform element classes, in the same way as traditional soil resources surveys are designed (Speight, 2009). The main innovation is in substituting digital terrain analysis for physiographic interpretation from aerial photography.

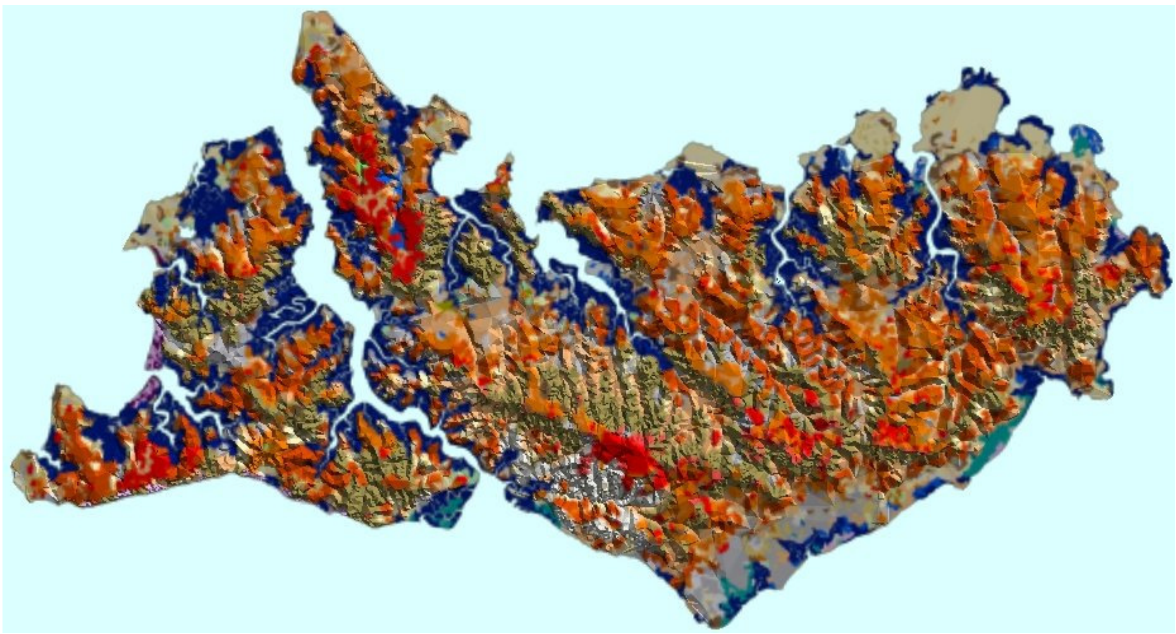


Plate 3-1 Soil mapping – reddish colours indicating Oxisols, grey Ultisols and blues Aquic soils

A general finding was that there is less error in the test and training data sets for vegetation class predictions than for soil and land unit class predictions (Tables 3-2 and 3-3). Plant communities tend to compensate for edaphic variation by changing their topographic positions (Guisan and Zimmermann, 2000) and the responses to landscape variation may not be consistent with soil patterns. Reducing the vegetation classes from 26 to 7 was made to rationalise land unit definitions. The general classification of vegetation communities that was produced is clearly more amenable to prediction from the data that was available. Some factor information could be improved as new environmental layers for geology based on gamma-radiometric mapping have become available that reveals the distribution of bedrock and regolith materials with sufficient detail to show variations at local scales (Minty et al., 2009).

Table 3-4 Vegetation classification tree evaluation

Training data

Total	**Errors**	**(a)**	**(b)**	**(c)**	**(d)**	**(e)**	**(f)**	**(g)**	**(h)**	**(i)**	**(j)**	**<- Vegetation classified as**
92	2	90			2							(a): Eucalypt forest
7	6	2	1	1	3							(b): Melaleuca forest
6	2	2		4								(c): Mangrove forest
10	0				10							(d): Vine Forest
0	0											(e): Sparse woodland
4	0						4					(f): Grassland/sedge
1	1	1										(g): Coastal woodland
120	**11**											**Error rate = 9%**

Test data

Total	**Errors**	**(a)**	**(b)**	**(c)**	**(d)**	**(e)**	**(f)**	**(g)**	**<- Vegetation classified as**
81	14	67		3	11				(a): Eucalypt forest
14	14	11		1	2				(b): Melaleuca forest
0	0								(c): Mangrove forest
9	7	7			2				(d): Vine Forest
3	3	2			1				(e): Sparse woodland
0	0								(f): Grassland/sedge
1	1	1							(g): Beaches/cheniers
108	**39**								**Error rate = 36%**

The simplification of the land unit classification that were needed to apply digital soil mapping techniques to land unit mapping on the Tiwi Islands demonstrate the level of inference that can be drawn reliably from land unit survey information. The mapping legend derived in this study is more general than the land unit classification obtained from legacy soil surveys. It is likely that land unit map legends contain inaccuracies, often associated with duplication, derived from mapping by different soil surveyors and tend to overstate the environmental inferences that can be made, particularly in regard to vegetation communities. However, a digital soil mapping approach to updating legacy soil survey data and mapping would appear to have many advantages in the digital age not the least being the ability to integrate new knowledge with the knowledge accumulated over the years. Our test data set comprised site information extracted from legacy soil surveys conducted 20 to 30 years ago. The inaccurate location of legacy soil survey data is an issue when this data is used to develop and test predictive models (Bui and Moran, 2001).

Issues associated with the inaccuracy of legacy land resource survey data and the complexity of the land unit model limit its value and reliability for selecting natural analogue areas to guide landscape restoration and conservation strategies. Consequently, test area surveys are needed to check and develop environmental inferences that will support reliable interpolation of soil and vegetation properties across the landscape. Significant landform effects on vegetation communities were identified using a DSM methodology based on the land unit concept. However, a high level of survey support will be needed to predict species distributions. Focussing this survey support in landscapes that are analogous to the restoration goals of mine rehabilitation rather than distributing them to assess the broad range of regional environmental variation will simplify the ecological modelling and improve the reliability of the predictions that are made.

Chapter 4 Natural analogue landform selection for planning mine landscape restoration

4.1 Introduction

It was recognised from the regional mapping of land units using digital soil mapping methods reported in Chapter 3 that focussing environmental surveys on analogous areas of natural landscapes was likely to produce more reliable statistical predictions of vegetation pattern. Also, it was recognised that the reliability of the inferences that could be made from regional ecological mapping such as land units was low, due to the lack of sufficient survey support for a complex conceptual model. Consequently, staged ecosystem survey approach that has a stronger basis in ecological theory is now developed. A landscape classification and natural analogue selection method is developed in this chapter that applies *extent* and *grain size* concepts used in habitat classification in landscape ecology. This approach relates the *extent* of mining disturbance and the *grain size* of hillslope environmental processes to the type and range of habitats that open cast mining impacts upon at a particular site. The approach is staged from broad reconnaissance in this chapter to detailed survey (reported in Chapter 6) to focus survey efforts in order to support accurate inference.

Environmental investigations at the Ranger case study site are presented in Chapters 4 to7 that support an ecological design approach using natural analogues. This approach could be extended where a restoration approach to rehabilitation is required. In Chapter 5, the soil profile in a constructed waste rock landform at Ranger uranium mine is characterised and water and solute balance investigated and compared with published information on natural soils.

In Chapter 6, an ecological landform design based on the geomorphic properties of a natural analogue area is independently assessed using deterministic modelling methods and a detailed, second stage investigation of soil and plant community patterns that leads directly on from Chapter 4 is reported on. In Chapter 7, species distribution modelling that used survey support from Chapter 6 is undertaken so as to test whether mine landform design could be expected to create similar vegetation patterns found in an analogous natural landscape. Because of the low level of predictability of vegetation and soils reported in

Chapter 3 from land unit mapping, the work reported in Chapter 7 was the key development from this thesis.

In the current chapter, natural analogue areas and environmental correlation of landform with plant community and soil morphological properties is investigated using digital terrain analysis and patch analysis methods that reflect environmental *grain size* and *extent* properties of habitats in hill slope environments.

4.1.1 *Landform scale*

Scale has been described as the fundamental conceptual problem in ecology (Levin, 1992) that is reflected in the spatial dependence of vegetation pattern on landform (Miller et al., 2007). Since digital raster analysis methods were introduced to landscape ecology, scale has been conceptualised in terms of *grain size* and *extent* (Wu et al., 2002) rather than as a map scale. A similar conceptual change, albeit with different terminology, is repeated in other landscape sciences such as digital soil mapping (McBratney et al., 2003). This has happened in response to high resolution raster, environmental data becoming available over extensive areas as an improvement on thematic information.

However, in spite of this conceptual development in gridded landscape models, analogue or reference ecosystem selection for mine rehabilitation is typically qualitative and subjective, and based on thematic land unit mapping (Zonneveld, 1989) or other regional environmental information (Hobbs and McIntyre, 2005). For example, natural analogue selection to set mine rehabilitation objectives at the Ranger case study site initially focussed on similarities in surface geology (Uren, 1992), geochemical factors (Riley and Rich, 1998) and rocky habitats represented in land units with regional context (Brennan, 2005). This has happened although local gradients in plant available water supply and soil drainage are known to affect short range ecosystem variation (Williams et al., 1996) and would be better represented in terms of *grain size* and *extent* properties of analogous natural landscapes.

In the context of Ranger mine, the environmental range that was sampled by Brennan (2005) to identify analogues was limited to well drained eucalypt woodland habitats 100 hectares in extent on land units with rocky substrates. The environmental properties associated with landform extent and relief at vegetation plot survey sites was overlooked and hill slope environmental gradients in water supply and drainage were not considered in the survey design. The very detailed vegetation information produced from plot studies without a detailed landscape context comparable to the mine landscape may lead to unforeseeable outcomes in the rehabilitation design.

For instance, where an analogue selection approach based on land units was used at the now closed Nabarlek uranium mine (150 kilometres to the north east of Ranger mine), Klessa (2000) reported that native woodland failed to establish on the reconstructed landscape. Unexpected ecological outcomes such as this are common on mined land restoration projects (Nicolau, 2003) and may be due to neglecting the *extent* of mining impacts in selecting the analogue area(s) and the *grain size* of hillslope environmental processes that affect the vegetation patterns to be restored, and failing to design landforms that accommodate this pattern.

The landscape model used affects the analogue selection process (Fischer et al., 2004; Lindenmayer et al., 2008). However, the topographic detail relevant to site restorations has been difficult to identify and factor into environmental restoration plans for a number of reasons outlined in Hobbs & McIntyre (2005). Landscape environmental variation is complex and crosses different dimensions that affect species distributions at local and broad regional landform scales. For instance, hillslope curvature affects concentrated water flow and distribution locally, while aspect and landform relief influence the radiant energy environment affecting vegetation growth and potential energy driving geomorphic processes over broader landform patterns, lands systems and bioregions (Franklin et al., 2000; Gessler et al., 2000; Nelson et al., 2007; Urban et al., 2002; Wilson and Gallant, 2000b). To address this landscape scale is considered in standard methods of soil and landscape assessment (Speight, 2009) and the local and regional factors in landscape classification and habitat variation are reflected in hierarchical habitat classification systems (Cushman and McGarigal, 2002; Mackey et al., 2008).

The natural ecosystem variation that results from landform variation is typically patchy, combining both discrete and continuous variation resulting from endogenous ecological processes as well as environmental heterogeneity related to landscape. The stochastic nature of ecological processes requires a statistical approach to survey design, comprising replication and environmental stratification across local landscapes. However, landscape characteristics need to be determined in an ecologically meaningful and methodologically sound way (Wagner and Fortin, 2005). To accomplish this, landscape ecologists have used different patch based metrics depending on landscape scale (extent and grain size) to develop environmental associations to characterise habitat variation (Neel et al., 2004; Wu and Levin, 1997; Wu et al., 2002).

A key concern in applying patch analysis methods is that the grain size and patch metrics need to link with underlying hillslope environmental processes (Bowman and Minchin, 1987; Ludwig et al., 2002; Ludwig et al., 2000b; Wu and Archer, 2005) to be of practical value for landscape restoration (Li and Wu, 2004). Also a level, or levels, of regionalisation is required to address questions of landscape utility, i.e. in the case of this study what part, or parts, of the natural landscape reflect the habitat type range that applies to restoring a mined, waste rock landscape. However, it must be recognised that landscapes function in an environmentally integrated way. Consequently, the analogue selection process that is most likely to reflect this will include an interconnected range of habitats found in natural hillslope environments that match the extent of mined landscape restoration task.

When conceptual ecological mapping frameworks (such as land units) are used as a regionalising factor to landform in habitat analysis, hillslope topographic details are overlooked and subjective and sometimes arbitrary boundaries are introduced (see Chapter 3). Alternatively, grid overlays have been used to represent the *extent* of landscape utility with respect to foraging animal range (Rempel and Kaufmann) or soil property mapping (Hollingsworth et al., 2007; McBratney et al., 2003) in quantitative landscape analysis.

Selecting a particular landform extent, as well as environmentally appropriate patch metrics, is a key consideration in answering the ecological questions being asked in mine restoration based on natural analogues. It is proposed that localised mining activity is analogous to foraging animal range in the way that the extent of landscape utility is defined. However, the habitat concepts being applied in mined land restoration may not be linked to the range of any particular species, but rather the distribution of plant communities in local hillslope environments. These habitat concepts need to be defined and this is done in the next section.

4.1.2 *Habitat concepts*

There are two meanings of the term habitat in use, one that is organism specific and another that is land based and it is important to distinguish how the term is being applied

(Miller and Hobbs, 2007). In the first instance, habitat is typically defined as an area containing the particular combination of resources and environmental conditions that are required by individuals of a given species or group of species to carry out life processes. Odum (1971) provided an alternative, broader definition of habitat as "*The area or environment where an organism or ecological community normally lives or occurs*". In this case, an ecosystem is a community of interacting organisms and their physical environment while habitat refers specifically to the physical environment in which they live.

An extension of the second usage of the term, the concept of *habitat types,* provides a general framework for understanding the ecological processes that operate at landscape scales. The composition of habitat types in a landscape and the physiognomic or spatial arrangement of those habitats are the two essential features that are required in an ecological description of any landscape (Dunning et al., 1992). Since digital terrain analysis underpins current methods in species and plant community distribution modelling (Austin, 2007; Guisan and Zimmermann, 2000), extending the concept of habitat type in this thesis to include patterns derived from digital terrain variables that represent environmental variation across landscapes (Wilson and Gallant, 2000a) is reasonable. Using digital terrain analysis to develop habitat type classifications advances the procedure of identifying natural analogue landscapes to represent restoration objectives for mine landscape reconstruction.

4.1.3 *Patch analysis*

Multivariate classification of fine grained environmental surfaces is the basis of habitat type classification used in this approach (McGarigal et al., 2009). The approach termed patch analysis averages variation in habitat type, described by cluster analysis of continuous fine grained environmental surfaces, into patches and derives ecologically meaningful patch metrics, according to the extent of a dependant variable, or object of interest across an extensive landscape (Wagner and Fortin, 2005). In this method, a hexagonal grid, or tessellation, is overlaid on the patch mosaic to represent the extent of local landscape structure that can be related to habitat use (Rempel and Kaufmann). Other grid shapes could be used, but hexagonal grids stack well, project onto the spherical surface of the earth with minimal distortion and are more representative of average conditions (because the perimeter is more or less equidistant from the centre) than rectangular grids for spatial averaging (Sahr et al., 2003). The extent of the grid elements used varies with the context of the study in relation to species foraging range (Rempel and Kushneriuk, 2003; Rempel and Kaufmann, 2003).

Patch metrics based on vegetation pattern and land use have been criticised as lacking ecological significance, which limits the understanding of underlying environmental process that can be derived (Li and Wu, 2004; Miller and Hobbs, 2007). Surface metrics derived from terrain and image analysis have been used that address this shortcoming, (Cushman and McGarigal, 2004a; Cushman and McGarigal, 2004b; McGarigal et al., 2009; Neel et al., 2004). Patch area metrics have been found to be relatively robust across a range of landscape structural extents (McGarigal et al., 2009; Wagner and Fortin, 2005). Also, landscape metrics related to class area may have strong relationships with species dispersal (Neel et al., 2004). Although, the links between the landscape gradients being measured by surface metrics and ecological consequences still need to be understood.

Similarity (or dissimilarity) may be assessed at sequentially higher levels in the patch analysis process — starting with classification of continuous fine grained environmental surface properties into a range of habitat types, followed by classifying landscape structural extents on the basis of contained habitat types so as to select natural landscapes analogous to an object of interest. Ecological surveys designed to assess the environmental factors that drive landscape environmental variation, represented by surface metrics, are required (Li and Wu, 2004). However, landscapes of similar extent with similar patch metrics are assumed to represent a similar habitat range. Consequently, the analogue selection process leads into reconnaissance ecological surveys that are structured around inferred environmental variations and are aimed at confirming the selection process and identifying environmental processes that affect species composition and community variation in analogue areas. This approach assumes that the habitat type classification can be related to measured ecosystem variation and that the direct impact from a mined landscape is analogous to foraging animal range.

4.1.4 *Environmental Factors*

In hillslope situations, where physiography exerts strong control over vegetation patterns (Bowman and Minchin, 1987; Ludwig et al., 2002; Wu and Archer, 2005) terrain properties connected to environmental processes such as water balance, drainage, sedimentation and erosion are correlated with plant community variation (Franklin et al., 2000). Fire frequency and intensity can also determine suitable conditions for a given species in seasonally dry savannah woodlands such as those that occur across much of northern Australia (Williams et al., 2003b). Although Bowman & Minchin (1987) concluded that the edaphically determined vegetation controlled fire regime rather than the converse. This study identified strong topographic control between *Melaleuca* communities from those dominated by eucalypts in a very gently undulating landscape with subtle topographic variation.

Bowman & Minchin (1987) used indirect gradient analysis of vegetation survey data in the Australian monsoon tropics near Darwin to reveal strong relationships between the vegetation pattern and edaphic variables that reflected two aspects of the moisture regime – water availability during the dry season and the degree of inundation during the wet season. Moisture availability was determined by topographic position, through its relationship with soil texture and water table depth. Poor drainage during the wet season separated *Melaleuca* communities in drainage lines from eucalypt communities on slopes, while shrubby and grassy open forests were differentiated by the intensity of the winter drought. There is an alternative point of view that fire can affect vegetation pattern rather than the reverse (Fensham, 1990; Gill et al., 2003; Grant and Loneragan, 2001). Also gradient analysis methods that directly constrain the ordination by environmental factors (Leps and Smilauer, 2003) are popular for environmental analysis of species distribution, although underlying assumptions of linearity are not robust and can artificially distort the results (McCune and Grace, 2002).

4.1.5 *Statistical methods*

4.1.5.1 Cluster analysis

Cluster analysis for groupings of similar sample sites identifies discontinuity and can be used in combination with gradient analysis and network analysis (connecting groups based on similarity) for comprehensive multivariate evaluation of ecological pattern in field survey data (Margules and Austin, 1991). Clustering can imply a simple partitioning (a set of non-overlapping clusters) or a more complex hierarchical pattern. A general form of hierarchical clustering based on the Bray Curtis measure of dissimilarity (Bray and Curtis, 1957) and UPGMA (group average) called flexible UPGMA allows the variation of a dilation parameter, β, that influences success in cluster recovery. Flexible UPGMA with a small negative β value has been shown to recover cluster structure well from noisy data (Belbin and McDonald, 1993).

While some hierarchical algorithms may be robust for ecological survey data, all of these methods are limited in terms of the number of sites that can be analysed, because a matrix of all pair-wise dissimilarities among sites must be calculated. For large data sets (1000 objects or more) such as remote sensing, extensive digital elevation models and their derivatives, non-hierarchical methods that search directly for a fixed number of groups, e.g., *k-means* clustering are more effective and reliable (Belbin, 1987; Bui and Moran, 2001).

4.1.5.2 Gradient analysis

Several environmental factors can determine species and plant community distribution in any particular situation and either direct (Austin, 2002a; Oksanen and Minchin, 2002) or indirect (Margules and Austin, 1991; McCune and Grace, 2002) gradient analysis methods have been developed to investigate relationships in community and landscape ecology. With direct gradient analysis sample sites are positioned according to measured environmental factors to elucidate species response to specific gradients of interest. Univariate response can be measured by regression (Austin, 2002b; Guisan et al., 1999) and multivariate responses are measured in terms of the statistical significance of ordination scores and correlations between axis scores and external environmental variables (Margules and Austin, 1991; McCune and Grace, 2002).

In contrast, indirect gradient analysis that ordinates survey sites according to covariance and association among species, with the benefit of reflecting the environmental space the way the biotic community interprets, is widely accepted (Austin, 1987; Austin, 2002a; Belbin, 1995; Ben Wu and Archer, 2005; Bio et al., 1998; Bowman and Minchin, 1987; Mackey et al., 2008; Margules et al., 1987; McCune and Grace, 2002; Minchin, 1987a). Ordination is first performed on just the community data, and then secondarily related to the ordination of the environmental variables, followed by an independent assessment of the importance of the environmental variables. Although quantitative relationships are not produced, indirect gradient analysis using nonmetric multidimensional scaling (NMS) it considered to accurately represent species space (Margules and Austin, 1991; McCune and Grace, 2002).

Indirect analysis of multivariate gradients is, however, difficult to visualise. Consequently, hybrid methods (Leps and Smilauer, 2003) particularly canonical correspondence analysis (CCA) that constrain the ordination in species space by correlations with environmental variables are popular in community ecology (Grant and Loneragan, 2001; Rayfield et al., 2005; ter Braak, 1996; Zhu et al., 2005).

The assumptions of simple linear relationships between species and environmental space used in hybrid analysis are often questioned and the methods themselves can create distortion (McCune and Grace, 2002; Minchin, 1987a). While detrending can be used to remove distortions (Enright et al., 1994) this may also confound clear interpretation (McCune and Grace, 2002). Hybrid ordination methods also ignore community structure that is unrelated to the environmental variables and the results can be unreliable as a consequence (Guisan et al., 1999). In general, direct gradient methods provide better species specific-models; hybrid methods such as CCA may provide a broader overview of multiple species, while indirect gradient analysis best represents species space and subjectively assesses environmental gradients.

Gradient analysis is highly sensitive to survey design (Austin and Meyers, 1996). Stratification that takes into account both the *grain* and *extent* of the pattern of patches across the landscape and replicated sampling of environmental factors across the habitat range of interest is needed to support statistical evaluation of environmental variation (Cushman and McGarigal, 2002; Fischer et al., 2004; McKenzie and Ryan, 1999; McKenzie et al., 1989).

4.1.6 *The way forward from thematic landscape models*

Understanding the landscape characteristics that underlie environmental processes and determine environmental heterogeneity in natural landscapes underlies a more reasoned approach to analogue area selection to for landscape restoration (Holl et al., 2003; Palik et al., 2000). Although, species responses depend on the resolution and choice of response variables and patch arrangement in the landscape (Ben Wu and Archer, 2005; Ludwig et al., 1999) quantitative landscape models have been used to map soils (McBratney et al., 2003), habitats (Austin, 2002b; Li et al., 2001) and species distribution (Guisan and Zimmermann, 2000) that rely on continuous, high resolution environmental information from remote sensing and digital terrain analysis (Wilson and Gallant, 2000b).

4.2 Methods

A first stage, reconnaissance investigation is presented designed to identify appropriate natural analogue landforms for planning landscape reconstruction at the Ranger uranium mine case study site in the Top End of the Northern Territory of Australia.

4.2.1 *Study area location*

Ranger uranium mine is located in the Ranger project area (79 km^2 in area) at 12^0 41' Lat 132^0 55' Long, 250 km east of Darwin and is surrounded by 19 803 km^2 of coastal plain and escarpment conserved in the world heritage listed Kakadu National Park (Figure 4-1).

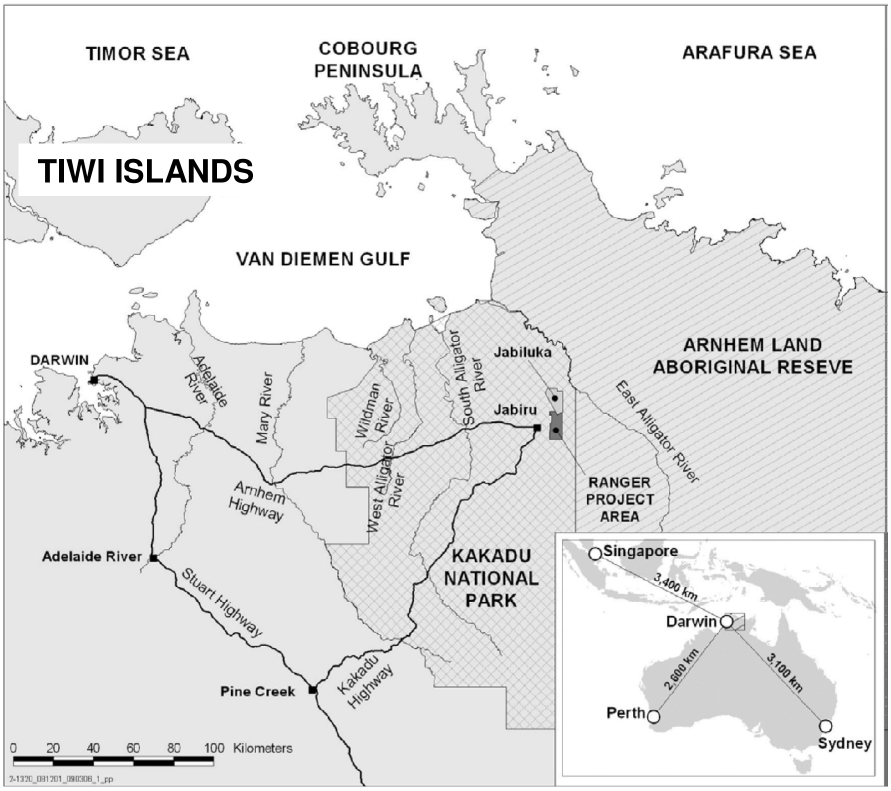


Figure 4-1 Ranger location map

The climate of the region is monsoonal with a wet season from October to March during which 95% of the annual rainfall occurs (mean 1700 mm), followed by a prolonged dry season characterised by little or no rainfall (Heerdegen and Cook, 1999; Taylor and Tulloch, 1985). Low plant available water supply and impaired soil drainage affect short range ecosystem variation in a local climatic environment that is marked by seasonal drought (Williams et al., 1996). The climate is similar to that of the Tiwi Islands described in Chapter 3.

4.2.2 *Vegetation*

Savanna communities dominate the region. A discontinuous tree canopy layer with a seasonal understorey of grasses characterises these communities. *Eucalyptus miniata-E. tetrodonta* savannas form the framework within which other important vegetation communities occur (Schodde et al., 1987). Savanna vegetation type is associated with lateritic peneplains in undulating terrain where soils are typically deep, well-drained and light textured. In the vicinity of Ranger Mine, this vegetation occurs over deeply weathered metamorphic rocks and granite. On rises and upper slopes where soils are shallow and more gravely, rusty-barked bloodwood (*Eucalyptus bleeseri*) is a major component. On gentle slopes in the undulating lowlands over metamorphic rocks, granite and dolerite *Eucalyptus tectifica- E. latifolia* savannas are the most widespread vegetation, except on very poorly drained soils. Soils tend to be heavier textured loams and clay loams, shallower, and less well drained.

4.2.3 *Geology, geomorphology and soils*

Kombolgie Formation Sandstone, Nanambu Formation gneisses and schists, and Cahill Formation schists and quartzites crop out in the area (Needham, 1988). Extensive coastal alluvial sediments are associated with a backswamp of the East Alligator River (the Magela Plain). Freshwater sediments are associated with the tributary creeks, and sandy, colluvial sheet wash fans radiate out from the base of the Arnhem Plateau and the Jabiluka outlier in the north, east and south and cover low lying parts of the landscape. A lateritised surface, known as the Koolpinyah Surface (Nott, 1994b) extends to the south of the Jabiluka outlier of the Arnhem Plateau, and relatively small areas of ferruginized sandstone occur on the outlier itself.

The Koolpinyah Surface can have a thick, weathered (ferruginized) zone that may extend up to 60 m into the bedrock below the present soil cover. The unit includes gradational and uniform Red and Yellow Earth soils on lateritic remnants with minor skeletal and gradational Yellow Earths on slopes. Dissected, lateritised remnants of the lowland

Koolpinyah Surface are also associated with scree slopes and outwash fans at the base of the plateau outlier between 20 and 50m elevation (Hollingsworth et al., 1998).

The Supervising Scientist for the Alligator Rivers Region (Wasson, 1992) undertook detailed geomorphic and pedological study of the Magela Creek Plain. They described acid peats overlying clays of estuarine origin and interpreted the evolution of the plain and the implications for contaminant management. White and Gigliotti (1982) and (Morley et al., 1985) measured detailed chemistry on profiles of selected soils in association with the Pancontinental proposal to mine the Jabiluka deposit. Their work also concentrated on the Magela floodplain. Hollingsworth *et.al.* (1998) described components of the Swift Creek alluvial system, and the Jabiluka system comprising the sheet wash fans from the plateau surfaces, and the peneplain of lateritised schists, or Koolpinyah system that extends south of the Jabiluka outlier, and the Magela Plain to the Ranger mine.

The soils in the Swift Creek and Jabiluka land systems are sandy throughout and show minimal soil profile development apart from organic matter accumulation at the surface and iron oxide accumulation in the sub-soil. The soils in the Koolpinyah land system have profiles with clay accumulation in the sub-soil and typically have an iron cemented (ferricrete) pan within 1 meter of the surface. The soils in the Magela land system are very poorly drained and have sulfidic and sulfuric horizons, commonly within 1 meter of the surface. The limited soil development is related to the siliceous nature of the parent materials, the lack of weatherable minerals and the erosivity and high drainage flux associated with the monsoonal rainfall environment (Duggan, 1991). Although it seems incongruous, the residual soils have formed in a stable environment over a long period of time (Nanson et al., 1993; Nott, 1994a).

4.2.4 *Fire environment*

Fire frequency and intensity is also an important environmental factor in these ecosystems. Eucalypt dominant woodlands tend to be fire resistant (Andersen et al., 2005) and frequent, high intensity fires produced by late dry season burning reduce biodiversity, principally in the understorey and groundcover (Bowman et al., 2003; Russell-Smith et al., 2003; Williams et al., 2003b). Consequently, differences in fire frequency and intensity between analogue areas and surrounding national park could produce differences in vegetation pattern that may be unrelated to topographic variation.

4.2.5 *Terrain analysis*

The process was used to construct the DEM of the Ranger site and it's surrounds was the same as that used for the Tiwi Islands study (described in Section 3.2.2), except that more accurate contour data (1 metre versus 5 metre) derived from corrected and contoured laser altimetry data was available from mapping contractors for ERA Ranger mine. This potentially improved the resolution of the DEM. A DEM extending northwards from the Ranger site in a swath approximately 6 kilometres wide and 25 kilometres long was created using ANUDEM 4.6 software (Hutchinson, 1997). Drainage lines mapped at 1:50 000 were overlaid with contour lines and spot heights from orthophoto mapping at 1: 10 000 standards of accuracy and a splined elevation surface was generated by ANUDEM that respected the natural surface drainage. The model included a topographically reconstructed mine landform that respected the original relief and drainage density. The model was checked for sinks, which were filled.

The higher resolution contour data and increased spatial resolution for the Ranger elevation data lead to a preliminary assessment of the optimum DEM resolution to be used in the analysis that follows. The effects of DEM resolution on the ability to make environmental inference has been studied elsewhere (Gallant and Hutchinson, 1997; McBratney, 2004; Wilson and Gallant, 2000b). The optimum resolution may vary with landscape relief to some extent (McKenzie et al., 2000) and can be estimated from the minimum root mean square (RMS) slope of all DEM points associated with the elevation data (Kienzle, 2004; Wilson and Gallant, 2000b). RMS analysis for the Ranger data indicated that the optimum DEM resolution was 10 metres in high relief plateau and escarpment landscapes and was something greater than this (approximately 30 metres) for low relief. The 20 metre grid resolution used to fit the terrain model was within the capability of the source data and able to resolve hill slope variation in low relief landscapes.

The suite of terrain attributes shown in Table 4-1 were generated using TAPESG (Gallant and Wilson, 1996) and have been hypothesised to be important determinants of the distribution of plants and ecological communities (Parker and Bendix, 1996). Newer terrain analysis software such as SAGA (Cimmery, 2007) provide graphical user interfaces, can generate wider range of secondary terrain attributes and are able to process laser altimetry (Lidar) data rather than contours (Ryan and Boyd, 2002). New terrain attributes have been specifically developed that describe flow patterns more accurately in low relief landscapes (Gallant and Dowling, 2003) but these are not critical for representing flow in hillslopes.

In TAPESG, different options are provided for determining flow direction (D8, Rho8, or Frho8), flow accumulation (D8, multiple flow paths, and Costa-Cabral' method), and slope gradient (D8 or finite difference). Other related programs in the bundle include DYNWET for calculating wetness index, EROS for soil loss and erosion and deposition potential, SRAD for estimating solar radiation. In summary, TAPESG is still one of the more versatile software packages available in terms of flow routing algorithms that can be used on different surfaces and the environmental attributes that are generated in hillslope environments are still valid (Nelson et al., 2007).

Table 4-1 Environmental attributes calculated by terrain analysis from DEM data

Attribute	**Module**	**Environmental interpretation**
primary terrain attribute		
elevation	TAPESG	climate, vegetation, potential energy, geomorphology
slope	TAPESG	overland and subsurface flow velocity and runoff rate, precipitation, vegetation, geomorphology
aspect	TAPESG	solar insolation, flora and fauna distribution, evapotranspiration
profile curvature	TAPESG	flow acceleration, erosion/deposition rate, geomorphology
plan curvature	TAPESG	converging/diverging flow, soil water content, soil characteristics
secondary terrain attribute		
erosion–deposition index	EROS	erosion/sedimentation rate, nutrient supply, landscape function, vegetation, soil depth and texture
static topographic wetness index	DYNWETG	plant available water, soil drainage status, vegetation, groundwater dependant ecosystems

4.2.6 *Habitat class mapping*

Continuous surfaces for slope, elevation, wetness index, and erosion/deposition index at 20 metre grid resolution were output from terrain analysis (422 892 grid cells in each environmental layer) representing the *grain* of environmental variation across an extensive lowland and highland landscape (Figure 4-2).

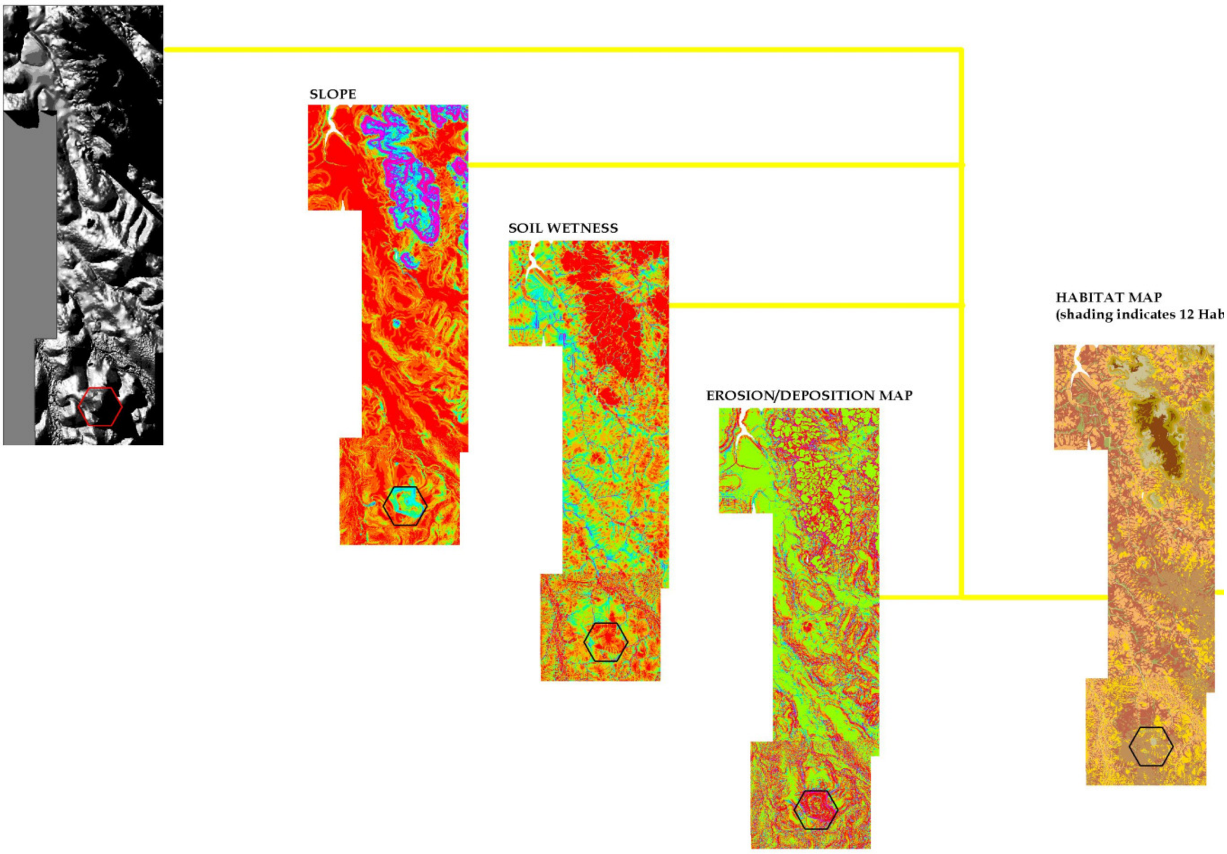


Figure 4-2 Continuous environmental surfaces for elevation, slope, soil wetness index, erosion/deposition index, and derived (k-means clustering) habiat type map with a hexagon (500 ha) outlining the Ranger landform

Habitat was classified from this data, using a non-hierarchical, *k-means*, classification strategy (Mackey et al., 2008; Mackey et al., 1988) in the PATN 3.03 (Belbin, 2008) software package to produce more than the desired number of classes. Hierarchical classification routines in PATN were used to evaluate structure in the initial classification to identify a desirable number of classes from the shape of the dendrogram before rerunning the non-hierarchical classification (Belbin, 1991). 11 habitat classes were classified from the environmental layers. The functions within PATN facilitate multivariate pattern analysis on large and complex data sets. The parameter values for the classifications are provided in Table 4-2.

Table 4-2 Environmental attributes and codes used in the analysis

No.	Attribute	Attribute codes
	Land surface	
1	drainage	1=very poor 2=poor 3=imperfect 4=moderate 5=well 6=rapid
2	runoff	0=none 1=very slow 2=slow 3=moderate 4=rapid 5=very rapid
3	permeability	1=very slow 2=slow 3=moderate 4=high
	Substrate	
4	substrate depth	in metres
5	substrate lithology	1=colluvium(CL) 2=ferricrete(FC) 3=schist(ST) 4=sandstone(SA) 5=alluvium(AL)
	Soil classification	
6	soil order	1=kandosol(KA) 2=hydrosol(HY) 3=rudosol(RU) 4=tenosol(TE)
7	soil sub-order	1=red(AA) 2=orthic(DS) 3=oxyaquic(DT) 4=brown(AB) 5=leptic(CY) 6=yellow(AC) 7=redoxic(ED) 8=stratic(ER)
8	soil great group	1=regolithic(GF) 2=petroferric(EA) 3=mesotrophic(AG) 4=paralithic(DU) 5=dystrophic(AF)
9	soil sub-group	1=basic(AR) 2=ferric(BU) 3=melanic(DK) 4=rendic(EF) 5=haplic(CD) 6=mottled(DQ) 7=bleached(AT)
	Soil morphology	
10	A horizon thickness	1=thin(A) 2=medium(B) 3=thick(C)
11	surface gravel	1=none(E) 2=slight(F) 3=common(G) 4=moderate(H) 5=very(I)
12	A horizon texture	1=sandy(K) 2=loamy(L) 3=clay loamy(M)
13	B horizon texture	1=sandy(K) 2=loamy(L) 3=clay loamy(M) 4=clayey(O)
14	soil depth	1=very shallow(T) 2=shallow(U) 3=moderate(V) 4=deep(W)
	Digital terrain analysis	
15	aspect	TAPESG, octant coded for slopes greater than 0.5%
16	wetness	TAPESG (DYNWETG) dynamic wetness index
17	elevation	tapesg (m)
18	slope	tapesg (%)
19	erosion	TAPESG (EROS) erosion/sedimentation index
20	easting	E location of site from SW origin
21	northing	N location of site from SW origin
	Landform description	
22	landform element	1=crest(HCR) 2=drainage depression(DDE) 3=swamp(SWP) 4=slope(HSL) 5=talus(TAL) 6=valley flat(VLF) 7=bench(BEN)
23	slope morphology	1=crest(C) 2=lower slope(L) 3=flat(F) 4=vale(V) 5=upper slope(U) 6=mid slope(M)

PATN calculates a measure of the dissimilarity in the association metric used between a set of objects based on the values for the variables used to describe the objects. Each environmental variable was standardised, prior to classification, dividing by its range to remove bias due to differences in the units of measurement. In this case, the objects were

20 metre grid cells and the variables were several environmental variables. The output was a habitat class map. Following field ecological surveys this could be considered to be a habitat type map.

4.2.7 *Natural analogue selection*

A hexagonal mesh of 500-hectare polygons was overlaid on the habitat class map to constrain the analogue landform selection process to natural landforms similar in extent to the mine landform. One element of the mesh covered the Ranger mine landscape. The habitat class map was intersected with the 500-hectare mesh and the areas of each habitat class were summed in each mesh element. The landscape classification derived in this way identified two landforms of the same extent as the mine landform and with a similar range and proportion of habitat classes. These landforms were assumed to support similar levels of ecosystem variation, which was assessed from subsequent field surveys described below in Section 4.2.9.

4.2.8 *Fire environment*

Fire frequency and intensity in the analogue areas was compared with surrounding areas of Kakadu National Park using publicly available, time series remote sensing. The monthly fire scar maps, derived from MODIS satellite data for the Northern Territory from January 1996 until July 2003 were acquired from the West Australian Department of Land Information web site (Marsden, 2003). Fires prior to 1 August were classified as less intense early fires and those after were categorized as more intense late fires. The fire frequencies were normalised (years in 10) for early and late fires and an analogue area was compared to similar habitats elsewhere in KNP (Schodde et al., 1987) to check whether there were likely to be obvious differences in the fire environment between the analogue area and other areas surrounding the mine.

4.2.9 *Environmental survey design*

A stratified survey design (Austin and Heyligers, 1989) with three replicate transects at two toposequence sites in the each of two selected analogue landforms was applied to describe soil, land surface and plant species presence-absence according to standard reconnaissance survey methods (McDonald et al., 2009). Sites were selected along each toposequence to sample different mapped habitat classes from the crests to the valley flats. The patterns in the field survey data were initially evaluated using indirect gradient methods including multivariate classification, ordination and network analysis methods in the PATN program.

4.2.10 *Indirect gradient analysis of environmental variation*

4.2.10.1 Classification

Two site data sets were input separately into PATN: (i) vegetation data as defined by species presence–absence and, (ii) environmental data derived from the terrain analysis and derived from soil and landscape descriptions taken during the field survey work. Species presence–absence was scored for each site in a species data matrix of 1's and 0's. Species that occurred only once were masked out of the analysis. Species that are this sparsely distributed do not add information to a classification process based on pair-wise dissimilarity measures and classification improves if they are removed. Initially there were 123 species recorded at 35 sites. Masking out species with single occurrences left 79 species with multiple occurrences. The environmental data set contained a mixture of ratio, ordinal and nominal observations recorded at each site. The nominal and ordinal variables were coded to integer values to facilitate numerical analysis. The environmental attributes and coding used are shown in Table 4-2.

The association measure used was the Czekanowski coefficient. This is the presence-absence equivalent of the measure attributed to Bray and Curtis (1957). The values range from '0' meaning identical to '1' inferring that the two sites have no species in common. This measure has been shown to be appropriate for the type of data being analysed here (Faith et al., 1987). Hierarchical polythetic clustering based on the association measure (Czekanowski) was used to fuse sites into groups. The unweighted pair-group method (UPGMA) from Sneath and Sokal (1973) was used to provide a best fit between the input association measures and what is termed the ultrametric distances implied from the dendrogram. The flexible version of the UPGMA implemented in PATN incorporates a variable (β=-0.1) that allows for a slight dilation in the fusion strategy, adjusting for insensitivity that occurs when sites have few of no species in common.

The site group definitions were copied from the species to the environmental dataset and Significant environmental differences between site groups were identified by the Kruskal-

Wallis statistic. The sites and then species were clustered to define site by species groupings in a two–way table. The data file for species presence–absence data file was edited to mask out singular species occurrences that did not add information to the pairwise association measure.

The species data matrix was then transposed to classify the species on the basis of where they occurred. This revealed species that tended to occur together. A two-step association measure was applied to account for the lack of 'duality' in the site as by species classification process (Belbin, 1991). The site and species classifications were then imposed on the data matrix to create a two-way table. This imposed the dendrogram ordering, with the defined groups, back on the raw data and was a useful check of the results of the classification process.

4.2.10.2 Ordination

Overall trends are more identifiable using ordination than classification. Non-metric multi-dimensional scaling was used because it is well suited to data that are non-normal, or are on arbitrary, discontinuous scales and is the method of choice for ecological community data (Margules and Austin, 1991; McCune and Grace, 2002; Minchin, 1987a). Ordination placed the sites in a reduced number of dimensions that reflected the Bray Curtis measures of association. Multiple linear regressions of the environmental variables on the ordination vectors were performed to determine vectors of maximum environmental correlation with ordination space. Visual confirmation of the relationships between groups and environmental vectors was obtained by presenting the data in a minimum spanning tree of site classes in ordination space.

4.3 Results

4.3.1 *Terrain modelling and habitat class maps*

The terrain modelling results, and the combined habitat class map that was derived from these results was depicted in Figure 4-2. Typically slopes were less than 3% and the erosion/deposition index varied between -1 and 1 typical of low relief, water shedding surfaces. However, upland plateau landform dominated the landscape to the north and west of the study area. The upland plateau was associated with the Arnhem Land escarpment and contrasted with the lowland peneplain on which the mine site was located. A three dimensional drape of the habitat class map shown in Figure 4-3a distinguished the upland plateau landscape in the north of the study area from the peneplanated lowlands typical of the area comprising the mine landscape.

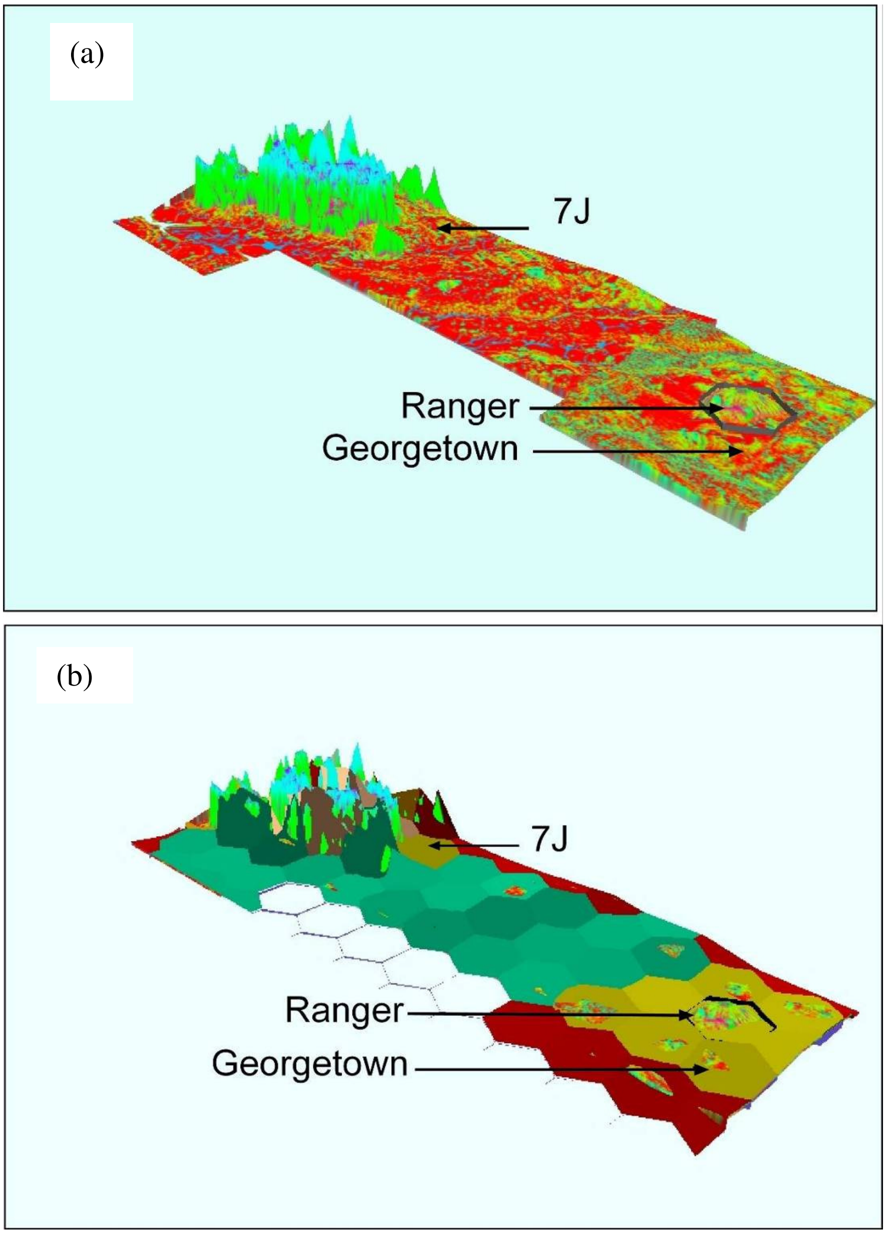


Figure 4-3 (a) Habitat classes in three-dimensional view, showing the extent of the Ranger final landform (10 x vertical exaggeration); (b) colour coded landscape classification in three-dimensional perspective view. Analogous landform polygons are tan coloured

Significant differences occurred between median values of slope, elevation, wetness index and erosion–deposition index between habitat classes. These are highlighted using box plots of the range in significantly different (according to the Kruskal Wallace statistic) terrain parameters for each habitat class in Figure 4-4.

The interpreted geomorphic process in each of the 11 habitat classes is given in Table 4-3. Habitat classes 2, 3 and 10 were likely to be aggraded by sediment according to their geomorphic attributes and erosion-deposition index. Habitat classes 1, 3, 8 and 11 were likely to be wetter than other classes according to wetness index ranges.

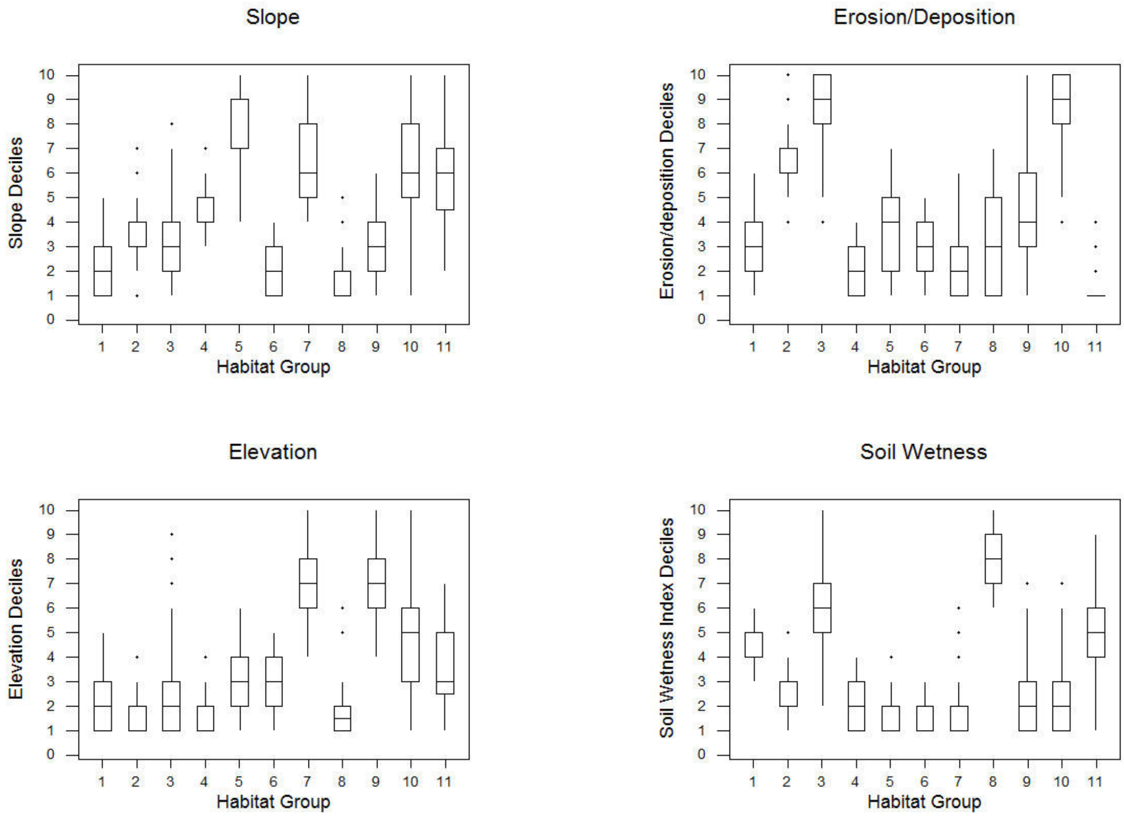


Figure 4-4 Box plots showing the range in environmental attributes for each habitat class mapped. The inner quartile range shown by the boxes highlights differences between the classes. Whiskers indicate outer quartile range and points indicate outliers.

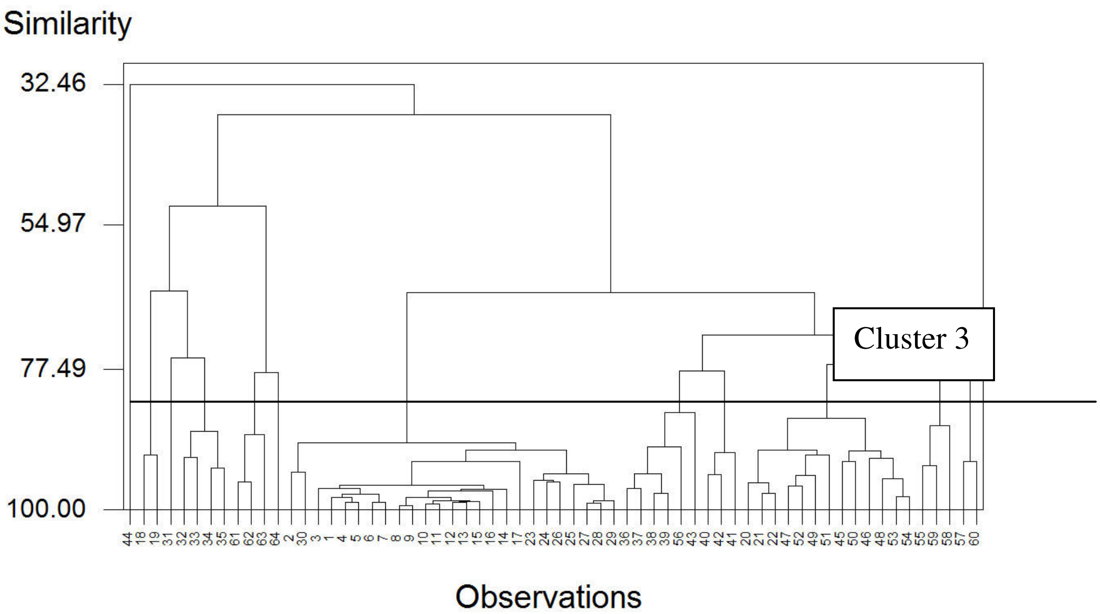
Table 4-3 Geomorphic description of habitat classes

Habitat class	Slope (%)	Geomorphic environment	Land surface	Wetness	Landform Description
1	low	erosional	lowland	not inundated	crests
2	gentle	erosional	lowland	not inundated	slopes
3	gentle	depositional	lowland	seasonally inundated	lower slope
4	gentle	erosional	lowland	not inundated	shoulder slopes
5	steep	erosional	upland	not inundated	plateau escarpment
6	gentle	erosional	lowland	not inundated	crests of low hills
7	steep	erosional	upland	not inundated	upper slopes
8	gentle	depositional	lowland	regularly inundated	drainage depressions
9	gentle to moderate	deposition and erosion are active	upland	not inundated	plateau surface
10	moderate to steep	depositional	upland	seasonal inundation	talus slopes and fans
11	moderate to steep	erosional	lowland and upland	inundated	stream channels

4.3.2 *Landscape classification natural analogue selection*

The landscape classification was made based on the relative areas of each of the 11 habitat classes described above in each 500 hectare tile (64 covering the study area) representing landscape structure. The dendrogram of the association matrix was assessed visually and a cut was made into 9 groups (Figure 4-5a). The part of the dendrogram showing linkages between the Ranger mine and similar landscape areas (Cluster 3) is shown in Figure 4-5b. The Ranger mine are and landscapes of the same extent that were similar to it in terms of habitat mix comprised the most extensive part of the overall landscape (27 observations/hexagons). The choice of two prospective natural analogue areas – Georgetown and 7J, as indicated on Figure 4-3b, reflects their close linkage to the habitat properties of the Ranger mine area, based on the association measures that are used. Other reasons were that the 7J area is physically remote from the Ranger site while the Georgetown area is adjacent to the mine. Both areas have similar underlying parent materials to the mine according to regional geological mapping (Needham, 1988). The depiction of the proposed analogue areas on Figure 4-3a, b indicates their geographic separation by approximately 10 kilometres.

(a)



(b)

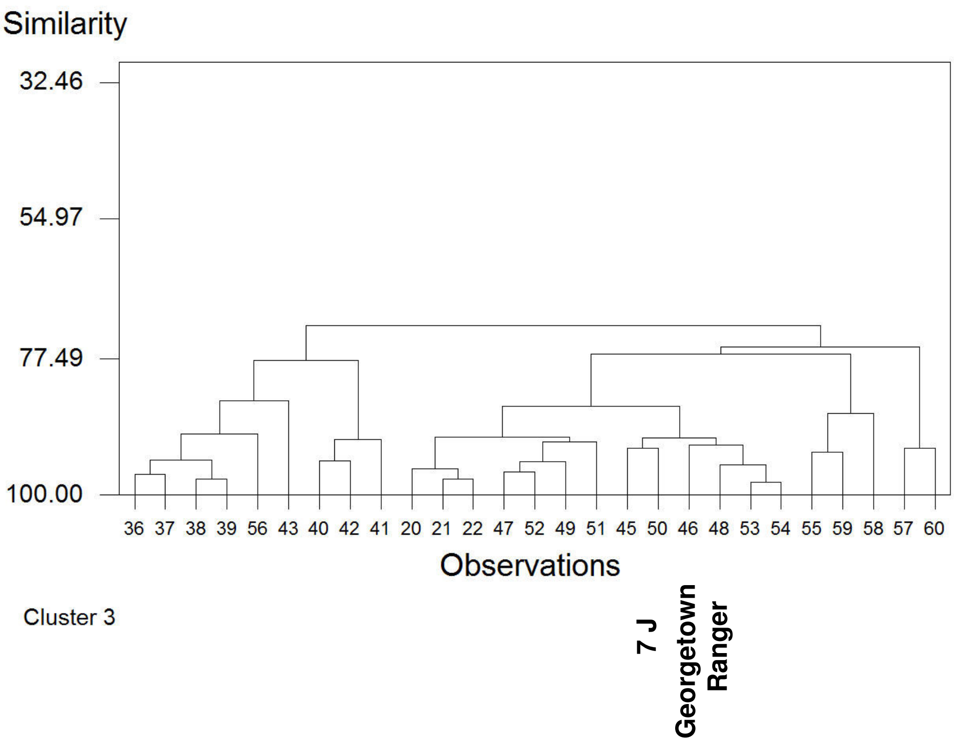


Figure 4-5 (a) Dendrogram showing linkages between 500 hectare hexagons representing landscape structural extent (observations, horizontal cut line for 6 groups); (b) detailed dendrogram for Cluster 3 comprising the Ranger mine and similar areas

Geographic separation was identified as an issue that determined some properties of these areas in the subsequent environmental survey and analysis. Both of the selected landforms were observed to be rocky rises, similar in scale and topography to the Ranger site. The *7J* analogue area abutted a high relief plateau 10-km to the north of the study area while the *Georgetown* area was immediately adjacent to Ranger mine. The most extensive landscape component in the selected landform patterns were the gently sloping erosional surfaces.

4.3.3 *Fire environment*

There were three habitat types that were defined by regional (KNP) habitat mapping (Schodde et al., 1987), represented in the analogue landforms (Figure 4-6). Brighter hues in Figure 4-6 indicated less frequent fires. Less frequent early dry season fires and more frequent late dry season fires occurred in the analogue area than in the surrounding parts of KNP.

The total dry season fire frequency was similar in the analogue area and selected habitats in KNP. In summary, late dry season fires were more frequent in the analogue area than in similar habitats in KNP, but the fire environments and habitat range were similar.

4.3.4 *Vegetation patterns*

123 species were recorded in the environmental survey. After masking out singular observations (90% annuals) that occurred once, the remaining 79 species were used in the analysis (Table 4-4). Multivariate classification distinguished nine site groups shown in Table 4-5. Site groups 2 and 5 were represented by one site each on sandy outwash fans in the *7J* analogue area. Classifying the transposed sites by species table produced six groups of species that commonly coexisted shown in Table 4-5.

The common or extensive plant community types such as species group 1 occurred in all site groups except site group 2 (Table 4-5). Other species groups were well-represented in particular environments. Matrix cells with notable species-site associations in Table 4-5 are highlighted by shading. Species group 3 was well represented in site group 1, which were all on sandy substrates. Species group 4 was well represented on the wet sandy substrates in site group 2. Species group 6 was well represented in poorly drained areas with ferruginous subsoil horizons in site groups 4 and 6. Apart from their absence in site group 1 (sandy substrates) the pattern of site distributions for species groups 2 and 5 was indistinct, i.e.

they were poorly defined by the habitat variation that was sampled. Otherwise dense localised patterns of species and site groups in the two-way table indicated that the analysis was robust.

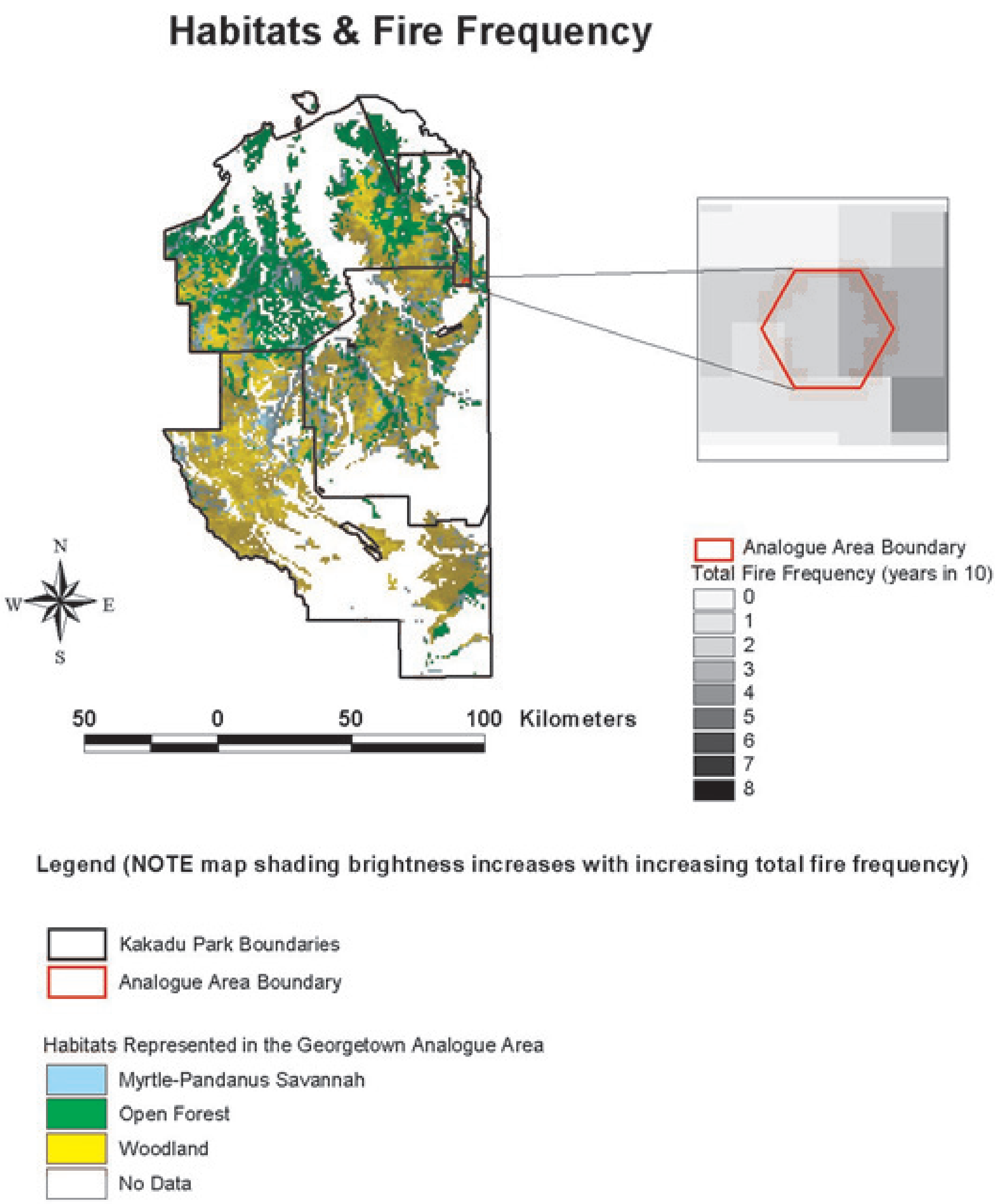


Figure 4-6 KNP habitat mapping units (Schodde, 1987) represented in analogue areas with fire frequency mapping showing detail in the Georgetown area

Table 4-4 Common analogues area species — names, abbreviations, group classification and frequency

Species group	**Species abbreviation**	**Species number**	**Species name**	**Frequenc y**
1 (34 members)	ACACMIMU	1	*Acacia mimula*	25
1	AMPEACET	5	Ampelocissus acetosa	13
1	ARISSPEC	7	Aristida species	3
1	BRACMEGA	8	Brachychiton megasperma	12
1	BUCHOBOV	9	Buchanania obovata	30
1	CASSFILI	13	Cassytha filiformis	3
1	COCHFRAS	15	Cochlospermum fraseri	18
1	CROTARNH	18	Croton arnhemicus	2
1	ERYTCHLO	23	Erythrophleum chlorostachys	16
1	EUCABLEE	24	Eucalyptus bleeseri	14
1	EUCACONF	25	Eucalyptus confertiflora	9
1	EUCAMINI	27	Eucalyptus miniata	18
1	EUCAPORR	28	Eucalyptus porrecta	16
1	EUCASETO	29	Eucalyptus setosa	6
1	EUCASPEC	30	Eucalyptus species	12
1	EUCATECT	31	Eucalyptus tectifica	9
1	EUCATETR	32	Eucalyptus tetrodonta	24
1	GARDMEGA	35	Gardenia megasperma	12
1	GOODSPEC	37	Goodenia species	15
1	GREVDECU	39	Grevillea decurrens	12
1	HAEMCOCC	43	Haemodorum coccineum	4
1	HETETRIT	46	Heteropogon triticeus	14
1	INDISAXI	47	Indigofera saxicola	8
1	LIVIHUMI	48	Livistonia humilis	17
1	MURDGIGA	52	Murdannia gigantea	8
1	PETAQUAD	59	Petalostigma quadriloculare	17
1	PLANCARE	60	Planchonia careya	24
1	POUTARNH	61	Pouteria arnhemica	11
1	SORGBRAC	65	Sorghum brachypodum	20
1	SPERSPEC	67	Spermacoce species	21
1	TACCLEON	69	Tacca leontopetaloides	14
1	TERMFERD	70	Terminalia ferdinandiana	30
1	WEDEURTI	76	Wedelia urticifolia	3
1	XANTPARA	78	Xanthostemon paradoxus	25
2 (8 members)	ANTIGHES	6	Antidesma ghesaembilla	4
2	CALYACHA	11	Calytrix achaeta	3
2	CARTSPIC	12	Cartonema spicatum	4
2	DROSSPEC	21	Drosera species	6

Table 4-4 continued

Species group	Species abbreviation	Species number	Species name	Frequency
2	ERIAOBTU	22	Eriachne obtusa	3
2	GRASSPEC	38	Grass species	5
2	PAVEBROW	56	Pavetta brownie	2
2	SOWEALLI	66	Sowerbaea alliaceae	6
3 (15 members)	ACACONCI	2	Acacia oncinocarpa	4
3	ALLOSEMI	4	Alloteropsis semialata	2
3	COLERETI	16	Coleospermum reticulatum	5
3	COMMENSI	17	Commelina ensifolia	5
3	DENHOBSC	19	Denhamia obscura	4
3	DOLIFILI	20	Dolichandrone filiformis	9
3	GARDSCHW	36	Gardenia schwarzii	4
3	GREVGOOD	40	Grevillea goodii	3
3	HETECONT	45	Heteropogon contortus	3
3	OWENSPEC	53	Owenia species	4
3	PACHJUNC	54	Pachynema junceum	5
3	PERSFALC	57	Persoonia falcata	6
3	PETAPUBE	58	Petalostigma pubescens	13
3	SETAAPIC	64	Setaria apiculata	3
3	THECPUNI	73	Thecanthes punicea	8
4 (6 members)	CLEOSPEC	14	Cleome species	2
4	GARDRESI	34	Gardenia resinosa	2
4	GREVPTER	42	Grevillea pteridifolia	2
4	LOPHLACT	49	Lophostemon lactifluus	2
4	MELANERV	50	Melaleuca nervosa	3
4	VERTCUNN	75	Verticordia cunninghamii	2
5 (6 members)	ACACSPEC	3	Acacia species	7
5	CALYEXST	10	Calytrix exstipulata	7
5	EUCALATI	26	Eucalyptus latifolia	6
5	RHYNLONG	62	Rhynchospora longisetis	2
5	SYZYEUCA	68	Syzygium eucalyptoides	2
5	TERMPTER	72	Terminalia pterocarya	8
6 (10 members)	FICUOPPO	33	Ficus opposita	2
6	GREVMIMO	41	Grevillea mimosoides	3
6	HAKEARBO	44	Hakea arborescens	4
6	MELAVIRI	51	Melaleuca viridifolia	8
6	PANDSPIR	55	Pandanus spiralis	5
6	SEDGSPEC	63	Sedge species	3
6	TERMGRAN	71	Terminalia grandiflora	2
6	UTRISPEC	74	Utricularia species	2
6	WRIGSALI	77	Wrightia saligna	11
6	XYRISPEC	79	Xyris species	2

Table 4-5 Two-way table of sites groups (rows) verses vegetation species groups (columns) with notable species-site associations highlighted by shading

```
                  Group 1               Group    Group 3     Group  Group  Group 6
                                          2                    4      5
    AAABBCCCEEEEEEEEEGGGHHILMPPPSSTTWX|ACCDEGPS|AACCDDGGHOPPPST|CGGLMV|ACERST|FGHMPSTUWX
    CMRRUAORRUUUUUUUUAORAENIUELOOPAEEA|NAARRRAO|CLOOEOAREWAEEEH|LAROEE|CAUHYE|IRAEAEETRY
    APIACSCOYCCCCCCCCROEETDVRTAURECRDN|TLROIAVW|ALLMNLRETECRTTE|EREPLR|ALCYZR|CEKLNDRRIR
    CESCHSHTTAAAAAAAADDVMEIIDANTGRCMET|IYTSASEE|COEMHIDVENHSAAC|ODVHAT|CYANYM|UVEADGMIGI
    MASMOFFACBCMPSSTTMSDCTSHGQCABSLFUP|GASSOSBA|OSREOFSGSSJFPAP|SRPLNC|SELLEP|OMAVSSGSSS
    ICPEBIRRHLOIOEPEEEPEORAUIUARRPEERA|HCPPBPRL|NEENBICOPPUAUPU|PETAEU|PXAOUT|PIRIPPRPAP
    MEEGOLANLENNRTECTGECCIXMGARNAEORTR|EHIETEOL|CMTSSLHOEENLBIN|ESECRN|ESTNCE|PMBRIEAELE
    UTCAVISHOEFIROCTRACUCTIIADEHCCNDIA|SACCUCWI|IIIICIWDCCCCECI|CIRTVN|CTIGAR|OOOIRCNCIC
                                      |        |               |      |      |
Group 1
B1  *  **   *   *   * *      **  *    |        |   *    *     *|      |      |
B3  *  ***  *  *    *         *    *  |        | * **   **  *  |      |      |
C1  *   * * *   * * * *       ** ***  |        |* ****** ******|      |      |
C2  *   * * *   * * * *       ** ***  |        |* ****** ******|      |      |
C3  *   * * *   * * * *       ** ***  |        |* ****** ******|      |      |
C4      *   *   *   *         *  * *  |        |*     *     *  |     *|      |    *
    ----------------------------------+--------+---------------+------+------+----------
B4  Group 2                           |   *   *|            *  |******|      |
    ----------------------------------+--------+---------------+------+------+----------
Group 3
D1  ** *   *** **   *     ** **** ** *|        |     *    *    |      |      |*
D2  ** ** ** * **   ** *   * * ***** *|   *    |           *  *|      |      |        *
D3  **  * *  * **   ** *  *  * ***** *|        |               |      |      |
E1  *   * *  ** *   ****  ** ** *  * *|        |               |      |**   *|
E2  *   * *  *  *   ****  ** ** *  * *|        |               |      |*    *|
F1  ** ** * ** ** * *     *     **** *|        |            * *|      |*     |
F3  ** **    * *  ***     **  * * ** *|        |     *      *  |      | *    | *
F4  *   * *  * ** * *     ** ***** * *| *      |               | *    |*  *  |
K2  **  *    ***  *      * ********* *|   * *  |              *|      |*    *|
K3  *   *   **** *  **     **** *    *|        |        *      |      |      |
    ----------------------------------+--------+---------------+------+------+----------
Group 4
D4        *           *    *  *  * *  |*    * *|     *      *  |      |      |  ***** *
K4                    *       * ** * *|       *|               |      |     *|   *** **
L4      *             *    *  *    *  |  *    *|               |  *   |  *   |  **   **
    ----------------------------------+--------+---------------+------+------+----------
E3  Group 5 *    ***    * *   **  * ** *  | *******|           *   |    * | *   *|        **
    ----------------------------------+--------+---------------+------+------+----------
Group 6
H1  **  * * *  *   *** * *  ** *** * *|        |   *           |      |      |
H2      * * ** *   * * * * * *       *|*     * |  *            |      |      |
H3  **  * * ** *  ** * * * * *  *  * *|*       |               |      |      |
M2  **  *    ******** ****   ** ** * *| *      |  *            |      |      |        *
M3  ** ** * *  *  ****** *   ** ******| * **   |   * *         |      |  *   |
    ----------------------------------+--------+---------------+------+------+----------
Group 7
K1  * * * *   ***** ** *    **     * *|        |              *|      | *    |
L1  *   *      ***  *   ** **      * *|        | *          *  |      |      |        *
L2  * *          *  ** ***  *   *    *|    *   |            *  |      | **   |        *
M1  * *****    ****  * *** ***** * ***|        |           *   |      | *    |        *
    ----------------------------------+--------+---------------+------+------+----------
Group 8
E4  *   *   * *     *    *   **  *** *|* ** * *|          *    |      |     *|   *
K5     ** *        ** *  *    * ** * *|     *  |            *  |   *  |******|   *
L3   * **   * *       *  *    *  *** *|  *     |            *  |    * |* * **|   **
    ----------------------------------+--------+---------------+------+------+----------
Group 9
L5   * **          *     * **  ****  *|        |     *         |*     |  *   |***** * **
M4   * **** * *   **  *  * * ** * ****|        |     *         |      |      | **     *
```

Best fit attributes and corresponding correlations (according to vector lengths) were identified by displaying the fit of environmental factors to NMDS plots for axis 2 versus axis 1 (Figure 4-7a) and axis 3 versus axis 1 (Figure 4-7b). Soil texture (A and B horizon), drainage class, soil great group, soil permeability class, substrate depth, runoff rate and wetness index were all highly correlated with species ordination space.

A- and B-horizon texture, drainage status and runoff rate were prominent in the left-bottom quadrant of the ordination plot (Figure 4-7a). Groups 3 and 6 were the most extensive site groups in this quadrant and occurred on slopes and crests of low rise landforms where the soils were well drained. Wetness index and substrate depth, acted in the top-right quadrant of the ordination plot (Figure 4-7b). Groups 1, 2, 4 and 8 sites populated this quadrant. Group 2 is a single site group occurring in the swamps located on deep colluvial sands. However, group 1 sites were better drained on the same sandy colluvium. Group 4 occurred on the lower slopes of the low rises formed in deeply weathered mica schist, while group 8 occurred on alluvial valley flats. Groups 7 and 5 sites populated the north east quadrant of the ordination. No strong environmental associations were evident in this quadrant. Group 9 occurred in the south west quadrant and occurred in level or very gently sloping landform elements with a cemented ferricrete layer between 0.5 and 1.0 m depth. Ferricrete substrate occurred at both group 9 and group 5 sites.

The minimum spanning tree network (Figure 4-8) showed site group linkages and the directions of strong environmental gradients in the ordination space, highlighting similar ecological groups and environmental driving factors. The minimum spanning tree accurately represents the distance between close objects, a slightly different focus to ordination. Group 2 was very distinct, although quite similar to group 5. Both these groups occurred in poorly drained locations on sandy colluvial substrate. Group 5 occurred on a transition between sandy colluvium and rocky rises with lateritised substrate rocks of mica schist. Consequently, group 5 was linked to group 3, which was the most extensive environment type surveyed.

Group 3 occurred in both analogue survey areas on slopes and crests of rises formed in weathered mica schist. Group 3 was linked to groups 6, 1 and 8 (Figure 4-8). The linkage was closest with group 6, predominantly on gentle to moderately rocky slopes. Group 1 is distinctly different from the rest of the groups and occurred on crests and slopes on the sandy colluvial landforms.

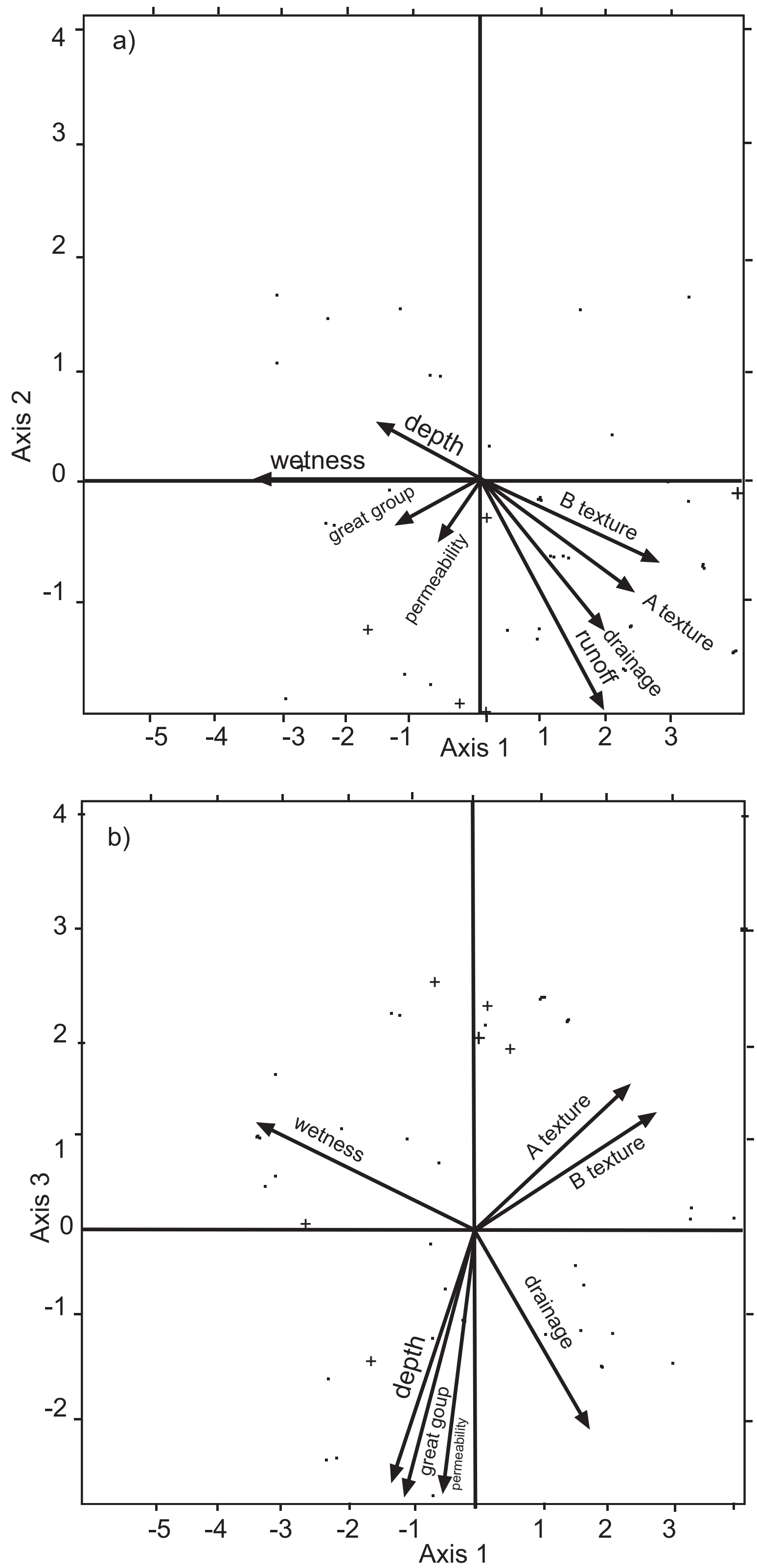


Figure 4-7 Scattergrams from ordination showing strongly correlated environmental variables

Group 8 was also linked to group 4 and occurred on swampy valley flats and on riparian areas respectively. Additionally, group 6 was linked to groups 3, 7 and 9 along the gradient of eucalypt woodlands on the rocky substrates, although group 7 occurred predominantly on the crests of rocky outcrops in the Georgetown survey area. However, group 9 occurred mainly on mainly very, gently sloping to level sites with a laterite pan at between 0.5 and 1.0 m depth.

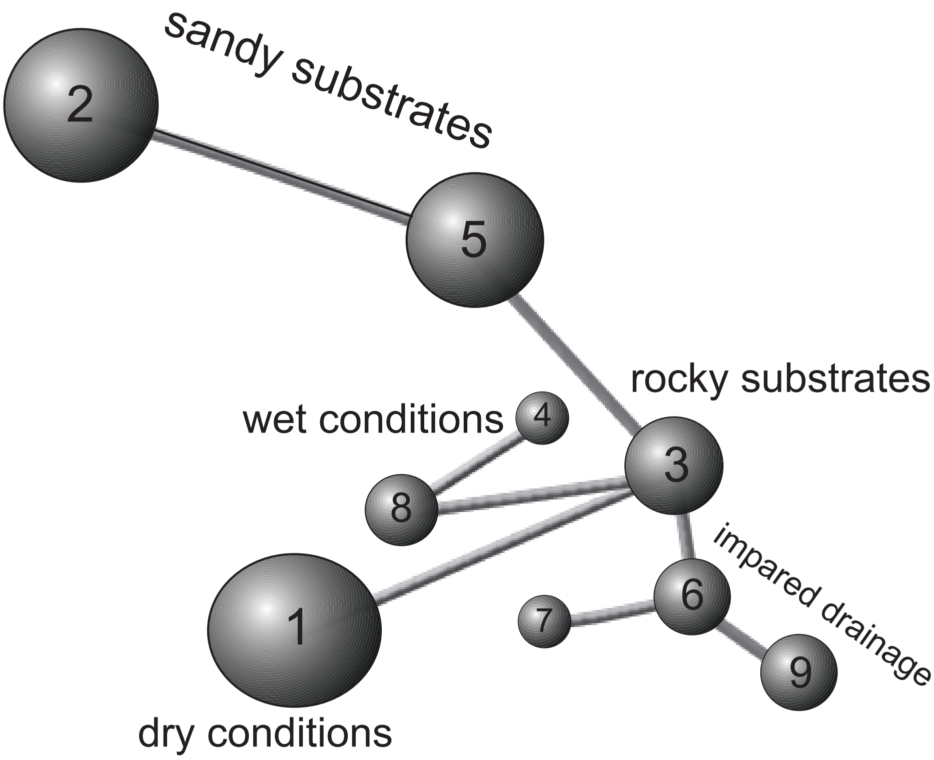


Figure 4-8 Minimum spanning tree depicted as a ball and stick model of ecological groups in ordination space, with vectors indicating directions and strength of environmental correlation with principle components

4.4 Discussion

For the Ranger case study site, the index of erosion–deposition indicated that most of the landscape acts as a transport surface for materials detached by rainfall and moved in overland flow (Moore et al., 1991). The analysis makes sense in terms of catchment water balance with extensive wetlands supported on seasonal runoff (Vardavas, 1987), environmental processes related to the energy environment (Odum and Odum, 2003) and local variation in landscape ecology (Ludwig et al., 2002).

Environmental processes that underlie ecological pattern are resolved and then aggregated to a scale that represents the system being studied. The high resolution landscape metrics that reflect the hill slope hydrology are robustly correlated with landscape function and the ability to retain material resources. This is supported elsewhere for perennial vegetation in Australian rangelands and savanna woodlands (Bastin et al., 2002; Bowman and Minchin, 1987). Applied at a high enough resolution to identify environmental range in hill slope environments and aggregated to describe ecological systems that are in context with the mine landscape, these methods could be adapted from a broad regional context to quantifying local environmental variation specific to mining disturbance and selecting natural analogues for mine restoration planning.

Analogue areas comprised broad variation in habitat class across hill slope environments. Environmental properties of sites in habitat classes in selected analogue areas varied in plant community (used to represent ecosystem type), soil drainage and substrate properties such as texture and gravel content that directly relate to plant available water storage. This corroborates the intuitive understanding of vegetation patterns in the savanna woodland environments in KNP (Wilson et al., 1991) related to water and sediment balance (Bowman et al., 1993).

All of the extensive savanna communities in the larger KNP area (Schodde et al., 1987) were represented in the reconnaissance survey. Out of the nine site groups (used to represent ecosystem type) five were broadly categorised as lowland *eucalypt* woodland and mixed *eucalypt* woodland and a sixth (group 4) as *myrtle-pandanus* savanna (sensu Schodde 1987). Three groups (groups 1, 2 and 5) were associated with sandy sheet wash fans emanating from the base of the upland Arnhem Plateau in the *7J* analogue area, while all of the measured environmental range on weathered schist substrates was present in the *Georgetown* analogue area. This gives confidence in focussing environmental surveys on the *Georgetown* analogue area. The environmental range represented in this analogue landform was wider than that sampled in previous analogue surveys for Ranger mine over a wider geographic extent (Brennan, 2005).

These findings emphasise the importance of including hill slope variation in selecting an analogue landform for the purpose of providing environmental support for restoring mined landscapes. The approach addresses the requirement expressed by Bell (1997) that landscape structures need to be introduced as a central theme in environmental variation to aid restoration efforts. Also required is the appropriate landscape context that is essential for selecting natural analogues. The importance of a landscape context has been recognised before (Duque et al., 1998; Ehrenfeld and Toth, 1997; Hobbs and McIntyre, 2005; Holl et al., 2003; Huttl and Bradshaw, 2001). However, this study presents more robust analogue selection methods that were focussed on the mined landscape rather than regional conservation objectives, which would have confounded rather than completed the picture of restoration objectives.

The concept of similar areas, identified from multivariate analysis of continuous environmental data derived from a DEM (Bryan, 2003; McGarigal and Marks, 1995) is adapted to specify the environmental pattern within at the scale of the mine landscape that needs to be restored. Understanding environmental pattern and scale is a central problem where basic ecology needs to be applied to describe phenomena that vary in space and time (Levin, 1992). In landscape restoration, the function of the larger landscape assemblage depends on patchiness in the distribution of resources in smaller units (Ludwig et al., 2004). Landscape assembly and linkage is intrinsic to the methodology used here. The range of analogue ecosystems was physically linked in a functioning landscape and based on detailed (20-m grid) environmental data. For instance the wetland and riparian sites were successfully discriminated from the woodlands on rock outcrops and mixed eucalypt woodland sites that were contiguous on hill slopes. These hill slope environmental features have been overlooked in past analogue studies focussed on a regional context (Brennan, 2005; Grant, 2006; Hobbs and McIntyre, 2005). Although these features are known to be important (Bowman et al., 1993; Russell-Smith, 1995; Wardell-Johnson and Horwitz, 1996) they have been difficult to assess (Dirnbock et al., 2002).

The current study also identified environmental gradients that are more important in a regional rather than local landscape context. For instance an analogue area, 10 kilometres away from the mine, had more extensive areas of vegetation communities in which *Corymbia bleeserii* is common. This woodland species is associated with relatively drier conditions (Williams et al., 1996) and its prevalence may be due to local rain shadow effects from surrounding plateau areas. Consequently, analogue data from areas that are remote from the mine site may introduce some complexities into the analysis and quantitative assessment based on environmental correlation and regression (Austin, 2002b). Care must be taken to exclude extraneous environmental gradients if regional environmental surveys are used to select analogues.

The frequent late dry season fires in the analogue and surrounding areas of KNP are a common feature of savanna woodland environments in the Top End dominated by eucalypt woodland (Andersen et al., 2005). The management of fire frequency and intensity is likely to have profound effect on the survival of eucalypt species early in the revegetation program (Williams et al., 1999). However, similarities in fire environment in analogue areas and surrounding areas of KNP (based on the review of fire frequency in Section 4.3.2) influence similar mature woodland communities in terms of species distribution and community structure.

The selected natural analogues areas are representatives of lowland environmental variation in context with the mine landform — without sacrificing environmental variation found over an extensive area. This is a very useful result considering the fact that previous analogue surveys were far more extensive and covered a narrower environmental range (Humphrey et al., 2005). Appreciating the connectedness of landscape is important, as upland landforms can exert strong localised influences on soils, vegetation and probably local climate in surrounding lowlands. The approach developed here preserved the context of hill slope environmental variation at the mine scale and carefully excluded extraneous escarpment environments that fringe upland plateau landscapes.

Inappropriate analogues or reference sites associated with broader landscape diversity can be identified and avoided by ensuring that the reconnaissance environmental survey sampled each habitat class to provide a clear focus on the environmental range relevant to the mine area. Generalising environmental variation using regional topographic references as proposed by Hobbs & McIntyre (2005) would not have provided an accurate context for selecting analogue areas in this instance.

4.5 Conclusions and further studies

4.5.1 *Conclusions*

Terrain analysis resolved patterns of hill slope environmental variation in low relief landscapes and identified extensive water shedding surfaces and wetter run-on areas. However, variation in substrate properties was linked to colluvial geomorphic processes in the surrounding landscapes. Using a landform extent and patch metrics based on the component areas of different habitat types answered the key question of what part, or parts of the natural landscape were appropriate to the mine restoration task. In this respect, localised mining activity was considered to be analogous to foraging animal range in the way that the extent of landscape utility is defined. The habitat concepts being applied were not linked to the range of any particular species, but rather the distribution of plant communities in local hillslope environments – particularly between *Melaleuca* and eucalypt woodland communities and discriminated environmental gradients in eucalypt woodland type that were related to soil and landscape properties reflecting plant water supply and drainage.

Landscape factors related to water and sediment distribution, drainage and plant available water supply in the soil exerted significant control over the distribution of vegetation types in seasonally dry savannah environments. Individual natural analogue landforms may contain a wide range of ecosystem types, representing the environmental range that occurs on hill slopes over a much more extensive area. Selecting analogues using a combination of terrain analysis, multivariate landscape classification and structured environmental surveys is a practical approach. This set of procedures is structured to provide a clear context with the scale of a mine landform and high resolution of environmental variation based on a DEM. Other environmental data sets can be introduced as required. For instance substrate mineralogy may be an issue that can be described from remotely sensed passive radiometric data (Bollhöfer et al., 2008). Multi-spectral and hyper-spectral data also have potential application for assessing a range of environmental factors (Ustin et al., 2004).

However, understanding factors that affect ecosystem variation within hill slope environments appears from this analysis to be more appropriate for planning landscape restorations following opencast mining than geochemical, geological or regional context. Landscape structural extent and environmental grain size are useful concepts – replacing concepts of map scale and resolution from thematic mapping – to develop landscape models for environmental analysis and analogue selection. Also, the proximity of natural analogue areas to the site to be restored may be a consideration in cases where there are steep rainfall and climatic gradients.

A quantitative approach to selecting analogue landforms is developed here that is based on multivariate statistical and patch analysis methods that are in general use in conservation and landscape ecology. Continuous surfaces of high resolution environmental information and multivariate analysis methods that were developed to select reserve habitats for conservation and landscape planning (Bryan, 2003; Margules and Austin, 1991; McGarigal and Marks, 1995; Rempel and Kushneriuk, 2003; Rempel and Kaufmann, 2003) could be applied to select natural analogue areas. Although, the resolution and scale used to describe ecosystem variation will be critical to adapting the method to the particular application (Levin, 1992).

4.5.2 *Further studies*

The analogue selection approach and multivariate analysis of environmental pattern presented in this chapter leads to more detailed survey supporting direct gradient analysis in Chapter 7. However, the natural analogue approach that has been presented implies that the properties of the surface cover in the mine landform can resemble the range of natural soils. Restoring comparable plant available water storage and drainage conditions in the constructed mine soils appears to be particularly important to demonstrate support for comparable woodland communities. The implications of land surface and landform cover construction on soil properties and revegetation response are investigated in Chapter 5.

Chapter 5 Mine landform cover design and environmental evaluation

5.1 Introduction

5.1.1 *Background*

Some physical aspects of landform cover design and soil reconstruction that relate to ecosystem reconstruction are required to identify an appropriate waste rock landform for mine site reconstruction. This chapter presents a common situation — whereby the operational nature of mineralised waste rock stockpiling and cover may limit the applicability of ecological research to documenting environmental processes rather than influence the design. Thus, operational constraints and critical factors other than ecosystem restoration may be influential on the design and construction outcomes. Restoring the capacity of the soil zone to support natural ecosystems is an important aspect of ecosystem reconstruction (Callaham et al., 2008; Heneghan et al., 2008). However, the work reported in this chapter highlights the constraints that operational decisions may impose on similar restoration projects and identify where a different approach may be required to support the endemic flora and biodiversity found in natural analogue landscapes.

Some background information is required to establish the practical context of landform cover design and to identify the critical issues in regard to ecosystem reconstruction. The objectives for designing landform cover systems can include control of dust and water erosion, chemical stabilisation of acid-forming mine drainage (through control of oxygen ingress), contaminant release control (through control of infiltration) and providing a growth medium for vegetation establishment (O'Kane and Wels, 2003). The last objective is perhaps the most important one for meeting the broad range of environmental objectives at closure. However, environmental restoration (if required) is one of a number of secondary design considerations.

There are several variations on the base method cover system design shown in Figure 5-1 that typically aim to limit deep drainage. "*Barren waste*" in typical cover configurations shown in Figure 5-1 is non-reactive waste rock or tailings produced and stockpiled from the mining operation. Barren waste is a logical cover material source available to most mine sites. "*Oxidized waste cover material*" is near surface barren waste rock that is relatively free of reactive minerals and weathering products through natural weathering over geological time.

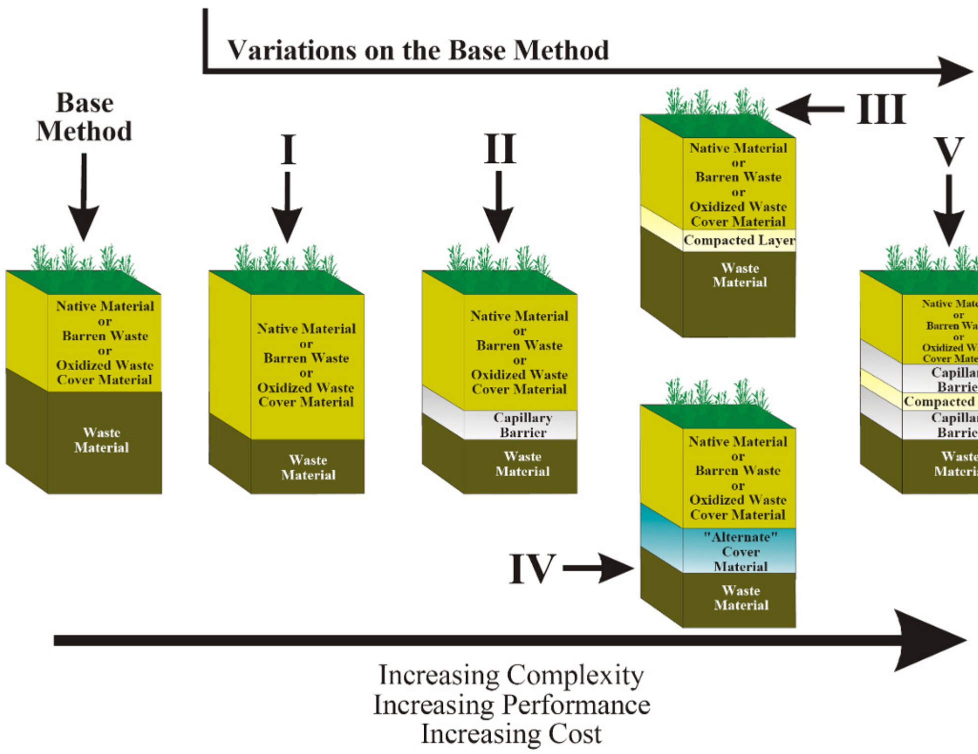


Figure 5-1 Schematic illustration of the base method cover system and variations of the base method (O'Kane et al., 2003)

While vegetation type and form is known to depend on the growth medium properties, cover design guidelines (O'Kane et al., 2003) aim to constrain revegetation outcomes and restrict tee root penetration so as to limit deep drainage (Gatzweiler et al., 2001; Taylor et al., 2003). Water that percolates through mineralised waste tends to contain high concentrations of solutes (Morin and Hutt, 2001). Consequently, regulatory authorities usually prescribe conservative design approaches with performance criteria based on a single factor such as drainage flux (O’Kane and Wels, 2003). Thus, established design practice maintains cover integrity against biological intrusion rather than provide support for appropriate ecosystems. These two aims may not be mutually exclusive if the cover has the range of appropriate drainage and water retention properties required to support selected ecosystems.

A store-and-release cover is a special case of either of the base method configurations shown in Figure 5-1 that consists of one or several layers designed to maximise root penetration and soil moisture storage (O'Kane and Wels, 2003). This type of cover relies on the moisture retention and storage characteristics of the cover material to 'store' infiltration for subsequent removal by evapotranspiration, the main process giving long term control of drainage flux. The root zone is not limited to the cover layer but may extend into the upper layers of the mine waste. In this case, the cover material would primarily serve as a medium for initiating plant growth and to avoid wind and water erosion of the underlying mineralised waste material to deleterious effect on the surrounding environment.

Surface erosion control is a key issue for maintaining the integrity of the cover system over time and reducing the risk of contamination of the surrounding environment (Evans, 2000; Pickup et al., 1987; Wasson, 1992). Engineering guidelines have been developed using the concept of graded, earthy rip-rap material for erosion resistant cover design over uranium mine waste material (Johnson, 2002). The earthy rip-rap of graded mine waste confers erosion resistance, water storage and drainage limitation properties on a basic type of store and release cover. The key design considerations are: (i) the size and durability of the coarse rip-rap material that is required to provide an erosion resistant bridging structure; and (ii) the amount and texture of the fine material infilling voids — providing unsaturated water storage capacity and a porous medium conferring matrix flow properties that reduce the risk of rapid water movement to depth via voids. There is evidence that protective gravel lags form on both natural and waste rock terrain over time limit erosion to a similar extent (Moliere et al., 2002). Therefore, ecological design considerations at mine sites where the natural analogue soils on erosional terrain are gravelly are consistent with constructing the cover according to these uranium waste containment guidelines.

5.1.2 *Cover design assessment*

The Ranger mining department designed and constructed a mineralised waste stockpile with a non-mineralised cover between 1998 and 2001 with the intention of reducing sediment and solute loads leaving the site and supporting revegetation. The design assessment made in this chapter was based on a water and solute balance of the store and release cover. The primary cover design criteria were the placement of sufficient thickness of non-mineralised, weathered rock material to provide an effective barrier to radon emanation and an erosion resistant surface of un-weathered rock to maintain containment over time. The results originated from confidential consultancy reports to ERA Ranger Mine (Hollingsworth, 2005; Hollingsworth et al., 2003). Comparisons between the constructed cover and observed properties of native soils in analogue areas were made to identify critical issues to enhance ecosystem support in the cover design process.

The Ranger case study site design parameters (in terms of cover thickness and the drainage limiting layer) were determined and the cover constructed before predictive modelling of design options and risk based impact assessment. Consequently, the assessment presented here documents the level of control of drainage and support for revegetation from the particular cover configuration that was used. Also, issues where cover design may not match critical environmental processes observed in native soils (Cook et al., 2002; Cook et al., 1998; Hutley et al., 2000; Kelley, 2002; Kelley et al., 2007) are highlighted for further consideration. The water balance is a key environmental process that determines terrestrial environmental pattern (Rodriguez-Iturbe et al., 2007).

5.1.2.1 Water balance

The soil water storage and evapotranspiration terms of the water balance of a store-and-release cover design are critical for limiting drainage flux through underlying material and for supporting the native vegetation (Crees et al., 1994). The water balance can be represented as a mass balance by equation 5-1:

$$P_n = E_t + R_o + \Delta D_e + D_r \qquad \text{(equation 5-1)}$$

, where, P_n is precipitation, E_t is evapotranspiration, R_o is net runoff, ΔD_e is change in soil water storage and D_r is drainage.

Drainage to depth can be measured directly using lysimeters (O'Kane et al., 2003) or determined indirectly from hydraulic gradients and soil properties (Cresswell and Paydar, 1996). Acquiring accurate measurements using lysimeters in heterogeneous waste rock can be problematic (Bennett and Plotnikoff, 1995) — particularly with respect to sampling a large enough area to be representative (Bews and Barbour, 1999). Alternatives include modelling drainage (Snow et al., 1999) or solving the water balance for drainage when the other parameters (evapotranspiration, runoff, soil water storage) are known (Dunin et al., 2001). One-dimensional drainage flux is calculated from monitored water potential profiles, the soil water retention curve and the soil hydraulic conductivity water potential relationship according to Darcy's' Law (equation 5-2).

$$q = (-K(\psi))\left(1 + \left(\frac{\Delta\psi}{\Delta z}\right)\right) \qquad \text{(equation 5-2)}$$

, where, q is flux (m.h^{-1}), K is hydraulic conductivity (m.h^{-1}), ψ is matric potential (m) and z is depth (m).

Equation 5-2 assumes that laminar flow conditions through the fine grained matrix dominate water movement. This may not apply in heterogeneous waste rock material if water can drain through large void spaces between rocks at water potentials close to saturation, leaving the bulk material relatively dry (Bennett and Plotnikoff, 1995). However, a subsoil drainage limiting layer that retards vertical water movement can change to more uniform local flow conditions that conform to the Darcian assumptions (Kuo et al., 1999). Ultimately effective control of deep drainage depends on how effective the revegetation is in evapotranspiring rainfall infiltration (Crees et al., 1994).

5.1.2.2 Evapotranspiration and soil water storage

Typical evapotranspiration rates from eucalyptus dominated savanna woodlands that occur at Ranger and extend across northern Australia range from 0.8 to 1.1 mm.day^{-1} for the tree overstorey (Hutley et al., 2001; O'Grady et al., 1999). Trees reduce their leaf area during seasonal drought to maintain relatively constant water use per unit area throughout the year (O'Grady et al., 1999). Past studies suggested that trees, shrubs and grasses obtain most of their water from the zone to 2.5 m (Cook et al., 1998) and therefore do not rely on deep groundwater in the tropical savanna environments of northern Australia (Hatton et al., 1998). A number of water use studies indicate that trees extract over 90% of their water requirement from the surface 2.5-3 m of soil (Eastham et al., 1990; Eastham et al., 1994; Kelley, 2002; Kelley et al., 2007; Schenk and Jackson, 2005; Soares and Almeida, 2001), although there is evidence that plant water use from as deep as 5 m is required in drought conditions (Kelley et al., 2007). The temporal changes in evapotranspiration from revegetation of waste rock with these native woodlands will be related to how effectively they can grow and extend their root systems to extract rainfall that infiltrates into and is retained in the landform cover system.

5.1.2.3 Temporal changes

Physical changes can occur in waste rock covers within ten years to fifty years of being constructed that alter their drainage limitation, water retention and erosion resistance properties. The dynamics are associated with fine material derived from intense weathering of waste rock freshly exposed from the mining process, root zone development, faunal activity (Taylor et al., 2003) and erosion and sedimentation in the young landscape (Moliere et al., 2002). Material physical changes and tree root growth can lead to water balance conditions that vary from when the cover was constructed. However, temporal changes in covers have not been well documented (O'Kane et al., 2003). Difficulties arise in evaluating temporal changes in covers because physical measurements on intact cores in waste rock are difficult to obtain and direct measurements of root depth are tedious, costly and inconclusive. Consequently, indirect methods of investigating soil physical changes and root growth are preferred.

Root growth is linked to soil water extraction in various studies (Cook et al., 2002; Dunin et al., 2001; Kelley, 2002; Kelley et al., 2007; Murphy and Lodge, 2006; Rose and Stern, 1967). Repeated measurements of water content and water potential profiles have been used to establish root depth over time from the depth to which significant seasonal decrease in soil water content occurs (Dunin et al., 2001). Murphy (2006) qualified measures of significance and measured profile water content changes in agricultural soil to within ±0.01 $m^3.m^{-3}$ using neutron moisture meters. However, the information on root growth that can be derived from repeated soil water content measurements in waste rock could be confounded by the fact that the soil water characteristic or water retention curve (the amount of water retained in soil at matric potentials between 0 and –10 metres) depends on capillarity and pore size distribution (Cresswell and Paydar, 1996), which may change over time as the primary rock material weathers rapidly. These physical changes in waste rock covers could be assessed indirectly by monitoring changes in the soil water characteristic describing soil water retention.

Soil water characteristic curves can be determined from *insitu* measurements of soil water content and soil water potential profiles (Rose and Krishnan, 1967) using a range of continuous monitoring methods. Repeated measurements of soil water retention over time can then be used to indicate physical changes induced by nascent soil forming processes in the waste rock material. This is in a sense the reverse problem to that encountered in developing pedotransfer functions used to estimate soil hydraulic properties from texture and other cheaper, more commonly measured parameters (Schaap et al., 2001; Schaap et al., 2004). For the purpose of the current study, knowledge of change in the water retention curve is key information for inferring the underlying soil texture changes.

A range of functions can be used to describe the drainage water retention curve (Schaap et al., 2004). The two parameter power model of Campbell (1974) is relatively easy to fit to water retention data and produces comparable results to other mathematical functions fitted to the draining water characteristic in soils (Brooks and Corey, 1966; Campbell, 1974; van Genuchten, 1980; van Genuchten and Nielsen, 1985) and waste rock materials (Fredlund and Xing, 1994). The Campbell model (equation 5-3) also has physically meaningful parameters that are related to the pore size distribution.

$$\Psi = \Psi_e \left[\frac{\theta}{\theta_s} \right]^{-b} \qquad \text{(equation 5-3)}$$

, where ψ_e is the air-entry potential (m), θ is the volumetric soil water content, θ_s is volumetric water content at field saturation ($m^3.m^{-3}$) and b is a fitted constant. If the water retention function can be represented by equation 5-3 then the hydraulic conductivity is given by equation 5-4.

$$k = k_s \left[\frac{\theta}{\theta_s} \right]^{2b+3} \qquad \text{(equation 5-4)}$$

, where k_s is the saturated hydraulic conductivity. The change in the fitted parameter, *b*, over time can be analysed statistically by testing for co-linearity (homogeneity of slopes) using an analysis of variance. However, the initial rate of change in hydraulic performance is likely to be relatively rapid in response to rapid weathering conditions.

5.1.3 *Ranger waste rock*

Weathering processes change the physical properties of primary, or unweathered, waste rock at Ranger once it is exposed at the land surface (Fitzpatrick et al., 1989; Milnes, 1989). The fines created by weathering (silt and fine sand) reduce infiltration and the large residual particles form a protective gravel lag that reduces the erosion rate. Below the compact surface layer, bulk density decreases and within the first two years of waste rock placement rocky soils form with gravel lags and vesicular crusts over less dense layers with an accumulation of translocated fine material (Fitzpatrick, 1986). Smectite, kaolinite, haematite and ferrihydrite are common weathering products from the schists (Milnes and Fazey, 1988). Even though the sulfide levels are very low in waste rock (on average < 0.04% S) values as high as 3.51% S have been measured in individual samples exhibiting acid weathering features (Milnes and Fazey, 1988). Under these circumstances, there was evidence that uranium can be mobilised in solution and discharged via seepage (Richards, 1987).

Emmerson and Hignett (1986) measured water retention, bulk density, infiltration rate and rock density of mine soil profiles at Ranger. Infiltration into recently placed, track compacted, waste rock ranged from 1 to 30 $mm.hr^{-1}$. Saturated infiltration rate (reached after one hour) ranged from 1.1 to 2.8 $mm.hr^{-1}$. The particle density of unweathered rock ranged from 2 410 to 2 650 $kg.m^{-3}$, typical of silicate rocks, while the dry bulk density of waste rock that had been track compacted and left for several years was 2 380 $kg.m^{-3}$. This dense condition needs to be ameliorated by deep ripping to create a medium that promotes infiltration and plant root growth.

The initial erosion rate off the waste rock landform is likely to be higher than in the surrounding landscape (Riley, 1995b). Simulation modelling of an eroding waste rock stockpile at Ranger without vegetation (Willgoose and Hancock, 1998) suggested that gully erosion of between 7 and 8 metres could occur on the structures within 1,000 years. However, vegetative cover on the same landform could reduce erosion to about 5.8% of that (Evans et al., 1999; Evans et al., 1998). Evidence from other sites in the region indicate that runoff and erosion rates equilibrate to those of the natural land surface within 50 years (Hancock et al., 2000). Natural erosion rates in the region around Ranger mine have been estimated to be 40 ± 5 $mm.Kyr^{-1}$ (Wasson, 1992).

The dissolution rate of weatherable minerals tends to decline over time as source terms diminish concomitant with an increase in the relative concentrations of resistant and stable mineral end products (Morin and Hutt, 2001). As a consequence, solute loads also decrease over time and solution composition tends towards an equilibrium that reflects background conditions. Observations of a rapid decline in dissolved uranium load over three to five years from mineralised waste stockpiles in the catchment to retention pond number 4 at Ranger mine support this view (Figure 5-2). Perturbations in the declining trend in

Figure 5-2 were associated with earthmoving activities, which were assumed to have exposed fresh weathering surfaces in the stockpiled material.

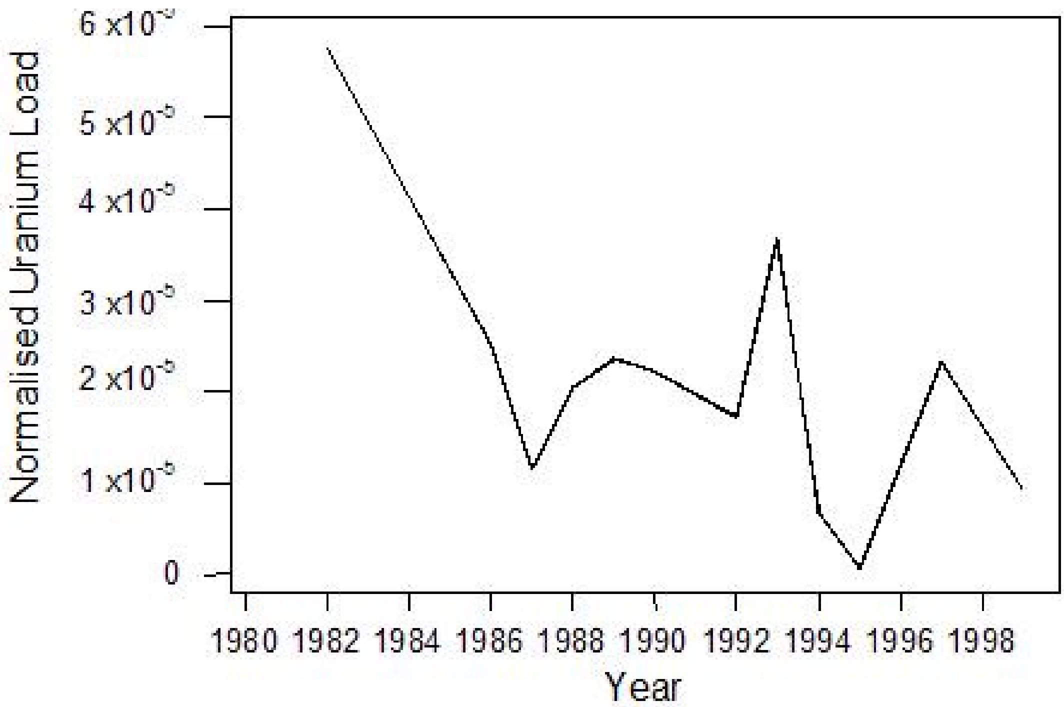


Figure 5-2 Changes in annual U load normalised with respect to total rainfall (mm) and waste rock storage (m3) in stockpile seepage over time

Apart from changes in material properties over time, drainage flux varies principally with year to year rainfall variations. Consequently, monitoring studies over one or several years may not be an accurate estimate of long-term drainage. Numerical water balance simulation models run over the historical rainfall record (or synthetic rainfall records where there is less than 20 years of historical data) is required to make a risk based impact assessment of cover design options, principally cover thickness and hydraulic conductivity (Ho et al., 2002). Field trials in which the performance of selected cover options can be monitored under actual field conditions are used to support numerical modelling (O'Kane et al., 2003), which is also subject to uncertainty due to parameterisation and model definition.

5.1.4 *Objectives*

The objectives of this chapter were to measure the water balance and solute balance of a waste rock cover trial with aims to: (i) assess the sensitivity of deep drainage to cover thickness (the main design parameter available to the mining operation); and (ii) identify temporal changes due to plant root growth and rock weathering processes. The trial was an example of a store and release cover variant type III in the basic cover design suite shown in Figure 5-1. While the design did not refer to landform or soil conditions in natural analogue areas, the trial indicated what specifications mining operations could meet for constructed covers.

5.2 Methods

5.2.1 *Waste rock stockpile construction*

The stockpile of mineralised waste rock material and the barren waste rock cover material was completed in 1999 and 2000. The waste rock materials in the stockpile consisted of chlorite schist, pegmatite and quartzite (U_3O_8 < 0.12%) placed in 10 metre high lifts. Batter slopes were at an angle-of-repose (slope approximately 1:1.4). The barren waste rock cover (less than 0.03% U_3O_8) on the stockpile was considered at the time to be part of the final landform. The cover was partially revegetated in January 2001 with endemic plant species using tube stock planted in 100 m^2 islands covering 10% of the area. The plant species used to create isolated patches of vegetation were propagated from seeds and cuttings taken from the Georgetown analogue area that was described in Chapter 4.

The materials in the stockpile cover consisted initially of a drainage limiting layer that comprised oxidised barren waste, namely weathered schist and pegmatite from the highly oxidised, near surface bench extracted in the mining process. A single layer of this material was paddock dumped over mineralised waste then spread and track compacted. Finally, barren unweathered or primary rock material was placed in a 0.5-1 metre thick surface layer to confer erosion protection. A shallow broad trapezoidal channel with a gradient of 1:2 000 was constructed to direct drainage to a flume, which discharged at the natural ground surface 25 metres below. The covered landform was instrumented in January 2001 to monitor runoff volume, drainage volume and water quality. The catchment area and monitoring sites are illustrated in Figure 5-3.

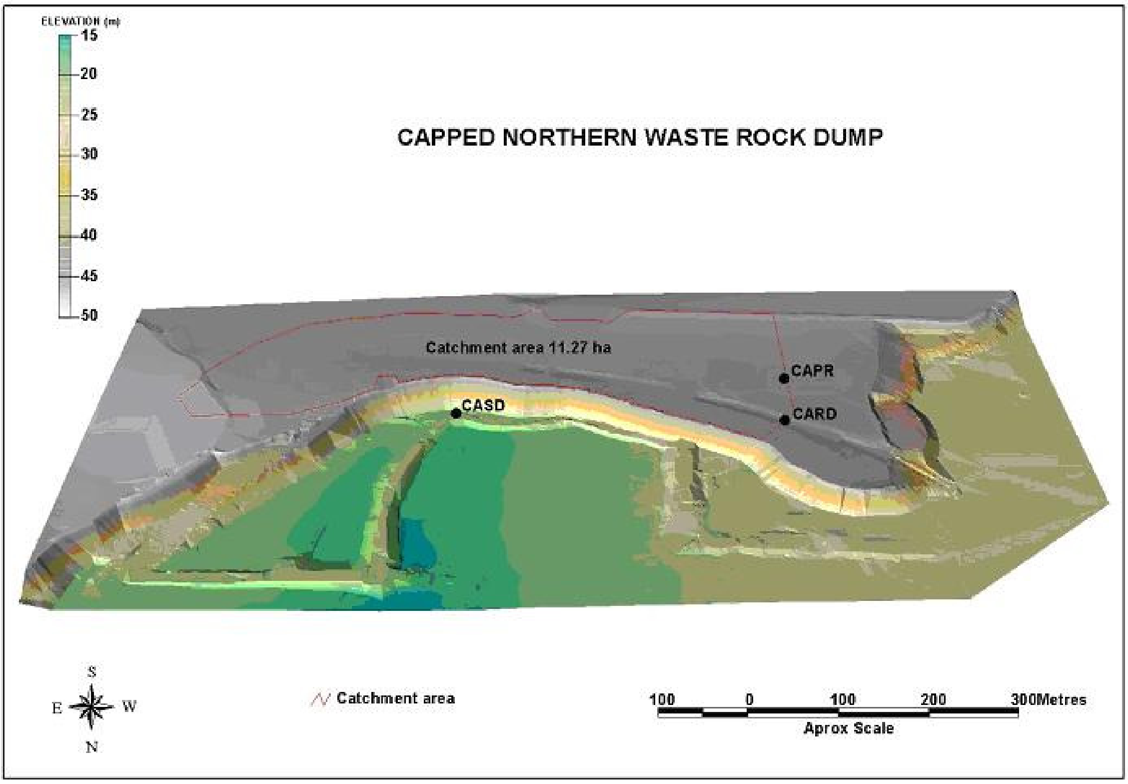


Figure 5-3 Landform cover trial area viewed from the north showing the monitoring sites and the catchment area (CASD – catchment area seepage drain monitoring point; CARD – catchment area runoff drain monitoring point; CAPR – catchment area profile monitoring and weather station)

5.2.2 *Environmental monitoring*

The catchment area depicted in Figure 5-3 was instrumented in January 2001 to monitor the water balance, with the following installations:

- runoff monitoring in the covered area runoff drain (CARD): Parshall flume measuring catchment runoff with a flow proportional sampling system
- seepage monitoring in the covered area seepage drain (CASD): V-notch weir measuring catchment seepage with an automated time-step (7 hourly from flow initiation) sampling system

- climate monitoring of the covered area profile revegetation (CAPR): a weather station measuring, rainfall, solar radiation, air temperature & relative humidity
- soil profile monitoring (CAPR): sensors to measure soil profile water content, soil profile water potential in revegetated and bare areas and near surface soil temperature.

5.2.2.1 Cover characterisation

The physical properties of the cover materials were characterised in terms of particle size, bulk density and water retention curves from cores taken during installation of soil moisture and water potential monitoring equipment at site CAPR. The cores were collected by driving a 50-mm diameter stainless steel pipe into the ground. The cover profile comprised four horizons, namely:

horizon 1 (0.0 - 0.3 m): a surface layer of barren (non-mineralised) hard rock with low hydraulic conductivity

horizon 2 (0.3 - 0.8 m): a sub-surface layer of barren hard rock with high hydraulic conductivity

horizon 3 (0.8 - 1.1 m): a sub-surface drainage limiting layer of track-compacted, weathered rock with lower hydraulic conductivity

horizon 4 (1.1 to 2.0 m): mineralised rock with the same hydraulic properties as horizon 2 above and extending for 20 m to the base of the stockpile.

The sampling design involved taking 24 cores on a one-metre grid pattern across bare and vegetated patches at the CAPR site (Figure 5-3). Soil water content and soil water potential sensors were installed in the core holes at depths designed to monitor each of the four horizons. Core sections obtained for each horizon were bagged, weighed and sub-sampled to determine the moisture content and the dry bulk density of each layer. Saturated hydraulic conductivity (K_s) was measured in triplicate the 50-mm diameter core holes for horizons 1 & 2 with a borehole permeameter (Talsma, 1987).

Infiltration rate through the surface was measured with a 1-metre diameter single ring infiltrometer (Marshall et al., 1996) in triplicate in the field. The infiltration ring dimensions were relatively large (approximately ten times the largest particle diameter) to reduce the effect of heterogeneity in the waste rock material on the infiltration measurement. A bentonite putty seal was formed on the inside of the rings to prevent lateral leakage. The infiltration ring was filled with water to a depth of 50 mm and infiltration rate was measured on an inclined ruler from the rate of head loss over time. Infiltration tests were run for two hours or until consecutive half-hourly measurements indicated that the rate was constant.

To characterise the water retention curves of the drainage limiting layer (horizon 3), 12 undisturbed, 75-mm diameter and 50-mm deep core samples were collected using a Tanner sampler (McIntyre and Loveday, 1974). Horizon 3 was accessed from the bottom of an excavated pit. The core samples were tested in the laboratory (Soil Water Solutions, 45a Ormond Avenue, Daw Park, South Australia, 5041). Bulk density cores were saturated in a tank of 0.01 M $CaCl_2$ solution for 24 h. Saturated hydraulic conductivity (K_s) was then measured at a constant 50 mm head. Cores were taken from the tank and weighed to obtain saturated weight, placed on porous ceramic plates set at a water potential of -5 kPa and then weighed again after 24 h. The equilibration and weighing process was repeated at water potentials of -10, -30 and -70 kPa. After equilibration, each core was oven dried and weighed to determine the dry soil bulk density and the gravimetric water content.

5.2.2.2 Climate and soil profile monitoring

A continuous logging system was installed to monitor the soil water content and water potential profiles as well as climate in terms of rainfall and evapotranspiration at the CAPR site (Figure 5-3) with the aim of estimating these terms and deep drainage in the water balance (equation 5-1). To achieve this two soil profiles were instrumented to measure soil water content (θ_v) and soil water potential (Ψ). One profile was in a bare area and the other in a revegetated area ten metres distant. The revegetated area had been ripped to 0.3 metres and planted in December 2000 with nursery grown plants that originated from the *Georgetown* analogue area described in Chapter 4.

Climate and soil profile monitoring installations are depicted in Plate 5-1. Cables to soil water content sensors, soil water potential sensors and the pluviometer were placed in conduit, and plastic enclosures were put around any exposed wiring to prevent damage by wild dogs (Plate 5-1e). Data logging and atmospheric monitoring equipment were strapped to a steel post (Plate 5-1f). The monitoring system had the following components:

- soil temperature sensor at 120 mm below the soil surface
- air temperature sensor
- relative humidity sensor with radiation shield
- solar radiation sensor (Middleton SK01-D pyranometer)
- Monitor Systems pluviometer with a 0.2 mL tipping bucket
- datalogger
- Magpie software (Measurement Engineering) for system and data management
- base station for gypsum blocks
- 4x gypsum block field stations (MEA2175) with lightning protection,
- 8x GBLite gypsum blocks (10 kPa to 200 kPa measurement range)
- 8x GBHeavy gypsum blocks (60 kPa to 600 kPa measurement range)
- 8x Theta probes (Delta-T, Type ML2x, dielectric measurement of soil water content ± 5% accuracy).

Daily potential evapotranspiration was calculated from continuously logged (15 minute interval) climate data using the Penman–Monteith formula for a standard reference crop (Grayson et al., 1997). All the equation parameters, apart from wind run, were measured at the CAPR site. Wind run measurements were obtained from a monitoring site 200 metres away (site OB3), which was maintained by Ranger mine.

Daily and hourly rainfalls were calculated using a pluviometer with a 50-mL tipping bucket. The pluviometer was placed 0.8 m above the ground to avoid rain splash from the land surface. Rainfall data from site OB3 (200 metres away) augmented these measurements when the cable to the pluviometer was damaged for periods from the 1st to the 25th January and between the 22nd February and the 1st March.

Gypsum blocks (*GBLite* and *GBHeavy*) for measuring soil water potential and *Theta* probes for measuring soil water content were installed in separate holes at the same four depths representing the near surface (horizon 1, 0.2 m), above the drainage limiting layer (horizon 2, 0.8 m), within the drainage limiting layer (horizon 3, 1.2 m) and below the drainage limiting layer (horizon 4, 2 m). Pairs of *GBLite* (10- 200 kPa range) and *GBHeavy* (60 – 600 kPa range) gypsum blocks were installed in separate holes at each depth to measure a combined soil water potential range from 0 to -600 kPa.

Good soil contact for the gypsum blocks was ensured by placing and surrounding them with a mixture of silica flour (<40 μm) and diatomaceous earth. Water was then poured into the installation hole to improve the contact with this material. Where installation holes penetrated through the drainage limiting layer (horizon 3) and into the mineralised rock underneath, sodium bentonite powder was used to reinstate horizon 3 and to ensure that any preferred pathways along the installation core hole were blocked.

Good soil contact for the measurement surfaces of the *Theta probe* sensor (four 60 mm long rods, Plate 5-1d) was ensured by inserting them into a bolus of moist fine material from the depth at which they were to be installed and the probe placed at the base of the installation hole. Sufficient fine waste rock material (< 2 mm) was poured into the hole to cover the probe completely, and then the hole was backfilled with the remaining extracted waste rock.

The drainage flux during the monitoring period from 25 January 2001 to 11 May 2005 was calculated continuously according to Darcy's Law (equation 5-2) from water potential gradient, unsaturated hydraulic conductivity and the water retention function (Cresswell and Paydar, 1996). Water potential and water content monitoring for horizon 2 (above drainage limiting layer), horizon 3 (drainage limiting layer) and horizon 4 (below drainage limiting layer) results were used in the calculations. Head gradients measured from water potential monitoring above and below a drainage-limiting layer combined with $K(\Psi)$ for this layer were used to calculate flux on an hourly interval. The maximum soil water content recorded above the drainage limiting layer (horizon 2) was used to calculate the field saturated water contents (θs) (equations 5-3, 5-4) each year for the bare and vegetated treatments.

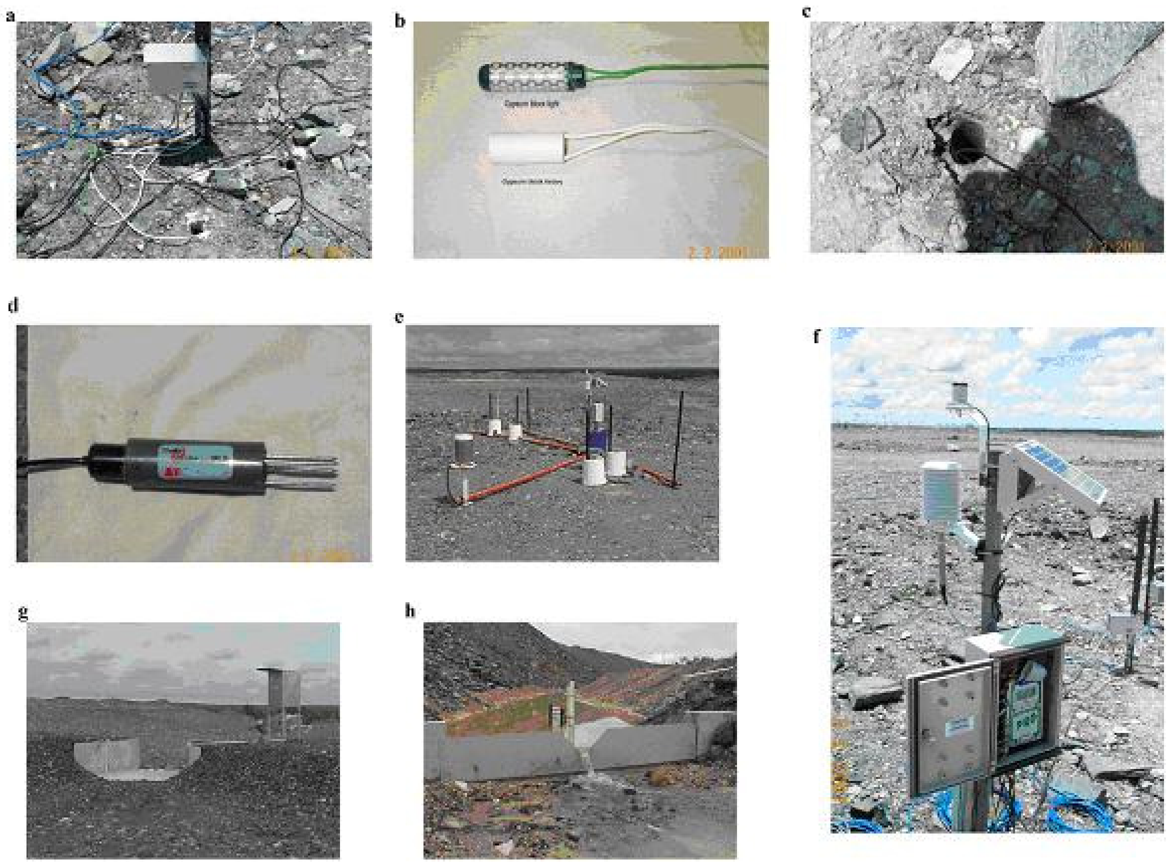


Plate 5-1 (a) gypsum block field station; (b) light and heavy gypsum blocks; (c) installation hole; (d) installation at site CAPR, (e) soil moisture (Theta) probe, (f) weather station; (g) Parshall flume installation monitoring runoff at site CAPR; (h) V-notch weir

The Campbell water retention curve parameters (equation 5-4) for the drainage limiting layer (horizon 3) were fitted to the data for each year to account for possible changes in water retention properties over time. Unsaturated hydraulic conductivity was calculated from the water retention function (equation 5-6) and used with the hourly measurements of hydraulic gradient between horizon 2 and horizon 4 (Δz = 1 m) in the Darcy equation (equation 5-2) to calculate drainage. Observed horizon thicknesses were used in the model. Fluxes estimated in this way close to saturation were cross-checked against catchment water balance measurements — estimating the drainage flux by difference from the water balance equation, using measured runoff and evapotranspiration.

Catchment runoff was measured with a Parshall Flume (Grant and Dawson, 1995) — a self-cleaning, gauged flow metering structure with well known flow characteristics accurate to ± 1 mm of measured flow – at site CAPR (Figure 5-3). The Parshall flume was designed with a 6 feet throat diameter to accommodate 1:20 year storm events for the time of concentration of the experimental catchment. The flume is depicted in Plate 5-1g. Drainage from the catchment was measured directly using a V-notch weir (Plate 5-1h) in the stockpile toe drain at site CARD (Figure 5.3). The drainage monitoring at this site was confounded to some extent by the lag between infiltration in the study catchment (25 m higher in elevation) and extensions of the drainage catchment outside of the surface catchment.

A one-dimensional unsaturated-saturated water balance simulation model SWIMv2.1 (Verburg *et al*. 1996) was used to predict the long-term drainage flux, and the sensitivity of this parameter to variations in properties of the cover was assessed. The following model boundary conditions were used in the model for bare ground:

i. initial soil matric potentials in the model were set from monitoring data
ii. no surface ponding was allowed
iii. at the bottom boundary (*i.e.* below the clay layer), seepage with a threshold matric potential of –100 kPa through time was specified
iv. the soil surface was bare (*i.e.* water losses through the surface were by evaporation and there was no effective transpiration by plants).

The top boundary conditions were revised for the revegetated cases and a simple power function was used to describe runoff to account for surface water storage in a ripped and revegetated land surface. Conversely, all surface water was sent to run off for the un-vegetated scenarios.

The bottom boundary condition ensured that water draining through the clay layer was lost to the system. This was consistent with measured water potentials and represented the situation in waste rock where water moves rapidly to depth and there is negligible upward moisture flux once the surface dries.

Water retention and hydraulic conductivity parameters for horizons in the cover were inferred from particle size measurements, bulk density and hydraulic conductivity measurements using the pedotransfer functions in the Rosetta program (Schaap et al., 2001).

5.2.2.3 Modelling scenarios

The sensitivity of annual drainage flux to different cover design scenarios was assessed using a one-dimensional, SWIMv2, model run over 22 years of historical rainfall records from 1980 (the full range of the available rainfall data). This stochastic assessment was based on the frequency distributions of annual drainage flux for the following cover configurations:

a. the current cover configuration, bare of vegetation
b. surface compaction of the current cover configuration to a surface bulk density of 2 000 $kg.m^{-3}$
c. a revegetated land surface with a 1.0-metre thick drainage limiting layer
d. a revegetated land surface with a 2.0-metres thick drainage limiting layer
e. the current cover configuration, revegetated.

The properties of the surface 0.3 m horizon (horizon 1) for the second cover scenario (b) were taken from reported measurements on compacted surface materials (Emerson and Hignett, 1986). A dry bulk density of 2 380 $kg.m^{-3}$ corresponding to a total porosity of 0.0876, with a measured infiltration rate of 1.08 $mm.hr^{-1}$ was used. Values for θ_s (saturated water content and θ_r (residual water content) were set at 86% of the total porosity and one third of the wilting point water content respectively for this modified case, following practical convention (Sorooshian et al., 1999). The volumetric water content at wilting point (1.5 MPa suction) for this material was 0.028 $m.m^{-1}$ (Emerson and Hignett, 1986).

The cover scenarios (c) and (d) above incorporated two vegetation layers. Layer one, represented woodland overstorey, extracting soil water down to 2.5 metres, while layer two represented shallow rooted grasses and shrubs extracting water down to 1.0 m. These water use patterns were based on published water use rates for native vegetation in the Darwin region (Hutley et al., 2000; Kelley, 2002).

5.2.3 *Measuring temporal changes in water retention characteristics*

The Campbell soil water retention function (equation 5-4) was fitted using least squares regression to draining soil water monitored over time at the profile monitoring site (CAPR). A statistical test of the homogeneity of slopes of the fitted lines was made using the Generalised Linear Modelling analysis of variance in the Minitab 14 statistical package. The statistical design was specified in terms of $ln\ (\theta)$ as the response variable and $ln\ (\Psi)$ as a covariate measurement nested within the surface treatment (bare or vegetated) and the year of monitoring. Surface treatment and year of monitoring were treated as crossed factors in the model. Monitoring data from 2005 did not include sufficient periods of soil drainage to fit a water retention function and was omitted from the analysis.

5.2.3.1 Stream flow monitoring

The sampling stations (CARD, CASD) were visited each week and, where possible, during storm events. The number of samples taken during any sampling period using ISCO automatic water samplers (carousel with 24 sample bottles) varied with the duration of the sampling period, the flow during the period and the selected pre-set volume step (CARD) or time step (CASD) between each individual sample. Individual autosampler bottles were unloaded from the carousels and then combined and mixed in the Ranger environmental laboratory to make up weekly composite samples. A grab sample was also taken if flow was observed.

Dissolved and particulate fractions were measured for composite and grab samples. For the solute and sediment load calculations, total load (L_T) was approximated by:

$$L_T = \sum_{1}^{n} C_n V_n \qquad \text{(equation 5-5)}$$

For station CARD, $V_1 = V_2 = V_3 \ldots = V_n = V$, where V is the preset volume step for triggering the sampler. Thus, the total load (L_T) passing the station for *n* samples was:

$$L_T = V\sum_{1}^{n} C_n \qquad \text{(equation 5-6)}$$

At station CASD, the volume step varied depending on the flow rate during the time interval between samples. The total load (LT) passing the station was estimated from the product of the concentration in the composite sample, C, and the volume flow between site visits, V, using the equation:

$$L_T = \sum_{1}^{n} C_n V_n \qquad \text{(equation 5-7)}$$

Loads calculated from time increment sampling at CASD incorporated sampling errors where the volume increment was too large to define the flow curve accurately. Major ions (Ca, K, Na, Mg, SO_4, SiO_2) were analysed by ICP-OES and trace metals by ICP-MS. A sub-sample was taken for dissolved trace metal analyses, filtered (< 0.45 µm) and acidified to pH < 2 with concentrated nitric acid (HNO_3). A second sub-sample was taken for general chemistry (pH, EC, TSS, alkalinity). A third sub-sample was taken, filtered (<0.45 µm) and used to measure dissolved nitrogen. Sub-samples for dissolved nitrogen (NH_4 and NO_3) analyses were stored at below 4°C. Total suspended sediment (TSS) was measured gravimetrically using glass fibre filter disks (<0.45 µm). Major ions and trace metals in suspension were analysed from *aqua regia* digests of TSS (total suspended sediment) samples. Dissolved loads were calculated by integrating measured concentrations over the volume increment being sampled in each composite sample.

5.3 Results

5.3.1 *Climate*

The rainfall record from January 2001 until April 2005 (Figure 5-4) depicted seasonal patterns and rainfall intensities to 140 mm.day^{-1}. Relative humidity, air temperature and soil temperature, and global solar radiation also showed strong seasonal patterns. As shown in Figure 5-4, the 2003 wet season was relatively dry. Annual climate variability qualifies the results of short-run monitoring studies such as this.

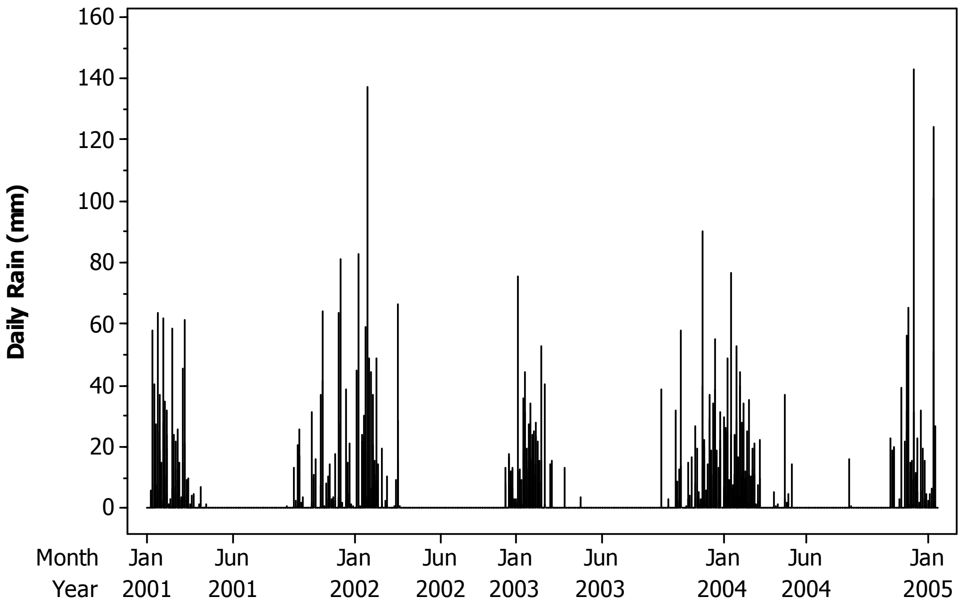


Figure 5-4 Daily rainfall 2001 to 2005

5.3.2 *Cover characterisation*

The measured thickness of the erosion resistant rock layer varied from 0.5 to 1.5-m while the thickness of the drainage-limiting layer (horizon 3) varied between 0.1 and 0.5-m. Horizon 3 had a relatively low clay content, around 5% (Table 5-1). This combined with the variable thickness (design thickness was 0.5-m) reduced its effectiveness as a drainage-limiting layer.

Table 5-1 Particle size analyses for horizon 3

Particle size fraction[1]	**Drainage limiting layer % Passing**	**Rock cover % Passing**
coarse gravel	8	13
medium gravel	29	30
fine gravel	18	20
coarse sand	13	14
medium sand	9	9
fine sand	8	6
silt	9	7
clay	5	0

[1]particle density = 2.64 $t.m^{-3}$

Soil physical characteristics of each cover horizon are shown in Table 5-2. Water retention at 700 kPa was estimated from bulk density, rock content and gravimetric water content of the fines (< 2 mm). The hydrological properties for horizon 3 the material comprised sparsely distributed macropores with a large effect on hydraulic conductivity near saturation.

Table 5-2 Physical and hydraulic property measurements

Hydraulic property (mm)		**Symbol**	**Units**	**Horizon 1 Cover rock**	**Horizon 2 Cover Rock**	**Horizon 3 Drainage limiting layer**
infiltration rate		I	cm.h-1	0.3		
saturated hydraulic conductivity		Ks	cm.h-1	0.125	100	0.125
saturated water content		θs	m3.m-3	0.30	0.47	0.46
bulk Density		ρb	t.m-3	1400	1190	1630
specific gravity		ρs	t.m-3	2640	2640	2630
clay percentage			%	0	0	5
rock percentage			%	63	63	55
water retention	0 kPa					0.46
	5 kPa	Ψ_5	$m^3.m^{-3}$			0.31
	10 kPa	Ψ_{10}	$m^3.m^{-3}$			0.28
	30 kPa	Ψ_{30}	$m^3.m^{-3}$			0.26
	70 kPa	Ψ_{70}	$m^3.m^{-3}$			0.23
	700 kPa	Ψ_{700}	$m^3.m^{-3}$			0.16

5.3.3 *Water balance*

The runoff coefficient (runoff as a proportion of rainfall) of the capped landform increased from 53% in 2001 to 78% in 2002 (Table 5-3). The 2002 season was relatively dry and the wet season monitoring in 2001 was incomplete.

Table 5-3 Runoff coefficients for 2001 and 2002 seasons

Measure	Units	2001	2002
rainfall	mm	936	747
runoff	mm	338	226
seepage	mm	290	65
evapotranspiration	mm	308	456
catchment area	m^2	112 700	112 700
runoff coefficient	%	53	78

The water retention curve parameters used in the hydrological input file for the SWIM program are shown in Table 5-4. In 2002, the mining operation applied a high energy compaction treatment to approximately 30% of the area to reduce infiltration though into mineralised waste. However, macropore and matrix parameters are provided for horizon 3 and input parameters for the compacted horizon 1 are given in Table 5-4.

Table 5-4 SWIMv2.1 hydrological input parameters

					Hydraulic conductivity		**Water retention**			**macro-porosity**
Material				**p1**	**p2**	**p3**	**p1**	**p2**	**p3**	**p1**
	θr $m^3.m^{-3}$	**θs** $m^3.m^{-3}$	**Ks** $cm.hr^{-3}$	**p**	**n**	**m**	**α**	**n**	**m**	**αex**
horizon 1	0.0321	0.2952	0.7	-1.208	1.5992	0.3747	0.0452	1.5992	0.3747	
horizon 1#	0.0375	0.2094	0.018	-1.2427	1.5076	0.3367	0.12131	1.5076	0.3367	
horizon 2	0.0348	0.4597	100	-0.7749	1.8043	0.4458	0.0508	1.8043	0.4458	
horizon 3										
matrix	0.0377	0.3574	0.3635	-0.3620	1.5992	0.3747	0.0102	1.5992	0.3747	
macropore			50	1						4.3
horizon 4	0.0348	0.4597	100	-0.7749	1.8043	0.4458	0.0508	1.8043	0.4458	

#compacted

The estimated drainage flux was insensitive to changes in the water retention characteristics, *i.e.* large changes in the moisture characteristics produced only small changes in the drainage flux below 2.0 metres.

The sensitivity of annual drainage flux below horizon 3 to surface compaction, revegetation and the thickness of the drainage limiting layer over the historical rainfall record is summarised in Figure 5-5. The highest range in annual drainage flux was associated with the current cover configuration. Effective revegetation of the current cover reduced the drainage flux by half (scenario *e* in Figure 5-5). Furthermore, increasing the thickness of the clay layer from 0.3 to 1 and 2 metres (scenarios *c* and *d* respectively in Figure 5-5) reduced drainage flux significantly (*$p<0.05$). The lowest annual drainage flux range was achieved for the surface compaction treatment.

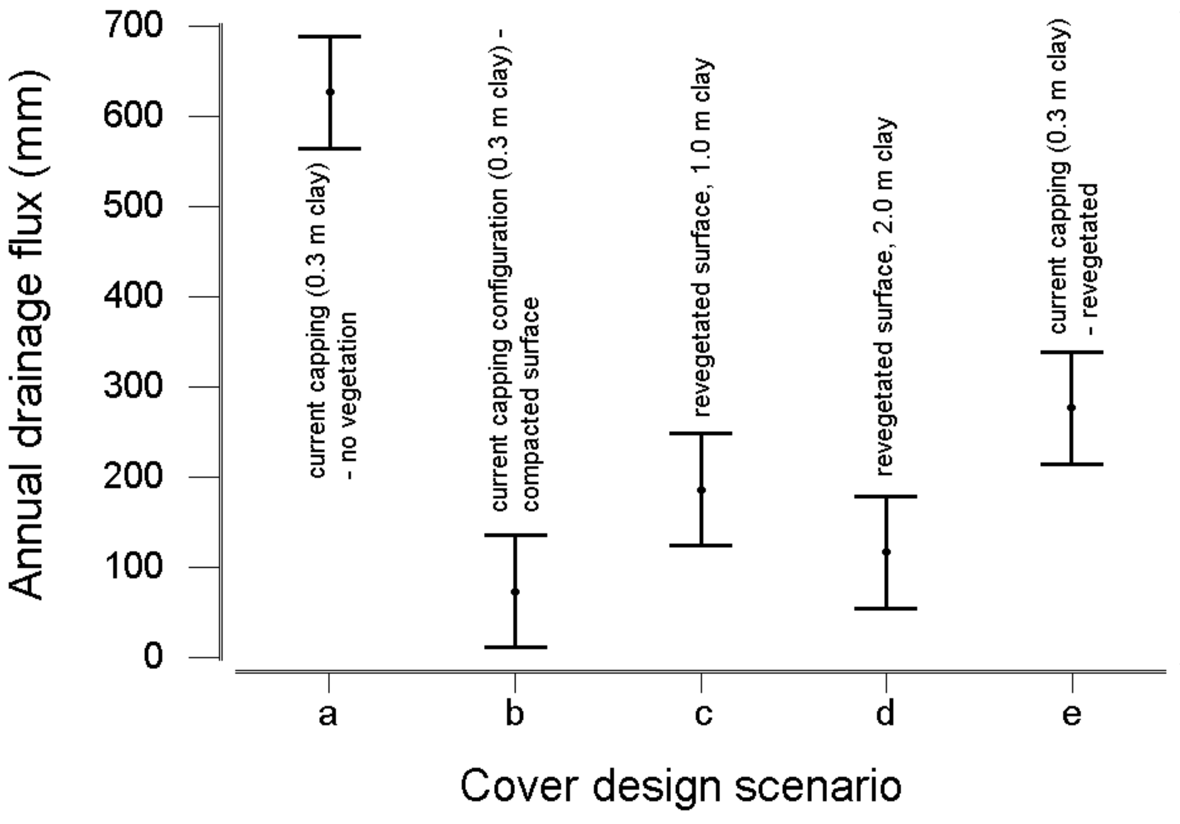


Figure 5-5 Average annual drainage flux showing 95% confidence intervals for different cover design configurations

5.3.4 *Water chemistry*

Major ion chemistry and general parameters differed between runoff and seepage water (Table 5-5). While Mg was the major cation in both runoff and seepage water, followed in that order by Ca, K and Na, SO_4 was the major anion followed by Cl, SiO_2, and Na. The major ion chemistry of the composite and grab samples was similar. Although, pH and TSS of seepage water grab samples was lower than the composite samples, alkalinity was higher in the grab samples.

Table 5-5 Major ions and general parameters in runoff and seepage water

Description	General parameters				Cations				Anions		
	pH	EC µs/cm	Alkalinity	TSS mg/L	Ca mg/L	Mg	K	Na	SiO_2 mg/L	Cl	SO_4
Runoff											
Composite Samples											
Average (n=20)	0.0	27	4	540	0.9	14	0.5	0.3	4	8	52
Maximum	0.0	40	6.1	1872	4	85.4	1.7	1.2	11.5	28	345
Minimum	3.50	2	2.72	56	0.2	2.5	0.3	0.1	1.7	5	5.5
Grab Samples											
Average (n=11)	6.4	96	4.2	388	0.6	11	0.5	0.4	4.5	5	38.7
Maximum	6.54	446	10.06	873	2.4	48.8	1.3	1.1	9.8	5	187.2
Minimum	6.23	15	2.4	183	0.2	2.5	0.3	0.1	1.7	5	5.5
Seepage											
Composite Samples											
Average (n=20)	6.0	498	3.3	583	3	63	1.5	1.0	10	11	233.7
Maximum	6.64	701	2.54	1027	4.4	100.4	2.5	1.3	14.9	31	366.8
Minimum	5.28	42	1.21	110	0.5	4.2	0.3	0.2	1.3	0.1	15.7
Grab Samples											
Average (n=11)	4.9	779	12.7	24	4.6	99	2.4	1.4	14	9	384
Maximum	5.51	1087	73.21	68	6.4	138.3	3.3	1.6	16.2	31	535.6
Minimum	4.26	442	–	–	2.8	58.5	1.2	1.1	9.6	1.2	217.9

Trace metal and nitrogen ion concentrations in runoff and seepage waters are shown in Table 5-6. While U, followed closely by Al, were the dominant trace metals in solution in both runoff and seepage, the concentration of U relative to Al was higher for the seepage water. Also, NO_3 concentrations were much higher in the seepage water.

The order of element concentrations in composite runoff samples was U>Al>Fe>Cu>Mn>Pb>Zn, and for grab samples Al>U>Fe>Mn>Cu>Zn>Pb. The order of elements in composite seepage samples was U>Al>Mn>Cu>Fe>Pb>Zn and for grab samples U>Cu>Al>Mn>Pb>Fe>Zn. According to the guidelines for protecting water quality in the receiving environment the values for U, Al, Pb, Mn and Cu in the runoff water and for Zn and NO_3 in seepage water, exceeded the acceptable levels.

Table 5-6 Trace metals and nutrient concentrations in runoff and seepage

Description	**Cu** **µg/L**	**Fe**	**Al**	**Mn**	**Pb**	**U**	**Zn**	**N-NH$_3$** **µg/L**	**N-NO$_3$**
Runoff									
Composite Sample									
Average (n=6)	96.3	839	1001	35.2	11.4	1159	6.9	50	50
Maximum	381	4899	6237	164	36.4	6635	13.3	–	–
Minimum	0.59	20	20.3	0.58	0.13	32.0	1.5	–	–
Grab Sample									
Average (n=7)	9.61[2]	328	630	17.9	0.96	498	2.5	50	50
Maximum	37.5	674	1307	64.9	1.8	2632	6.8	–	–
Minimum	1.48	20	19.6	3.82	0.33	18.7	1	–	–
Seepage									
Composite Sample									
Average (n=5)	65.5	58	117	101	18	3280	9.9	–	–
Maximum	122	210	485	163	78.84	5984	17.2	–	–
Minimum	1.15	20	11.5	1.2	0.07	40.8	1.3	–	–
Grab Sample									
Average (n=6)	603	36.5	324	203	43.2	8625	21.5	50	1154
Maximum	1196	106	595	356	86.2	15041	35.7	50	3500
Minimum	92.7	20	39.2	93	4.07	3504	8.6	50	50
Environmental guideline									
Site based trigger level[1]				11	0.33	0.3	-	-	-
ANZECC 95% trigger level	1.4		55	1900	3.4	20*	8.0	900	700

[1]downstream receiving environment monitoring point MG009

Fe and Al were the predominant trace metals in suspended sediments in both the seepage and runoff waters (Table 5-7). However, U concentrations in sediment from seepage were higher than that found in sediment sampled from runoff.

The trace metal loads in runoff and seepage for selected solutes for 2001 and 2002 are shown in Table 5-8. Uranium load was higher in seepage, while major ion loads (Mn, Mg and SO_4) were higher in runoff. There appears to be no real reduction in loads for these trace metals between 2001 and 2002.

5.3.5 *Soil profile monitoring*

The temporal variation of soil water content and water potential for horizon 1 (0 – 0.3 metres) between 2001 and 2005 for bare and vegetated plots are shown in Figure 5-6. As would be expected the soil water content and water potential were lower during the dry season than during the wet season. Also, the soil water content was consistently lower in the ripped and vegetated plot than in the bare plot. This effect was attributed to the ripping treatment during tree planting operations that caused decompaction and a coarser pore size distribution.

Table 5-7 Suspended sediment chemistry

	TSS mg/L	**Ca**	**K**	**Mg**	**Na**	**S**	**SiO_2**	**Cl**	**Al µg/L**	**Cu**	**Fe**	**Mn**	**Pb**	**U**	**Zn**
Runoff															
Composite sample (n=12)															
n	12	12	12	12	12	12	12	12	10	8	2	6	2	6	2
Average	344	0.2	1.6	16.2	0.1	0.16	3.4	0.6	12723	58	4749	50	30	12.2	141
Maximum	387	0.3	2.6	22.6	0.1	0.3	4.2	1.0	57	85	32	109	44.2	178.4	250
Minimum	300	0.1	0.7	9.8	0.1	0.03	2.7	0.2	51563	32	39454	0.57	16.5	65.7	32.8
Seepage															
Composite sample															
n	17	18	13	13	13	13	13	11	12	2	12	11	2	12	2
Average	1158	3.9	1.8	95	1.4	385	7.7	1.2	18304	19	17475	67.5	2.0	926	1.1
Maximum	7911	11.6	3.3	240	2.4	997	18.9	1.8	54819	24	60368	219	2.9	2789	1.3
Minimum	0.1	0.1	0.1	0.2	0.1	0.17	0.2	0.3	24	14	137	0.9	1.11	41	0.9

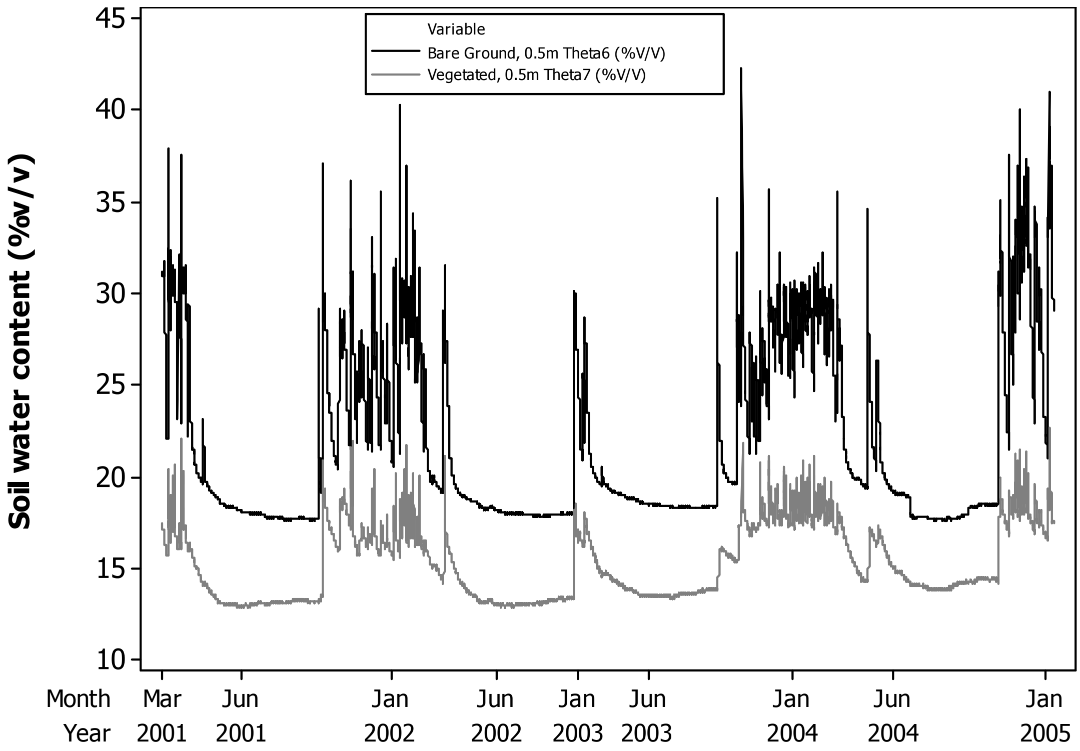
Table 5-8 Solute loads in runoff and seepage in 2001 and 2002

	U (kg)	Mn (kg)	Mg (kg)	SO4 (kg)
seepage				
2001	13.0	0.4	782	3029
2002	7.1	16.2	521	2094
runoff				
2001	2	0.05	1563	7512
2002	4	0.03	1777	6607

Soil water potential generally trended down during the dry season (Figures 5-6 to 5-9) attributable to evapotranspiration and deep drainage. Although the water potential during the 2001 dry season was relatively higher, probably due to imperfect contact between buried sensors and the surrounding soil matrix. After 2001, the wetting and drying patterns were more consistent and regular. The water potential profiles of the bare and vegetated treatment were similar in 2001 and 2002. However, the soil water potentials decreased during the subsequent wet seasons to –300 kPa under both bare and vegetated treatments. This was still in the range of drained water potentials reached under gravity. However, water potentials as low as -600 kPa in the 2003 and 2004 dry seasons in the vegetated plots indicated that plant roots had begun to extract water throughout the surface soil (Figure 5-6 to 5-8). In summary, ripping appeared to reduce fine porosity in the vegetated surface in the first season and water extraction by vegetation began to be measured after two wet seasons in horizon 1 and horizon 3.

The soil water content and water potential for horizon 2 (0.3 – 1.0 m) were consistently lower in the ripped and vegetated treatment compared with the bare treatment (Figure 5-7).

However, by the fourth wet season (2004) the maximum water contents were similar for bare and vegetated plots and by 2005 the maximum water contents in the bare treatment were less than the vegetated treatment. The maximum water content approximates the saturated water content and the porosity of the soil. Consequently, ripping and re-vegetation appears to have generated a stable porosity while the bare earth treatment has self-compacted over time leading to reduced porosity in horizon 2.



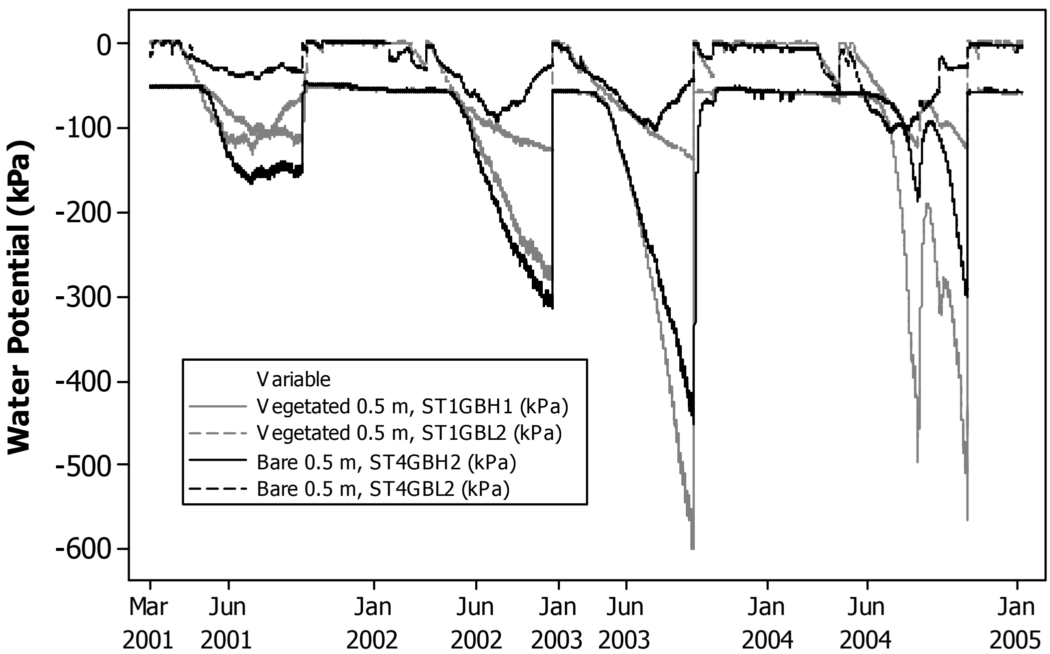
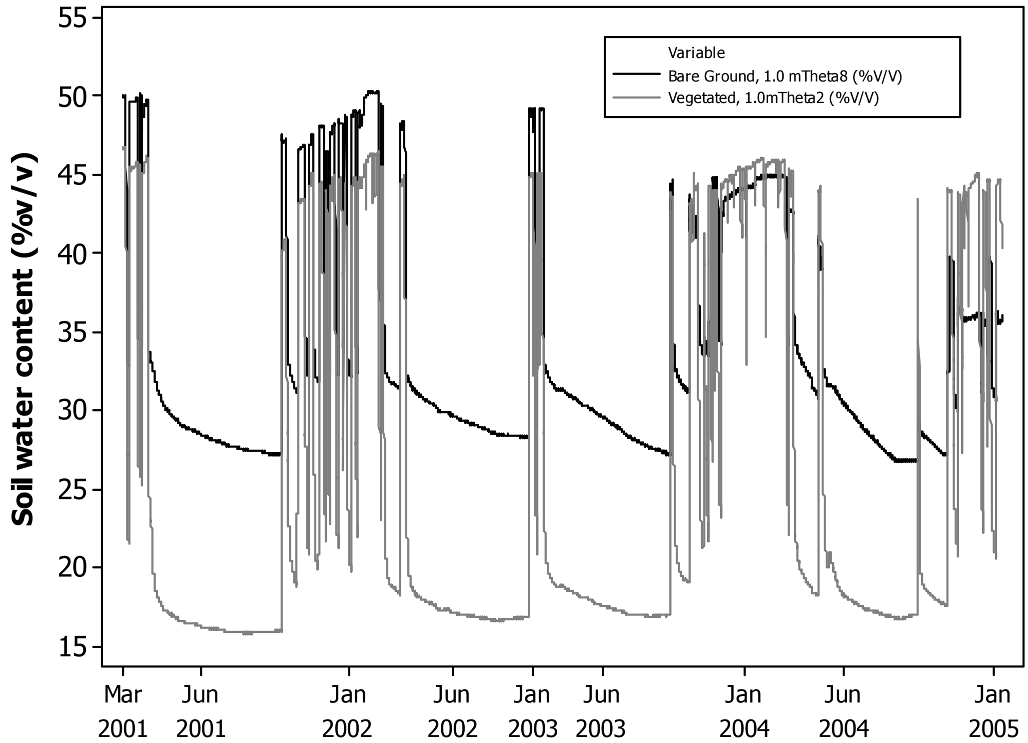


Figure 5-6 Soil water content and water potential monitoring for horizon 1 (0-0.5 m)



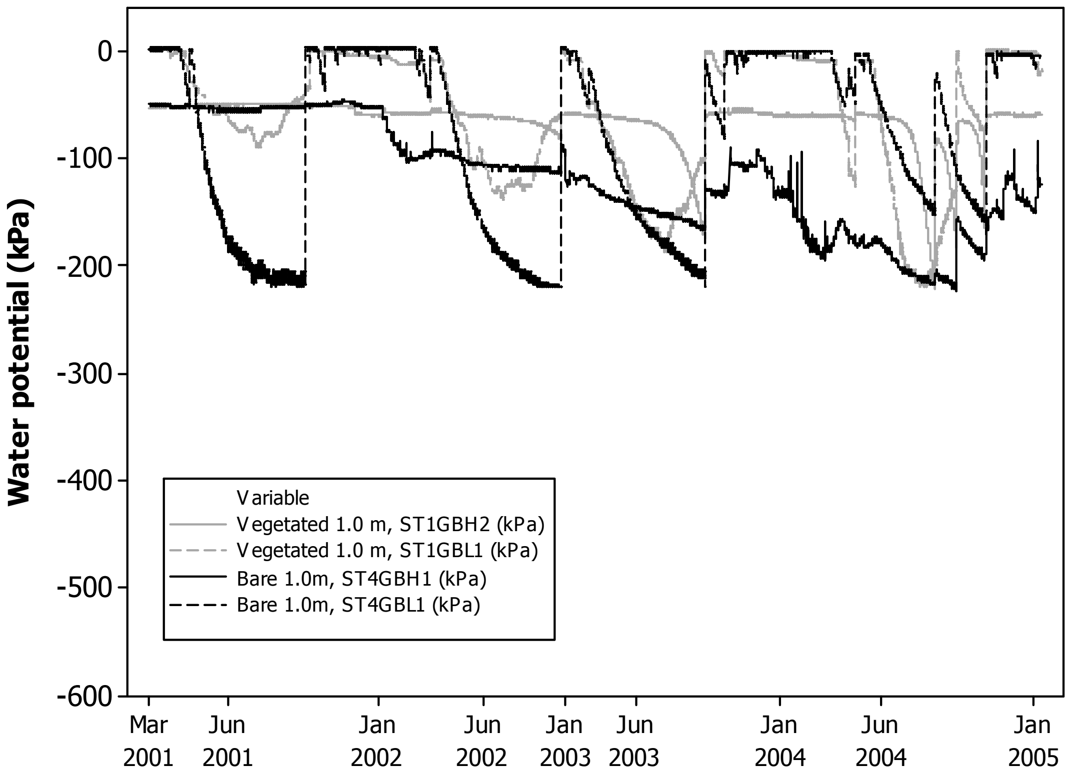


Figure 5-7 Soil water content and water potential monitoring for horizon 2 (0.5 – 1.0 m)

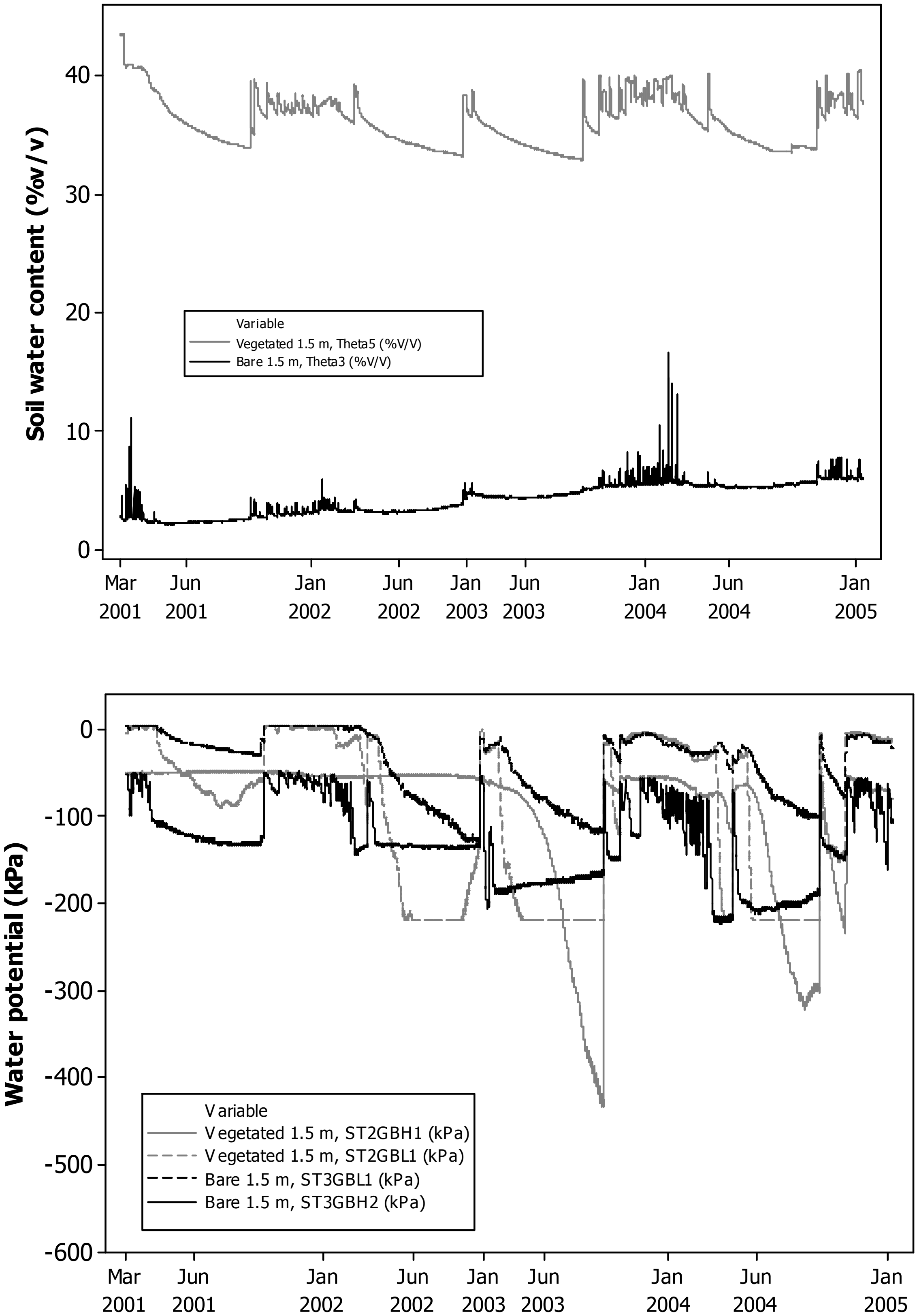


Figure 5-8 Soil water content and water potential monitoring for horizon 3 (1.0 – 1.5 m)

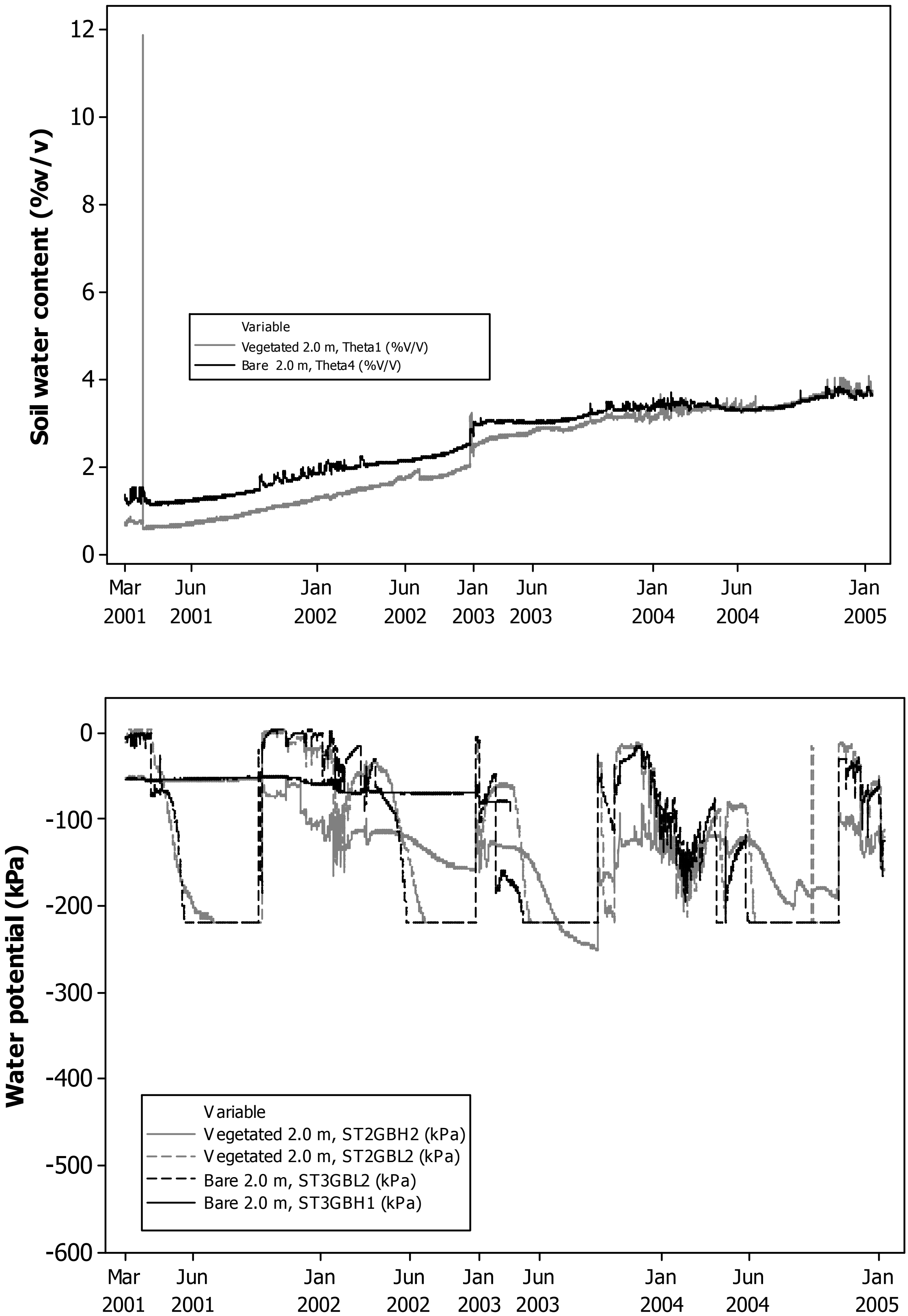


Figure 5-9 Soil water content and water potential monitoring for horizon 4 (1.5 – 2.0 m)

Seasonal water potential variation for horizon 2 indicated that the bare treatment drained freely over the monitoring period, while drainage in the vegetated treatment increased over subsequent dry seasons until potentials of -200 kPa were reached in 2004. This indicates an increased permeability over time in the underlying drainage limiting layer (horizon 3) in the vegetated treatment and a lack of effective drainage limitation from horizon 3 in the bare treatment. This difference between treatments with respect to horizon 3 was attributed to uneven construction of this layer across the trial.

The soil water content and water potential for the drainage limiting layer, horizon 3 (1.0 – 1.5 m) were much lower in the bare treatment than the vegetated treatment (Figure 5-8). Rapid macropore flow would have contributed only a small amount of water to the fine-grained matrix. Horizon 3 was poorly constructed in the bare treatment.

In summary, uneven construction of horizon 3, particularly in the bare treatment, created preferred pathways for water movement to depth. Also, the range in water potential in horizon 3 increased over time in the vegetated treatment, which indicated improved drainage and increased plant water extraction by the 2003 dry season.

There was no difference between bare and vegetated treatments in terms of the soil water content and water potential for horizon 4 (1.5 – 2.0 m) shown in Figure 5-9. Rapid water movement occurred along preferred flow through macropores. Soil water contents at this depth were low, increasing gradually over time and responding to individual rainfall events. The gradual increase in water content could indicate infilling of fine particles from the overlying horizon 3 in the vicinity of the probes. The water potential measurements show seasonal wetting and drying with some evidence by the 2003 dry season of water uptake by plant roots in the vegetated plots.

5.3.6 *Temporal change in water retention*

The water retention curves were similar for horizon 1 and horizon 2 (Figure 5-10). Water retention in the bare treatment was typical of a sandy clay loam and in the vegetated treatment sand. This difference was consistent with observations of a shallower drainage limiting layer (containing fines from oxidised rock material) in the bare treatment than the vegetated treatment. However, water retention curves for horizon 3 (Figure 5-11) were typical of a sandy clay loam for the vegetated treatment, while water retention in the bare treatment was typical of a sandier texture. From this it was concluded that the base and upper surface of the drainage limiting layer (horizon 3) were uneven across the vegetated and bare treatment plots.

Over time, the volume of water retained at high water potentials increased slightly and may be attributed to the release of fines by physical weathering of waste rock. However, the oxidised rock material used to construct the drainage limiting layer (horizon 3) had the most significant effect on soil water retention.

In the short term, barren rock produced sandy to sandy loam material. The oxidised rock used in the drainage limiting layer produced sandy loam to sandy clay loam material with higher water retention properties.

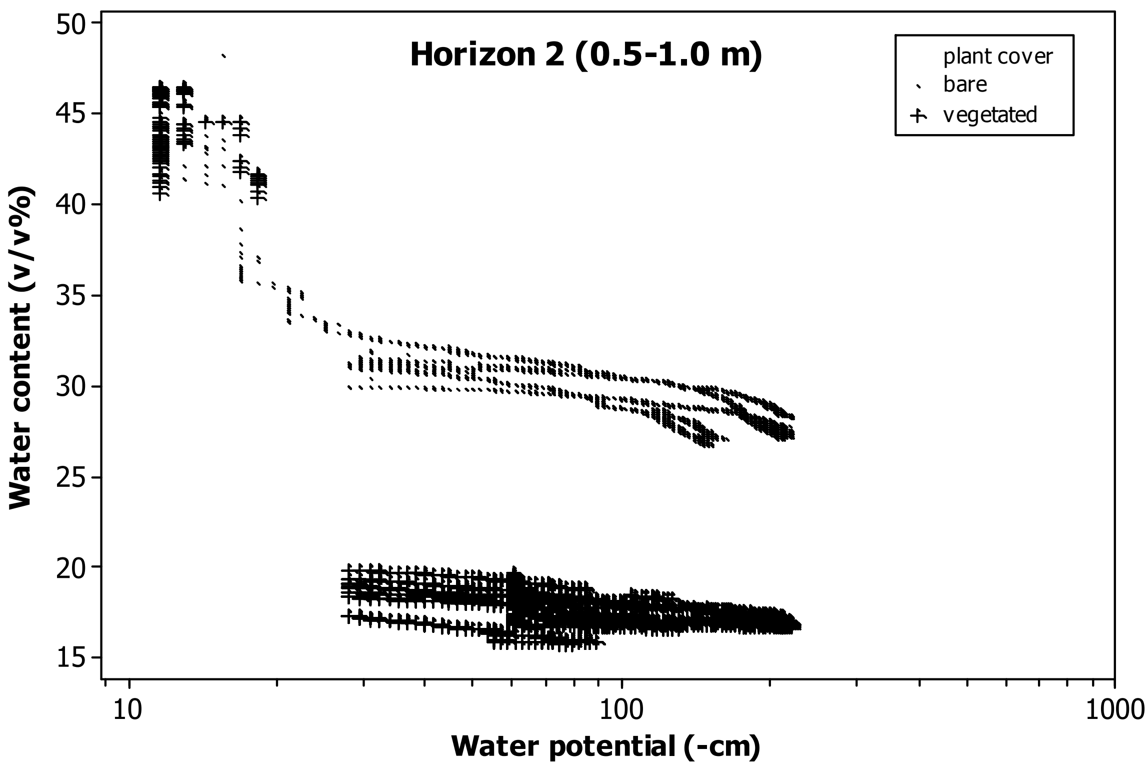


Figure 5-10 Water retention curves for horizon 2

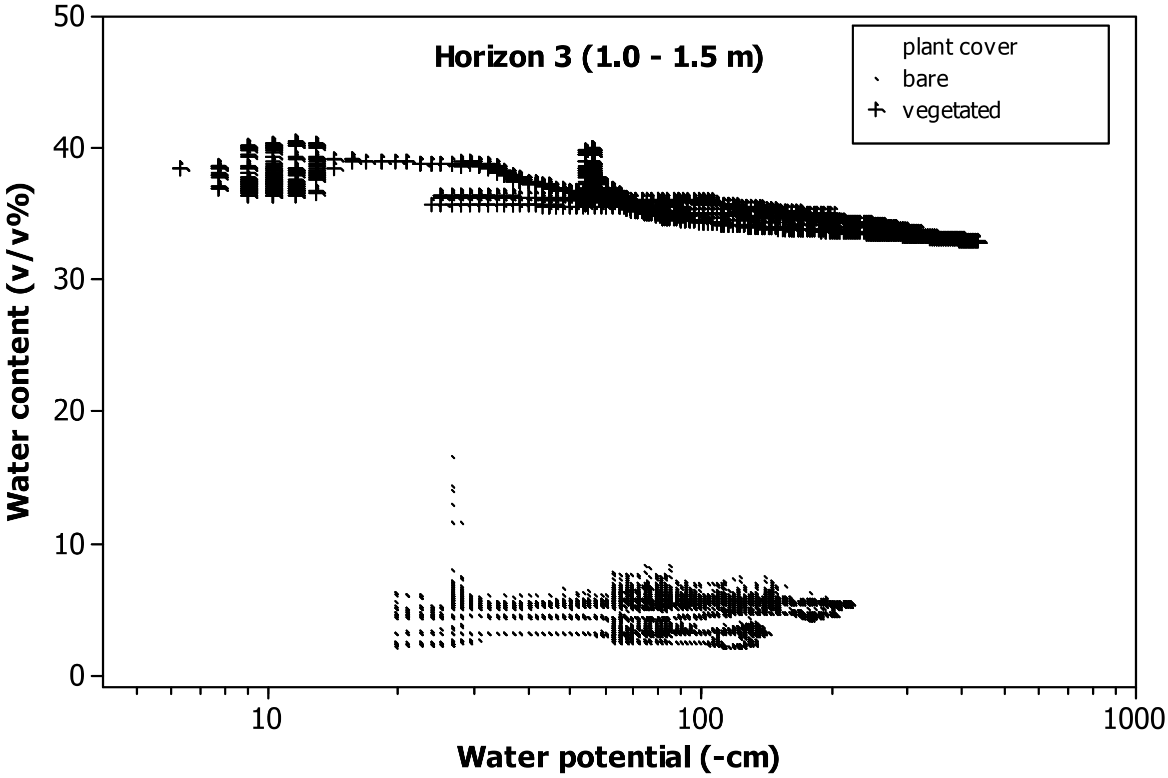


Figure 5-11 Water retention curves for horizon 3

Figures 5-12 and 5-13 show differences in the fitted water retention curve characteristic, *b*, (equation 5-2) between treatments namely, vegetated and unvegetated plots and over time for horizon 1 and horizon 3 (drainage limiting layer). increases in the water retention curve slope characteristic, *b,* with plant cover and over time indicate increase in the proportion of finer pores (related to texture and bulk density) that drain over the measured water potential range. Water retention for horizon 2 was similar to that in horizon 1 (Figure 5-12). However in horizon 3, water retention increased in the bare treatment over time while there was little change in the vegetated treatment (Figure 5-13). This was attributed to the effect of uneven construction that compromised the integrity of horizon 3 in the bare treatment where translocation of fine material downwards with infiltrating rainfall from overlying weathered waste rock added fines to this layer over time.

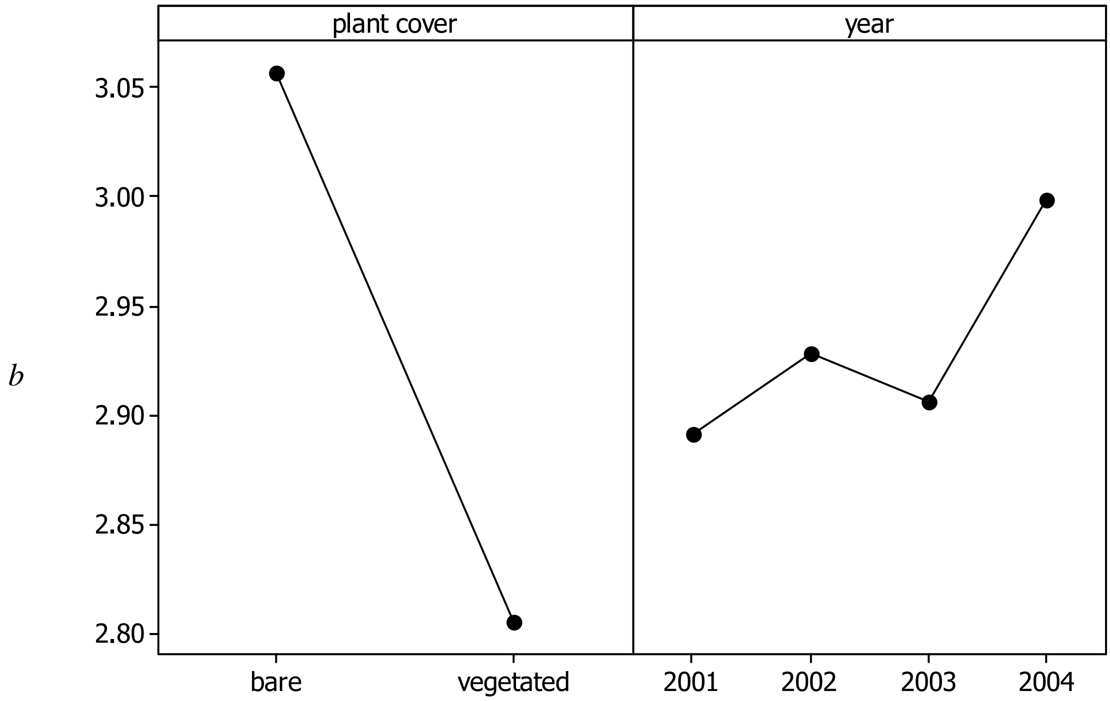


Figure 5-12 Main effects plot for the Campbell water retention coefficient, *b* (horizon 1, 0 - 0.3 m)

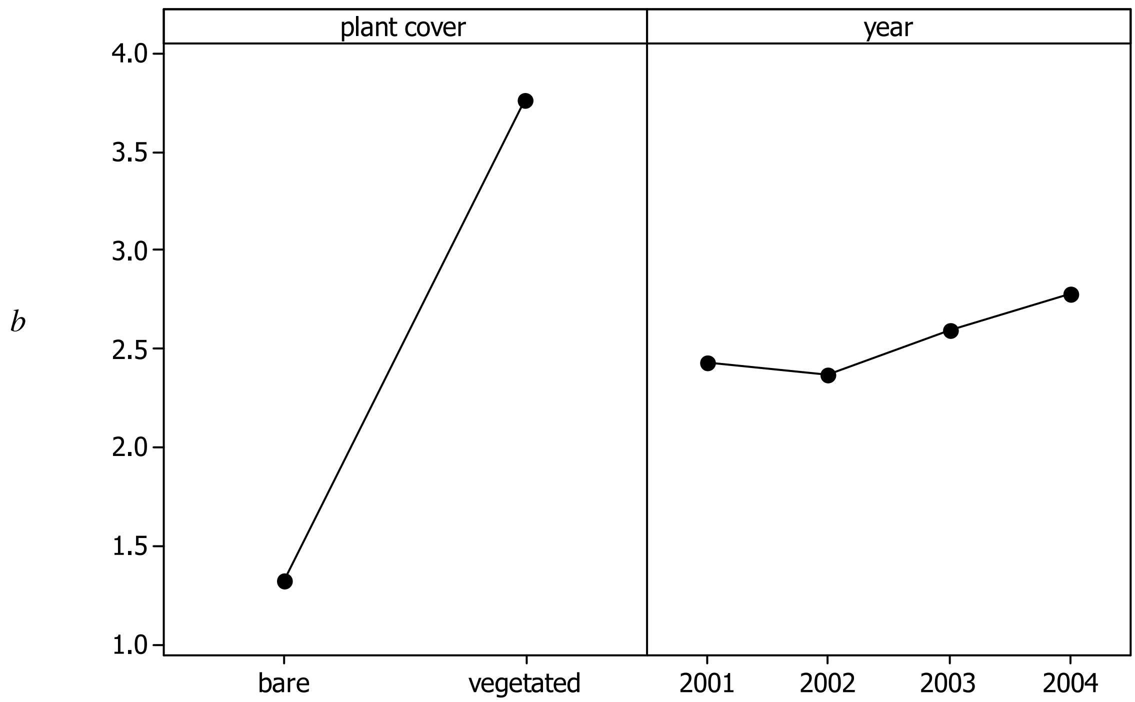
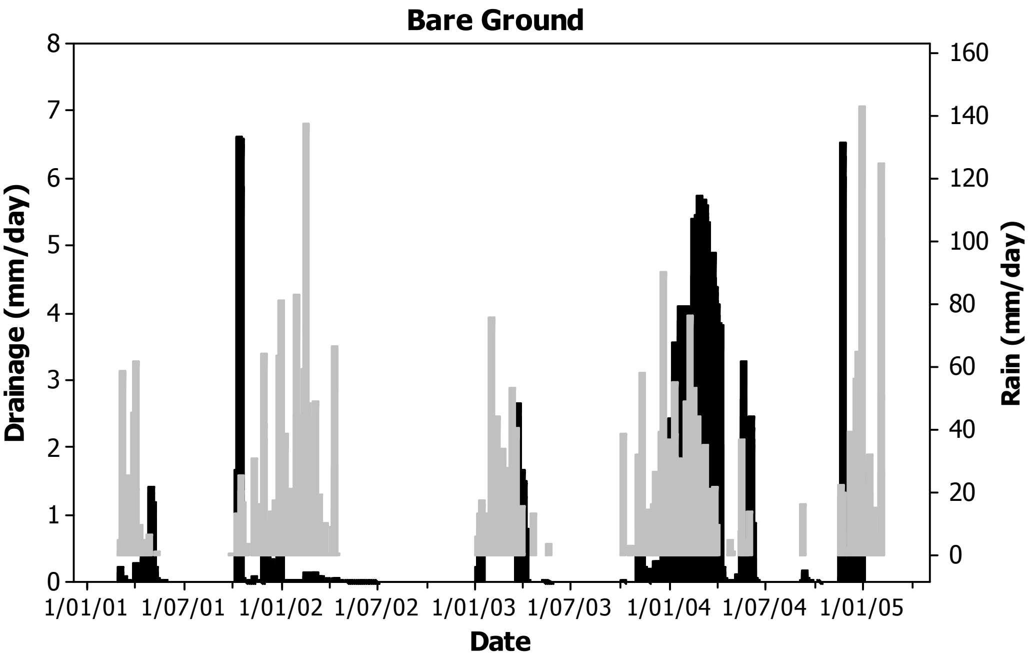


Figure 5-13 Main effects plot for Campbell equation parameter, *b* (horizon 3, 0.8 – 1.5 m)

5.3.6.1 Drainage Flux

The drainage flux estimated using the Darcy equation (equation 5-2) through horizon 3 and into the underlying waste rock stockpile for bare and vegetated areas is plotted with daily rainfall in Figure 5-14.



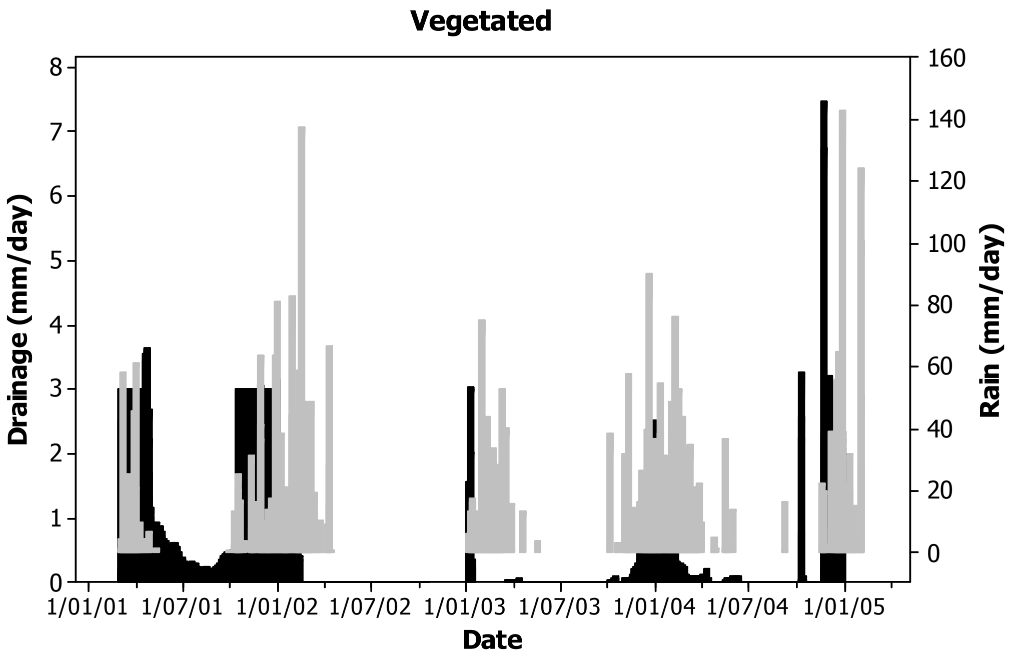


Figure 5-14 Daily drainage flux (black) and daily rainfall (grey) for bare and vegetated waste rock

The high saturated hydraulic conductivity of the overlying layers (Table 5-2) and macro-porosity in horizon 3 (Table 5-4) resulted in very little lag between rainfall and drainage flux. Drainage flux was closely associated with daily rainfall in both vegetated and bare plots, except for the vegetated plot in 2002. However, in the 2004 wet season, which was relatively wet, the drainage flux was greater in the bare plot, which didn't have the plant water use component of the vegetated treatment.

The retention of moisture above the drainage limiting supported the high and sustained drainage flux during the 2002 dry season and into the following wet season in the vegetated plot. The surface ripping would have initially enhanced infiltration into the vegetated plot. This effect was lost in subsequent wet seasons as a surface crust developed, rainfall ingress decreased and root growth compromised the integrity of the drainage limiting layer. Plant growth and root water uptake in the vegetated plot appeared to reduce the drainage flux by the 2004 wet season. Further changes in the water balance are likely to occur as plant roots grow through the drainage limiting layer (horizon 3) and affect its ability to control drainage into the underlying waste rock and retain water in the root zone.

Annual drainage and drainage as a proportion of rainfall (drainage: rainfall) are shown in Table 5-9 for bare and vegetated plots. Deep ripping and revegetation enhanced drainage as a proportion of rainfall compared with bare ground. However, the amount of drainage under revegetation declined (relatively) in subsequent years. In prolonged wet seasons, such as that in 2004, drainage as a proportion of rainfall was less in the vegetated plot compared with the bare plot. Evapotranspiration in the vegetated plot would account for this difference.

Table 5-9 Annual drainage as a proportion of rainfall
for bare and vegetated plots

Rainfall year1	**Rain (mm)**	**Vegetated plot drainage (mm)**	**drainage: rain (%)**	**Bare ground drainage (mm)**	**drainage: rain (%)**
2001	3912	281	7.2	20	5.0
2002	1590	300	19	98	6.0
2003	920	12	1.0	24	3.0
2004	1807	68	4.0	422	23
2005	1034	129	1.2	38	4.0

[1] measured between September and August

5.4 Discussion

5.4.1 *Cover characterisation*

Some of the performance issues with the constructed cover were related to the integrity of the construction process. Mining operations were unable to construct a drainage layer less than 1-m thick. This limitation needs to be considered in designing drainage limiting layers constructed by mining operations without engineering control. A similar cover configuration constructed on stockpiled waste at the former Rum Jungle mine in 1983 incorporated a compacted, 0.5 metres thick, clay drainage limiting layer designed to limit drainage flux to 5% of incident rainfall. This design target was met in the first 10 years. Subsequently the drainage flux increased to 10% of incident rainfall (Kuo et al., 2003). Ant and termite activity combined with desiccation cracks and root penetration increased permeability in the drainage limiting layer (Taylor et al., 2003). The thickness of the cover medium is clearly a key consideration for long-term performance of a revegetated cover.

5.4.2 *Water quality*

Although drainage waters are generally circum–neutral, elevated concentrations of U, Al, Pb, Mn and Cu are an issue. The situation of high metal concentrations in effluent streams from mining operations has been documented at many sites. Weathering and passivation processes in the stockpiled waste will lead to declining metal concentrations in drainage water streams over time (Morin and Hutt, 2001). There was evidence presented in Figure 5-2 that this would be the case at Ranger. However, it will be some time before the chemistry of water emanating from the monitored stockpile at Ranger mine will meet the closure criteria envisaged for this site (Johnston, 2002). Whether or not this is a critical issue for closure will depend on where the compliance point is set.

An extended runoff and seepage monitoring program is needed to predict the time required for mineral weathering processes to produce background water quality conditions adjacent to waste rock landscapes. Operational monitoring data from waste rock stockpiles (Figure 5-3) indicated that at least a three to five year period is needed to assess trends in water chemistry. Unambiguous monitoring results may require longer term landform trials (Morin and Hutt, 2001).

5.4.3 *Water balance*

In summary, plant water uptake in the store and release cover construction has extended down to two metres within two years and lead to declining effectiveness of the drainage-limiting layer to reduce deep drainage. While most of the published data relating water extraction patterns to root growth is for pasture plants, water extraction has been measured below two metres within two years of planting Lucerne a perennial pasture legume (Dunin et al., 2001). Perennial grasses may be less deep rooted than this (Murphy and Lodge, 2006). Water use from eucalypts planted for recharge control has been found to exceed that from perennial pastures and to extend down to 5 metres within 3 years, compared with water extraction to 2.5 metres under perennial pasture in the Western Australian wheat belt (Eastham et al., 1994) – evidence that native trees initially establish deep root systems. Most of the tree water use came from the top 3 meters. These published results corroborate the findings from water extraction patterns under native vegetation presented here. Uneven construction of this layer limited its effectiveness from the start in the bare treatment. Revegetation appeared to create a stable porosity in the surface horizons that is lacking in bare areas that subsequently slump when wet and set hard when dry.

The newly constructed store and release cover is an infiltration limited desert in a high rainfall environment, *i.e.* most of the water runs off. The revegetation was too sparse to be effective and the runoff coefficients were so high as to limit the amount of water available to support local woodland vegetation. The catchment runoff coefficients for the waste rock stockpile cover at Ranger (Table 5-3) were much higher than for a natural landscape. This is similar to the findings by Duggan (1991). Also, Evans (2000) used a runoff coefficient of 0.36, based on catchment measurements reported in Duggan (1991) for the natural land surface in the vicinity of Ranger mine. However, the proportion of rainfall occurring as either runoff or groundwater recharge has important effects on landscape ecology, determining how and where water is available to ecosystems. Surface ripping can be used to increase porosity and infiltration, but effect disappears rapidly without effective revegetation to generate stable porosity around root voids (Dexter, 2004; Kew et al., 2007; Mando, 1997). Stable macropores that allow water to infiltrate into the surface and be available to plant roots will be critical to maintaining rainfall ingress to support ecosystem development. The vigour and density of the revegetation needs to be increased for an effective store and release cover system. Slight canopy separations measured in the native analogue area woodlands (Chapter 4) indicate that effective revegetation will provide a fairly continuous canopy cover.

Significant changes in the soil water store of the constructed cover over a 4 year period were associated with plant water extraction from the developing root system. While better control over the construction of the sub-surface drainage limiting layer could improve drainage control the layer is unlikely to remain intact unless it is installed beyond the influence of tree roots. Similar interactions between tree roots, soil fauna and constructed covers have occurred at other sites (Taylor et al., 2003). Based on published information on water extraction patterns under native vegetation, drainage limiting layers may need to be installed below 5 metres to be beyond the influence of tree roots in this environment (Eastham et al., 1994; Kelley, 2002; Kelley et al., 2007). The thicker cover will depend on evapotranspiration from the revegetation to control drainage otherwise persistent saturated soil conditions at depth will drive deep drainage.

The level of control that was achieved in operational handling of weathered zone material could not achieve layer thicknesses less than 1 metre. Increasing the thickness of the drainage limiting layer to 2 metres could also halve the drainage flux according to water balance modelling estimates. This design modification has wider support (O'Kane et al., 2003). Again, the effectiveness of the revegetation in establishing transpiring leaf area will be critical to achieving the predicted improvement in drainage control. Without effective evapotranspiration infiltration will still flux down through a drainage limiting layer, albeit at a slower rate.

5.4.4 *Environmental support*

Waste rock covers designed for ecosystem support may need to provide for plant available water store in the surface five metres (Kelley, 2002; Kelley et al., 2007) and a 400-500 mm deficit at the end of the dry season (Cook et al., 1998). Some of this water supply may be provided in a thicker drainage limiting layer. While the surface 2 to 3-m contains most of the root activity, deeper profile water retention may be critical to tree survival in periods of extreme drought. While there is evidence that over storey woodland plants can grow well on mine waste, the eventual introduction of annual grass understorey will significantly increase evapotranspiration demand (Hutley et al., 2001) and put additional demand on water retention and water stress on plant communities.

The water balance for the ripped and revegetated monitoring plots showed enhanced drainage initially that declined over three years. Because the cover was not effectively revegetated the general condition was of low infiltration through a compact and massive surface. Low infiltration through the surface acted as a throttle limiting the water available to plant growth, a critical factor in ecosystem function (Ludwig et al., 2000a) particularly in water limited environments (Cook et al., 2002). This constraint over plant water supply could affect the successful establishment of comparable ecosystems to those in the surrounding natural landscape.

In *Eucalyptus tetrodonta* savannas that occur widely in the natural landscape around Ranger mine, growth of fine surface roots is restricted to the wet season and concentrated in the upper 0.5 m (Chen et al., 2004). These roots probably supply most of the nutrients taken up by the tree, as well as water during the wet season. However, the fine surface roots rapidly die off during the dry season, during which tree roots can exploit the top five metres of soil to supply transpiration requirements (Cook et al., 2003; Kelley et al., 2007). It seems likely that evergreen eucalypts then produce fine roots at depth to access subsoil water, but these fine subsoil roots would possibly become inactive or die during the wet season because of elevated ground-water levels and consequent hypoxic conditions. Local water tables rise from 7–10 m depth in the dry season up to 0.5–2 m in the wet season in *Eucalyptus tetrodonta* savannas (Cook et al., 1998). Thus, evergreen water uptake may alternate between surface fine roots in the wet season and subsoil fine roots in the dry system. This natural, seasonal variation in phreatic water tables may not be reinstated in the waste rock landscape without some drainage limitation at 3 to 5-m depth.

5.5 Conclusions

Cover design for ecosystem reconstruction is a special case of the store-and-release cover where the potential water storage and drainage conditions within the root zone can support the re-establishment of self-sustaining natural vegetation communities, typified in the natural analogue landscape. The design of the plant growth medium needs to demonstrate the capacity to support naturally occurring woodland ecosystems and restore a natural water balance to the mine landscape, as well as resist erosion. The revegetation will indirectly affect landform integrity since the evapotranspiration capacity of the revegetation, and the protection it affords from rainfall erosion significantly influence drainage flux, runoff and erosion rates.

A basic cover design concept that uses unweathered, or primary, waste rock material over oxidised waste to provide erosion resistance, water retention for plant growth and drainage limitation could be refined to conform with natural analogue conditions essentially by increasing the thickness of the component layers. High energy compaction of the drainage limiting layer may improve its performance. Surface treatment activities such as deep ripping also need to be combined with effective revegetation. Otherwise the stable soil porosity that is associated with biological activity and is essential for ecosystem function will not develop.

Demonstrating similarities in ecosystem support properties between the final landform cover and the soil zone in a natural analogue area will underpin an ecosystem restoration methodology. Long–term monitoring (5 to 10 years) of a final landform trial designed according to the environmental properties of natural analogue areas could improve cover design specifications and be used to develop a predictive capacity that ensures on-site and off-site environmental outcomes will be achievable within a particular time frame.

5.5.1 *Further studies*

The geomorphology, vegetation, soils and hydrology of a natural analogue area, the *Georgetown* analogue area identified in Chapter 4, are described in Chapter 6 to support a design approach for landscape restoration following mining. Landform designs can be validated using erosion and hydro-ecological models and by checking the geomorphic and cultural context of the landscape. However, the capacity to predict ecosystem patterns in the reconstructed terrain is needed to demonstrate an ecological design methodology. Consequently, a detailed study is made in Chapter 6 of environments in the a natural analogue area selected in Chapter 4 to support species distribution models that are developed in Chapter 7. These quantitative models can be used to assess whether a landform design will support similar vegetation patterns to surrounding natural landscapes.

Chapter 6 Analogue landform environmental survey, design criteria and evaluation

6.1 Introduction

6.1.1 *Background*

Consultation and independent evaluation are front and centre of any mine closure planning process because the context of landscape restoration has to be addressed from several different stakeholder perspectives (ICMM, 2006). The consultation and evaluation process has the current life of mine as the starting point. The life of mine plan accounts for the volumes and quality of waste the capacity of pits and the schedule of operations to closure. In this chapter, the geomorphic properties of a selected natural analogue area have been applied to life of mine plan estimates of waste rock to design a waste rock landscape at the Ranger uranium mine site. The design is also informed by consultation with Traditional Owners of the land and by independent technical evaluation.

A schematic outline of the program of consultation and independent technical evaluation leading from mine planning to construction is presented in Figure 6-1.

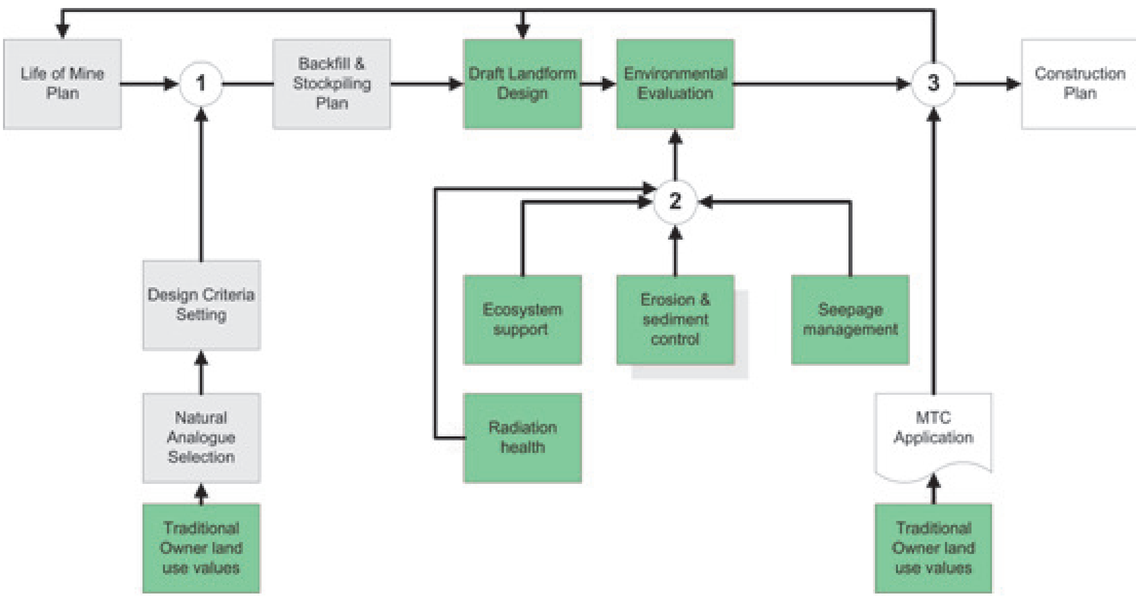


Figure 6-1 Landform design process, numbered circles indicate consultation points

This is an iterative process that is recommenced following significant changes to the mine plan. The stages of the program that were completed in this thesis are highlighted in green in Figure 6-1. Points of consultation are indicated by numbers in Figure 6-1.

6.1.2 *Landscape design*

Relief and hillslope morphometry are topographic properties of an ecosystem, which when integrated with stable soil and the plant communities, connect the surrounding ecosystems (Carpenter, 1998). Thus topography, soil and vegetation can be reconstructed together to optimize the restoration of key ecological processes. The methodological development being proposed here is therefore to design mine landform relief and hillslopes from the *grain size* and *extent* attributes of natural analogue landforms that were investigated in Chapter 4. The purpose is to restore natural ecosystem patterns and environmental processes at the landscape level. However, the analogue natural ecosystems need to be described in sufficient detail to support the design process and validate species distributions in the reconstructed landscape.

The focus on ecological objectives is aimed to differentiate the proposed methodology conceptually from hydrological approaches to topographic reconstruction, such as the basin relief approach (Toy and Chuse, 2005). The latter is directed primarily to protecting the off-site receiving environment rather than the quality of the on-site environment after mine closure. Although reference, or analogue sites, are used routinely to set closure criteria for mine rehabilitation following opencast mining (Bell, 2001), this approach lacks landscape ecological context. A range of ecological landform design methodologies have been proposed that recommend physical evaluation methods based on numerical erosion and hydrological models (Nicolau, 2003; Toy and Chuse, 2005). Previous studies assert that methods to restore natural biodiversity in mined landscapes need to be developed (Ehrenfeld, 2000; Ehrenfeld and Toth, 1997).

The failure of the ecological design may be partly due to complex ecosystem composition and variation (Holl et al., 2003). However, analytical approaches that have become widely applied in landscape ecology, conservation biology and wildlife management have yet to be adopted in restoration ecology. The *grain size* and *extent* attributes of ecological scale in landscapes that were used in Chapter 4 to identify analogue landforms and describe underlying environmental processes of erosion-sedimentation and water distribution are applied here to geomorphic design of the waste rock landform. This will partly address the issue of a lack of ecological context. The critical aspects of cover design or soil reconstruction affecting plant available water and drainage that were identified in Chapter 5 also need to be understood.

6.1.3 *Soils on waste rock and natural analogues*

The waste rock materials at Ranger are generally richer in P and have lower in N than natural soils in the area (Ashwath et al., 1994). The rock materials can occasionally be saline and in some situations may produce acid drainage with high metal concentrations (Mg, Mn, U and Pb), as evident in Chapter 5. Salinity caused by magnesium sulfates produced from weathering of primary rock material has been found at levels not toxic to plants (Ashwath et al., 1994). However, tolerant plant species may be required in situations where there is drainage discharge.

The retention of nutrients is a key measure of ecological integrity (Ludwig et al., 2004), as nutrient fractions such as labile or plant available phosphorous, a limiting factor for the establishment of woodland savanna ecosystems (Short et al., 2000), tend to be concentrated in the topsoil under natural woodland. The retention and recycling of nutrients are closely linked to amount of organic matter in the near surface of soils overlying deep weathered regolith that supports eucalypt woodland (Koch and Hobbs, 2007; Schwenke et al., 2000a).

The soil morphological patterns associated with drainage status were observed to be linked to gradients in woodland vegetation type. Past studies of root distribution in northern savanna woodlands indicate that 70% of the root biomass is at < 0.2 m soil depth and the rest of the root mass is accounted for between 0.3 < 1.4 metres (Werner and Murphy, 2001). Although, the dry season water extraction patterns occur predominantly in the top 2 metres some water extraction can occur below 4.5 meters (Chen et al., 2004; Kelley et al., 2007). Also, the morphologies of gradational texture profiles and rocky soils were not inconsistent with soil design specifications that have been produced for erosion resistant covers that use a mix of materials to produce erosion resistance, drainage control and growing medium for plants over uranium milling waste (Johnson, 2002). This concept seems compatible with the analogue soil morphology — described in Chapter 4 — as massive and earthy, gravelly, sandy loams at the surface to sandy clay loams at depth, with a root-limiting layer in the form of weathered rock or ferricrete hardpan within 1.5 to 3.0- m of the land surface.

6.1.4 *Eco-hydrology*

Eco-hydrology is an important aspect of mine-site rehabilitation (Section 2.5.3). The interactions between groundwater and surface water in the natural landscape occur mainly in the surface five metres (Vardavas, 1993) where woodland vegetation is consuming a major part of the available water through evapotranspiration (Cook et al., 2002; Hutley et al., 2000). However, the hydraulic conductivity in the subsoil in the waste rock would be much higher than in the natural regolith where there is a conductivity throttle at 2 – 3 metres below the surface that generates throughflow to the creeks (Vardavas, 1993). The hydraulic conductivity throttle in the natural regolith is analogous to the drainage-limiting layer in the constructed landform cover evaluated in Chapter 5. The water balance of the shallow hydrological system may determine the ecological outcomes. The shallow hydrological system needs to be placed in perspective and reasonable bounds on the components of the water balance need to be set with respect to the natural environment and what may happen following rehabilitation.

A significant hydrological study of eucalypt woodland in the Northern Territory was undertaken at Howard Creek near Darwin between 1996-98 (Cook et al., 1998; Hutley et al., 1997; Kelley, 2002). The Howard Creek study measured the evapotranspiration (ET) components of the water balance and used these to estimate the rates of groundwater recharge. The technique involved heat-pulse monitoring of overstorey transpiration, chamber estimates of understorey transpiration and soil evaporation, and then the total ET measurement was obtained via eddy covariance. There was also some measurements of soil water dynamics (Kelley et al., 2007). Cook *et.al.* (1998) undertook a hydrographic water balance that differed from that of Hutley *et.al.* (2000). These differences are summarised in Table 6-1.

Another study was done by Cook et al. (1998) in which they estimated ET from interpretation and rescaling of catchment and groundwater hydrographs rather than direct measurements. Hutley et al. (2000) reported the actual measurement results, but their study did not extend beyond ET and a limited number of soil water dynamics measurements and their ET rates appear to be underestimated. The total ET of 1,110 mm.yr^{-1} estimated from the catchment water balance by Cook et al. (1998) was higher. However, the Hutley et al. (2000) measured interception separately (88 $mm.yr^{-1}$) while Cook et al. (1998) lumped interception into their total ET. Consequently, the combined understorey and soil evaporation of Cook et al. (1998) and Hutley et al. (2000) were similar (Table 6-1).

Vardavas (1998) outlined a parametric model of the Magela Creek catchment, where Ranger mine is located, and observed that interflow was important in stream flow generation. This study concurs with the fact that interflow can be a larger component of stream flow than surface runoff. This study also reported that the wet season runoff was saturation controlled by a variety of below-ground processes, including interflow, soil-water-storage capacity and bedrock percolation rates.

Table 6-1 Comparison of water balance estimates at Howard Creek reported by (Cook et al., 1998) and (Hutley et al., 2000)

Component	**(Cook et al., 1998) ($mm.yr^{-1}$)**	**(Hutley et al., 2000) ($mm.yr^{-1}$)**
Rainfall	1 720	1 750
Overstorey – wet season transpiration	370	160
Overstorey – dry season transpiration	175	153
Overstorey total transpiration	545	313
Understorey and soil evap. – wet season	440	381
Understorey and soil evap. – dry season	125	176
Total understorey and soil evaporation	565	557
Leaf interception	0	88
Total ET (sum of above ET components)	1 110	958
Stream flow	590	590
Net groundwater recharge	20	—
Residual (rain – ET – flow – recharge)	25	202
Stated study uncertainty	-20 to +120	—

Previously, Vardavas (1993) reported that a reasonable volume of water enters and leaves the groundwater system each year (150 to 200 $mm.yr^{-1}$) virtually all the water movement was vertical with water entering as rainfall during the wet season and being lost as ET during the dry season. Woods (1994) estimated similar natural recharge rates in the order of

65 to 130 mm.yr^{-1} (5 to 10% of rainfall) under natural conditions from a drawdown analysis of the Jabiru town water supply bore field (Pidsley 1990).

6.1.5 *Ecosystem survey support*

The ecological outcomes of a mine landform design need to be predictable in order to validate the landform design being proposed (Nicolau, 2003). The number of observations and the range and scale of analogue landscape variation would be factors in designing ecosystem surveys to support predictions. Landscape topography can spatially influence patterns of water loss and accumulation thereby dampening or amplifying vegetation patterns (Ben Wu and Archer, 2005). Therefore, a grid survey design with a site spacing that is a fraction of the flow path length from crest to valley floor should resolve variation in hill slope environments that affect plant species distribution.

Resolution of the hill slope factors causing environmental variation probably requires a minimum of 100 survey observations (Stockwell and Peterson, 2002). The common and abundant species that account for most of the ecosystem space also need to be identified, as these species will contribute the most value to the rehabilitation design. Collecting species abundance data requires additional effort to recording presence absence. Detailed study plots and plant area counts are needed. Environmentally stratified sampling designs have been used to assess spatial organisation of ecosystems in landscapes (Ludwig and Tongway, 1995; Rempel and Kushneriuk, 2003). The six ecosystem types identified in Chapter 4 could be used to select representatives from the grid survey observations across the analogue landform, thereby ensuring that plant community descriptions had an explicit landscape context with the mine landscape restoration task.

6.1.6 *Erosion*

Rainfall in the Alligator Rivers Region is some of the most intense and erosive in Australia, and hence water erosion is a significant process (Duggan, 1991) that needs to be addressed in rehabilitation design. In such a situation, average denudation rates may not accurately reflect the risk to containment integrity from gully erosion. The Australian guidelines for the construction of covers over radioactive waste material recommend a structural life of at least a 1 000 years (Anon, 1987). To meet these guidelines, a sediment transportation model is required. SIBERIA is a deterministic three-dimensional erosion and sediment transport model, which can be used to simulate topographic evolution over long periods based on transport limited erosion rates (Willgoose et al., 1991).

The SIBERIA erosion model has been calibrated for materials and conditions at Ranger uranium mine (Evans et al., 2004; Evans and Willgoose, 2000; Willgoose and Riley, 1998) and elsewhere (Boggs, 2003; Hancock et al., 2000; Hancock et al., 2008; Hancock and Turley, 2006) to assess containment risks associated with the final landform design. The D8 flow routing algorithm used in SIBERIA to route surface runoff is a computational simplification that does not accurately represent overland flow divergence and soil water conditions (Wilson et al., 2005). While SIBERIA may not accurately represent variation in soil conditions, it nevertheless describes how a catchment would look, on average, at a given time. It is thus used as a tool for validating landform designs (Hancock et al., 2008; Hancock and Turley, 2006).

The aims of this chapter were twofold: (i) develop and undertake independent validation of a landform design for closing Ranger uranium mine; and (ii) describe the ecological context for the waste rock landform based on the environmental patterns in a selected natural analogue area.

6.2 Methods

6.2.1 *Ranger mine landscape*

An oblique air photo of the Ranger mine site obtained in 1974 (Plate 6-1) before mining began shows the site to be located on a peneplanated surface, bounded by streams and backed by high relief sandstone plateau. Another aerial oblique photo of the site taken in 2001 and looking north (Plate 6-2) shows land use associated with the operating mine including waste rock stockpiles, open pits, ponds, tailings storage facility and industrial facilities. This gives an indication of the extent of the restoration task. An outline of the proposed design and evaluation process that refers to the Ranger mine site and follows the scheme presented in Figure 6-1 is developed in the sections that follow.

Ranger mine has important indigenous owner land use values. Ecosystem reconstruction is important to the Mirrar traditional owners of the Ranger mining lease. The results of initial consultation with the Mirrar on their expectations for landscape reconstruction and revegetation set the boundary conditions for landscape restoration at this site. Consultation on the broad vision for landscape reconstruction was made with the Traditional Owners and their representatives (Gundjehmi Aboriginal Corporation and the Northern Land Council)

In a site visit to the *Georgetown* analogue landform area (identified in Chapter 4) in 2002 the Mirrar verbally supported the vision of the rehabilitated final landform. The next (but first recorded) consultation on the closure process with the Mirrar occurred in 2005 at Mula in Kakadu National Park. Final land use objectives were discussed including aspects of final landform scheduling, backfilled pit landforms, land surface rockiness, tailings storage facility rehabilitation and water course reinstatement. The Mirrar placed particular emphasis on the following issues: (i) leaving no additional water bodies; (ii) providing a land surface that would support hunting and gathering activities; and (iii) reinstating woodland plants.

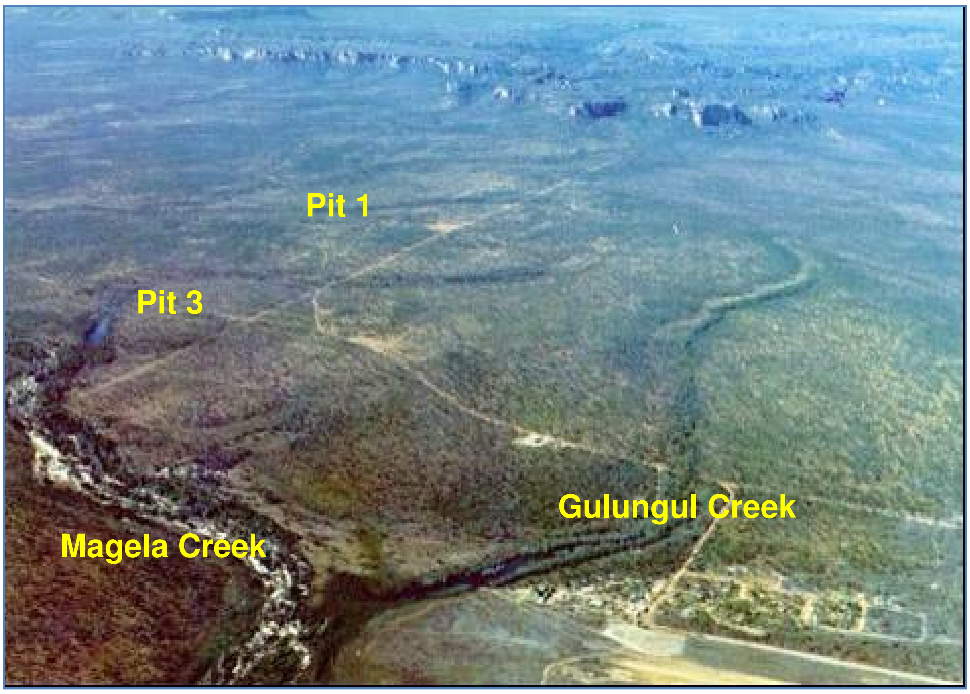


Plate 6-1 Oblique air photo of the Ranger site to the south before mining began

An extract from the summary notes from the meeting included the following statement:

"A stable, natural environment that is safe to access and capable of sustaining the variety of animals and vegetation (including native food and medicinal species) endemic to the surrounding environment. The preferred underlying landform would be as flat and low in relief as possible to minimise the possibility of erosion."

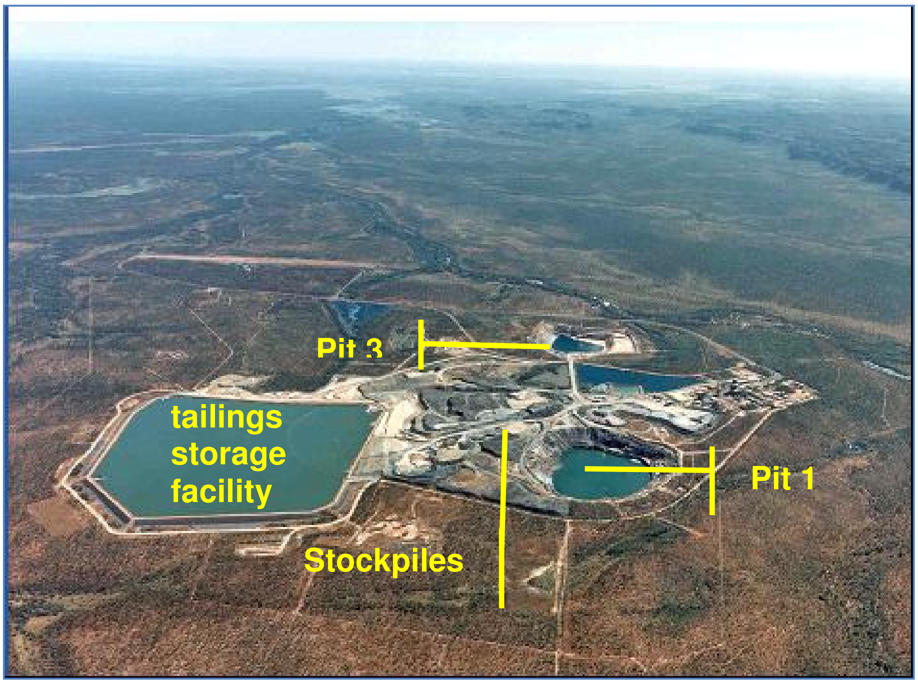


Plate 6-2 Oblique air photo of the Ranger site to the north in 2001

6.2.2 *Landform design*

Landform design criteria at the Ranger mine case study site were derived from the central tendency and dispersion measures of terrain attributes in the *Georgetown* analogue area. The design criteria were applied to mine planning estimates to draft a mine landform design. The geomorphic similarity of the landform design to the natural landscape was then evaluated. Published landscape evolution modelling and hydro-ecological modelling methods were used by independent investigators to assess whether the landform design restored natural erosion and water balance processes. In addition, a detailed environmental survey of the *Georgetown* analogue area is made and vegetation presence-absence, abundance, and soil properties described to support ecological modelling in Chapter 7.

6.2.3 *The natural analogue areas*

A method, used for selecting the natural analogue areas and identifying the ecosystem gradients with reference to the scale and geomorphology of the mine landscape, was developed in Chapter 4. Two analogue landforms were selected — similar in shape, size and habitat complexity to the final landform — and the environmental range was classified into six ecosystem types on rocky substrates. One analogue landform, *Georgetown*, a rocky rise southeast of the mine was preferred because it comprised a complete range of environmental variations of the rocky substrates that could be incorporated into the mine landform. Thus the natural analogue selection was based on a reconnaissance survey to identify the context and range of environmental variations appropriate to reconstruction of the mine landscape.

6.2.4 *Topographic reconstruction*

Assuming the natural landscape is in equilibrium with environmental conditions, then the acceptable gradients of the landform design properties can be set from their numeric ranges obtained from the natural analogue area. Toy and Chuse (2005) proposed a mine landform design method that restored drainage-basin area, weighted mean slope, and drainage density for the pre-disturbance or analogue landscape. A major limitation of their proposal is that the selection process for the analogue areas was not specified. However, the proposed major focus of this thesis was to use digital terrain analysis and computer aided design techniques to create "*steady state*" landscape configurations typical of mature natural landforms. Another limitation is that soil properties are not specified. A problem with specifying soil property development in waste rock is that there will be changes over time (see Chapter 5). There have been recent developments in predicting soil formation from parent materials in hillslope environments from erosion, sedimentation and mineral weathering processes(Minasny et al., 2008). However, these are yet to be applied to landscape restoration.

The design methodology developed here differs from Toy and Chuse (2005) in that it is iterative and specifies the environmental range in an analogue area (Chapter 4) and uses terrain properties from terrain analysis to incorporate scale dependent ecological processes rather than average catchment characteristics. In this methodology the primary terrain attributes (Table 4-1) related to the landscape energy level, geomorphic processes (Allen et al., 2003), erosion, sedimentation and drainage (Moore et al., 1991) were specified from the natural analogue area. As digital terrain attributes tend to be non-normally distributed (Gallant and Wilson, 1996), the central tendency and acceptable range in landform criteria were defined more reliably from non-parametric measures such as median and quantile distribution.

6.2.5 *Terrain analysis*

The construction of the elevation grid (DEM) was described in Chapter 4. A range of topographic parameters: erosion and sedimentation indices, wetness index, relief, catchment size, slope, flow path length, and landscape curvatures were used to describe variation in the physical environment that supports ecological diversity. Slope length is an important parameter that is very difficult to measure because it can be measured from different start and end points (Renard et al., 1997). The flow path length in TAPESG was applied here. This flow length measure is the longest flow path at a point, i.e. it is the longest distance to the catchment divide. Curvature is an important landform attributes that affects slope stability (Hancock et al., 2003) and water and sediment distribution in hillslope environments.

There are two types of curvature measurement. The first is profile curvature is the curvature of the surface in the direction of maximum downward slope, that is, along the flow line through a point. A second type is the plan curvature, which is the curvature of a contour drawn through a point. In general, curvatures are determined by the reciprocal of the radius of a curve. A gentle curve has a large radius and hence a low curvature, while a tight curve has a small radius and large curvature. As the curvature values are small, so they are multiplied by a hundred to give more comfortable values. The units of curvature are radians per metre, the change in orientation typically measured by travelling 1 metre along the respective line. Curvature values are typically less than one and are multiplied by 100 for easier interpretation. Thus a curvature of 5 is 0.05 1/metres, and corresponds to a radius of 20 metres. Negative curvature values represent concavity and positive values convexity.

The upslope contributing area, measured in square meters, is the area contributing runoff flow at a point in the landscape. TAPESG calculates this area from flow path and flow accumulation in the landscape. The FD8 or Rh08 flow routing algorithms are implemented in TAPESG allowing flow dispersion to be represented. Flow is distributed to multiple nearest-neighbour nodes in upland areas above defined channels and D8 or Rho8 algorithms below points of presumed channel initiation. Based on visual assessment of high-resolution air photography in the analogue landform, an area of 10 000 m^2 to define stream channel initiation was used. A maximum cross grading area of 10 000 m^2 was used that corresponded with the area defining stream channel initiation. The cross-grading algorithm is applied when the contributing area is less than the cross-grading threshold. In low relief landscapes, this prevents dispersion of flow lines in the bottom of valleys.

6.2.6 *Environmental evaluation of the reconstructed landform*

In order to compare the landscape attributes of the analogue and reconstructed landform, a terrain model of the mine landform that was based on the life of mine plan estimates of pit and waste volumes that were valid in 2001was created was created using Vulcan (Maptek) mine planning software. The quantity of material remaining on the land surface after the backfill of the pits consisted of 17.41 Mm^3 of non-mineralised (< 0.02% U_3O_8) material and the tailings dam walls, which currently form a water retaining structure. Pre-existing catchment boundaries were reinstated and non-mineralised stockpiles were re-contoured according to the design guidelines to create a draft design that comprised water-shedding waste rock landforms. Primary terrain properties of elevation, slope and curvature that were based on their ranges in the *Georgetown* analogue area were used as topographic design criteria. The range of the elevation and slope specifications was slightly widened to allow for consolidation over time in the backfilled pits.

6.2.6.1 Terrain evaluation

The landscape attributes of elevation, slope, wetness index, erosion index (LS factor from the USLE erosion model), and stream power index and plan curvature in the built landscape were compared to their range in the surrounding natural landscape. These indices were selected to describe the energy level of the landscape and the patterns of water, sediment and erosion distribution that defined landscape dynamics and ecosystem function. A perspective view of the reconstructed landform showing different digital terrain attributes in the analogue area was checked visually in ArcGIS. The quartiles of the distribution of analogue terrain attributes were mapped to identify values that were outside the analogue area range and the design was modified.

6.2.6.2 Landform evolution

Landform evolution over 1000 years for a DEM that comprised the post-mining landform and the Georgetown analogue landform was independently assessed using the SIBERIA model (Lowry et al., 2006). The DEM of the proposed post-mining landform had to be resampled to ensure that it fell within the grid array limitations (250 cells by 250 cells) of the SIBERIA program (Willgoose and Riley, 1998). The locations of catchment outlets from the drainage network generated from the DEM were identified. Also in the process, two surface conditions were applied: areas which had a slope of greater than 10%, which were regarded as being likely to be treated with rock batters and mulching; and the remainder of the area was regarded as possessing *'natural'* characteristics. Erosion and hydrology parameter values from Evans (2000) were used to describe the surface condition of each of the two areas.

6.2.6.3 Ecohydrological evaluation

An independent evaluation of catchment water balance was made for a DEM comprising the post mining landform and the Georgetown analogue landform using WEC-C (Croton and Dalton, 2006), a distributed deterministic catchment model (Croton and Barry, 2001) that simulates stream flow generation from runoff and below ground processes, *i.e.* interflow and groundwater discharge. The model was described in detail by Croton and Barry (2001) and has been applied in a range of eucalypt woodland environments to assess ecohydrological effects of mine rehabilitation (Croton and Reed, 2007), land clearing (Croton and Bari, 2001) and forestry development (Bubb and Croton, 2002). WEC-C is particularly useful for mining studies in that it can incorporate soil excavation, and other profile changes, as inputs during the simulation. The unit of time for input of evaporation and rainfall data is daily; however, to maintain stability and accuracy the model operates on an hourly internal time-step.

Initial modelling of possible soil and vegetation covers for the rehabilitation of the Ranger mine used a simple one-dimensional (1-D) model of the profile. This modelling indicated the likely range of water balance components for the rehabilitated mine. A weakness of 1-D modelling is that interflow is not allowed and so outputs are restricted to ET, surface runoff and deep recharge. The soil layering for the base case used in 1-D modelling of the soil profile is shown in Table 6-3. These characteristics were partly based on information from the landform trial results reported in Chapter 5 and on experience with the WEC-C model in eucalypt woodland environments in south Western Australia.

Table 6-2 Parameters for the base-case 1-D model

Layers	Total Thickness (m)	Soil Type	Ks (mm/day)	Vegetation Roots
1 & 2	1.0	cover	50	yes
3 & 4	1.0	clay liner	10	yes
5 to 7	3.0	waste rock	24,000	yes
8 to 11	7.0	subsoil	10	yes to 8m
12	3.0	drainage layer	10	no

The soil profile given in Table 6-3 was run for four vegetation cases, each having an annual sinusoidal variation in LAI (leaf area index) to reflect the growth and senescence of vegetation through the wet and dry seasons. The base case had an LAI of 0.8-1.6 from overstorey and understory LAI reported by Hutley (2000). To test sensitivity to LAI, three cases using simple ratios of this base case were simulated: 0.4-0.8 (half base), 0.6-1.2 (three quarters base) and 1.0-2.0 (one and a quarter base). The model was also run for four variations of K_s for the clay drainage limiting layer: 5 mm.day^{-1}, 10 mm day^{-1} (base case), 20 mm day^{-1}and 40 mm day^{-1}.

The 3D water balance model formulation was based on a combination of the parameterisation of the WEC-C model for natural, uncovered areas, comprising a catchment called Corridor Creek and with the parameterisation of the 1D mine rehabilitation WEC-C model for the covered areas. Corridor Creek is a monitored catchment on the southern side of the Ranger mine that has been affected by the mining operations — there are wetland filters, small dams, as well as sections of a tailings dam and a mine pit within the catchment. There has also been controlled release of mine water into the catchment. The gauged catchment is 257 hectares in area and stream flow data was available from 1 Jan 1994 until 14 Apr 2003. The total area of the simulated catchment is 306 hectares.

The initial WEC-C 3D model was prepared for the Corridor Creek catchment at a 50-m grid spacing and a six layer profile. Water flow in Corridor Creek is a combination of surface runoff during intense rainfall events and near-surface interflow during the balance of the wet season. Generally flows cease by early May for all years in the period 1994 to 1998; this implied that deep groundwater made little or no contribution to stream flow and that stream flow depended on surface runoff and interflow. This model of the groundwater system was consistent with the studies by Vardavas (1993).

6.2.7 *Environmental surveys*

Two environmental surveys of vegetation and soils were made to establish: (i) plant species distribution; and (ii) soil morphology and soil physical and chemical characteristics that are important to ecosystem reconstruction. Firstly, a grid survey of the Georgetown analogue landform (500 hectares) was undertaken to describe soil morphology and species presence–absence. A grid spacing of 200 metres was used to sample 102 sites. This spacing gave a resolution that was similar to the third quartile of the range in slope lengths (160 metres), *i.e.* one quarter of the slope lengths in the analogue area were longer that the grid spacing and three quarters were shorter. Some resolution of hill slope variation may have been lost, however this spacing produced a reasonable number of sites to survey in the time available and was well within the range of hill slope lengths (300 – 400 m).

At each site the woodland vegetation was visually classified (based on the dominant overstorey species) into 1 of the 6 ecosystem types on rocky substrates (Table 6-4) determined from the multivariate analysis in Chapter 4. Soil morphology was described at each site according to standard methods (McDonald et al., 2009) from auger profiles down to contact with the substrate or 1.5 metres, whichever was shallower.

Table 6-3 Six vegetation communities on schist derived substrates in the Georgetown analogue area

Ecosystem type code	Ecosystem type description
3	Tall Eucalyptus tetrodonta/E. bleeseri/mixed open woodland with a sparse mixed tree midstorey and mid-dense Sorghum intrans/Petalostigma quadriloculare understorey.
4	Mid-high Melaleuca viridiflora open woodland with a very sparse mixed tree midstorey and dense mixed grass/Cyperaceae understorey.
6	Tall Eucalyptus tetrodonta/E. miniata open woodland with a sparse mixed tree midstorey and mid-dense Heteropogon triticeus/Sorghum intrans understorey.
7	Tall Eucalyptus tetrodonta/E. miniata/E. porrecta open forest with a sparse mixed tree midstorey and mid-dense Sorghum intrans/Petalostigma quadriloculare understorey.
8	Tall Eucalyptus foelscheana/mixed open woodland with a sparse mixed tree midstorey and sparse Heteropogon triticeus/Themeda triandra understorey.
9	Mid-high Eucalyptus tectifica/E. tetrodonta/E. confertiflora open woodland with a sparse mixed tree midstorey and sparse Sorghum intrans/Eriachne sp. /Petalostigma quadriloculare understorey.

Secondly, three representative sites of each ecosystem type listed in Table 6-4 were randomly selected from the survey grid to provide detailed information on plant species and abundance and soil chemical and soil physical properties. A replicated plot design was used that comprised 3, 20 x 20-m permanent quadrats that were marked at the centre with a steel post and the location recorded with a hand held GPS.

The ecology of plant communities was characterised in terms of community structure, plant diversity and abundance. A biodiversity index for each of the ecosystem types is presented. Common species were tabulated with an importance index and an indication of the range of their occurrence across ecosystem types. Ecosystem structure, including plant density and ground cover information for each stratum were described.

Vegetation community composition and structure was described in terms of overstorey, mid-storey and groundcover strata. Overstorey was defined as those trees greater than 8 metres high. Midstorey was characterised as the 2-4 metre layer and understorey comprised all plants below 2 metres. Overstorey canopy cover, species, height, health and diameter at breast height (DBHOB) for each individual tree over 2 metres was recorded. Understorey measurements of species, density and foliage projective cover were made in 10, 1 x 1 metre quadrats evenly spaced along two transects running through the larger quadrat.

Furthermore, the framework species were identified in terms of their dominance in overstorey, mid-storey and groundcover strata of each ecosystem type. Dominance was assessed from: (i) coverage (basal area or projected foliage cover), which is a measure of the amount of space occupied by individuals of a given species; (ii) density (number of stems or individuals per unit area), which is a measure of the numerical dominance relative to other species; and (iii) frequency (how many samples contain individuals of a given species), which is a measure of the commonness of the species.

A standard suite of soil properties reflecting soluble and cation exchange chemistry and organic matter content were measured to characterise soil fertility in vegetation plots. Two additional sites in *Melaleuca viridiflora* open woodland (ecosystem type 4 described in Chapter 4) were sampled in the soil survey to make 20 sites in total. The depth increments were selected with respect to rooting patterns of woodland plants (Chen et al., 2004; Werner and Murphy, 2001). The sampling and analysis plan are presented in Table 6-4.

Table 6-4 Analogue soil sampling and analysis plan

Sample depth	Analyte
Soil chemical properties	*Major nutrients*
Topsoil:	Total N: Kjeldahl: Method 7A2*
1. 0-0.05	Mineral nitrogen: Method 7C2
2. 0.05 - 0.1	Total (Organic) C: Method Leco
3. 0.1-0.2	Total S: Method Leco
Sub-soil	Extractable P: Method Hedley Fractionation
4. 0.8-1.0	Bicarbonate extractable P: Method 9B2
5. B2 Horizon sample to confirm base status for classification	Bicarbonate extractable K: Method 18A1
	Water soluble nutrients
	soluble bases (in the saturation extract) Ca, K, Mg, Na: Method 14H1
	Micro-nutrients
	+DTPA extractable Cu, Zn, Mn, Fe: Method 12A1
	Exchange Properties
	Exchangeable bases(silver thiourea extract); Exchangeable Al; CEC: ECEC : Method 15F1; 15F2; 15F3
	Base saturation percentage: Method 15L1
	Exchangeable sodium percentage: Method 15N1
	General Parameters
	pH saturation extract: Method 14C1
	Electrical Conductivity saturation extract: Method 14B1
Soil physical properties	1. particle size distribution
Topsoil	2. gravel content
Subsoil	3. infiltration rate
	4. hydraulic conductivity
	5. water retention
Number of samples = 76	

* method codes refer to Rayment and Higginson(1992)

6.3 Results

6.3.1 *Landform design*

The descriptive statistics for each landform property are shown in Table 6-5. The large coefficient of variation of the mean indicates non-normal distribution of each property. The range between the first quartile and the third quartile represent half of the overall range between the minimum and maximum. Curvature values are presented as radius of arc in metres. Median values of profile and plan curvature were very low in the natural landscape. Hill slope profiles are virtually straight (median curvature 0 metres) and radius's of arc within the range of –5 000 (concave) to 5 000 metres (convex) appear to be acceptable in a constructed hill slope. The radii of arc for plan curvatures are in the order of 100 metres for both concave (vale) and convex (ridge) landform elements.

The relief and curvature properties of the natural landscape were used as geomorphic design criteria by the Ranger mine department to produce a stockpile and backfill 3D model that accounted for waste from mining and processing, and void volumes in backfilled pits.

Table 6-5 Descriptive statistics for terrain properties in the Georgetown analogue area

Variable	Units	Mean	Coeff. Variation (%)	Minimum	Median	Maximum	1st Quartile	3rd Quartile
slope (%)	%	2.0	50	0.0400	2.0300	6.4300	1.2500	2.7500
elevation (m)	m	9.3	50	0.0	8.7600	22.6300	5.5200	12.5275
profile curvature	radius (m)	-12,500	na	-250	0.0	400	-5,000	5,000
plan curvature	radius (m)	11,111	na	-0.4	1000	1	-100	83
flow length (m)	m	149	205	0.0	60.0	3596.5	20.0	169.7
LS factor[1]		0.55	200	0.0	0.3	22.2	0.1	0.6
erosion/deposition		-0.1	na	-572.0	-1.0	503.0	-3.0	0.0
catchment area	m^2	25300	622	400	2943	2793817	1613	4974

[1] slope length and steepness factor from the universal soil loss equation (Renard et al., 1997)

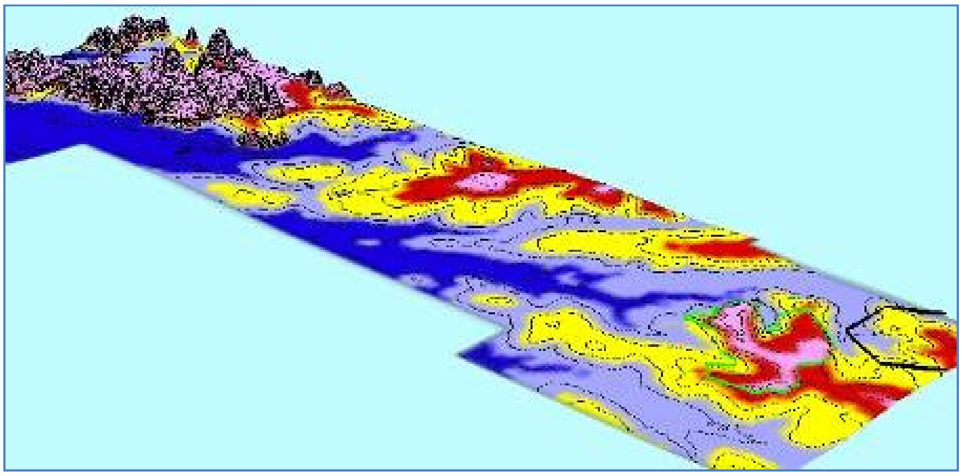
The landform design criteria given in Table 6-6 were based on the measured ranges in terrain properties in the natural analogue area (Table 6-5). Elevation and slope criteria used in Table 6-6 were slightly higher than the maximum values in the analogue area. This was needed to allow for 10 metres of subsidence in the backfilled pits.

Table 6-6 Landform design criteria

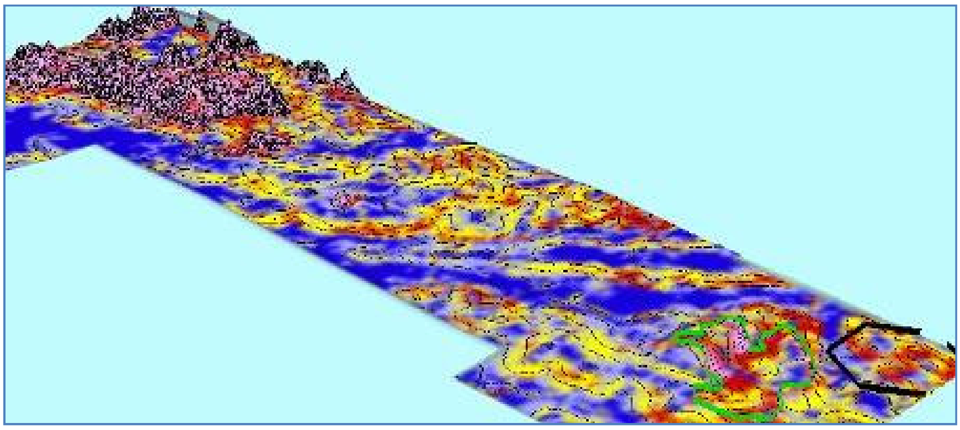
Variable	Units	Range
slope (%)	%	0 – 6.5
relief (m)	m	25
profile curvature	radius (m)	-5000 (concave) to 5000 (convex)
plan curvature	radius (m)	-100 (vale) to 100 (ridge)

6.3.2 *Design evaluation*

The three dimensional perspective views of the landform parameters elevation and slope shown in Figure 6-2 depict ranges of these parameters in broad landscape surveyed in Chapter 4, the Georgetown analogue area used to develop design criteria and the waste rock landform produced by the Ranger mine department. The quartile ranges in both parameters are indicated by different colours in Figure 6-2.



Elevation



Slope

Legend

1st quartile

2nd quartile

3rd quartile

4th quartile

outside the range in the *Georgetown* analogue area

Georgetown analogue area extent

waste rock landform extent

Figure 6-2 Three dimensional perspective views of terrain properties highlighting analogue area ranges

The inter-quartile range in slopes was associated with hill slope environments. Elevation and slope parameters were slightly out of range of the *Georgetown* analogue area. A magenta shading indicates slopes and elevations in the final landform design and the broader landscape that are outside the range of these parameters in the Georgetown analogue area. Erosion issues associated with these areas of the landform design are highlighted in subsequent independent evaluation of erosion.

The reconstructed terrain appeared to be similar in oblique aerial view to the pre-mining landscape (Plate 6-3).

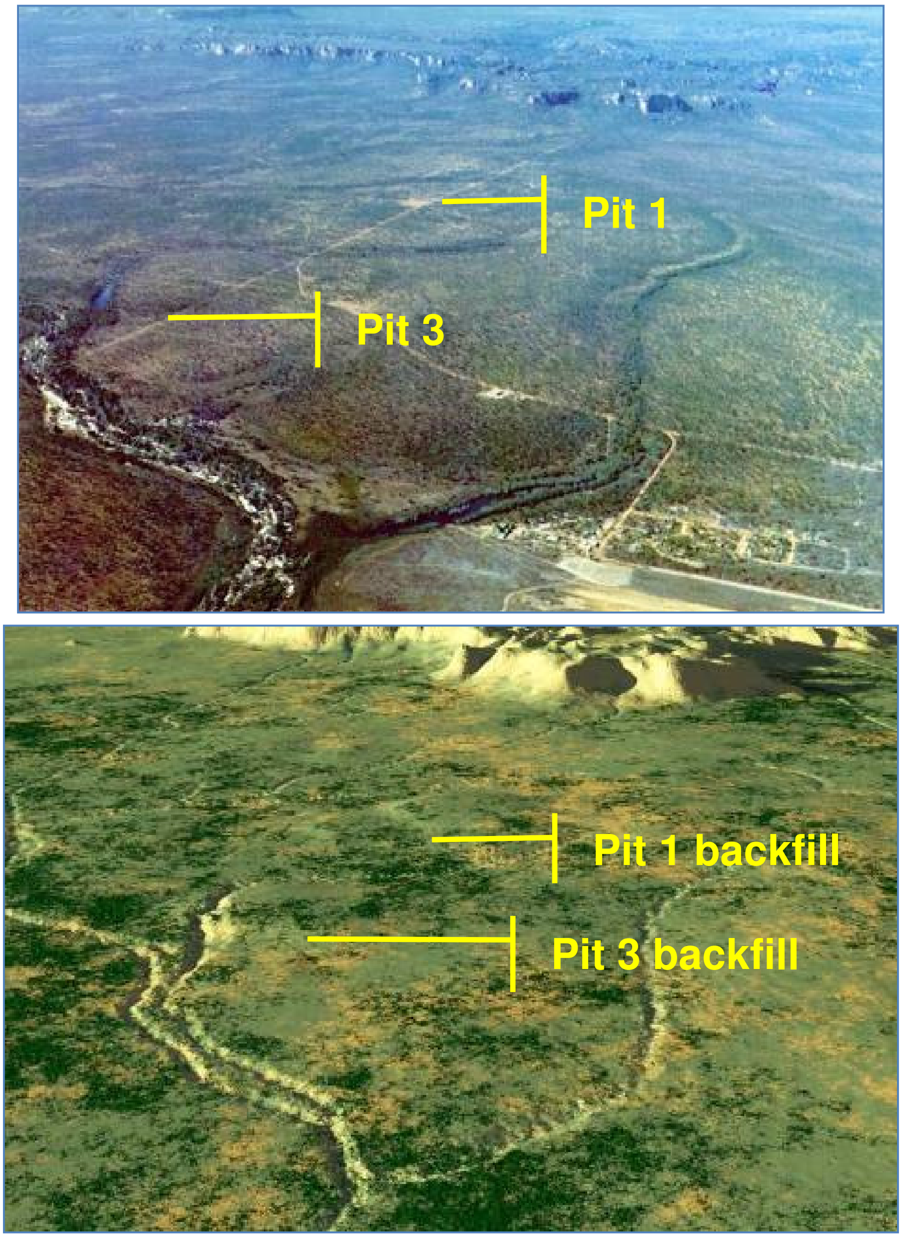


Plate 6-3 Ranger mine area air photo before (top) and after (bottom) computer simulated view (south)

Viewing a computer rendered image of the landscape design with the 2001 mine landscape (Plate 6-4) confirmed the backfilling of the pits, reshaping of residual waste stockpiles and the tailings storage facility and the removal of mine water ponds.

Ten meters of surcharge of waste rock over the Pit 1 backfill created a steep batter slope in the foreground of Plate 6-4.

Plate 6-4 Ranger mine in 2001 (top) and reconstructed landscape visualisation (bottom) – north view

This surcharge was designed to allow for 100 metres of tailings in the backfilled pit to consolidate over time and preserves a water shedding surface over the reconstructed landscape.

6.3.2.1 Landform evolution

The difference between the initial elevation of the reconstructed landscape and elevation after 1 000 years of simulated erosion (Lowry et al., 2006) is depicted in Figure 6-3. Gullies, three to five metres deep were predicted in steeply sloping areas abutting the creek lines for natural (a) and worst case (b) modelling scenarios, while three to four metre deep gullies were also predicted in the natural analogue area over the same period.

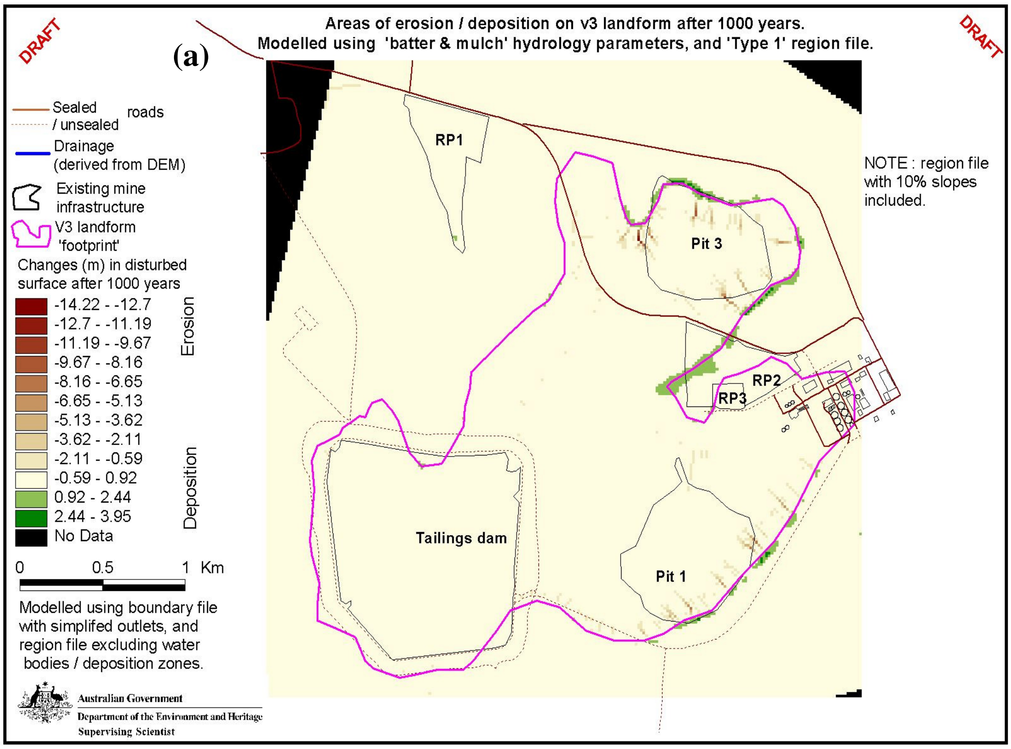
6.3.2.2 Eco-hydrology

An impeding layer, defined as the upper section of the C horizon in the soil profile, was introduced into the 3D model to generate stream flow, as the hydrographic data implied that groundwater was at depth across most of the catchment. All vegetation was as per the 1D base-case, *i.e.* an LAI range of 0.8-1.6, with the seasonal LAI shown in Figure 6-4.

Observed and simulated stream flows for the water years 1994 to 1998 are compared in Table 6-7. While the model did not fit the observed data well for any given year, the overall averages were close enough for simulated (350 mm.yr-1) and observed (361 mm.yr-1) stream flow.

The area of the simulated catchment shown in Figure 6-5 is 306 hectares. The depth of material placed over the existing soil-surface during rehabilitation is also identified. The two deep areas of fill are mine pits and the fill plateau in the western section is the tailings storage facility.

The simulated ET components of the water balance are given in Table 6-8. There is only a small year-to-year variation in the ET components. The average total simulated ET is 1 263 mm per year with a range of 1 236 to 1 293 mm per year.



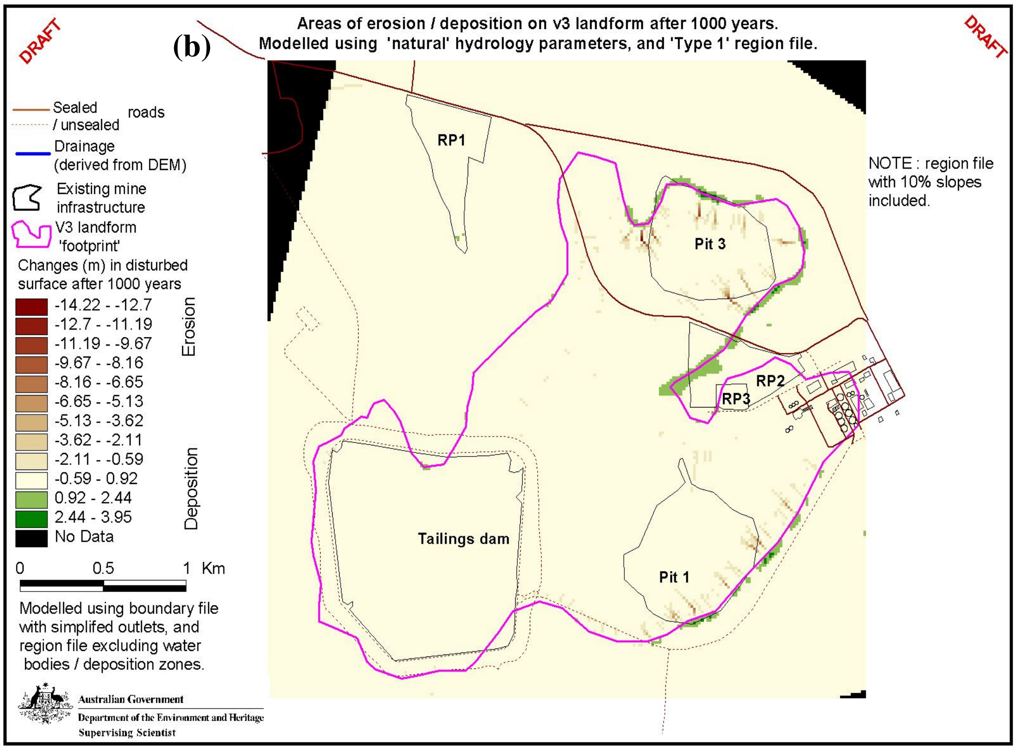


Figure 6-3 Areas of potential erosion — deposition after 1000 years using (a) batter/mulch hydrology parameters; and (b) natural hydrology parameters

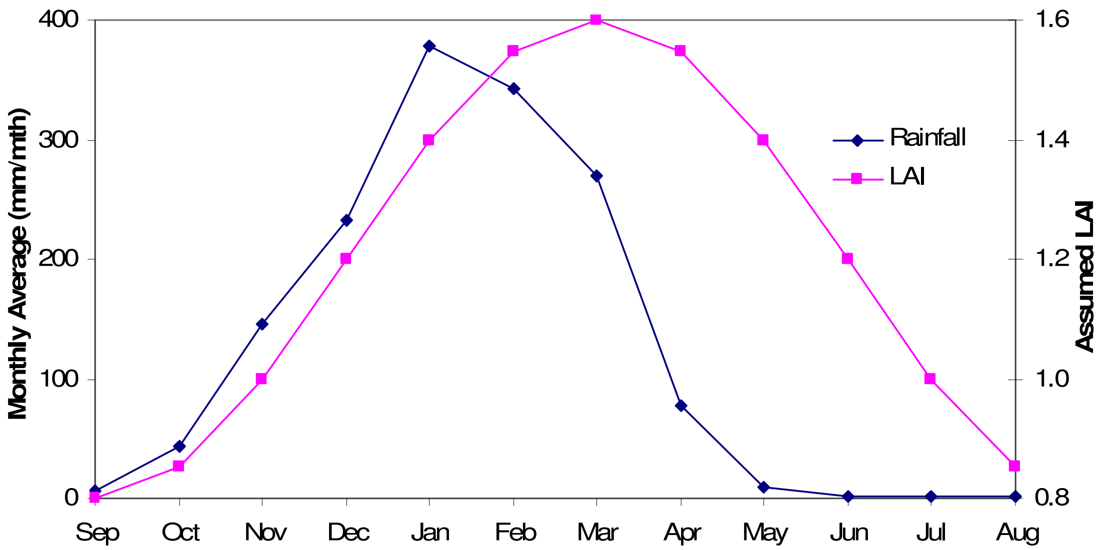


Figure 6-4 Assumed equivalent LAI distribution through the year with average monthly rainfall plotted for comparison

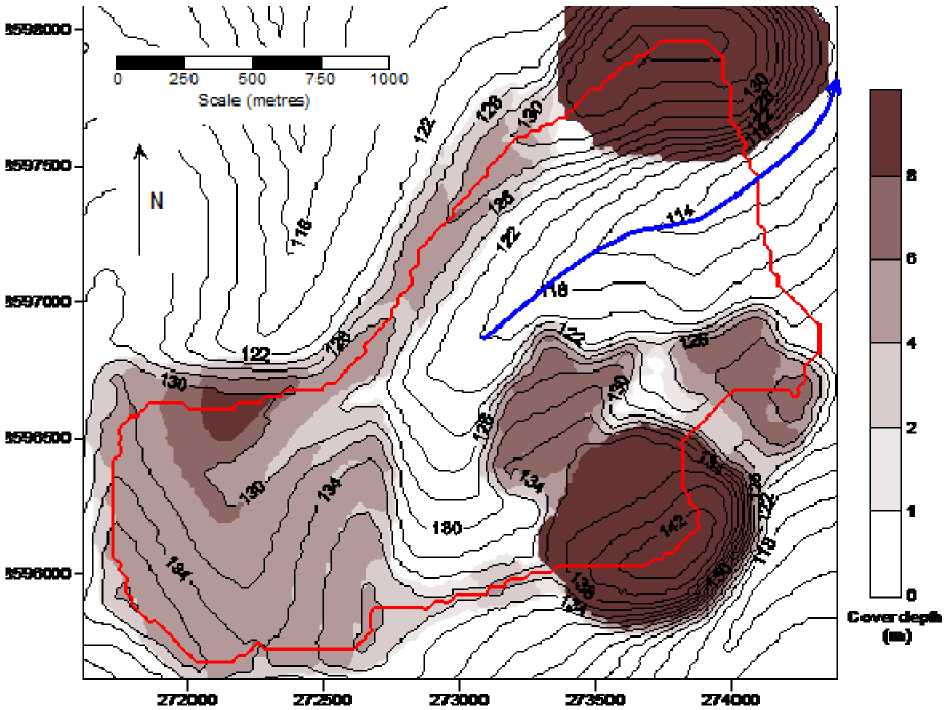


Figure 6-5 Ranger mine catchment used in the WEC-C 3D model

Table 6-7 Comparison of observed and simulated annual stream flows for Corridor Creek. The runoff coefficient is based on observed flows. Average rainfall was 1,681 mm/yr

Water Year	Observed Stream flow (mm/yr)	Simulated Stream flow (mm/yr)	Runoff Coefficient (%)
1994	350	282	22
1995	410	453	23
1996	211	75	15
1997	376	676	19
1998	456	265	27
Average	361	350	21

Table 6-8 Simulated annual ET components for Corridor Creek

Water Year	Rainfall	Pan Evap.	Overstorey ET	Understorey and soil ET	Total ET
1994	1,606	2,610	586	666	1,252
1995	1,754	2,486	579	669	1,248
1996	1,429	2,450	580	700	1,280
1997	1,950	2,548	582	654	1,236
1998	1,667	2,508	582	711	1,293
Average	1,681	2,521	582	680	1,262

The 3D model profile had nine layers and the main soil properties for the covered and uncovered areas are given in Table 6-9. The γ-θ and θ-K relations used for the various soils in the simulations are presented in Table 6-10. These relations were based on those used at a range of sites and adapted to Ranger soils, based on soil descriptions, conductivity values for cover materials (Chapter 5), and preliminary measurements for the natural analogue area.

Table 6-9 Layering and parameters for the 3D mine model

Layers	Total Thickness (m)	Soil Type	K_s (mm/day)	Vegetation Roots
Waste rock covered areas:				
1	1.0	cover	50	yes
2	1.0	clay liner	10	yes
3 & 4	variable	waste rock	24,000	yes
5	0.3	upper C horizon	2	yes
6 to 8	18.5	C horizon	10	yes in top layer
9	20.0	main aquifer	10	no
Natural	areas			
1	0.4	A horizon	1,000	yes
2	0.4	B1 horizon	100	yes
3 & 4	0.4	B2 horizon	10	yes
5	0.3	upper C horizon	2	yes
6 to 8	18.5	C horizon	10	yes
9	20.0	main aquifer	10	no

The average runoff, groundwater recharge (water discharging in the model via the drainage layer) and transpiration ratio results of the 1D hydrological model simulations are shown in Table 6-11. A transpiration ratio (actual: potential) equal to 1.0 in Table 6-11 indicated that the vegetation was not water stressed during the simulation and was able to transpire at potential rates; a transpiration ration less than 1.0 indicated plant water stress with demand exceeding soil water supply. For the Corridor Creek simulations, the transpiration ratio was close to 1.0 and varied with the sinusoidal variation in LAI (Figure 6-4).

An average transpiration ratio of less than 0.9 indicated significant periods of stress and implied that the available water could not maintain LAI. In Table 6-11 the average transpiration ratio was 0.96 for all cases having the base case LAI of 0.8-1.6, but all cases were below 0.9 for the higher LAI of 1.0-2.0, which implies that the vegetation would not maintain the higher LAI that would be closer to the 0.8-1.6 case. The 0.8-1.6 LAI base cases for runoff, groundwater recharge and transpiration ratio indicate that the vegetation establishment was successful.

Table 6-10 γ-θ and θ-K relations used for the various soils in the simulations. Units are mm of water for θ and mm/day for K

Natural soil		**Mine landform cover**	
γ-θ	**θ-K**	**γ-θ**	**θ-K**
A horizon		Erosion resistant surface	
0.002	0.002	0.033	0.033
0.31	0.02	0.32	0.30
0.34	0.3	0.34	0.35
0.35	0.35	0.35	
B1 horizon		Clay liner	
0.123	0.123	0.123	0.123
0.320	0.33	0.320	0.33
0.340	0.35	0.340	0.35
0.35		0.35	
B2 horizon		Waste rock	
0.123	0.123	0.002	0.002
0.320	0.33	0.31	0.02
0.340	0.35	0.34	0.3
0.35		0.35	0.35
Upper C horizon			
0.123	0.123		
0.320	0.33		
0.340	0.35		
0.35			
C and D horizon			
0.123	0.123		
0.320	0.33		
0.340	0.35		
0.35			

The variation in runoff (under 6–10% of rainfall) between the cases in Table 6-11 is small because the hydraulic conductivities of the cover and the clay liner, and the water holding capacities of these layers, control the runoff. The variation in hydraulic conductivity for the clay drainage limiting layer has a limited effect.

Table 6-11 Average runoff, groundwater recharge and transpiration ratio for the 1D model for different cover scenarios

Runoff (% rainfall):				
	K_s clay (mm/day)			
LAI	5	10 (base)	20	40
0.4 - 0.8	10	7	6	6
0.6 - 1.2	9	7	6	6
0.8 - 1.6 (base)	8	6	6	6
1.0 - 2.0	8	6	6	6
Groundwater Recharge (% rainfall):				
	K_s clay (mm/day)			
LAI	5	10 (base)	20	40
0.4 - 0.8	20	24	25	25
0.6 - 1.2	12	15	16	16
0.8 - 1.6 (base)	6	8	9	9
1.0 - 2.0	3	5	6	6
Transpiration Ratio:				
	K_s clay (mm/day)			
LAI	5	10 (base)	20	40
0.4 - 0.8	1.00	1.00	1.00	1.00
0.6 - 1.2	1.00	1.00	1.00	1.00
0.8 - 1.6 (base)	0.96	0.96	0.96	0.96
1.0 - 2.0	0.84	0.85	0.85	0.85

In the case of recharge, the range of values is larger, (3–25%) as both LAI and soil conductivity control groundwater recharge — the water that escapes use by the vegetation. Assuming successful revegetation,the range of groundwater recharge could have been 6–9% for the simulated soil profiles. However, the recharge could account for 25% of rainfall if the revegetation is unsuccessful (from Table 6-11). Table 6-12 provides some insight into the likely water balance components if the ideals embodied in Table 6-11 are not met. In the first column of Table 6-12 for the waste rock profile, runoff reduced to 0 meant recharge rose to 26% for the selected LAI of 0.6-1.2. Any water entering this type of profile would drain freely to groundwater due to the high hydraulic conductivity of the waste rock and most of the water will pass the rooting zone. In the second column, characterised by waste rock underlain with a sub-soil and assumed LAI equal to the base case (0.8-1.6), the recharge is still high due to the free-fall of water in the top 5 metres of profile. This scenario could occur where waste rock is placed over a natural land surface.

Table 6-12 Average runoff, groundwater recharge and transpiration ratio for the 1D model for the four cover scenarios. All cases use base-case soil values unless stated otherwise

LAI	1. All primary rock	2. Rock to subsoil	3. No subsoil	4. Shallow roots	5. Weathered rock
Runoff (% rainfall):					
0.4 - 0.8	0	0	7	7	7
0.6 - 1.2	0	0	7	6	7
0.8 - 1.6	0	0	6	6	7
Groundwater Recharge (% rainfall):					
0.4 - 0.8	32	32	24	24	23
0.6 - 1.2	26	24	15	17	14
0.8 - 1.6	22	8	10	13	7
Transpiration Ratio:					
0.4 - 0.8	1.00	1.00	1.00	1.00	1.00
0.6 - 1.2	0.92	0.99	0.99	0.95	1.00
0.8 - 1.6	0.81	0.94	0.92	0.85	0.97

As shown in Table 6-12, column 3, which is characterised by subsoil in layers 8 to 12 with highly conductive waste rock, did not affect runoff but increased the recharge under the LAI of 0.8-1.6 to 10%. It also caused the transpiration ratio to drop from 0.94 to 0.92. This case represents the situation where waste rock of considerable depth would cover the tailings in the base of the backfilled pits.

The fourth column in Table 6-12, the shallow-rooting case, highlighted the sensitivity to a poorly understood variable. Tree roots were assumed to reach 8–m in the base case (Table 6-4); in the shallower–rooting vegetation case they are assumed to reach 5–m, that is the waste rock–subsoil interface. The reduced rooting depth has caused the base case LAI (0.8-1.6) to have a transpiration ratio below the 0.9 water stress indicator. The reduced transpiration ratio for the shallow rooting scenario implies that a lower LAI (0.6-1.2) is likely and the recharge rate of 8% predicted in Table 6-11 could increase to 17% predicted in Table 6-12.The last column in Table 6-12 presents the results of the weathering of the waste rock and the reduction of its K_s from 24 000 mm.day^{-1} to 100 mm.day^{-1}. This large change only reduced the rate of groundwater recharge from 8% for the base case in Table

6-13 to 7% in Table 6-12. It takes only 12 days for a wetting front to pass through the 3 metres of the waste rock layer, and so, in terms of wet season time scales there is little real difference between the two cases, in spite of the much lower conductivity. The water holding capacity of the waste rock hasn't been changed, only its conductivity.

The average runoff coefficient (9.6% of rainfall) was comparable to expected runoff coefficients from the 1D modelling. However, because those areas with cover material had a surface K_s of 50 mm.day^{-1}, while those without had a surface K_s of 1000 mm.day^{-1}, a bimodal runoff response was simulated for extreme rainfall events. The daily simulated runoff distributions on 2, 3 and 4 January 1997 in Figure 6-6 illustrate this bimodal response. Rainfalls were 119 mm on the first day and 148 mm on the second. On 2 January 1997, runoff from the uncovered areas varied from 0 – 100 mm while for the covered areas it was a uniform 59 mm. The amount of soil moisture store available in the A and B Horizons controlled runoff on the uncovered areas, while the conductivity of the surface soil in combination with its matrix potential controlled runoff from the covered area. So uncovered (natural) and covered (waste rock) produced different runoff processes — saturation excess controlled and infiltration excess controlled respectively.

The 3-D simulations indicated that there was no interflow within the covered areas, *i.e.* seepage at the interface between the cover and the original soil-surface was completely absent. This is because the downward flux at the interface between the cover and the soil was less than the infiltration capacity of the underlying soil, and so, saturated conditions, which are required to generate the interflow, don't develop there. It is possible that a situation could develop where such seepage takes place; but this would require either the underlying soil to have a lower conductivity than the 2 mm.day^{-1} presently used, or the deep groundwater system would rise to this level and create a saturation excess at the interface. There was an average deep recharge rate of 9.4% of rainfall during the 13 years of the simulation (146 mm.yr^{-1} or 1,900 mm in total). The present model is a closed system and excess groundwater can only express itself as a groundwater rise. The simulated groundwater depth for 16 May 2003 is depicted in Figure 6-7.

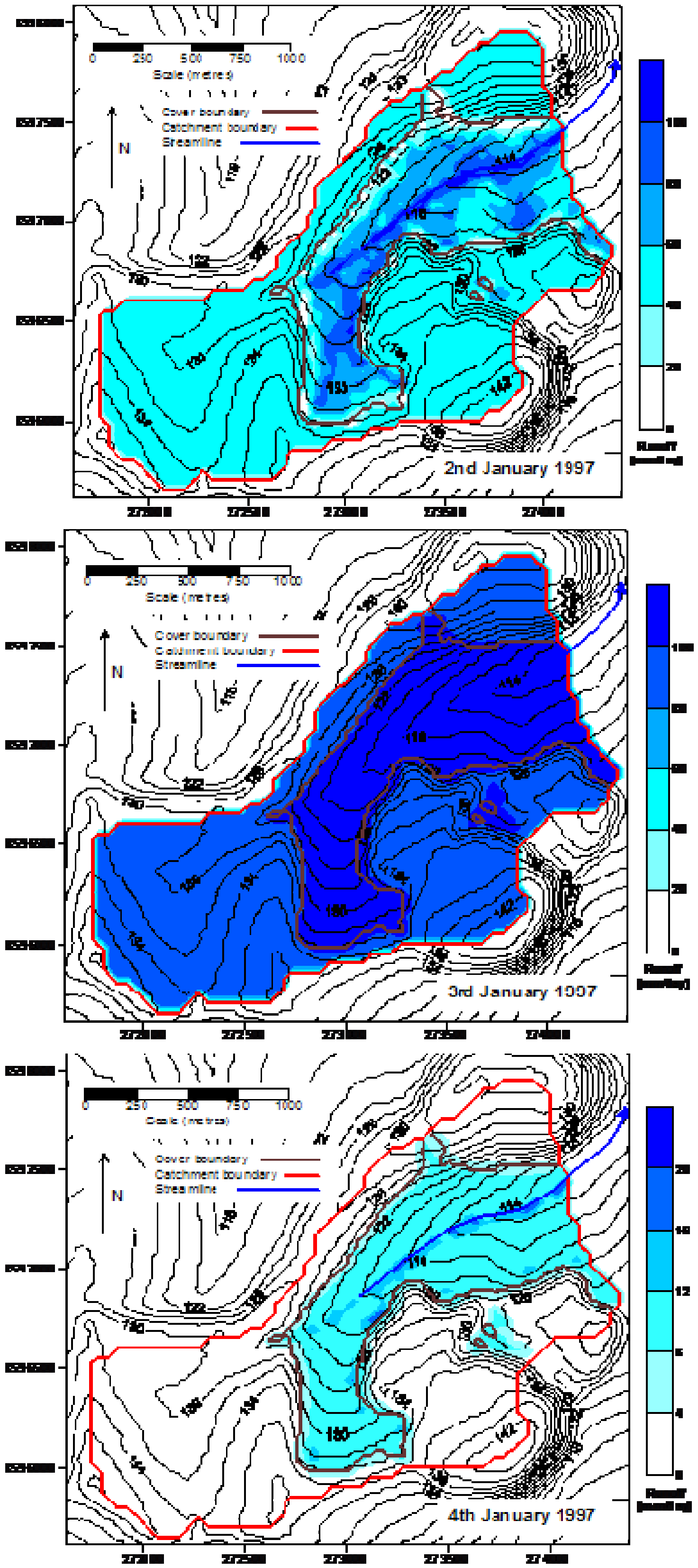


Figure 6-6 Daily simulated runoff distributions on 2nd, 3rd and 4th January 1997

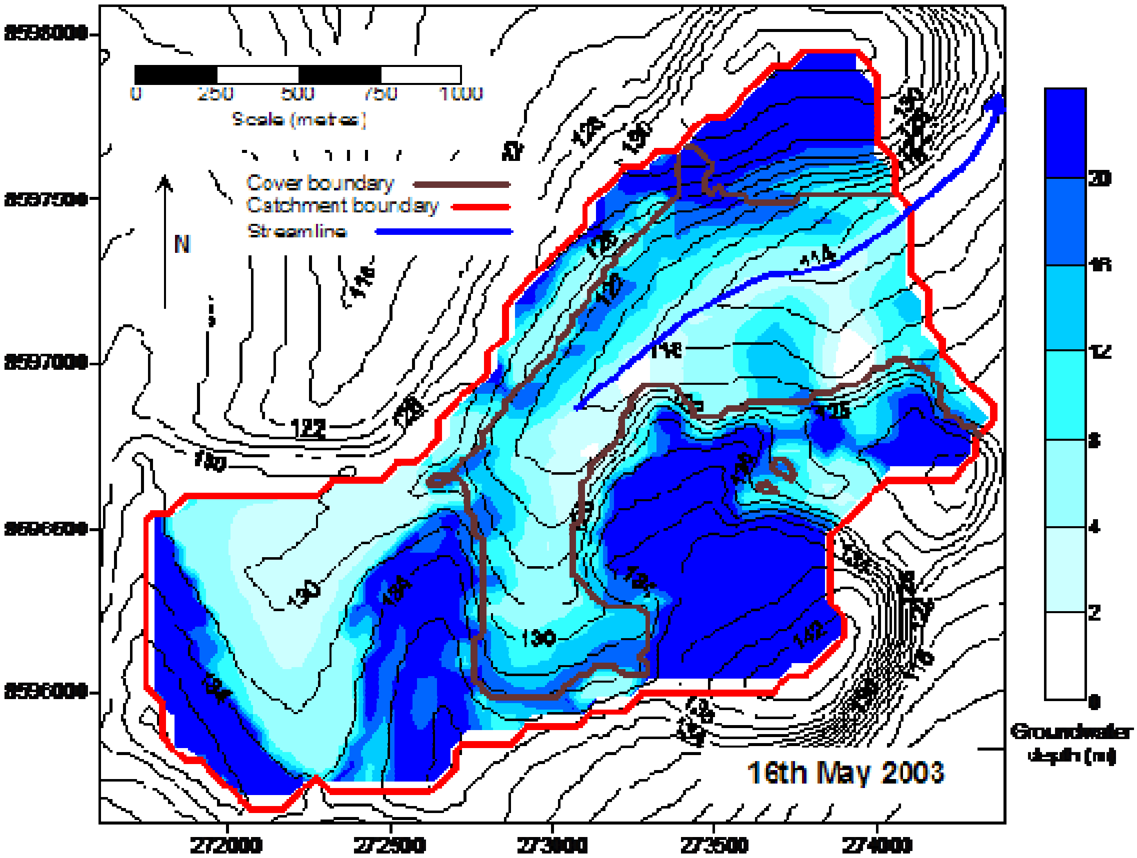


Figure 6-7 Simulated groundwater depths for 16th May 2003

The simulation started with a minimum groundwater depth of 20 metres and groundwater rose by 2003 to be close to the surface over a large proportion of the catchment. Continuing the simulations would produce groundwater seepage areas. If the post-rehabilitation groundwater discharge from the catchment can absorb the net recharge and negate the simulated rises shown in Figure 6-7 then discharge will not occur. However, more information on the hydrogeology of the area is needed before this can be simulated.

6.3.3 *Environmental surveys*

6.3.3.1 Soil properties

The ranges in soil drainage status, gravel content, A horizon depth, A horizon texture, B horizon texture and depth to rock for each ecosystem type or class across the Georgetown analogue area (defined from Chapter 4) are depicted as box plots in Figure 6-8.

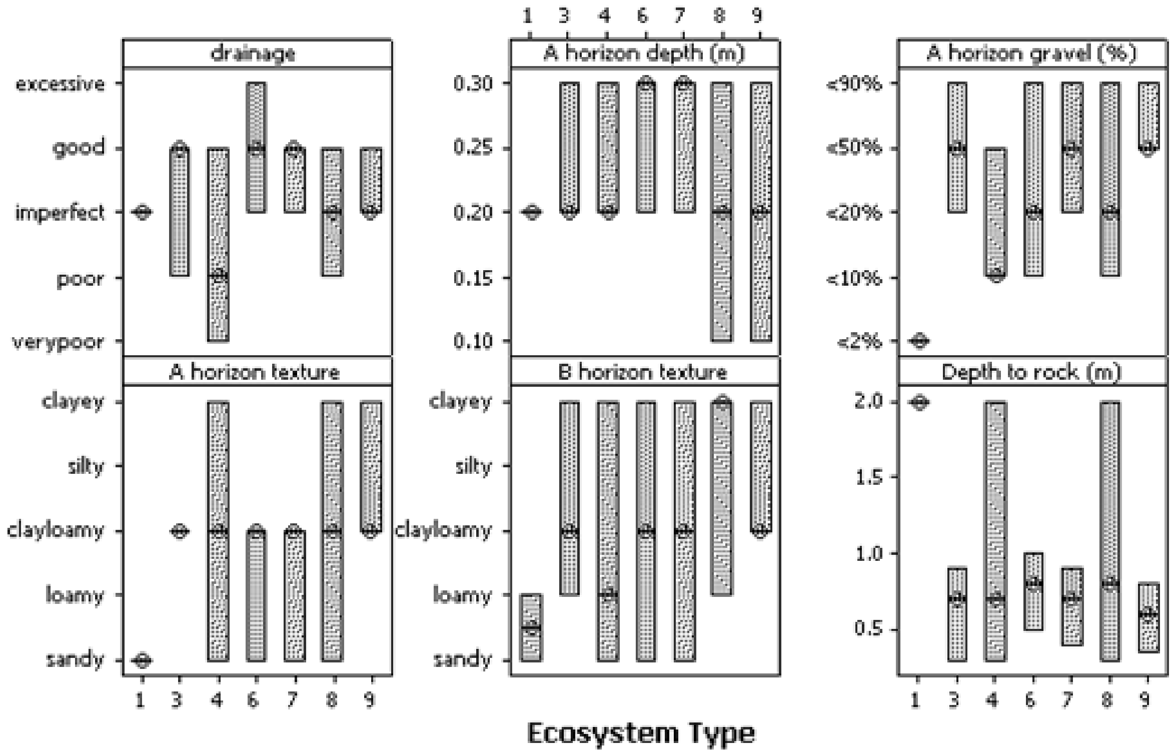


Figure 6-8 Range (boxes) and central tendency (median line) of soil morphological properties measured from 100 sites in the Georgetown analogue area

The analogue area comprised very poor and poorly drained flats (ecosystem 4), poor to imperfectly drained lower slopes (ecosystem 8), and imperfectly drained to well drained crests and slopes on rises (ecosystems 3, 7 and 9). The depth to a root barrier, such as rock or laterite, varied between 0.5 and 1.0 m for ecosystems formed on rocky substrates (ecosystem classes 3, 6, 7, and 9) while soil depths in ecosystems with some colluvial or alluvial influence (ecosystem classes 4, 8) were more variable. The soils were gravelly. The soils on slopes and crests (ecosystems 3, 6, 7, 9) tended to be moderately gravelly to very gravelly in the A horizon, where plant root activity is concentrated. Soils on flats and lower slopes formed on colluvium and alluvium (ecosystems 4 and 8) were less gravelly. Surface soil textures ranged from sand to sandy loam and subsoil textures ranged from sandy loam to clay.

Table 6-13 Summary statistics for soil physical and hydraulic properties

Variable	Unit	N	Minimum	Median	Maximum	Range
gravel	%	76	17.0	51.0	84.0	67.0
sand	%	76	44.5	82.5	93.5	49.0
silt	%	76	2.0	6.0	22.0	20.0
clay	%	76	3.0	10.5	46.5	43.53
infiltration rate (dry)	$mm.min^{-1}$	24	19.53	47.73	167.77	148.24
infiltration rate (wet)	$mm.min^{-1}$	24	5.47	17.53	75.50	70.03
sub-soil permeability	$mm.min^{-1}$	24	1.35	13.96	75.48	74.13
penetration resistance	MPa	24	0.780	1.305	2.610	1.830
water retention @ -10kPa	$m^3 \cdot m^{-3}$	24	0.15	0.19	0.35	0.20
water retention @ -1500 kPa	$m^3 \cdot m^{-3}$	24	0.03	0.04000	0.15000	0.12
plant available soil water	$mm.min^{-1}$	24	110	145	320	210
density	$t.m^{-3}$	24	1.08	1.4800	1.9600	0.88
porosity	%	24	26.23	44.18	59.10	32.87
aeration	%	24	7.35	25.28	36.16	28.81

Soil profiles are gradational, characterized by gradual increase in clay content with depth (Figure 6-9). The ranges of soil physical and hydraulic test statistics are presented in Table 6-13. Volumetric soil gravel contents ranged from 17 to 84%, which corroborates the field morphological descriptions. The gravel fraction is relatively inert and do not contribute significantly to water and nutrient retention or conduct flow.

In terms of soil fertility, the major nutrients, especially nitrogen and phosphorus, were concentrated in the surface 0.2 metres of the soil profiles with maximum levels in the very near surface 0 – 0.05 metres (Figure 6-10 a,b). However, potassium was more unevenly distributed in the profiles and appeared to be superficially related to cation exchange capacity (Figure 6-10 c,d).

The carbon profile in the soil was similar to distributions of nitrogen and phosphorus (Figure 6-11 a). The profile distributions of the micronutrients iron (Fe) and manganese (Mn) was also similar to the carbon profile (Figure 6-11 a,b), while copper (Cu) and zinc (Zn) profiles were different from the carbon profile and are less closely associated with root growth and nutrient cycling in the woodland vegetation.

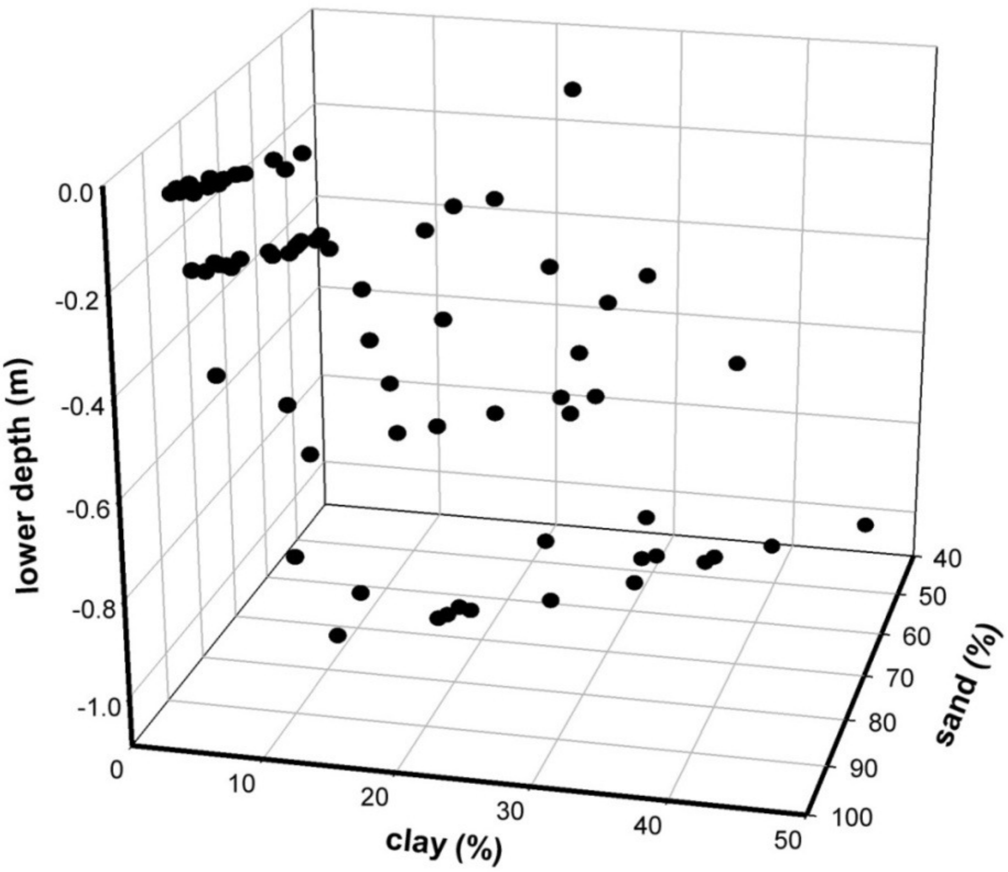


Figure 6-9 Sand and clay content with profile depth

6.3.3.2 Vegetation

Field survey work confirmed a savanna community framework of *Eucalyptus miniata-E. Tetrodonta*, comprising a discontinuous tree canopy with a seasonal understorey of grasses, within which a range of other vegetation communities occur. The tall *E. tetrodonta—C. bleeseri* mixed open woodland associated with ecosystem type 3 is depicted in Plate 6-5a. The density and canopy structure was variable. The mid-high *Melaleuca viridiflora* open woodland with a very sparse mixed tree midstorey and dense mixed grass—*Cyperaceae* understorey associated with ecosystem type 4 is depicted in Plate 6-5b. Isolated clumps of overstorey vegetation are evident. The tall *E. tetrodonta—E. miniata* open woodland with a sparse mixed tree midstorey and mid-dense *Heteropogon triticeus—Sorghum intrans* understorey associated with ecosystem type 6 is depicted in Plate 6-5c.

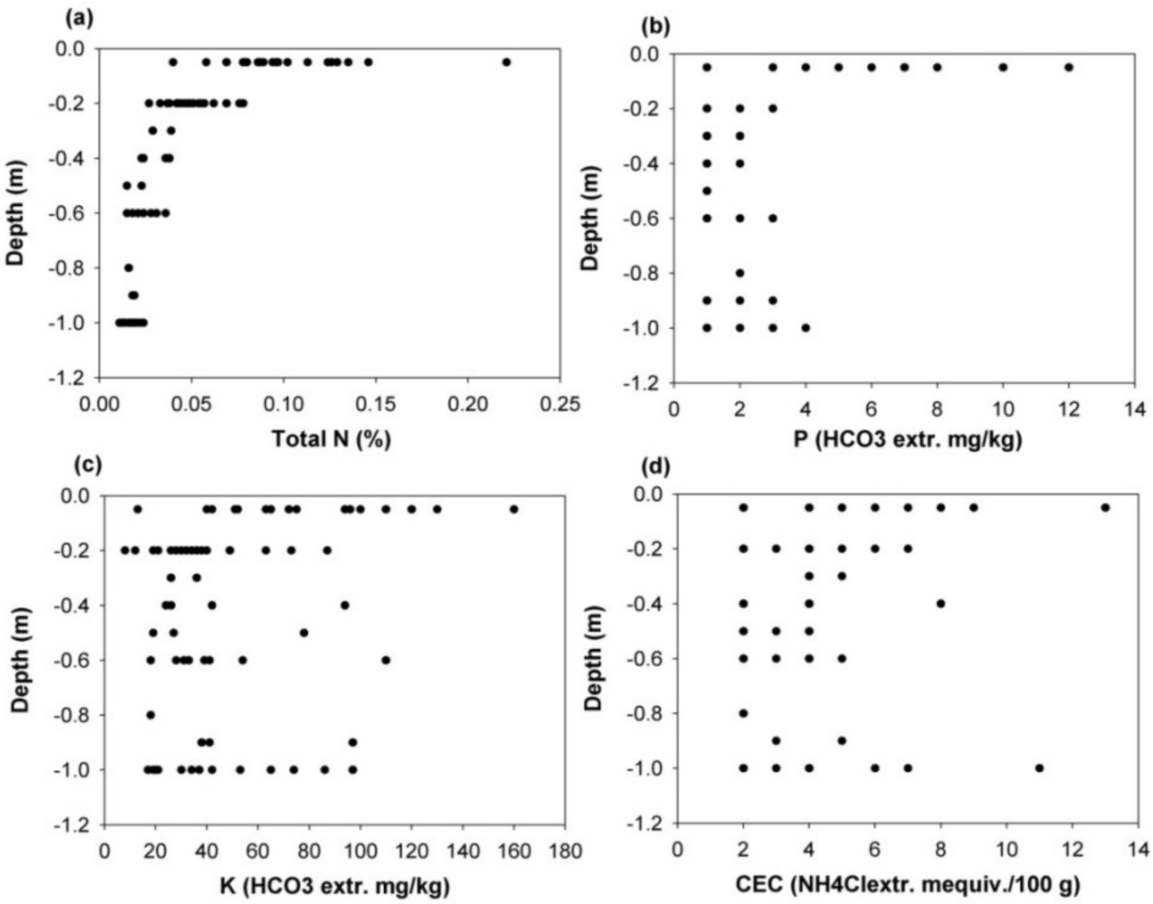


Figure 6-10 Major nutrient and cation exchange capacity profiles for analogue soils

There appeared to be differences in the density of recruiting overstorey species in the mid and understorey. Tall *E. tetrodonta—E. miniata—E. porrecta* open forest with a sparse mixed tree midstorey and mid-dense *Sorghum intrans—Petalostigma quadriloculare* understorey associated with ecosystem type 7 is depicted in Plate 6-5d. Tall *E. foelscheana* mixed open woodland with a sparse mixed tree midstorey and sparse *Heteropogon triticeus—Themeda triandra* understorey associated with Ecosystem type 8 is depicted in Plate 6-5e.

Mid-high *E. tectifica—E. tetrodonta—E. confertiflora* open woodland with a sparse mixed tree midstorey and sparse *Sorghum intrans—Eriachne spp. —Petalostigma quadriloculare* understorey associated with ecosystem type 9 is depicted in Plate 6-5f. A high level of patchiness in canopy structure within as well as between ecosystems was evident from the site photography.

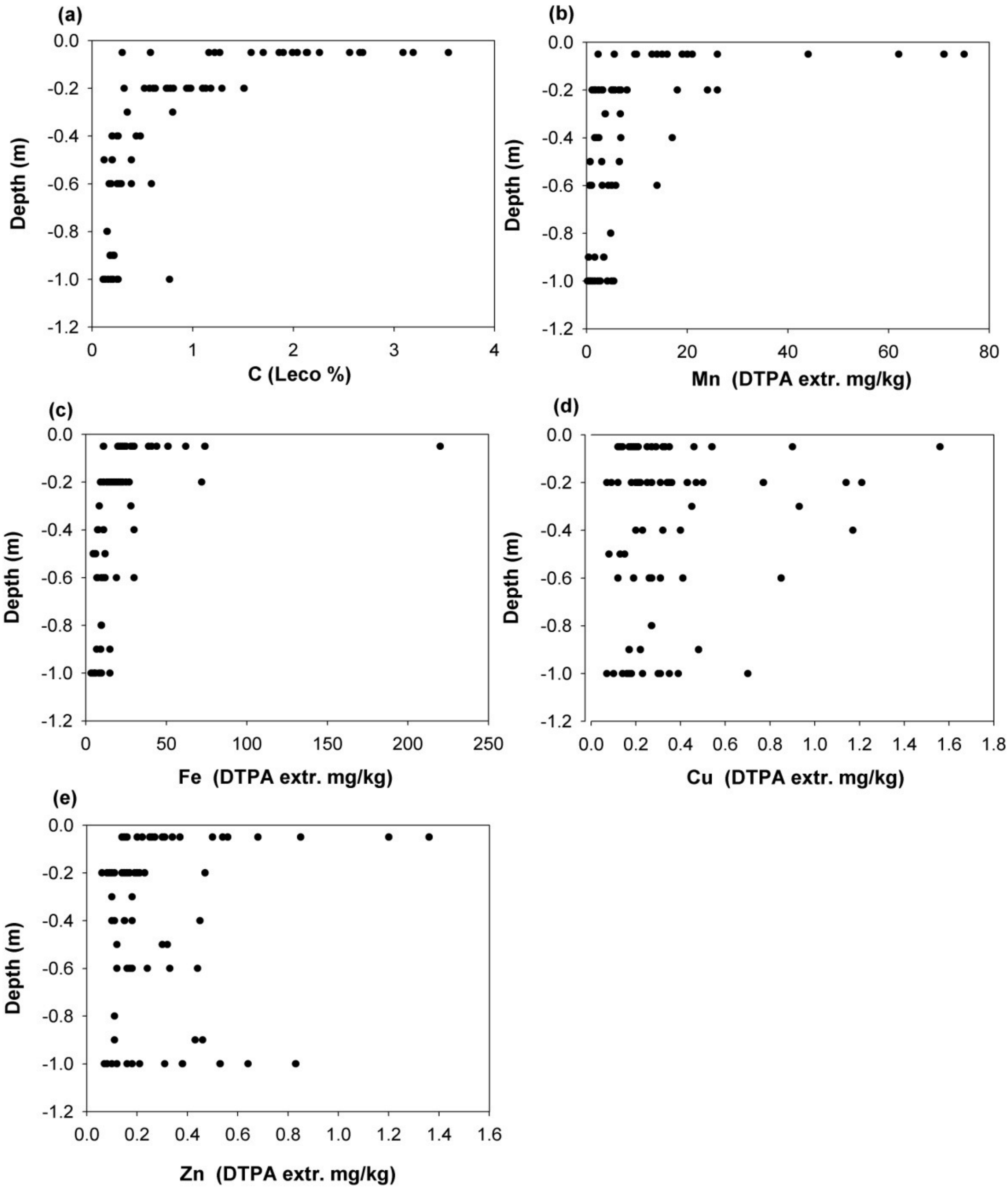


Figure 6-11 Total carbon and micronutrient profiles
for analogue soils

The patchiness in plant community structure across ecosystem types is described in Figure 6-12 a-f. Canopy cover varied from 60 to less than 20% while ground cover ranged from 40 to 70%. Isolated clumps of melaleuca woodland in the wetland ecosystem type 4 produced a wide range of overstorey stem density. Stem densities in the other ecosystem types (eucalypt woodlands) ranged between 200 and 400 stems per hectare. Woodland height ranged from about 10 to 17 metres. Stem diameters ranged between 10 and 25 centimetres. Even though there were 2 to 7 species in the overstorey at most sites the understorey contained many more species. Woodland ecosystems contained between 2 and 6 species in the overstorey (mostly eucalypts), between 2 and 11 in the midstorey and 11 to 40 understorey species. Species counts indicated that ecosystems 6 and 9 had the greatest number of species on average, and ecosystem 4 (the melaleuca wetland) the lowest species richness. However, the presence of relatively uncommon species in this ecosystem type increased the Shannon-Weiner biodiversity index (Figure 6-12 i).

The framework species, their dominance (importance values) and their occurrence across vegetation community (ecosystem) types are listed in Table 6-14. These species occurred at more than one site in a particular ecosystem and across more than one ecosystem type. In the overstorey, 8 species qualified as framework species. A further 14 species (including 3 overstorey species) qualified as framework species in the midstorey. An understorey species qualified as framework species if it did not occur in other strata but was found in more than one vegetation community. Ecosystem type 4 (*Melaleuca* grassland/sedge land) was unique in structure.

The relative importance value (IV) in Table 6-14 identified *Eucalyptus tetrodonta*, *E. miniata* and *C. foelscheana* to be the three most dominant framework overstorey species and one that would fit well in rehabilitated woodland ecosystems at Ranger mine.

Cochlospermum fraseri, *Acacia mimula*, and *Xanthostemon paradoxus* were the three most dominant in the midstorey stratum and the three most dominant understorey species were *Sorghum intrans*, *Petalostigma quadriloculare* and *Aristida holathera*. Of the 126 species recorded the presence of these 9 species occupied most of the available space across the ecosystem range in the analogue area.

(a) Ecosystem Type 3 (Sites 21, 73, 97): Tall Eucalyptus tetrodonta/E. bleeseri mixed open woodland with a sparse mixed tree midstorey and mid-dense Sorghum intrans/Petalostigma quadriloculare understorey

(b Ecosystem Type 4 (Sites 27, 42, 36): Mid-high Melaleuca viridiflora open woodland with a very sparse mixed tree midstorey and dense mixed grass/Cyperaceae understorey

(c) Ecosystem Type 6 (Sites 12, 56, 94): Tall Eucalyptus tetrodonta/E. miniata open woodland with a sparse mixed tree midstorey and mid-dense Heteropogon triticeus/Sorghum intrans understorey

(d) Ecosystem Type 7 (Sites 8, 23, 65): Tall Eucalyptus tetrodonta/E. miniata/E. porrecta open forest with a sparse mixed tree midstorey and mid-dense Sorghum intrans/Petalostigma quadriloculare understorey

(e) Ecosystem Type 8 (Sites 5, 64, 84): Tall Eucalyptus foelscheana/mixed open woodland with a sparse mixed tree midstorey and sparse Heteropogon triticeus/Themeda triandra understorey

(f) Mid-high Eucalyptus tectifica/E. tetrodonta/E. confertiflora open woodland with a sparse mixed tree midstorey and sparse Sorghum intrans/Eriachne sp./Petalostigma quadriloculare understorey

Plate 6-5 Ecosystem photos at vegetation plot survey sites

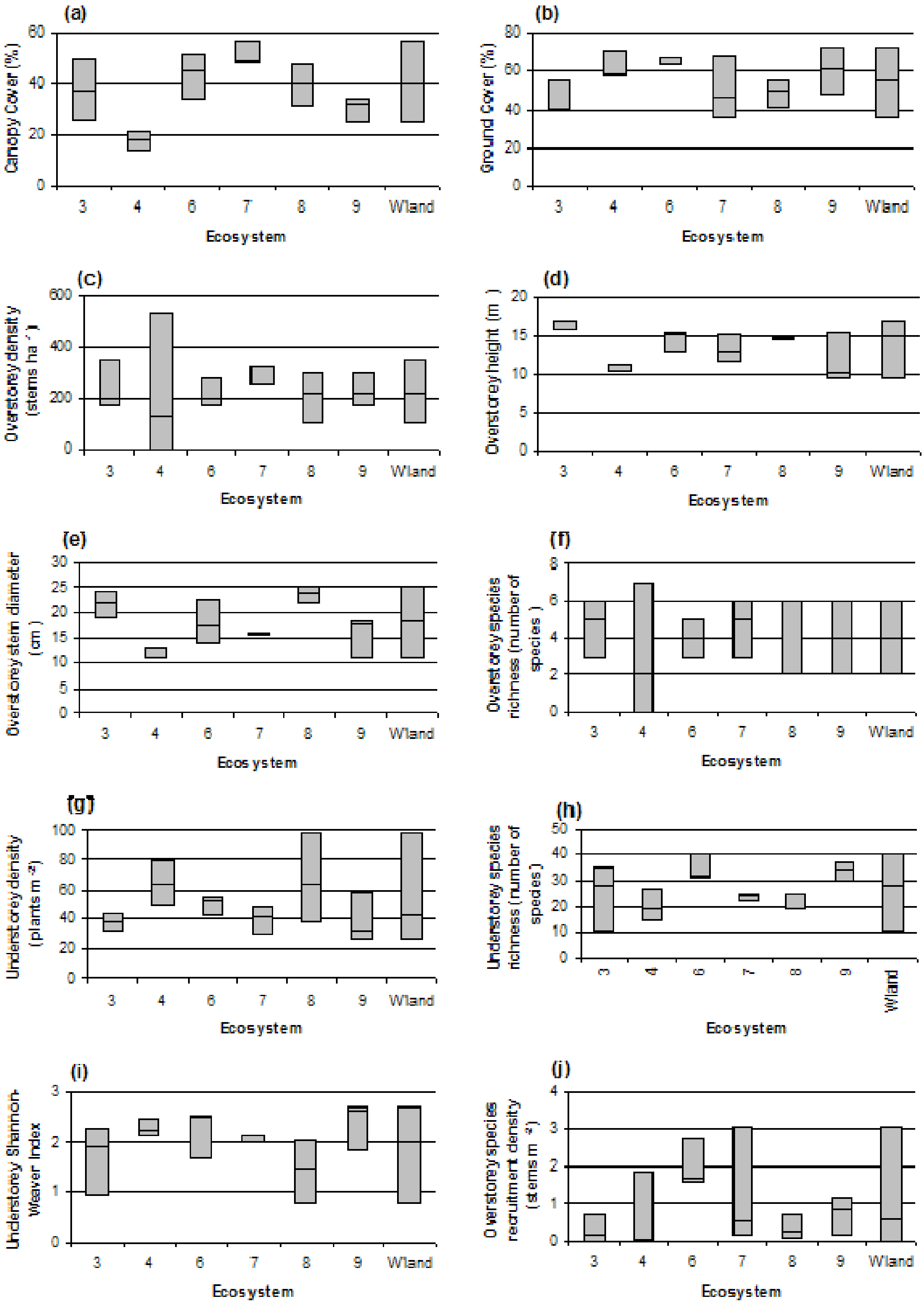


Figure 6-12 Ranges in overstorey and understorey parameters each ecosystem type and grouped woodland

Table 6-14 Framework species in overstorey and midstorey strata for each vegetation community type

Strata	Family	Species	IV[1]	3	4	6	7	8	9
Overstorey	Combretaceae	*Terminalia ferdinandiana*	0.09				x		x
(>8m)	Myrtaceae	*Corymbia bleeseri*	0.17	x					
		Corymbia foelscheana	0.68					x	x
		Corymbia latifolia	0.16						x
		Eucalyptus miniata	0.75	x		x	x		
		Eucalyptus porrecta	0.34	x			x		
		Eucalyptus tetrodonta	1.00	x		x	x		
		Xanthostemon paradoxus	0.37	x		x		x	
Midstorey	Bixaceae	*Cochlospermum fraseri*	1.00					x	x
(2-8m)	Caesalpiniaceae	*Erythrophleum chlorostachys*	0.07			x			
	Combretaceae	*Terminalia canescens*	0.10		x				x
		Terminalia ferdinandiana	0.03	x		x	x		
	Lecythisaceae	*Planchonia careya*	0.10		x			x	x
	Mimosaceae	*Acacia mimula*	0.63	x		x	x		
	Myrtaceae	*Eucalyptus confertiflora*	0.04			x			
		Eucalyptus porrecta	0.07				x		
		Eucalyptus setosa	0.10				x		
		Eucalyptus tetrodonta	0.02	x					
		Melaleuca viridiflora	0.09		x				
		Syzygium suborbiculare	0.05					x	
		Xanthostemon paradoxus	0.26	x		x			x
	Pandanaceae	*Pandanus spiralis*	0.05		x				
	Proteaceae	*Grevillea decurrens*	0.08			x			
		Hakea arborescens	0.04						x
		Persoonia falcata	0.04					x	

[1] IV - Importance Values

6.4 Discussion and conclusions

6.4.1 *Landform design*

The landform design method presented here provides a clear geomorphic and ecological context in the natural landscape, which facilitated communication of design issues at different levels to a range of stakeholders. The description of ecosystem variation across a natural analogue landscape that adequately represents the extent and grain size of environmental processes in the mine landscape (from Chapter 4) also supports a capability to extend the current topographic reconstruction approach (Toy and Chuse, 2005) to include ecological objectives. The implications of the detailed environmental survey of soils, vegetation and terrain properties in a selected natural analogue for landform design and evaluation are discussed below.

6.4.2 *Environmental evaluation*

Gully erosion, plant available water store and groundwater recharge were identified from landscape evolution and hydro-ecological modelling of a first pass landform design that was based on the ranges in slope, relief and catchment area observed in an analogue landscape. These issues can be addressed in the next iteration of the design. In some respects, the landform evolution and hydro-ecological modelling methods also need further development before reliable quantitative assessments can be made. Parameterisation of the SIBERIA landform evolution model (Lowry et al., 2006) did not have the flexibility to allow variation in cover properties that affect runoff. Also, gully initiation in convex shoulder slopes of the landform may to some extent be artefacts of the D8 flow routing algorithm used in the SIBERIA model. A number of enhancements to the model that address some of these issues have been made (Hancock et al., 2008). However, different modelling approach that accurately represents soil forming processes causing temporal changes in soil hydraulic properties, chemical fertility and plant available water could be considered. Mechanistic soil formation models (Minasny et al., 2008) may offer some insights in this respect.

The hydro-ecological simulations highlighted that recharge rates depended on the density of vegetation cover and that the vegetation density depended on the soil profile or landform cover, particularly with respect to the rooting depth, water retention and the downward flux of water. Due to variations in rainfall, both recharge and drought are episodic with low rainfall periods having no significant recharge and high rainfall periods averaging 17% rainfall recharge. Consequently a simulation approach, with support from monitored natural and reconstructed catchments, will be needed to assess the hydrological performance of the mine rehabilitation.

The mechanics of the processes controlling water balance components ET, stream flow generation and groundwater recharge, will determine whether the mine rehabilitation will restore the pre-mine water balance in the upper layers, or whether it will move towards a state with a different total ET and thereby different rates of stream flow and groundwater recharge to that of the natural landscape. Water balance simulation implied the long-term average recharge for the rehabilitated mine area will be something like 8 to 9% of rainfall for the areas overlaying a subsoil and 10% of rainfall for the areas overlaying the pits. In the natural situation, excess water vents from the system via stream flow, which accounts for more than 20% of the water budget compared with 34% for Howard River while estimated ET was 1 110 mm.yr-1 for Howard River (Cook et al., 1998) and 1 240 mm.yr-1 for Corridor Creek. This assumes that the revegetation reaches a density similar to that of the surrounding native areas.

Runoff generation processes in areas with, and without, waste rock cover material behave differently. This is consistent with studies of mine and natural catchment hydrology in the short term {Duggan, 1991 #730}. Over the longer term (approximately 50 years) runoff rates from mine landforms may trend towards that from the natural landscape as vegetation becomes established and rudimentary soils form {Moliere, 2002 #932}. However, initially the covered areas generate runoff during intense rainfall events via an infiltration excess process, while the uncovered areas have a combination of saturation excess and interflow processes. Also there appears to be no interflow from the covered areas at the cover–soil interface. The groundwater system must be able to absorb a net recharge rate of around 9.4% of rainfall to prevent groundwater rise. The rate of groundwater rise still needs to be determined.

6.4.3 *Environmental surveys*

The current work has broadened the scope of existing analogue studies for Ranger mine (Brennan, 2005), which documented well drained woodland sites across the region. The analogue design methods support a range of revegetation outcomes that are integrated with hill slope topography and soil pattern to extend topographic reconstruction methods (Toy and Chuse, 2005) so as to support ecosystem reconstruction in the mine landscape (Nicolau, 2003). This development has practical implications that could improve the success of mine rehabilitation using natural analogues. For instance at Nabarlek mine site nearby to Ranger mine, the topographically reconstructed mine landscape failed to meet the objective to establish uniform eucalypt woodland (Klessa, 2000) because the effects of hill slope hydrology on environmental pattern in the rehabilitated landscape were not resolved.

The soils in the natural analogue area at Ranger are rocky, moderately deep and are gradational in texture profile. These properties are similar to the waste rock cover materials reported on in Chapter 5. However, organic matter associated with enhanced levels of macro and micronutrients has accumulated near the land surface in the native soils that could take time to develop in the revegetated mine soils. This pattern has been observed elsewhere in tropical and temperate deep weathered landscapes (Parrotta and Knowles, 2001; Schwenke et al., 2000b). In noting this, soil organic matter accumulation is a key component in assessing functional integrity of ecosystems in general (Ludwig et al., 2004) and rehabilitated mine soils in particular (Koch and Hobbs, 2007). A solution to this slow accumulation of organic matter is by carefully conserving and using natural topsoil in rehabilitation to enhance the process (Bell, 2001; Parrotta and Knowles, 2001). At Ranger mine, mine scheduling and weed problems meant that topsoil was not conserved and different measures will be needed to restore soil fertility and ensure successful revegetation and rapid development of surface organic carbon content – the key to nutrient cycling across a range of low fertility soils.

The structure of natural plant communities was patchy and this patchiness increased in areas that were poorly or imperfectly drained. The number of species also declined as drainage was impaired from hill slope to valley flat. Consequently, revegetation objectives that do not account for the patchiness induced by hill slope environmental variation are unlikely to be achievable and terrain factors that affect drainage could be expected to influence revegetation outcomes.

6.4.4 *Further work*

Research into the eco-hydrology of natural and reconstructed landscapes to support ecosystem restoration tasks would improve the methodology presented here. Site based measurements of tree canopy characteristics, water use by vegetation and catchment water balance have not been presented and are essential. Further investigations are needed to support accurate hydrological assessment of the mine rehabilitation. The need for a paired catchment study of a constructed landform and a natural analogue catchment was raised in Chapter 5. This study could be designed to address the significant gaps in knowledge of hydrological processes. However, water retention in the root zone, sub-soil hydraulic conductivity and successful revegetation strategies supporting high canopy leaf areas will be key factors in restoring the landscape. Assuming that all of these factors can be addressed, then the landform design and the vegetation survey results presented in this chapter are a basis for predicting the distribution of common and abundant overstorey, or framework species, in the revegetated mine landscape. This is the objective of the next chapter.

A framework of common and abundant species is identified. In the overstorey, 8 species and in the mid-storey 11 additional species qualified as framework species, while 9 overstorey, midstorey and ground cover species occupied most of the available environmental space. One additional species, *Melaleuca viridiflora,* had a limited range — dominating the overstorey and mid-storey in less biodiverse low lying, poorly drained areas. Understanding the way that landform design affects the distributions of these species will underpin successful revegetation for the case study site. In Chapter 4, terrain was identified as an important factor affecting species distributions. Quantitative species distribution models based on the vegetation survey data from Chapter 4, Chapter 6 and regional vegetation surveys are developed in Chapter 7.

Chapter 7 Ecological modelling for landscape reconstruction

7.1 Introduction

7.1.1 *Background*

Landscape restoration following opencast mining requires a site-specific understanding of the biophysical factors that determine environmental patterns in natural landscapes. In the first stage, quantitative methods to select and analyse natural analogue landscapes was presented in Chapter 4. The broad range of ecosystems (represented by vegetation type) in selected natural analogue landscapes that were similar, from environmental terrain analysis, to a mine area were described. In the second stage presented in Chapter 6, the biophysical environment was described from grid and stratified environmental surveys in the selected, analogue landscape to support an ecological landform design evaluation process. Common and abundant plant species that provided an ecological framework in the analogue landscape were identified and basic data on species distribution at 200-m resolution was obtained. Chapter 7 uses this survey support to model the distribution of common and abundant woodland species and biodiversity in the landform design to test the analogue approach.

Elsewhere, the biodiversity surveys that support environmental restoration of landscapes have focussed on regional conservation values (Johnson and Miyanishi, 2008; Tischew and Kirmer, 2007), substrate and geochemical properties (Riley and Rich, 1998; Suksi et al., 1996; Uren, 1992), and regional land resources mapping (Brennan, 2005; East et al., 1994; Hobbs and McIntyre, 2005). The need to integrate landscape ecological science into restoration practice to achieve success has been recognised (Bradshaw, 1997; Holl et al., 2003; Temperton, 2007). Species distribution modelling is a means of achieving this integration.

The topographic variables derived from DEMs (Chapter 4) are used as environmental predictors in almost every published example of predictive mapping of terrestrial vegetation (Austin, 2007; Guisan and Zimmermann, 2000). However, the reliability that can be achieved from species distribution modelling depends on the combined effect of the ecological model to be tested, a data model and a statistical modelling approach (Austin, 2002b). A brief introduction to these three aspects affecting reliable prediction of species distribution follows.

7.1.2 *Ecological modelling*

The data models and methods used for conservation reserve planning are focused predominantly on rare and endangered species and have a regional context (Austin, 2002b; Ferrier and Guisan, 2006; Lehmann et al., 2002b). However, the data models and methods used in mine landscape reconstruction can be applied to common and abundant species that provide the ecological framework of locally extensive ecosystems in natural analogue landscapes. Many studies have modelled the influence of landscape factors on the composition and structure of plant communities (Austin, 1987; Austin et al., 1984; Austin et al., 1983; Coops and Catling, 2002; Woinarski and Fisher, 2003; Woinarski et al., 2005) and confirm the notion that plant communities change continuously along environment gradients.

There is a strong argument for modelling individual species distributions first and then assembling the predictions to account for vegetation community dynamics (Guisan et al., 2006). The most robust approach is to predict the distribution of individual species rather than plant communities (Ferrier and Guisan, 2006; Guisan and Thuiller, 2005; Guisan and Zimmermann, 2000; Lehmann et al., 2003). Ecosystem dynamics are then represented in the autocorrelated component. Although, significant autocorrelation can also identify that the theoretical ecological framework needs to be reassessed (Austin, 2007).

The strength of modelling individual species distributions is that it does not assume community structure to be stable over time (Ferrier and Guisan, 2006). Although equilibrium between the environment and observed species patterns is still assumed, modelling individual species rather than species assemblages and classifying their superimposed distributions afterwards is more logical than fitting *a priori* community classifications (Ferrier and Guisan, 2006; Guisan and Thuiller, 2005).

7.1.3 *Statistical method*

Statistical models that account for non-linearity in species presence–absence responses to environment, such as generalised additive models, or GAMs (Hastie and Tibshirani, 1990), are the preferred method for species distribution modelling (Austin, 2007; Austin, 2002b; Guisan et al., 2002; Guisan et al., 2006; Guisan and Thuiller, 2005). Generalised additive models (GAMs) allow almost any response curve to be fitted. This is an important advantage since species rarely present pure bell-shaped or linear response curves along environmental gradients (Guisan and Harrell, 2000).

Parsimonious variable selection for data models is an important issue to minimise error and uncertainty (Barry and Elith, 2006; Fielding and Bell, 1997). Maggini (2006) tested a range of variable selection methods and found that a cross validation approach (Stone, 1977) provided the best compromise between model performance and parsimony or model stability. For presence–absence data, area under the curve (AUC) methods, considered to be the current best practice for assessing model success (Austin, 2007), have been used extensively in species distribution models (SDMs) (Elith et al., 2006).

Another issue is the problem of autocorrelation. Incorporation of the autocorrelation of model residuals into predictive models may minimise model uncertainty (Maggini et al., 2006; Malanson et al., 1992), although, optimal parameterisation and sampling can be obscured when autocorrelation is included (Austin, 2002b). Lehmann (2003) offers a solution to this approach using GAM implemented in S+Plus2000 in the GRASP module (Generalised Regression and Simulation Package). A wide range of GRASP applications have been published spanning different groups and ecosystems, including terrestrial vegetation species (Cawsey et al., 2002; Clarkson et al., 2004; Lehmann et al., 2002a) and mapping of plant communities (Ferrier and Guisan, 2006).

7.1.4 *Data model*

Statistical modelling is sensitive to the quality and range of the survey data (Barry and Elith, 2006; Legendre et al., 2002; Van Niel and Austin, 2007). Restricted environmental sampling reduces the combinations of environmental variables under which statistical models of environmental correlation are calibrated and this limits the extent of their application (Thuiller et al., 2004). However, using species presence–absence data for species distribution modelling rather than abundance data (Cushman and McGarigal, 2004; Guisan and Zimmermann, 2000) is cheaper, quicker and does not appear to degrade results (Margules and Austin, 1991). The number and range of observations is more critical to the reliability of species distribution models than using abundance data.

The survey design needs to resolve local environmental pattern (Austin and Heyligers, 1989; Edwards et al., 2006) that are often scale dependant. Grid survey designs have been applied to targeted natural landscapes to resolve observed hill slope patterns affecting the environmental range (Ben Wu and Archer, 2005). This approach does not presume that ecological complexity, which affects short range variation, is fully understood. Extensive ecosystems would be over–represented and ecosystems with limited extent would have fewer observations. Using grid spacings less than the natural hillslope lengths would account for the effects of hillslope environmental process (erosion, sedimentation and water distribution) that drive diversity.

One hundred sites appear to be a minimum to quantify species distribution (Austin, 2007; Maggini et al., 2006; Pierce et al., 1993; Stockwell and Peterson, 2002). Environmental complexity, particularly anisotropic spatial autocorrelated variation, may increase the minimum number of survey sites to 250 (Webster and Oliver, 2001). Also, species prevalence (frequency of species' occurrences in the data set) can shift prediction errors from omission (under prediction of true presences) to commission (over prediction) with the best compromise occurring at about 0.5 (Liu et al., 2005). Other authors have suggested the use of weighting absences to adjust sample prevalence towards this value to reduce modelling errors (Lehmann et al., 2003; Maggini et al., 2006). The number of sample sites needed to quantify species distribution depends on the abundance (prevalence) of the species concerned and environmental complexity.

Statistical models need to be calibrated and validated. Calibration and validation of models using independent data sets is ideal (Fielding and Bell, 1997; Guisan and Harrell, 2000) but it can be difficult to justify excluding data to calibrate the model. An alternative cross-validation, or bootstrapping, approach tends to give the upper boundary of error probabilities (Fielding and Bell, 1997; Maggini et al., 2006). Consequently, K-fold cross-validation (Franklin et al., 2000) and bootstrapping methods (Guisan and Harrell, 2000) offer a compromise that allow model stability to be assessed without losing information during calibration.

7.1.5 *Objectives*

The primary objective of this chapter was to develop statistical models for the distributions of framework species in the context of an open-cast mine restoration project and apply these models to predict revegetation outcomes in a mine landform design. Supplementary objectives were to: (i) assess the effect on measured species response of using presence-absence or abundance data and sample size; (ii) investigate aspects of the predictor variables including geographic range and the significance of autocorrelation. The work focuses on the Ranger uranium mine case study site.

7.2 Methods

The grid environmental survey was made of the 500 hectare *Georgetown* analogue area adjacent to the mine (Chapter 6) and provided the core dataset for the GAM modelling of species distribution presented in this chapter. This area contained the environmental range found on rock substrates recorded over a much larger area (79 km^2, see Chapter 4). The reconnaissance survey data that was collected from two analogue areas (Georgetown and 7J) in Chapter 4 and information from a regional natural analogue survey of well drained woodland habitats (Brennan, 2005) has been introduced to assess the effect of increasing survey support with regional information to measure trends in species response to environment. Initially, multivariate analytical methods were used to test the null hypothesis that species distribution was unrelated to environmental gradients for presence-absence and abundance date. Finally, univariate (GAM) models of framework woodland species are developed and applied to a landscape that incorporates the draft Ranger mine landform design that was presented in Chapter 6.

7.2.1 *Field survey data*

The field survey data included: (i) 101 survey sites located 200 hundred metres apart in a square grid survey over the *Georgetown* analogue landform (500 hectares) on the Ranger mine lease (Chapter 6); (ii) 32 sites in a environmentally stratified sampling design over the 79 km^2 Ranger lease area (79 km^2, Chapter 4); (iii) 28 detailed plot survey sites over the East Arnhem Land (10 000 km^2) from Brennan (2005). The locations of the sites on the Ranger lease area are shown in Figure 7-1. The quantitative vegetation survey data of species abundance and community structure at 18 plot survey sites were presented in Chapter 6 for the *Georgetown* analogue area. This data is now supplemented with 10 vegetation survey sites over the Alligator Rivers region comprising Kakadu National Park and East Arnhem Land and covering 10 000 km^2. The common and abundant species across the environmental range in the *Georgetown* analogue area listed in Table 7-1 were selected for species distribution modelling.

Table 7-1 Selected common and abundant species in the overstorey and midstorey across analogue areas

Strata	ID	Species
Overstorey (>8m)	1	Eucalyptus tetrodonta
	2	Eucalyptus miniata
	3	Corymbia bleeseri
	4	Corymbia foelscheana
	5	Corymbia latifolia
	6	Corymbia porrecta
	7	Corymbia setosa
	8	Eucalyptus tectifica
	9	Xanthostemon paradoxus
Midstorey (2-8m)	10	Acacia mimula
	11	Melaleuca viridiflora
	12	Erythrophleum chlorostachys
	13	Petalostigma pubescens
	14	Pandanus spiralis

These are *'framework'* species (identified in Chapter 6) and their establishment is a key concern in the revegetation strategy for Ranger Mine. Although the revegetation strategy excluded acacias, *Acacia mimula* is included in the analysis because it is ubiquitous in the midstorey layer and provides habitat for small mammals (Woinarski et al., 1999a; Woinarski et al., 1999b). Also, although *Melaleuca viridiflora* was identified as a midstorey species in Chapter 6, it is the dominant overstorey species in Melaleuca woodland communities.

7.2.2 *Terrain analysis*

The factors and environmental correlates that influence environmental variation across the study area are summarised in Table 7-2. Landscape or terrain variables derived in Chapter 4 from a digital elevation model (DEM) were used as predictors; all other variables were treated as covariates.

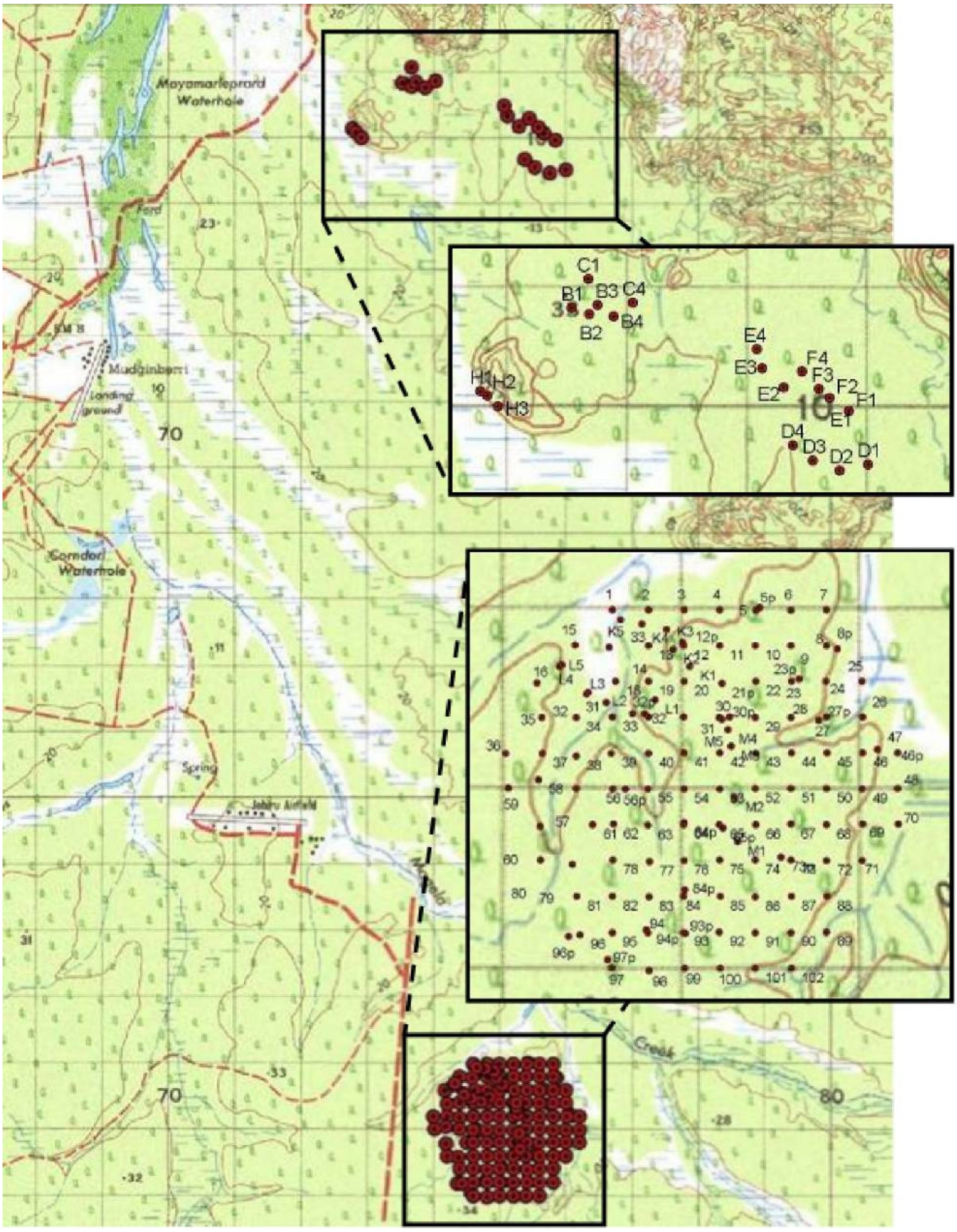


Figure 7-1 Sites in analogue areas on the Ranger lease surveyed for species presence-absence

Table 7-2 Environmental factors in the vegetation model

Variable influencing species distribution	**Significance and interpretation**	**Potential environmental correlates**
climate		
temperature	nature and rate of biological activity	climate surfaces
wetness	nature and rate of biological activity	climate surfaces and terrain attributes
parent material		
substrate	influences nutrient levels and drainage	geophysical remote sensing
erosion and deposition	influences water and nutrient supply	terrain attributes and geophysical remote sensing
time		
age of land surface	older relief may not reflect contemporary environmental conditions	empirical correlations with terrain attributes
biomechanical impact		
termites, ants	local scale disturbance (termites, ants, etc.) may control nutrient and water supply	no effective predictors
disturbance		
fire	fire frequency and intensity affect biodiversity	fire frequency mapping late in the dry season when intensive fire impacts occur
endogenous vectors		
	soil morphology affects local conditions	inherited soil profile features may impose limitations
	realised plant niche differs from idealised distribution	model species individually

The topographic variables used to predict species distribution are listed in Table 7-3. The environmental covariates that were used are listed in Table 7-4. Continuous variation in the mineral properties of parent materials was assessed using airborne gamma radiometric mapping from NT Geological Survey. The radiometric data show a strong U signal associated with outcropping schists particularly in the Arnhem Plateau and Pine Creek regions with clear anomalies at Ranger and the northern extent of the Arnhem Plateau region. The thorium signal is associated with exposed petroferric horizons and strong K signal associated with alluvial clays (Wilford, 2006) that are extensive in floodplains of the Darwin Coastal region.

The distribution of late dry season fires was also used as covariate in the statistical analysis and was mapped from monthly fire scar mapping (NOAH AVHR imagery). The resolution of the imagery (1 km^2) is coarse relative to the DEM (50-m).

Table 7-3 Topographic variables used for environmental prediction

ID	Environmental variable	Description	Source
Primary topographic attributes[1]			
1	elevation	elevation above sea level	DEM
2	slope	measured in percent	TAPESG
3	relief	absolute range in elevation within 600 m circle	TAPESG
4	aspect	N, NE, E, SE, S, SW, W, NW, level (zero slope)	TAPESG
5	plan curvature	rate of change of aspect, generated using a 3x3 window	TAPESG
6	profile curvature	rate of change of slope generated using a 3x3 window	TAPESG
Secondary topographic attributes[1]			
7	wetness index	static wetness index describes the spatial distribution and extent of zones of saturation	TAPESG
8	slope length index (LS)	sediment transport capacity index equivalent to the length-slope factor in the Revised Universal Soil Loss Equation	TAPESG
9	stream power index	measure of erosive power that predicts net erosion in convex areas and net deposition in concave areas	TAPESG
10	upslope flow path length	maximum distance of flow to a point	TAPESG
11	contributing area	upslope area above a point	TAPESG
12	dispersal area	dispersal area below a point	TAPESG

[1](Wilson and Gallant, 2000b).

7.2.3 *Multivariate analysis*

Multivariate hybrid gradient analysis methods detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) methods in the CANOCO VERSION 4.2 statistical package (ter Braak and Smilauer, 2002) were used to identify environmental trends in species abundance and presence-absence data. The multivariate analysis used a forward selection procedure and a Monte Carlo permutation to test variable significance. Two Monte Carlo permutation tests were performed to measure statistical significance using an F-test (0.05 probability level) of the first and combined canonical ordination axes (ter Braak and Smilauer, 2002). This tested the null hypothesis that species response was unrelated to environmental gradients.

Table 7-4 Environmental covariates

ID	Environmental variable	Description	Source or reference
Parent material			
13	Gamma-K	K band response from gamma radiometric mapping	NT Geological Survey :Kakadu II 1982, Alligator River 1976, Koolpinyah 1974 radiometric grids (150 m line spacing, www.minerals.nt.gov.au/ntgs)
14	Gamma-Th	Th band response from gamma radiometric mapping	As above
15	Gamma-U	U band response from gamma radiometric mapping	As above
Climate			
16	Prescott Index	Ratio of annual rainfall to annual evapotranspiration	Australian climate surfaces (www.bom.gov.au)
17	Fire frequency	Years in 10 frequency of late dry season fires	NT Bushfires Council fire scar mapping from monthly NOAA imagery
Biodiversity			
18	Bioregionalisation	Sub-region classification	National Bioregionalisation mapping (Thackway and Cresswell, 1995)
Soil morphology			
19	A horizon thickness class		Field survey
20	A horizon gravel class		Field survey
21	A horizon texture class		Field survey
22	B horizon texture class		Field survey
23	Depth to rock		Field survey
24	Drainage class		Field survey

7.2.4 *Modelling experiments*

The univariate methods were used to conduct a number of modelling experiments to test the effects of parameterisation and data model and to develop species distribution models (SDMs) for framework species. The univariate GAM models were calibrated using the GRASP Version.3.2 package (Lehmann et al., 2003) for S-PLUS Version 2000 (Insightful Corp., Seattle, WA, USA). The response variables were the common and abundant woodland species defined as binary variables, and N2 species richness (number of species) defined as a continuous variable with a Gaussian distribution. In using GAM for species presence-absence data, a quasi-binomial model was chosen; whereas for N2 species richness, a quasi-Poisson model was most appropriate (Lehmann et al., 2002b). In both cases, a stepwise 'cross-selection' method also termed as CROSS, was used to select the predictors. With CROSS a CVROC (cross–validated receiving operator characteristic) was calculated at each step of the selection procedure. The receiver operating characteristic (ROC) was used to test goodness of fit for the binomial model (Fielding and Bell, 1997). The selection stops when no more predictors can be added or removed according to a Bayesian information criterion (BIC), the most selective criterion available. CROSS then selects the model showing the highest value of CVROC.

Pearce et al. (2001) recommended using 2 degrees of freedom (df) in the GAM function for presence–absence vegetation survey data. However, using the default setting in GRASP (4 degrees of freedom) produced a more valid model (ROC 0.848; CROC 0.684). This result was with the default cross–validation selection direction, 'backwards'. Changing the search direction to 'both' caused model instability (ROC 0.93; CROC 0.679). The modelling experiments carried out on the analogue data are listed in Table 7-5.

Modelling alternatives and the effect of varying the data model to increase support (number of sites) were tested. The first experiment (E0) involved testing the differences in environmental correlation between presence-absence data (143 sites) and abundance data (28 sites) with the aim of identifying trends and modelling limitations with these data sets. The set of 12 explanatory environmental variables and their relationships with the presence-–absence of 60 species recorded at 143 sites were analysed based on a unimodal model in CCA. Analysis for the effect of covariates on gradient lengths was done before and after the inclusion of bioregional, climatic, parent material, fire frequency and soil morphological variation. All subsequent experiments (E1-E7) calibrated univariate GAM models.

The second group of experiments (E1-E4) tested the effect of weighting absences and removing absences outside the community range along the key environmental gradients, recommended practices in pre-processing presence-absence data in regional biodiversity modelling (Austin and Nicholls, 1997; Leathwick, 1998). E1 baseline models were developed using presence-absence data for 101 sites in the Georgetown analogue area with no weighting of absences or masking the environmental range to remove strings of absences.

Table 7-5 Modelling experiments conducted on analogue site data

Experiments	Description
Environmental correlation	
E0	Multivariate environmental model (presence-absence response) ~ [topography], [climate, geology, fire, soils]
	Multivariate environmental model (abundance response) ~ [topography], [climate, geology, fire, soils]
Univariate models	
E1 (baseline)	Univariate environmental models ~ topography; Georgetown sites
Weights (compared with E1)	
E2	Univariate environmental models ~ topography; Georgetown sites; weighted absences
Support size (compared with E2)	
E3	Univariate environmental models ~ topography; with weighted absences; 101 Georgetown grid sites + 32 Ranger Project Area sites = 133 sites
E4	Univariate environmental models ~ topography; with weighted absences; 101 Georgetown sites + 32 Ranger Project Area sites + 10 regional analogue sites = 143 sites
Spatial autocorrelation (compared with E2)	
E5	Univariate environmental models ~ topography + trend (tested); with weighted absences; 101 Georgetown sites + 32 Ranger Project Area sites + 10 regional analogue sites = 143 sites
E6	Univariate environmental models ~ topography + εenv ; with weighted absences; 101 Georgetown sites Spatial model: εenv ~ IK*
E7	Univariate environmental models ~ topography + IK* (tested); with weighted absences; 101 Georgetown sites

Experiment E2 tested the effect of weighting absences and masking environmental range. An overall prevalence of 0.5 was obtained by giving the presences a weight of 1, and the absences a weight defined by the ratio of the number of presences on the number of absences. A model was then fitted using these weights. Results were compared with E1 baseline models. Species absences at environmental extremities were masked to create environmental envelopes. Experiment E3 augmented the survey support to one hundred and thirty three sites with thirty two sites in the Ranger project area (Chapter 4). Experiment E4 further augmented the survey design with 10 sites remote from the Ranger mine from a regional analogue survey (Brennan, 2005).

The third group of experiments (E5-E7) tested the effect of spatial autocorrelation. In experiment E5, first-order spatial autocorrelation was incorporated into the models by adding a trend predictor. A first order trend surface was fitted using all 143 sites and added to the GAM model. Experiment E6 investigated second-order spatial models based on the 101 sites in the Georgetown analogue area. GAM analysis was performed and indicator kriged (IK) estimates of local auto correlation were calculated. The IK term of spatial autocorrelation was added to the list of predictors, which was tested using cross-validation selection criteria in GRASP. Experiment E7 investigated the local autocorrelation using an autoregression term (Maggini et al., 2006). This term was regressed against the residuals of the E1 model to account for the residual spatial autocorrelation still present after environmental correlation. The results were compared with the E1 baseline models using the Wilcoxon signed-rank test, a distribution-free test of pairwise difference measure. The final experiment (E7) tested the effect of extending the Georgetown analogue data set with the reconnaissance survey data of the Ranger project area (Chapter 4) making 133 sites.

7.2.5 *Validation*

Observed response values were plotted against predicted, by correlation (COR) and also by fivefold cross-validation of the model, to evaluate GAM models. The area under the ROC function (AUC) provides a single measure of overall accuracy independent of a particular threshold. An AUC value of 0.5 indicates the scores for two groups do not differ, while a score of 1.0 indicates no overlap in the distributions of the group scores. A value of 0.8 for the AUC means that for 80% of the time a random selection from the positive group would have a score greater than a random selection from the negative class. Cross-validation performance (CVROC) on randomly selected sub-samples of the data indicates instability in the model. The goodness of fit of the Poisson model of N2 species richness was assessed from correlation between actual and predicted values.

The effect of varying modelling options in GRASP on model performance was tested for *Melaleuca viridiflora* to optimise the setup before running the modelling experiments. *Melaleuca viridiflora* had a relatively low prevalence in the survey, 0.3 compared with Eucalyptus tetrodonta 0.69, while its dominance is definitive for a woodland community type (Melaleuca woodland). Wilcoxon signed-rank tests were used to compare validation statistics between pairs of modelling experiments. This test, which does not assume any particular distribution, can be applied as a one-way test of competing results.

Spatial predictions were built in S-PLUS using optimised GAM models for the 20 km^2 (50-m grid resolution) that covered the Ranger mine, including the final landform surrounded by undulating rises. The grid of prediction variables was generated in ArcGIS version. 9.2. The predicted species distributions were exported back to ArcGIS for display.

7.3 Results

7.3.1 *Multivariate environmental response*

Gradients longer than 4.0 in Table 7-6 for the first and second ordination axes indicated environmental trends for which unimodal methods (such as DCA and CCA) are likely to be appropriate (ter Braak, 1988). Transforming and detrending the species data did not alter or improve the analytical results. The first two extracted CCA axes provided the best species–environment correlations and interpretations and the species-environment correlation decreased only very slightly when covariates (soils, fire frequency, climate, parent material) were included in the analysis (partial CCA in Table 7-6). Apparently, covariates (non-terrain variables) were not significant factors in explaining species-environment relationships.

The unimodal models adequately described the environmental gradients for *Corymbia foelscheana* (CORYFOEL), *Eucalyptus tectifica* (EUCATECT) and *Corymbia latifolia* (CORYLATI) (Figure 7-2). The open ended, more linear models for *Eucalyptus tetrodonta* (EUCATETR), *Eucalyptus miniata* (EUCAMINI), *Corymbia bleeseri* (CORYBLEE), *Corymbia setosa* (CORYSETO), *Xanthostemon paradoxus* (XANTPARA) and *Corymbia porrecta* (CORYPORR) indicated that their environmental ranges were only partially sampled in this data set.

The combined canonical axes for CCA and partial CCA (Table 7-7) were significant (P = 0.05) meaning that the null hypothesis, based on the premise that species variation was unrelated to environmental gradients, was dismissed for the 148 samples of species presence–absence. However, the null hypothesis was accepted for the smaller species abundance sample (28 samples) based on the P-value shown in Table 7-7, even though the environmental trends in relation to the first and second ordination axes were investigated further.

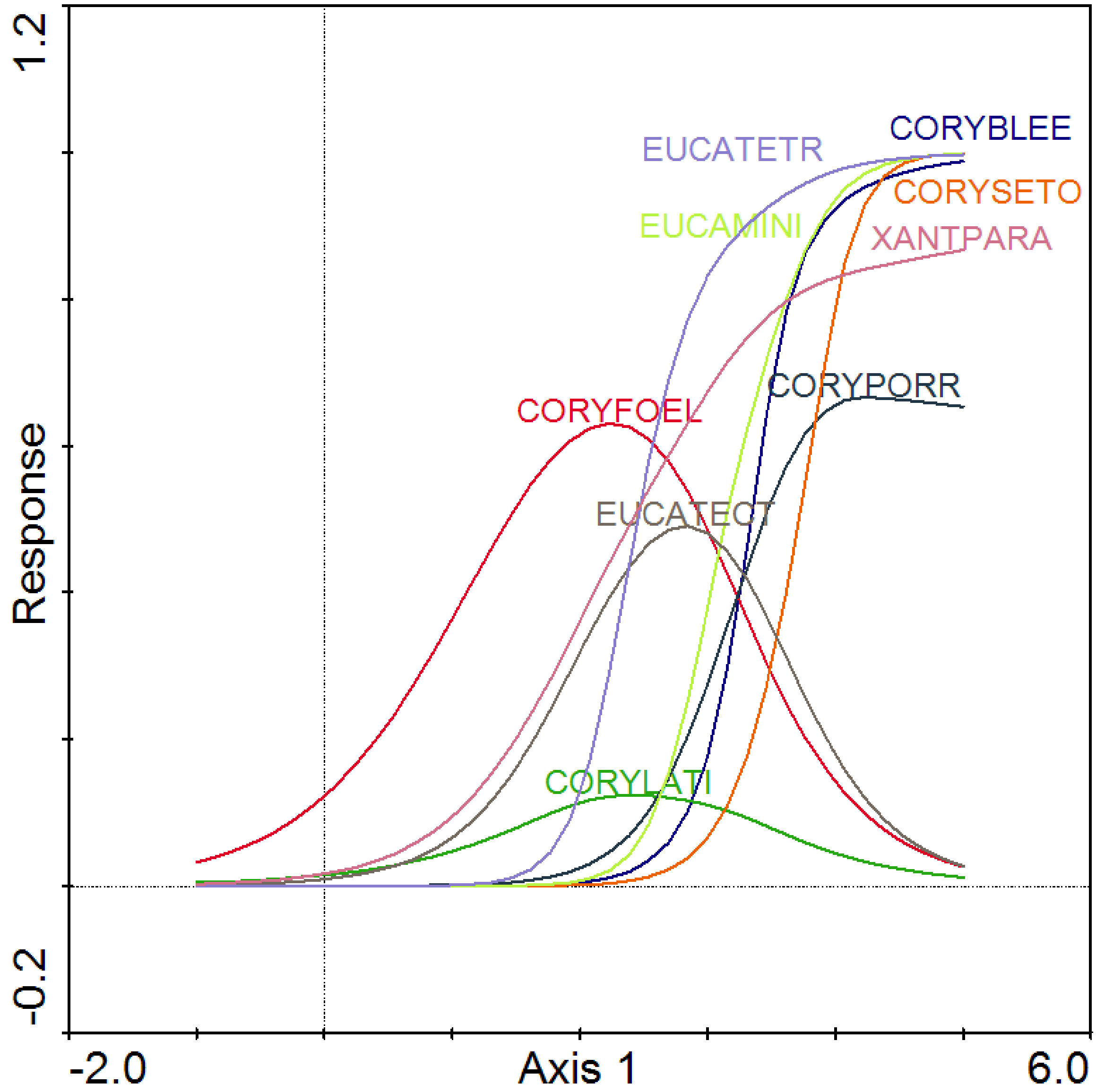


Figure 7-2 Species response (GAM) to 1st Ordination axis

The next objective was to identify significant environmental predictors of species distributions. Five environmental variables out of 12 were significant predictors ($P<0.05$, Table 7-8) for species presence-absence.

Table 7-6 Summary of multivariate analysis

Axes	**1**	**2**	**3**	**4**	**Total inertia**
Indirect Gradient Analysis: DCA (744-031.con)					
Eigenvalues	0.419	0.154	0.117	0.103	2.785
Lengths of gradient:	4.457	4.709	2.378	2.276	
Cumulative percentage variance of:					
species data	15.0	20.6	24.8	28.5	
Species-environment relation	22.7	31.9	36.8	40.9	
Sum of all eigenvalues					2.785
Direct Gradient Analysis: CCA (744-032.con)					
Eigenvalues	0.106	0.056	0.046	0.032	2.785
Species-environment correlations	0.539	0.668	0.649	0.648	
Cumulative percentage variance:					
species data	3.8	5.8	7.4	8.6	
Species-environment relation	29.9	45.6	58.5	67.6	
Sum of all eigenvalues					2.785
Sum of all canonical eigenvalues					0.354
Direct Gradient Analysis: Partial CCA (- covariates, 744-033.con)					
Eigenvalues	0.070	0.049	0.036	0.029	2.785
Species-environment correlations	0.501	0.543	0.680	0.669	
Cumulative percentage variance:					
species data	3.0	5.0	6.5	7.7	
Species-environment relation	25.4	43.2	56.2	66.6	
Sum of all eigenvalues					2.383
Sum of all canonical eigenvalues					0.277

Table 7-7 Summary of Monte Carlo test of CCA ordination

	Eigenvalue	**F-ratio**	**P-value**
Species presence-absence (143 samples) CCA (744-032.con)			
Test of significance of first canonical axis:	0.106	5.136	0.0040(**)
Test of significance of all canonical axes:	0.354	1.576	0.0020(***)
CCA + covariates (744-033.con)			
Test of significance of first canonical axis:	0.070	3.648	0.0400(*)
Test of significance of all canonical axes:	0.277	1.315	0.0080(**)

NS – not significant; ** - highly significant

Environmental trend results were different and less reliable in CCA for species abundance data (Table 7-9). Consequently, the regression analysis which follows was based on species presence-absence.

Table 7-8 Conditional Effects on species presence-absence in CCA

ID	Variable	λA1	P	F
2.	Slope	0.04	0.010 (*)	1.99
12.	Dispersal Area	0.03	0.008 (**)	1.99
1.	Elevation	0.03	0.040 (*)	1.66
5.	Plan curvature	0.03	0.022 (*)	1.76
11.	Contributing Catchment Area	0.02	0.148	1.26
4.	Aspect	0.02	0.250	1.15
10.	Upslope Flow Path Length	0.02	0.356	1.04
7.	Wetness Index	0.02	0.718	0.77
6.	Profile Curvature	0.03	0.036 (*)	1.77
8.	LS factor	0.01	0.502	1.00
9.	Stream Power	0.02	0.822	0.74
3.	Relief	0.01	0.968	0.58

[1] λA1, Eigenvalue

Table 7-9 Conditional effects on species abundance in CCA

ID	Variable	λA1	P	F
3	Relief	0.34	0.014 (*)	2.36
9	Stream Power	0.24	0.048 (*)	1.85
11	Contributing Catchment Area	0.23	0.050 (*)	1.73
6	Profile Curvature	0.14	0.366	1.11
4	Aspect	0.18	0.144	1.44
8	LS factor	0.13	0.350	1.11
2	Slope	0.13	0.420	1.03
7	Wetness Index	0.13	0.348	1.12
12	Dispersal Area	0.11	0.524	0.87
1	Elevation	0.09	0.526	0.75
10	Upslope Flow Path Length	0.09	0.618	0.65
5	Plan curvature	0.07	0.768	0.46

7.3.2 *Univariate data models*

On the whole, the environmental data were consistent with observed species responses. For example responses depicted by histograms in Figure 7-3 varied predictably between *Eucalyptus tetrodonta* and *Melaleuca viridiflora*. *Eucalyptus tetrodonta* preferred progressively wetter sites, tending towards lower elevations and lower slopes. *Melaleuca viridiflora* preferred wet areas at low elevations.

Upslope flow path length was correlated with contributing catchment area ($r=0.87$, $P < 0.05$) and stream power ($r=0.72$, $P < 0.05$). Consequently, to achieve parsimony in variable selection and to reduce collinearity, upslope flow path length was used and contributing catchment area and stream power were excluded from subsequent modelling.

The box plots in Figure 7-4 depict the performance parameters of GAM models for modelling experiments E1-E4 (Table 7-5). A horizontal line indicates the median on boxes representing the inner quartile range. Whiskers depict the range, excluding outliers. Visual examination of box plots did not reveal a consistent improvement in validation (ROC) or reliability (cross validation, CVROC) with increasing survey support (E1-E4).

The statistics in Figure 7-4a denote model performance across all of the species listed in Table 7-1. The baseline model (E1) appeared to perform as well as, or better than, the extended data models that were tested. Weighting species absence and masking environmental ranges (E2) reduced the inner quartile range but did not increase the median ROC. Adding reconnaissance survey data (E3) did not appear to alter the test parameters over the baseline models (E1). Adding regional survey data (E4) from Brennan (2005) reduced model stability, as measured by the range in CVROC. E2 and E4 models were also more complex as indicated by higher degrees of freedom (DFR).

Boxplots depicting the performance parameters for modelling experiments testing spatial autocorrelation (E5-E7) and the base case (E1) in Figure 7-4b shows that introducing a regional trend predictor (E5) increased DFR. By introducing local autocorrelation, as autocorrelated residuals of the GAM model (E6) and as a kriged variable (E7), appeared to have improved the ROC, explained deviance (D2) statistics and reduced the DFR.

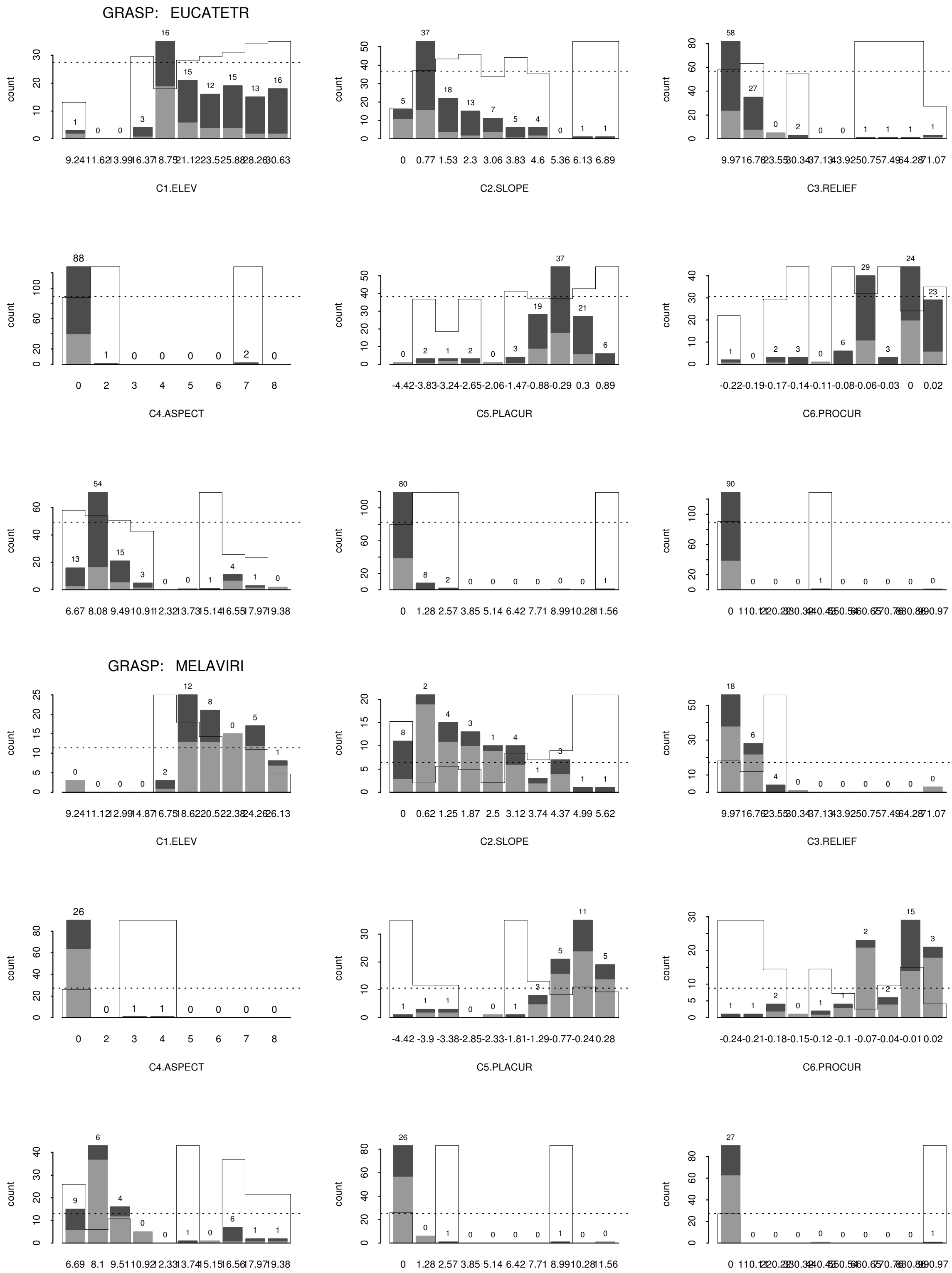


Figure 7-3 Histogram showing environmental
response for *Eucalyptus tetrodonta* and *Melaleuca viridiflora*

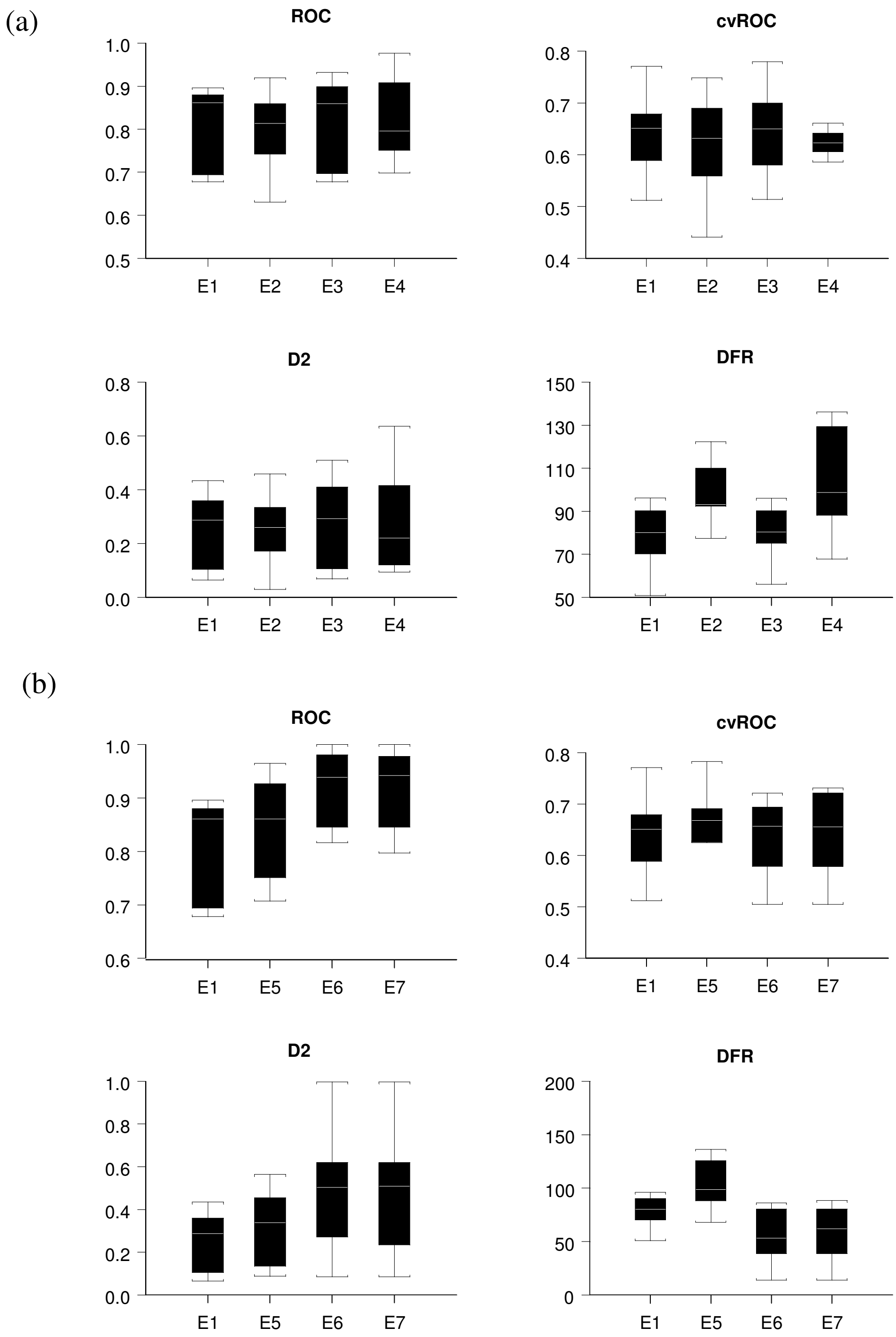


Figure 7-4 Receiver operating characteristic (ROC); cross-validated ROC (CVROC); explained deviance (D2) and residual degrees of freedom (RDF) for (a) models in experiments E1; E3-E4; (b) E1; E5-E7

Detailed analysis of the Wilcoxon signed-rank test results (Table 7-10) confirmed the assessment of baseline model performance. A difference in CVROC was indicative of differences in model stability and predictability and differences in DFR signified variation in model size and complexity. Values for the area under the receiving operator characteristic (AUC) of greater than 80% were indicative of stable and reliable models (Section 7.2.5).

CVROC for the baseline model E1, with no weighting of absences, masking of environmental range, extended survey support or autocorrelated variables, was the most reliable or stable. Extended survey support (E2, E4) increased the DFR statistic by increasing the number of samples used to fit the model rather than fitting simpler models.

According to the validation statistics for selected SDMs (Table 7-11), the models obtained for *E. tetrodonta, C. bleeseri, C. foelscheana, A. mimula, M. viridiflora* and *P. spiralis* showed good discrimination and stability according to the interpretation of combined ROC and CVROC values (0.5-0.7, poor discrimination ability; 0.7-0.9 , reasonable discrimination; 0.9-1, very good discrimination) given by Swets (1988).

Table 7-10 Significance of the Wilcoxon signed-rank test (n.s., >0.1; *, <0.1; **, <0.05; ***, <0.01) based on receiver operating characteristic (ROC), cross-validated ROC (CVROC), explained deviance (D2) and residual degrees of freedom (DFR)

	ROC			**CVROC**			**D2**			**DFR**		
	E2	**E3**	**E4**	**E2**	**E3**	**E4**	**E2**	**E3**	**E4**	**E2**	**E3**	**E4**
E1	n.s.	n.s.	n.s	n.s.	n.s.	n.s	n.s.	n.s.	n.s	***	n.s	**
E2	-	n.s.	n.s.	-	n.s.	n.s.	-	n.s.	n.s.	-	n.s	n.s.
E3	-	-	n.s.	-	-	n.s.		-	n.s.	-	-	***

	E5	**E6**	**E7**	**E5**	**E6**	**E7**	**E5**	**E6**	**E7**	**E5**	**E6**	**E7**
E1	n.s.	***	***	n.s.	n.s.	n.s.	*	***	***	***	n.s.	n.s.
E5	-	***	***	-	n.s.	n.s.	-	**	*	-	n.s.	n.s.
E6	-	-	n.s.	-	-	n.s.	-	-	n.s.	-	-	n.s.

Species with marginally acceptable levels of model stability (CVROC) included *E. miniata*, *E. tectifica* and *X. paradoxa*. The CVROC values of the models for the remaining species (*C. latifolia, C. setosa, E. chlorostachys*, and *P. pubescens*) indicated instability.

Species with prevalence values above about 0.4 were modelled successfully. Exceptions were *M. viridiflora* and *P. spiralis*, which had relatively predictable distributions in spite of being under–represented in the survey design (low prevalence). The species richness (N2) was not significant, although species richness was negatively correlated with wetness index (P<0.001, Table 7-11).

Table 7-11 Validation statistics for selected species distribution models

Statistic Response	ROC	CVROC	D2	DFR	Prevalence	Validation	Model
N2	0.00	0.00	0.46	69.00		poor	E1
EUCATETR	0.97	0.72	0.66	61.92	0.69	good	E1
EUCAMINI	0.94	0.69	0.09	96.20	0.45	good	E7
CORYBLEE	0.94	0.74	0.51	66.04	0.42	good	E7
CORYFOEL	0.85	0.72	0.24	85.37	0.49	good	E7
CORYLATI	1.00	0.67	0.85	62.53	0.06	poor	E1
CORYPORR	0.95	0.64	0.52	58.07	0.31	poor	E1
CORYSETO	0.75	0.59	0.12	92.06	0.31	poor	E1
EUCATECT	0.76	0.69	0.12	92.08	0.32	poor	E1
XANTPARA	0.96	0.69	0.57	61.74	0.6	poor	E1
ACACMIMU	0.96	0.73	0.58	66.09	0.5	good	E1
MELAVIRI	0.92	0.71	0.43	85.28	0.22	good	E1
ERYTCHLO	0.69	0.56	0.10	92.07	0.32	poor	E1
PETAPUBE	1.00	0.61	0.97	63.46	0.06	poor	E1
PANDSPIR	0.86	0.69	0.33	88.31	0.22	good	E1

7.3.3 *Species models*

The analysis of variance statistics for each SDM selected on CVROC and ROC statistics is shown in Table 7-12. The baseline data model (E1) provided most of the good or successful statistical models as measured by AUC values of 80% or more, apart from those for *E. miniata, C. porrecta* and *E. tectifica.* The SDMs for these three species were based on the *Georgetown* sites and included a significant autocorrelated (IK) variable in the E7 data model, While there was a significant regional trend in the SDMs for *C.*

foelscheana, C. setosa, C. porrecta and E. tectifica (in the E5 data model) they were better predicted using the baseline data model (E1). The distribution of *E. tectifica* was unusual in having a positive correlation with slope.

Table 7-12 Environmental predictors included in selected SDMs (Y) indicating the significance of sequential removal from the model based on a chi-squared test statistic (n.s., >0.1; *, <0.1; **, <0.05; ***, <0.01)

Predictor / **Response**	**C1.elev**	**C2.slope**	**C3.relief**	**C4.aspect**	**C5.plancur**	**C6.procur**	**C7.wi**	**C8.ls**	**C9.sp**	**C10.usfpl**	**C11.uca**	**C12.uda**	**trend**	ε_{ENV}	**IK**	**Model**
N2	Y n.s	Y ***	Y ***	Y ***	Y ***	Y ***	Y ***	Y **								E1
EUCATETR	Y ***	Y n.s.	Y *	Y ***	Y n.s.	Y n.s.	Y n.s.	Y **	Y ***	Y n.s.						E1
EUCAMINI	Y **							Y n.s.							Y*	E7
CORYBLEE	Y *		Y **			Y *			Y ***							E1
CORYFOEL	Y *	Y n.s.	Y **	Y *									Y ***			E1
CORYLATI	Y n.s	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y n.s.						E1
CORYPORR			Y *		Y *								Y **		Y *	E7
CORYSETO	Y n.s	Y n.s.											Y **			E1
EUCATECT	Y n.s	Y **											Y **		Y *	E7
XANTPARA	Y n.s	Y n.s.	Y ***	Y n.s.	Y n.s.	Y *	Y n.s.	Y n.s.	Y n.s.	Y **						E1
ACACMIMU	Y ***	Y n.s.	Y *	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y n.s.	Y *							E1
MELAVIRI	Y ***	Y *	Y *	Y *												E1
ERYTCHLO	Y *	Y *														E1
PETAPUBE	Y *	Y *	Y *	Y ***												E1
PANDSPIR	Y **	Y **	Y **													E1

The primary topographic attributes of elevation, slope, aspect and relief were important predictors of species distribution, based on the number of models incorporating them in Table 7-12). The secondary topographic attributes of profile curvature, slope length factor, plan curvature, stream power and wetness index were significant contributors to relatively fewer SDMs. Wetness index (C7.wi in Table 7-12) was chosen in five models, although its inclusion in the N2 species richness model was the only instance that registered a significant contribution as indicated by the analysis of variance.

The predicted distributions of four woodland species (*E. tetrodonta*, *M. viridiflora*, *C. foelscheana* and *E. tectifica)* in and around Ranger mine (using the baseline data model, E1) were mapped and overlaid (Figure 7-5). Apart from *E. tectifica* the distributions depicted in Figure 7-5 are for relatively common species that are represented across a range of habitat types in the vicinity of Ranger mine (Schodde et al., 1987).

7.4 Discussion

7.4.1 *Key predictors of species distribution*

The significance of species distribution correlated to the local terrain attributes was tested. Climate, soil morphology, parent materials and fire frequency did not significantly account for significant variation in the distribution of framework species. This supported the assumed stationarity in the measured environmental response, that the spatial component of variance was not significant for the statistical modelling. This is due in part to the influence of landform on soil properties, but of course the general adaptation of woodlands in this region to fire has been cited as a significant factor (Williams et al., 2005). The predictability of species distributions depended on sample size and species prevalence. Species presence-absence data were adequate for species distribution modelling.

7.4.2 *What model?*

Quantifying vegetation responses to environmental variation integrates landscape design with revegetation planning and supports reasonable revegetation objectives. Otherwise, objectives can be based on un-testable hypotheses of what constitutes a similar natural environment and unreasonable or impractical goals for landscape restoration. While 100 sites, selected based on environmentally stratified survey design, is the minimum recommended for species distribution modelling as was done in this study and others (Stockwell and Peterson, 2002), it is not as demanding as it might seem if species presence-absence data, rather than species abundance data are acquired (Margules and Austin, 1991).

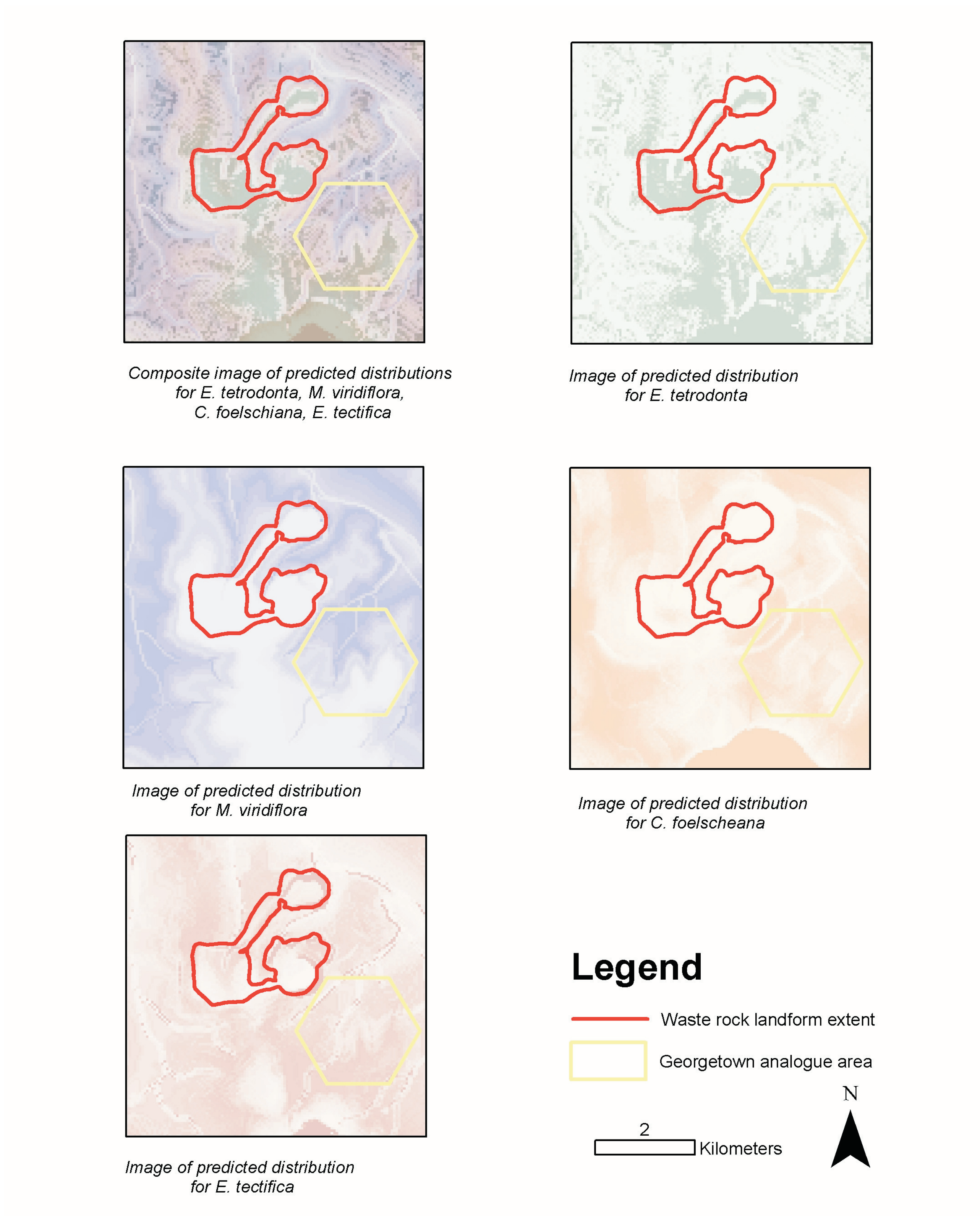


Figure 7-5 Predicted species distributions for *Eucalyptus tetrodonta, Melaleuca viridiflora, Corymbia foelscheana, Eucalyptus tectifica*

Measuring species abundance and cover from detailed plot surveys would require additional effort and expense (Austin and Heyligers, 1989), which is unnecessary unless revegetation dynamics in responses to ecological processes such as disturbance and dispersal need to be predicted.

In modelling the species-environment interrelationships, weighting species absences to increase prevalence and masking species absences beyond the environmental envelope, which are recommended methods to remove model bias and increase precision (Austin, 2002b; Lehmann, 2005; Maggini et al., 2006), did not improve model performance or stability. Weighting absences dampened the species response and removed information from a small data set. Therefore, the data models and statistical models developed in conservation ecology needed to be applied slightly differently in a mined land restoration where environmental gradients are better defined.

No significant model was attained when using species with prevalence values less than about 0.4. This corresponds to 40 positive observations of species occurrence in a survey of 100 sites. Species that were an exception (*M. viridiflora* and *P. spiralis*) were very clearly defined by topography. More effort is needed to predict the distribution of less common species and arguments for extending survey support to predict them will be based on either ecological or cultural values.

Extending data models geographically away from a mine site may not significantly improve overall performance or add information if regional geographic factors come into play. Also, as reported elsewhere (Austin, 2002b; Bui and Moran, 2001), unstructured observational survey data introduces bias and model instability. Essentially, sites remote from the mine tend to be less analogous and selecting analogue sites based on geographic range rather than context in the mine landscape causes analytical issues that are difficult to resolve.

7.4.3 *Species prediction*

In contrast to the case of prevalence values of less than 0.4, significant species distribution models (SDMs) were developed for common species (with prevalence values greater than 0.4). While this was expected (Maggini et al., 2006) there were exceptions for species strongly controlled by landform properties (Melaleuca viridiflora prevalence and Pandanus spiralis prevalence 0.22). Elevation, slope, relief and aspect had the broadest predictive value for the species being modelled. These attributes define the energy environment and set the broad context for landscape environmental variation (Allen et al., 2003; Ehrenfeld and Toth, 1997; Odum and Odum, 2003). Other studies have shown that plant communities vary with landform type in Kakadu National Park (Russell-Smith, 1995) and elsewhere in Australia (Austin and Meyers, 1996; Ryan et al., 2000), and overseas (Mackey et al., 2000; Maggini et al., 2006). However, this study quantified topographic control of ecosystem variation in a way that is applied to designing a mine landform. The primary landform attributes that were selected extend topographic reconstruction (Toy and Chuse, 2005) to support particular ecological outcomes in the mine landscape.

Slope curvatures and slope lengths affected the distributions of particular species, while increasing site wetness (poorer drainage) significantly reduced species richness. This finding is corroborated elsewhere in northern Australia for savanna woodlands (Bowman and Minchin, 1987). Building natural environmental function into the landform may require incorporating plan curvature to reinstate more subtle vegetation patterns associated with water distribution in hill slopes. Significant correlation of biodiversity with water distribution in peneplanated landscapes was the case here as elsewhere in Australia (Wardell-Johnson and Horwitz, 1996) but there is a lack of predictive studies (Dirnbock et al., 2002). Therefore a capacity to manage biodiversity in reconstructed landscapes was not previously developed.

Significant regional trends (experiment E5) in the distributions of *C. foelscheana, C. setosa, C. porrecta* and *E. Tectifica were modelled*. However, the models for these species were unreliable (Table 7-11) and a larger survey support is warranted. In the case of the data model on the *Georgetown* analogue area with (E7) an autocorrelation component produced the best validation statistics for these species. These findings are similar to those described elsewhere (Russell-Smith, 1995; Williams et al., 1996), although appropriate sampling design may be required to resolve these influences convincingly.

7.4.4 *Landform design and revegetation planning*

The water balance properties of the surface 3 to 5 metres of the mine landform were not included from the species distribution modelling. Creating a cover consisting of a well graded porous medium (Johnson, 2002) with similar water holding and erosion resistance to natural soils will lead to analogous conditions, albeit more freely drained. The water retention and hydraulic conductivity of this zone were linked to vegetation density and stream flow studies reported in Chapter 6. All these parameters are important for consideration in landform design.

Conditions associated with soil formation (such as hard pans that restrict root growth and free drainage) affect vegetation and catchment hydrology in the natural environment (Vardavas, 1993) but may not be present in the mine landform. However, the investigation of the hydraulic properties and water balance of a constructed cover that incorporated a drainage limiting layer that was reported in Chapter 5 indicated that low conductivity horizons could be constructed from a range of waste rock materials. Constructing a low conductivity subsoil layer at 2 to 3 metres in the mine landscape may be analogous to natural conditions where throughflow is a significant contributor to stream flow (see Chapter 6).

Land management factors apart from landform, such as wild fire intensity and frequency, have been the focus of may comparable studies of ecosystem restoration in degraded landscapes (Brennan et al., 2003; Chaffey and Grant, 2000; Fisher et al., 2003; Koch, 2007; Ludwig et al., 2004). Studies have shown that late fires alter the structure and diversity of mid storey and ground cover components of native ecosystems (Andersen et al., 2005; Fisher et al., 2003; Gill et al., 2003; Ross et al., 2004; Yates and Russell-Smith, 2003). Mature savannas are highly adapted to fire (Williams et al., 2003b) and by studying the common and abundant overstorey and midstorey species in prevailing fire environments, ecosystems in static equilibrium with the prevailing fire environment are described. Further work could be needed to predict the distributions of fire sensitive species and to identify whether fire needs to be managed beyond initially excluding it from revegetated areas to ensure that all of the desired species can be established. Individualistic species responses to fire, weeds and climate change are limitations to current modelling methods that need to be addressed to predict and manage revegetation outcomes (Austin, 2007).

7.5 Conclusions

In conclusion, an ecological approach to mine landform design based on natural analogues can be used to extend topographic reconstruction methods to encompass ecosystem restoration on waste rock landforms after opencast mining. However, successful application of ecological principles to mine landform design will depend on the following:

- ecological, data and statistical models that are used to select and describe natural analogues to plan a restoration approach must be appropriate to the scale of the mine disturbance and the nature of local environmental gradients
- where the strategy used to select and sample natural analogue areas is not designed to support quantitative environmental analysis of common species the results and recommendations will be unreliable
- static vegetation models can be used to predict distributions of common and abundant species as a test of the ecological design
- a straightforward approach to species distribution modelling using species presence-absence data is recommended that concentrates on the range of ecosystems present in analogous natural landscapes carefully selected for their ecological diversity and representativeness of the mine landscape
- more complex species distribution modelling approaches that incorporate options for weighting species absences and masking environmental ranges that can improve predictions in large regional biological surveys, are not as effective when used with natural analogue data
- predicting the distributions of species that are less prevalent, such as rare or threatened species that may be poorly represented in natural analogues will require additional effort and may be a worthwhile focus for further survey work
- focussing supplementary survey designs for particular species in selected landform analogue areas, rather than expanding the geographic range should ensure that the context of landscape reconstruction is maintained.

Digital terrain information is the core data set of every quantitative species distribution model referenced. Not all of the environmental predictor information that is desirable to support the analogue selection and species distribution modelling is likely to be available at a site. Terrain data is however, extensively available at high enough resolution (<30m) to support detailed environmental analysis of terrain variation globally.

The response data for these models may be presence-absence of a species. Although the geographic range of the sampling program constrains the extent over which the statistical modelling will be accurate. Also, a survey design that provides a detailed description of appropriate hillslope environmental variation occurring in the vicinity of the mine site is likely to be far more informative for a restoration project than geographically extensive biodiversity survey data. This data and the geographic proximity of the analogue to the mine site are key considerations in analogue selection and environmental survey data selection for supporting quantitative modelling of species distributions in mine restoration projects.

Chapter 8 General discussion, conclusions and future research

8.1 General discussion

The first broad aim of this thesis was to test the hypothesis that natural analogue landscapes can be used to develop practical ecological design methodologies for restoring landscapes constructed from waste rock following opencast mining. Landscape classification and analogue selection were considered to be critical to the overall methodological approach. Previous studies indicated that analogue selection based on thematic land unit classifications (White and Walker, 1997; Zonneveld, 1989) or on regional ecological context (Aronson and LeFloch, 1996b; Bryan, 2003; Palik et al., 2000) may compromise the accuracy and validity of the selection process. In this thesis an analogue selection method was developed that linked variation of hill slope environmental processes (determined from digital terrain analysis) to variation in landforms of similar configurations as the rehabilitated mine landforms. Thus the thesis addressed the need for realistic goals providing for a range of restoration outcomes (Ehrenfeld, 2000), framed on an ecosystem approach (Ehrenfeld and Toth, 1997) and incorporating quantitative methods developed in landscape and conservation ecology to develop landscape-level restoration specifications (Holl et al., 2003) has been addressed.

While land units that represent an integrated ecological view of the landscape have been generally recommended to select reference sites for landscape restoration studies (White and Walker, 1997; Zonneveld, 1989) the limitations of regional, thematic environmental mapping are recognised in the literature (Holl et al., 2003) and this may have constrained the ecosystem restoration methodology recommended for mine rehabilitation to adaptive management actions (Hobbs and McIntyre, 2005; Wu and Hobbs, 2002). Regional environmental survey data based on a thematic land unit classification produced unreliable predictions of environmental pattern in this thesis. However, poor predictive performance was linked to a subjective and over specified conceptual model. The digital terrain analysis and statistical methods used in landscape and conservation ecology offered an alternative landscape classification methodology that was compatible with ecological theory, with the advantage that reliability was estimated explicitly during model selection and calibration. This landscape classification based on continuous environmental surfaces supported environmental survey designs that were framed on concepts of ecological scale, namely the *grain size* of environmental processes and *extent* of the mine landform to encompass the broad variation in hill slope ecosystems within analogous natural landscapes. The environmentally stratified survey design supported the selection and detailed definition of the range of ecosystems relevant to landscape restoration at a mine site. Also, extraneous environmental variation associated with geomorphic processes unrelated to the mine landscape were identified and removed from further predictive analysis. Relief determines a range of flux conditions for sediment, surface and groundwater flow (Ehrenfeld and Toth, 1997) and is a key landform design criterion (Nicolau, 2003; Toy and Chuse, 2005). Relief was also a key terrain parameter affecting species, plant community distribution and underlying substrate in the study.

Quantitative ecological modelling was used to assess the ecological design methodology. Generalised additive regression models of common and abundant tree species that form the ecosystem framework in natural analogue landforms were predictable functions of terrain properties. Vegetation pattern was strongly related to hydrological variation in hill slopes and design criteria such as plan curvature that affect runoff and drainage redistribution will influence revegetation outcomes. Vegetation pattern associated with landscape hydrological processes has been identified by others (Bastin et al., 2002; Ben Wu and Archer, 2005) and associated with landscape functional integrity (Ludwig et al., 2004). However, in this study observed ecological pattern is linked to hydrological process (determined from terrain analysis). This linkage has been a critical issue in landscape ecology (Fischer et al., 2004) that has limited the ability to incorporate landscape pattern in goals for landscape restoration (Fischer et al., 2004; Hobbs, 1997).

The similarity of natural ecosystems to the ecosystems at a mine site for a particular landform analogue may decrease with increasing distance from the site. The distance effect could be associated with regional variables, possibly rainfall environment and insolation, unrelated to landform. Regional environmental gradients that affect ecosystem pattern will need to be quantified to understand the distance effect. An assumption that the landscape is in static equilibrium with environment has been made. While this was reasonable in the context of an ancient, deep-weathered landscape, climate change could introduce new habitats for which there are no current analogues (Williams et al., 2007). Predicting individual species distributions is more robust to equilibrium assumptions than predictions of community distribution (Ferrier and Guisan, 2006). However, a different modelling approach will be needed to incorporate dynamic environmental factors such as fire and species competition (Austin, 2007).

Biodiversity predictions (measured as the number of species recorded) were unreliable although strongly correlated with site drainage or wetness at a case study site. Survey support based on 100 sites was the lower limit required for species distribution modelling for common and abundant species. Rare species, or species with low prevalence in analogue landforms, are generally poorly predicted. Although, uncommon species with distributions that are strongly associated with terrain properties need less survey support. The minimum level of survey support will need to be increased if rare species, biodiversity, competitive effects (Austin, 2007; Maggini et al., 2006) or regional trends are modelled (Webster and Oliver, 2001).

The second broad aim of the thesis was to develop methods for landscape design to restore natural ecosystems. The analogue landform approach provided a clear context in the natural landscape that facilitated communication and understanding of critical issues. Eco-hydrological modelling will have a key role, along with more routinely used erosion and sedimentation models and hydraulic design models (Nicolau, 2003; Toy and Chuse, 2005), in developing methods for landscape restoration at specific mine sites. Successful ecosystem restoration may depend on restoring water retention for ecosystem support in the landform cover, or reconstructed soil zone. Surface and near surface water relations will be particularly important where water supply is a limiting factor (Croton and Reed, 2007), or ecosystems depend on groundwater (Johnson and Miyanishi, 2008). Further work concentrated on understanding ecohydrological processes in paired catchments — comparing the performance of a trial landform with analogue landform(s) is recommended rather than standard agronomic plot trials that lack a relevant landscape context.

The overlap between soil science, geomorphology, and landscape and conservation ecology underpins a scientific approach to landscape restoration presented in this thesis. The key issue for setting achievable and valid design criteria depends on the objectivity applied to selecting natural analogues and describing processes that drive environmental variation.

8.2 General conclusions

Plant community variation is a predictable function of landform and there will be an inevitable shift in mine rehabilitation away from revegetation of landforms designed from environmental engineering principles to ecologically engineered landforms that support biodiversity. This shift requires the development of quantitative ecological models with acceptable accuracy and reliability. Thematic regional environmental mapping does not provide reliable and accurate information needed for site specific ecosystem restoration. The problems are associated with complex conceptual ecological models that can't be validated and a lack of resolution in relation to hill slope environmental processes related to the distribution of water and sediments that clearly underpin environmental pattern. Quantitative statistical modelling approaches based on data models and ecological models that use high resolution terrain data offer a practical alternative to selecting analogue landforms and quantifying ecological variation as a function of landform.

Concepts and methods used in conservation ecology have been adapted and applied here to select natural analogue landforms that represent the range of environmental variation likely, or able to be included in the mine landform. Landforms similar in size and shape to the mine landform are reasonable candidates, with some qualifications. The geomorphic process in the analogue needs to conform to the mine landform. Although the mine landform is constructed from poorly sorted waste rock it does in fact resemble a residual landform where erosion is the dominant geomorphic process. Alluvial or colluvial landscapes that are aggraded by sedimentation are not likely to be suitable analogues supporting analogous vegetation. Relief is a key landform parameter defining the potential energy of a landscape and the dominant geomorphic processes. High relief landscapes can influence geomorphic processes and environmental pattern over large areas of lower relief, e.g. sheet wash fans surrounding high relief plateaux and escarpments. Consequently, environmental variation associated with extraneous high relief landscapes needs to be identified and excluded from the analogue areas.

Environmental surveys must be designed to sample the maximum environmental variation associated with landform in context with the mine. The reliability of quantitative species distribution models will depend on a representative sampling strategy. Sampling programs designed to cover a wide geographic distribution of analogue sites may not provide a clear resolution of hill slope environmental variation and can introduce geographic effects on ecosystem variation that can't be quantified.

Quantitative statistical modelling of ecological outcomes provides an important validation of ecological design principles for the mine landform. Generalised additive modelling using presence–absence data for key overstorey species and quantitative landform parameters from terrain modelling can be used to develop a reliable prediction of species distribution. More intensive sampling campaigns will be needed to predict the distributions for less common species and species with poorly defined environmental range.

The hydraulic properties of the landform cover or reinstated soil zone, could have a critical influence on the eco-hydrology of the revegetated mine landscape. In humid tropical environments, cover design for landscape restoration may not need to be more prescriptive than engineering design principles based on a soil rip-rap concept that incorporates weathered regolith within a bridging structure of more competent waste rock and can be constructed using mine waste rock material. The soil and catchment hydrology in the natural analogue area could provide cover design criteria for ecosystem support; principally cover thickness, infiltration, and hydraulic conductivity and water retention characteristics.

8.3 Future research

Detailed understanding of the eco-hydrology of constructed and natural landscapes will underpin the successful restoration of ecosystem processes that affect woodland revegetation and associated riparian environments. In the humid tropics, the capacity to regenerate the evapotranspiration capacity of the native vegetation is likely to be the key factor that determines the hydrology of the mine landscape. Key principles that relate to the ability of the cover design to support woodland evapotranspiration and natural stream flow regimes need to be confirmed and refined.

Landscape restoration is a broad field and this investigation has concentrated on species distribution without going into detail with regard to environmental and ecological processes that underlie the environmental patterns measured by vegetation survey. Further work that linked understanding of salient environmental processes with species distribution is warranted. To this effect:

- investigating of leaf area index and community structure in relation to seasonal and management induced (fire) ecosystem dynamics in natural and rehabilitated landscapes would improve our understanding of water balance processes, which will be critical to revegetation success and receiving environment water quality outcomes
- incorporating remotely sensed information may be a more practical way to evaluate ecosystem dynamics and demonstrate support since cover and leaf area index can be estimated from multispectral data
- modelling leaf area index as a function of landform and soil cover design, then applying this information to a catchment water balance model would link ecological planning with engineering design.

Re-establishing ecosystems on a restored mine landscape is a dynamic process, distinctly different in some respects from the static presentation of ecosystems in the natural analogue areas that is presented in this thesis. Quantifying the ecosystem dynamics in terms of species competition, wildfire and climate change will require more observations in space and time and more focus on species abundance than has been presented here. The additional effort needed to support a dynamic modelling approach may not be warranted for particular mine restoration projects. The environmental survey support needed for dynamic modelling is more demanding than is required to support the static modelling approach presented in this thesis. However, mine landscape restorations offer unique opportunities for collaboration with research institutions to extend knowledge of landscape function and test ecological theory. Paired catchment studies using a control–impact experimental design with monitoring of trial waste rock and natural analogue area catchments could be used to test and validate landform design concepts.

Bring in the Wine[3]

A glass goblet
Deep-tinted amber.
Crimson pearls drip from the wine-cask,
Boiling dragon and roasting phoenix weep jades of fat.
Silken screens and embroidered curtains close in the
scented breeze.
Blow the dragon flute,
Beat the lizard-skin drum.
White teeth sing,
Slender waists dance.
More than ever now, as the green spring nears its
evening,
And peach flowers scatter like crimson rain!
Be advised by me, stay drunken all your life:
Wine does not reach the earth on Liu Ling's grave.

Li Ho (791-817)

[3] Poems of the Late Tang. Penguin Classics Editor: Betty Radice. Penguin Books. New York 1965. Translated with an introduction by A.C. Graham.

Bibliography

Accad, A. and Neil, D.T., 2006. Modelling pre-clearing vegetation distribution using GIS-integrated statistical, ecological and data models: A case study from the wet tropics of Northeastern Australia. Ecological Modelling, 198(1-2): 85-100.

Allen, T.F.H., Giampietro, M. and Little, A.M., 2003. Distinguishing ecological engineering from environmental engineering. Ecological Engineering, 20(5): 389-407.

Andersen, A.N. et al., 2005. Fire frequency and biodiversity conservation in Australian tropical savannas: implications from the Kapalga fire experiment. Austral Ecology, 30(2): 155-167.

Andersen, A.N., Cook, G.D. and Williams, R.J., 2003. Synthesis: Fire Ecology and Adaptive Conservation Management. In: A.N. Andersen, G.D. Cook and R.J. Williams (Editors), Fire in Tropical Savannas, The Kapalga Experiment. Ecological Studies. Springer, New York, pp. 153-164.

Andersen, A.N., Fisher, A., Hoffmann, B.D., Read, J.L. and Richards, R., 2004. Use of terrestrial invertebrates for biodiversity monitoring in Australian rangelands, with particular reference to ants. Austral Ecology, 29(1): 87-92.

Anon, 2006. Mine Rehabilitation, Leading Practice Sustainable Development Program for the Mining Industry. Commonwealth of Australia, Canberra.

Anon, 2008. Planning for integrated mine closure: toolkit. International Council on Mining & Metals, London.

ANZECC, 2002. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. National Water Quality Management Strategy, 1. Australian and New Zealand Environment and Conservation Council.

ANZMEC, 2000. Strategic Framework for Mine Closure, Canberra, pp. 19.

Appelberg, M. and Svenson, T., 2001. Long-term ecological effects of liming - The ISELAW programme. Water Air and Soil Pollution, 130(1-4): 1745-1750.

Araujo, F.S.d., Martins, S.V., Meira Neto, J.A.A., Lani, J.L. and Pires, I.E., 2006. Structure of the shrub and tree vegetation colonizing an area degraded by kaolin mining, in Bras Pires, MG, Brazil. Revista Arvore, 30(1): 107-116.

Archer, J., Hudson-Edwards, K.A., Preston, D.A., Howarth, R.J. and Linge, K., 2005. Aqueous exposure and uptake of arsenic by riverside communities affected by mining contamination in the Rio Pilcomayo basin, Bolivia. Mineralogical Magazine, 69(5): 719-736.

Aronson, J. and LeFloch, E., 1996a. Hierarchies and landscape history: Dialoguing with Hobbs and Norton. Restoration Ecology, 4(4): 327-333.

Aronson, J. and LeFloch, E., 1996b. Vital landscape attributes: Missing tools for restoration ecology. Restoration Ecology, 4(4): 377-387.

Ashwath, N., Cusbert, P. and Bayliss, B., 1994. Characterisation of mine spoils for plant growth factors: comparison of mine soils with natural soils for chemical properties that may affect growth and long-term viability of native plants. EPA: 1990-94., Commonwealth EPA, Canberra.

Atega, P.C., 1992. Land unit mapping: a tool for integrated land use planning and management within the Marikina Watershed Reservation in Rizal, Philippines, Integrated land use management for tropical agriculture. Proceedings Second International Symposium, Queensland, 15 September to 25 September 1992.

Austin, M., 2007. Species distribution models and ecological theory: A critical assessment and some possible new approaches. Ecological Modelling, 200(1-2): 1.

Austin, M., Nicholls, A.O. and Margules, C.R., 1990. Measurement of the realized quantitative niche: environmental niches of five *Eucalyptus* species. Ecological Monographs, 60: 161-177.

Austin, M.P., 1987. Models for the Analysis of Species Response to Environmental Gradients. Vegetatio, 69(1-3): 35-45.

Austin, M.P., 2002a. Case studies of the use of environmental gradients in vegetation and fauna modelling: theory and practice in Australia and New Zealand. In: J. Scott et al. (Editors), Predicting Species Occurrences: Issues of Accuracy and Scale. Island Press, Covelo, California, pp. 73–82.

Austin, M.P., 2002b. Spatial prediction of species distribution: an interface between ecological theory and statistical modelling. Ecological Modelling, 157(2-3): 101-118.

Austin, M.P., Belbin, L., Meyers, J.A., Doherty, M.D. and Luoto, M., 2006. Evaluation of statistical models used for predicting plant species distributions: Role of artificial data and theory. Ecological Modelling, 199(2): 197.

Austin, M.P., Cunningham, R.B. and Fleming, P.M., 1984. New Approaches to Direct Gradient Analysis Using Environmental Scalars and Statistical Curve-Fitting Procedures. Vegetatio, 55(1): 11-27.

Austin, M.P., Cunningham, R.B. and Good, R.B., 1983. Altitudinal Distribution of Several Eucalypt Species in Relation to Other Environmental-Factors in Southern New-South-Wales. Australian Journal of Ecology, 8(2): 169-180.

Austin, M.P. and Gaywood, M.J., 1994. Current Problems of Environmental Gradients and Species Response Curves in Relation to Continuum Theory. Journal of Vegetation Science, 5(4): 473-482.

Austin, M.P. and Heyligers, P.C., 1989. Vegetation survey design for conservation: gradsect sampling of forests in northeastern New South Wales. Biological Conservation, 50: 13-32.

Austin, M.P. and Meyers, J.A., 1996. Current approaches to modelling the environmental niche of eucalypts: Implication for management of forest biodiversity. Forest Ecology and Management, 85(1-3): 95-106.

Austin, M.P. and Smith, T.M., 1989. A New Model for the Continuum Concept. Vegetatio, 83(1-2): 35-47.

Barry, S. and Elith, J., 2006. Error and uncertainty in habitat models. Journal Of Applied Ecology, 43(3): 413-423.

Bastin, G.N., Ludwig, J.A., Eager, R.W., Chewings, V.H. and Liedloff, A.C., 2002. Indicators of landscape function: comparing patchiness metrics using remotely-sensed data from rangelands. Ecological Indicators, 1(4): 247-260.

Belbin, L., 1987. The Use of Non-Hierarchical Allocation Methods for Clustering Large Sets of Data. Australian Computer Journal, 19(1): 32-41.

Belbin, L., 1991. The analysis of pattern in biosurvey data. In: C. Margules and M. Austin (Editors), Nature Conservation: Cost Effective Biological Surveys and Data Analysis. CSIRO Australia, Canberra, pp. 176-190.

Belbin, L., 1995. A Multivariate Approach To The Selection Of Biological Reserves. Biodiversity And Conservation, 4(9): 951-963.

Belbin, L., 2008. PATN. Blatant Fabrications Pty Ltd.

Belbin, L. and McDonald, C., 1993. Comparing three classification methods for use in Ecology. Journal of Vegetation Science, 4: 341-348.

Bell, L.C., 2001. Establishment of native ecosystems after mining -- Australian experience across diverse biogeographic zones. Ecological Engineering, 17(2-3): 179-186.

Bell, S.S., Fonseca, M.S. and Motten, L.B., 1997. Linking Restoration and Landscape Ecology. Restoration Ecology, 5(4): 318-323.

Ben Wu, X. and Archer, S.R., 2005. Scale-dependent influence of topography-based hydrologic features on patterns of woody plant encroachment in savanna landscapes. Landscape Ecology, 20(6): 733-742.

Bennett, J.W. and Plotnikoff, W.W., 1995. Hydrology of Waste Rock Dumps. C426, ANSTO, Sydney.

Beven, K., 1997. TOPMODEL: a critique. Hydrological Processes, 11: 1069-1085.

Beverly, C.R. and Croton, J.T., 2002. Formulation and application of the unsaturated/saturated catchment models SUSCAT and WEC-C. Hydrological Processes, 16(12): 2369-2394.

Bews, B.E. and Barbour, S.L., 1999. Lysimeter design in theory and practice, Tailings and Mine Waste '99. Balkema: Rotterdam, pp. 13 - 21.

Bio, A.M.F., Alkemade, R. and Barendregt, A., 1998. Determining alternative models for vegetation response analysis: a non-parametric approach. Journal Of Vegetation Science, 9(1): 5-16.

Bishop, T.F.A. and McBratney, A.B., 2001. A comparison of prediction methods for the creation of field-extent soil property maps. Geoderma, 103(1-2): 149.

Bollhöfer, A. et al., 2008. Airborne gamma survey of the historic Sleisbeck mine area in the Northern Territory, Australia, and its use for site rehabilitation planning. Journal of Environmental Radioactivity, 99(11): 1770.

Bonell, M. and Molicova, H., 2003. A hydrological approach to tropical forestation: Controversial issues and possible applications of topographic-wetness models. Journal of Tropical Forest Science, 15(3): 411-431.

Bowman, D. and Minchin, P.R., 1987. Environmental relationships of woody vegetation patterns in the Australian monsoon tropics. Australian Journal of Botany, 35(2): 151-169.

Bowman, D., Woinarski, J. and Menkhorst, K., 1993. Environmental Correlates of Tree Species Diversity in Stage-III of Kakadu National-Park, Northern Australia. Australian Journal of Botany, 41(6): 649-660.

Bowman, D.M.J.S., Zhang, Y., Walsh, A. and Williams, R.J., 2003. Experimental comparison of four remote sensing techniques to map tropical savanna fire-scars using Landsat-TM imagery. International Journal of Wildland Fire, 12(3-4): 341-348.

Bradshaw, A., 1997. Restoration of mined lands - using natural processes. Ecological Engineering, 8(4): 255-269.

Bradshaw, A., 2000. The use of natural processes in reclamation - advantages and difficulties. Landscape And Urban Planning, 51(2-4): 89-100.

Bradshaw, A.D., 2004. The role of nutrients and the importance of function in the assembly of ecosystems, Assembly rules and restoration ecology: bridging the gap between theory and practice, pp. 325-340.

Bradshaw, A.D. and Huttl, R.F., 2001. Future minesite restoration involves a broader approach. Ecological Engineering, 17(2-3): 87-90.

Bray, J.R. and Curtis, J.T., 1957. An ordination of the upland forest communities of southern Wisconsin. Ecological Monographs, 27: 325-349.

Brennan, K., 2005. Quantitative descriptions of native plant communities with potential for use in revegetation at Ranger uranium mine. 502, Supervising Scientist.

Brennan, K.E.C., Majer, J.D. and Koch, J.M., 2003. Using fire to facilitate faunal colonization following mining: an assessment using spiders in Western Australian jarrah forest. Ecological Management & Restoration, 4(2): 145-147.

Bridge, G., 2004. Contested terrain: Mining and the environment. Annual Review of Environment and Resources, 29: 205-259.

Brocklehurst, P., 1998. The History and natural Resources of the Tiwi Islands, Northern Territory. Chapter 4 - Vegetation, Parks & Wildlife Commission, N.T.

Brooks, R.H. and Corey, A.T., 1966. Properties of porous media affecting fluid flow. Journal of Irrigation Drainage Division of the American Society of Civil Engineers, 92 (IR2): 555-560.

Brown, J.R., 1994. State and Transition Models for Rangelands .2. Ecology as a Basis for Rangeland Management - Performance Criteria for Testing Models. Tropical Grasslands, 28(4): 206-213.

Brown, M.T., 2005. Landscape restoration following phosphate mining: 30 years of co-evolution of science, industry and regulation. Ecological Engineering, 24(4): 309-329.

Bryan, B., 2003. Physical environmental modeling, visualization and query for supporting landscape planning decisions. Landscape and Urban Planning, 65(4): 237-259.

Bubb, K. and Croton, J., 2002. Effects on catchment water balance from the management of Pinus plantations on the coastal lowlands of south-east Queensland, Australia. Hydrological Processes, 16: 105-117.

Bui, E.N., 2004. Soil survey as a knowledge system. Geoderma, 120(1-2): 17-26.

Bui, E.N. and Henderson, B.L., 2003. Vegetation indicators of salinity in northern Queensland. Austral Ecology, 28(5): 539-552.

Bui, E.N., Loughhead, A. and Corner, R., 1999. Extracting soil-landscape rules from previous soil surveys. Australian Journal of Soil Research, 37(3): 495-508.

Bui, E.N. and Moran, C.J., 2000. Regional-scale investigation of the spatial distribution and origin of soluble salts in central north Queensland. Hydrological Processes, 14(2): 237-250.

Bui, E.N. and Moran, C.J., 2001. Disaggregation of polygons of surficial geology and soil maps using spatial modelling and legacy data. Geoderma, 103(1-2): 79-94.

Bui, E.N. and Moran, C.J., 2003. A strategy to fill gaps in soil survey over large spatial extents: an example from the Murray-Darling basin of Australia. Geoderma, 111(1-2): 21-44.

Burke, A., 2003. Practical measures in arid land restoration after mining - a review for the southern Namib. South African Journal of Science, 99(9/10): 413-417.

Busby, J.R., 1991. BIOCLIM - A Bioclimate Analysis and Prediction System. In: C.R. Margules and M.P. Austin (Editors), Nature Conservation Cost Effective Biological Surveys and Data Analysis. CSIRO, Australia, pp. 64-68.

Cairns, J., 1994. Rehabilitating damaged ecosystems. In: J. Cairns (Editor), Rehabilitating damaged ecosystems. Lewis Publishers Inc, Boca Raton; USA

Callaham, M.A., Rhoades, C.C. and Heneghan, L., 2008. A Striking Profile: Soil Ecological Knowledge in Restoration Management and Science. Restoration Ecology, 16(4): 604-607.

Campbell, G.S., 1974. Simple method for determining unsaturated conductivity from moisture retention data. Soil Science, 117(6): 311-314.

Carlsson, E. and Buchel, G., 2005. Screening of residual contamination at a former uranium heap leaching site, Thuringia, Germany. Chemie der Erde - Geochemistry, 65(Supplement 1): 75.

Carpenter, S., 1998. Ecosystem ecology. In: A.T. Dodson S , Carpenter S , Ives A , Jeanne R , Kitchell J , Langston N , Turner M (Editor), Ecology. Oxford University Press, Oxford, pp. 123-162.

Chaffey, C.J. and Grant, C.D., 2000. Fire management implications of fuel loads and vegetation structure in rehabilitated sand mines near Newcastle, Australia. Forest Ecology and Management, 129(1-3): 269-278.

Chapman, M. and Underwood, A., 2000. The need for a practical scientific protocol to measure successful restoration. Wetlands, 19(1): 28-48.

Chartier, M.P. and Rostagno, C.M., 2006. Soil erosion thresholds and alternative states in northeastern patagonian rangelands. Rangeland Ecology & Management, 59(6): 616-624.

Chen, X.Y., Eamus, D. and Hutley, L.B., 2004. Seasonal patterns of fine-root productivity and turnover in a tropical savanna of northern Australia. Journal of Tropical Ecology, 20: 221-224.

Choi, Y.D., 2007. Restoration ecology to the future: A call for new paradigm. Restoration Ecology, 15(2): 351-353.

Christian, C. and Stewart, G., 1964. Methodology of integrated surveys, Principles and methods of integrated aerial surveys of natural resources for potential development. Unesco, Toulouse, pp. 146.

Cimmery, V., 2007. User Guide for SAGA (Version 2.0).

Cook, G.D., Williams, R.J., Hutley, L.B., O'Grady, A.P. and Liedloff, A.C., 2002. Variation in vegetative water use in the savannas of the North Australian Tropical Transect. Journal of Vegetation Science, 13(3): 413-418.

Cook, P.G., Favreau, G., Dighton, J.C. and Tickell, S., 2003. Determining natural groundwater influx to a tropical river using radon, chlorofluorocarbons and ionic environmental tracers. Journal of Hydrology, 277(1-2): 74-88.

Cook, P.G. et al., 1998. Water balance of a tropical woodland ecosystem, northern Australia: a combination of micro-meteorological, soil physical and groundwater chemical approaches. Journal of Hydrology, 210: 161-177.

Coops, N.C. and Catling, P.C., 2002. Prediction of the spatial distribution and relative abundance of ground-dwelling mammals using remote sensing imagery and simulation models. Landscape Ecology, 17(2): 173-188.

Coops, N.C., Smith, M.L., Jacobsen, K.L., Martin, M. and Ollinger, S., 2004. Estimation of plant and leaf area index using three techniques in a mature native eucalypt canopy. Austral Ecology, 29(3): 332-341.

Costa-Cabral, M. and Burges, S., 1994. Digital elevation model methods (DEMON). A model of flow over hillslopes for comuptation of contributing and dispersal areas. Water Resources Research, 30(6): 1681-1692.

Crees, M.R., Smettem, K. and Orr, M., 1994. Evaluation of the potential of artificial soil covers to reduce infiltration, Water Down Under '94. The Institution of Engineers, Australia, 11 National Circuit, Barton, A.C.T., Adelaide, pp. 539-544.

Cresswell, H.P. and Paydar, Z., 1996. Water retention in Australian soils. I. Description and prediction using parametric functions. Australian Journal of Soil Research, 34: 195-212.

Croton, J. and Dalton, J., 2006. Modelling the soil/vegetation cover and water balance at Ranger Uranium Mine. Confidential Report to EWL Sciences, Water & Environment Consultants.

Croton, J.T. and Ainsworth, G.L., 2007. Development of a winged tine to relieve mining-related soil compaction after bauxite mining in Western Australia. Restoration Ecology, 15(4): S48-S53.

Croton, J.T. and Bari, M.A., 2001. Using WEC-C, a distributed, deterministic catchment model, to simulate hydrologic responses to agricultural clearing. Environmental Modelling & Software, 16(7): 601-614.

Croton, J.T. and Barry, D.A., 2001. WEC-C: a distributed, deterministic catchment model - theory, formulation and testing. Environmental Modelling & Software, 16(7): 583-599.

Croton, J.T. and Reed, A.J., 2007. Hydrology and bauxite mining on the Darling Plateau. Restoration Ecology, 15(4): S40-S47.

Cummings, J., Reid, N., Davies, I. and Grant, C., 2005. Adaptive restoration of sand-mined areas for biological conservation. Journal of Applied Ecology, 42(1): 160-170.

Cushman, S.A. and McGarigal, K., 2002. Hierarchical, multi-scale decomposition of species-environment relationships. Landscape Ecology, 17(7): 637-646.

Cushman, S.A. and McGarigal, K., 2004a. Hierarchical analysis of forest bird species-environment relationships in the Oregon Coast Range. Ecological Applications, 14(4): 1090-1105.

Cushman, S.A. and McGarigal, K., 2004b. Patterns in the species-environment relationship depend on both scale and choice of response variables. Oikos, 105(1): 117-124.

Davis, A.L.V., Aarde, R.J.v., Scholtz, C.H. and Delport, J.H., 2002. Increasing representation of localized dung beetles across a chronosequence of regenerating vegetation and natural dune forest in South Africa. Global Ecology and Biogeography, 11(3): 191-209.

Dexter, A.R., 2004. Soil physical quality: Part I. Theory, effects of soil texture, density, and organic matter, and effects on root growth. Geoderma, In Press, Corrected Proof.

Dirnbock, T., Hobbs, R.J., Lambeck, R.J. and Caccetta, P.A., 2002. Vegetation distribution in relation to topographically driven processes in southwestern Australia. Applied Vegetation Science, 5(1): 147-158.

Doran, J.W. and Parkin, T.B., 1994. Defining and assessing soil quality. In: J.W. Doran, D.C. Coleman, D.F. Bezdicek and B.A. Stewart (Editors), Defining Soil Quality for a Sustainable Environment. SSSA Special Publication. SSSA, USA, pp. 3-22.

Dormann, C.F., 2007. Effects of incorporating spatial autocorrelation into the analysis of species distribution data. Global Ecology And Biogeography, 16(2): 129-138.

Douglas, I. and Lawson, N., 2000. Material flows due to mining and urbanization. In: U. Ayers and L. Ayers (Editors), A Handbook of Industrial Ecology. MA: Elgar, Cheltenham, UK/Northampton,, pp. 351–364.

Duggan, K., 1991. Mining and erosion in the Alligator Rivers Region of Northern Australia. Doctor of Philosophy Thesis, Macquarie University, Sydney, 265 pp.
Dunin, F.X. et al., 2001. Water balance changes in a crop sequence with lucerne. Australian Journal of Agricultural Research, 52(2): 247-261.
Dunning, J.B., Danielson, B.J. and Pulliam, H.R., 1992. Ecological Processes That Affect Populations in Complex Landscapes. Oikos, 65(1): 169-175.
Duque, J.F.M., Pedraza, J., Diez, A., Sanz, M.A. and Carrasco, R.M., 1998. A geomorphological design for the rehabilitation of an abandoned sand quarry in central Spain. Landscape and Urban Planning, 42(1): 1-14.
Eamus, D., 2003. How does ecosystem water balance affect net primary productivity of woody ecosystems? Functional Plant Biology, 30(2): 187-205.
Eamus, D. and Froend, R., 2006. Groundwater-dependent ecosystems: the where, what and why of GDEs. Australian Journal Of Botany, 54(2): 91-96.
East, T.J. and Cull, R.F., 1988. Application of natural hillslope characteristics to rehabilitated mine structures, Supervising Scientist, Canberra.
East, T.J. et al., 1994. Erosional Stability Of Rehabilitated Uranium-Mine Structures Incorporating Natural Landform Characteristics, Northern Tropical Australia. Zeitschrift Fur Geomorphologie, 38(3): 283-298.
Eastham, J. et al., 1990. Tree/pasture interactions at a range of tree densities in an agroforestry experiment. II. Water uptake in relation to rooting patterns. Australian Journal of Agricultural Research, 41(4): 697-707.
Eastham, J., Scott, P.R. and Steckis, R., 1994. Components of the water-balance for tree species under evaluation for agroforestry to control salinity in the wheat-belt of Western Australia. Agroforestry Systems, 26(3): 157-169.
Edwards, J.T.C., Cutler, D.R., Zimmermann, N.E., Geiser, L. and Moisen, G.G., 2006. Effects of sample survey design on the accuracy of classification tree models in species distribution models. Ecological Modelling, 199(2): 132.
Ehrenfeld, J.G., 2000. Defining the Limits of Restoration: The Need for Realistic Goals. Restoration Ecology, 8(1): 2-9.
Ehrenfeld, J.G. and Toth, L.A., 1997. Restoration ecology and the ecosystem perspective. Restoration Ecology, 5(4): 307-317.
Emerson, W. and Hignett, C., 1986. Chapter 3: Water acceptance, inter-particle bonding and pore water composition of weathering rock piles. In: A.R. Milnes and A.B. Armstrong (Editors), Rehabilitation of waste-rock dumps, Ranger No. 1 mine, Jabiru, N.T. CSIRO Division of Soils, Adelaide.
Enright, N.J., Miller, B.P. and Crawford, A., 1994. Environmental correlates of vegetation patterns and species richness in hte northern Grampians, Victoria. Australian Journal Of Ecology, 19(2): 159-168.
Evans, K.G., 2000. Methods for assessing mine site rehabilitation design for erosion impact. Australian Journal of Soil Research, 38: 231-247.
Evans, K.G., Saynor, M.J. and Willgoose, G.R., 1999. Changes in hydrology, sediment loss and microtopography of a vegetated mine waste rock dump impacted by fire. Land Degradation & Development, 10: 507-522.
Evans, K.G. and Willgoose, G.R., 2000. Post-mining landform evolution modelling: 2. Effects of vegetation and surface ripping. Earth Surface Processes and Landforms, 25(8): 803-823.
Evans, K.G., Willgoose, G.R., Saynor, M.J. and House, T., 1998. Effect of vegetation and surface amelioration on simulated landform evolution of the post-mining

landscape at ERA Ranger Mine, Northern Territory. 134, Supervising Scientist, Canberra.
Faith, D.P., Minchin, P.R. and Belbin, L., 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetatio, 69: 57-68.
Fensham, R., 1990. Interactive effects of fire frequency and site factors in *Eucalyptus* forest near Darwin, northern Australia. Journal of Ecology, 15: 255-266.
Ferreira, S.M. and Aarde, R.J.v., 1997. The chronosequence of rehabilitating stands of coastal dune forests: do small mammals confirm it? South African Journal of Science, 93(5): 211-214.
Ferrier, S. and Guisan, A., 2006. Spatial modelling of biodiversity at the community level. Journal of Applied Ecology, 43(3): 393-404.
Fielding, A. and Bell, J., 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation, 24: 38-49.
Filet, P.G., 1994. State and Transition Models for Rangelands .3. The Impact of the State and Transition Model on Grazing Lands Research, Management and Extension - a Review. Tropical Grasslands, 28(4): 214-222.
Fischer, J. and Lindenmayer, D., 2007. Landscape modification and habitat fragmentation: a synthesis. Global Ecology and Biogeography, 16(3): 265-280.
Fischer, J., Lindenmayer, D. and Fazey, I., 2004. Appreciating Ecological Complexity: Habitat Contours as a Conceptual Landscape Model. Conservation Biology, 18(5): 1245-1253.
Fisher, R., Vigilante, T., Yates, C. and Russell-Smith, J., 2003. Patterns of landscape fire and predicted vegetation response in the North Kimberley region of Western Australia. International Journal of Wildland Fire, 12(3-4): 369-379.
Fitzpatrick, R.W., 1986. Chapter 2: Morphology, classification and genesis of "mine soils" on the rock dumps. In: A.R. Milnes and A.B. Armstrong (Editors), Rehabilitation of waste-rock dumps, Ranger No. 1 mine, Jabiru, N.T. CSIRO Division of Soils, Adelaide.
Fitzpatrick, R.W., Reddell, P., Milnes, A.R. and Beech, T.A., 1989. Description and classification of minesoils, natural undisturbed soils and stockpiled soils with respect to chemical, microbiological and vegetation characteristics. In: A.R. Milnes (Editor), Waste rock dumps, Ranger No. 1 Uranium mine, Jabiru, N.T. Comparison of minesoils with stockpiled and undisturbed natural soils: physical, chemical, microbiological and vegetation characteristics. Confidential Report for Ranger Uranium Mines Pty Ltd. CSIRO Division of Soils, Adelaide, pp. 6-22.
Franklin, J., 1995. Predictive vegetation mapping: Geographic modelling of biospatial patterns in relation to environmental gradients. Progress In Physical Geography, 19(4): 474-499.
Franklin, J., McCullough, P. and Gray, C., 2000. Terrain variables for predictive mapping of vegetation communities in Southern California. In: J.W.a.J. Gallant (Editor), Terrain Analysis: Principals and Applications. John Wiley and Sons, New York, pp. 381.
Fredlund, D.G. and Xing, A., 1994. Equations for the soil-water characteristic curve. Canadian Geotechnical Journal, 31: 521-532.
Gahegan, M., 2000. On the application of inductive machine learning tools to geographical analysis. Geographical Analysis, 32(2): 113-139.

Gallant, J. and Dowling, T., 2003. A multiresolution index of valley bottom flatness for mapping depositional areas. Water Resources Research, 39(12): 1347.
Gallant, J.C. and Hutchinson, M.F., 1997. Scale dependence in terrain analysis. Mathematics and Computers in Simulation, 43(3-6): 313-321.
Gallant, J.C. and Wilson, J.P., 1996. TAPES-G: a grid-based terrain analysis program for the environmental sciences. Computers and Geosciences, 22: 713-722.
Gatzweiler, R., Jahn, S., Neubert, G. and Paul, M., 2001. Cover design for radioactive and AMD-producing mine waste in the Ronneburg area, Eastern Thuringia. Waste Management, 21(2): 175-184.
Gessler, P., Chadwick, O., Chamran, F., Althouse, L. and Holmes, K., 2000. Modeling soil-landscape and ecosystem properties using terrain attributes. Soil Science Society Of America Journal, 64: 2046-2056.
Gessler, P.E., Moore, I.D., McKenzie, N.J. and Ryan, P.J., 1995. Soil-landscape modeling and spatial prediction of soil attributes. International Journal of Geographical Information Systems, 9: 421-432.
Gibbons, P. et al., 2008. Rapidly quantifying reference conditions in modified landscapes. Biological Conservation, 141(10): 2483-2493.
Gill, A., Allan, G. and Yates, C., 2003. Fire-created patchiness in Australian savannas. International Journal of Wildland Fire, 12(3-4): 323-331.
Gillson, L., 2004. Testing non-equilibrium theories in savannas: 1400 years of vegetation change in Tsavo National Park, Kenya. Ecological Complexity, 1(4): 281-298.
Grant, C., 2006. State-and-transition successional model for bauxite mining rehabilitation in the jarrah forest of Western Australia. Restoration Ecology, 14: 28-37.
Grant, C. and Koch, J., 2007. Decommissioning Western Australia's First Bauxite Mine: Co-evolving vegetation restoration techniques and targets. Ecological Management & Restoration, 8(2): 92-105.
Grant, C.D. and Loneragan, W.A., 2001. The effects of burning on the understorey composition of rehabilitated bauxite mines in Western Australia: community changes and vegetation succession. Forest Ecology and Management, 145(3): 255-279.
Grant, D.M. and Dawson, B.D., 1995. Isco Open Channel Flow Measurement Handbook. Isco Environmental Division, Lincoln, Nebraska.
Grayson, R.B., Argent, R.M., Nathan, R.J., McMahon, T.A. and Mein, R.G., 1997. Hydrological Recipes (Estimation Techniques in Australian Hydrology). CRC for Catchment Hydrology, Melbourne.
Griffith, J. and Toy, T., 2001. Evolution in revegetation of iron-ore mines in Minas Gerais State, Brazil. Unasylva, 207(52): 9-15.
Guevara, R., Rosales, J. and Sanoja, E., 2005. Pioneer vegetation on rocks, a biological potential for revegetation of degraded areas by iron-ore mining. Interciencia, 30(10): 644-652.
Guisan, A., Edwards, T.C. and Hastie, T., 2002. Generalized linear and generalized additive models in studies of species distributions: setting the scene. Ecological Modelling, 157(2-3): 89-100.
Guisan, A. and Harrell, F.E., 2000. Ordinal response regression models in ecology. Journal of Vegetation Science, 11(5): 617-626.
Guisan, A. et al., 2006. Making better biogeographical predictions of species' distributions. Journal of Applied Ecology, 43(3): 386-392.

Guisan, A. and Thuiller, W., 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters, 8(9): 993-1009.
Guisan, A., Weiss, S.B. and Weiss, A.D., 1999. GLM versus CCA spatial modeling of plant species distribution. Plant Ecology, 143(1): 107-122.
Guisan, A. and Zimmermann, N.E., 2000. Predictive habitat distribution models in ecology. Ecological Modelling, 135(2-3): 147-186.
Guo, X., Zhang, J., Gong, H. and Zhang, G., 2007. Interspecific associations of restored vegetation and related soil factors in Antaibao Mine, Shanxi. Biodiversity Science, 15(1): 46-52.
Hancock, G.R., 2004. The use of landscape evolution models in mining rehabilitation design. Environmental Geology, 46(5): 561-573.
Hancock, G.R. et al., 2000. Medium-term erosion simulation of an abandoned mine site using the SIBERIA landscape evolution model. Australian Journal of Soil Research, 38(2): 249-263.
Hancock, G.R., Grabham, M.K., Martin, P., Evans, K.G. and Bollhofer, A., 2006. A methodology for the assessment of rehabilitation success of post mining landscapes--sediment and radionuclide transport at the former Nabarlek uranium mine, Northern Territory, Australia. Science of The Total Environment, 354(2-3): 103-119.
Hancock, G.R., Loch, R.J. and Willgoose, G.R., 2003. The design of post-mining landscapes using geomorphic principles. Earth Surface Processes and Landforms, 28(10): 1097-1110.
Hancock, G.R., Lowry, J.B.C., Moliere, D.R. and Evans, K.G., 2008. An evaluation of an enhanced soil erosion and landscape evolution model: a case study assessment of the former Nabarlek uranium mine, Northern Territory, Australia. Earth Surface Processes And Landforms, 33(13): 2045-2063.
Hatton, T., Reece, P., Taylor, P. and McEwan, K., 1998. Does leaf water efficiency vary among eucalypts in water-limited environments? Tree Physiology, 18(8-9): 529-536.
Hatton, T.J., Ruprecht, J. and George, R.J., 2003. Preclearing hydrology of the Western Australia wheatbelt: Target for the future? Plant And Soil, 257(2): 341-356.
Hawkins, B.A., Diniz-Filho, J.A.F., Mauricio Bini, L., De Marco, P. and Blackburn, T.M., 2007. Red herrings revisited: spatial autocorrelation and parameter estimation in geographical ecology. Ecography, 30(3): 375-384.
Heerdegen, R.G. and Cook, G.D., 1999. Redefining wetness by dryness: the duration and extent of the North Australian Monsoon. In: M. Roche, M. McKenna and P. Hesp (Editors), Conference Proceedings 20th New Zealand Geography Conference. New Zealand Geographical Society Conference Series No. 20. John Walker Printers, New Zealand, pp. 139 - 140.
Heikkinen, J. and Makipaa, R., 2010. Testing hypotheses on shape and distribution of ecological response curves. Ecological Modelling, 221(3): 388-399.
Henderson, B.L., Bui, E.N., Moran, C.J. and Simon, D.A.P., 2005. Australia-wide predictions of soil properties using decision trees. Geoderma, 124(3-4): 383-398.
Heneghan, L. et al., 2008. Integrating Soil Ecological Knowledge into Restoration Management. Restoration Ecology, 16(4): 608-617.
Hewitt, A., McKenzie, N., Grundy, M. and Slater, B., 2008. Qualitative survey. In: N. McKenzie, M. Grundy, R. Webster and A. Ringrose-Voase (Editors),

Guidelines for Surveying Soil and Land Resources. Australian Soil and Land Survey Handbook Series. CSIRO Publishing, Melbourne, pp. 285-306.
Ho, C., Arnold, W., Cochran, J. and Taira, R., 2002. Development of a risk-based probabilistic performance-assessment method for long-term cover systems. SAND2002-3131, US Department of Energy.
Hobbs, R., 1997. Future landscapes and the future of landscape ecology. Landscape and Urban Planning, 37(1-2): 1-9.
Hobbs, R. and McIntyre, S., 2005. Categorizing Australian landscapes as an aid to assessing the generality of landscape management guidelines. Global Ecology and Biogeography, 14(1): 1-15.
Hobbs, R.J., 2007. Setting effective and realistic restoration goals: Key directions for research. Restoration Ecology, 15(2): 354-357.
Hobbs, R.J. and Cramer, V.A., 2003. Natural ecosystems: Pattern and process in relation to local and landscape diversity in southwestern Australian woodlands. Plant And Soil, 257(2): 371-378.
Hobbs, R.J. and Norton, D.A., 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology, 4(2): 93-110.
Hodges, C.A., 1995. Mineral Resources, Environmental Issues, and Land Use. Science, 268(5215): 1305-1312.
Hoffmann, B.D. and Andersen, A.N., 2003. Responses of ants to disturbance in Australia, with particular reference to functional groups. Austral Ecology, 28(4): 444-464.
Holl, K.D., 2002. Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. J Appl Ecology, 39(6): 960-970.
Holl, K.D. and Crone, E.E., 2004. Applicability of landscape and island biogeography theory to restoration of riparian understorey plants. Journal Of Applied Ecology, 41(5): 922-933.
Holl, K.D., Crone, E.E. and Schultz, C.B., 2003. Landscape restoration: Moving from generalities to methodologies. Bioscience, 53(5): 491-502.
Hollingsworth, I., 2005. Change over time in landform capping performance - ERA Ranger Mine. Confidential report to ERA Ranger Mine, EWL Sciences, Jabiru.
Hollingsworth, I., Cramb, G. and Ong, C., 1998. Jabiluka Soils. In: A. Milnes and A. Jackson (Editors), The Jabiluka Project. Final six monthly progress report to the Minister for Resources and Energy, October 1998. ERA Environmental Services Pty Ltd, Darwin, pp. 37pp.
Hollingsworth, I., Odeh, I., Bui, E. and MacLeod, P., 2007. Rule Based Land Unit Mapping of the Tiwi Islands. In: P. Lagacherie, A. McBratney and M. Voltz (Editors), Digital Soil Mapping: an Introductory perspective Chapter 29 Developments in Soil Science. Elsevier Amsterdam, pp. 404-414.
Hollingsworth, I.D. and Spain, A., 2000. Soil from spoil: a case study from Oaky Creek Coal Mine, Bowen Basin, Qld, Australia, Third AMEEF Innovation Conference "On the Threshold: Research into Practice", Brisbane, pp. 22-40.
Hollingsworth, I.D., Zimmerman, A. and Cramb, G., 2003. Evaluation of capping design effects on infiltration and runoff from the final landform. Confidential report to ERA Ranger Mine. Report 269, EWL Sciences.
Humphrey, C., Hollingsworth, I. and Fox, G., 2005. Development of predictive habitat suitability models of biological communities in aquatic and terrestrial ecosystems, Alligator Rivers Region Technical Committee Workshop. Environmental Research Institute of the Supervising Scientist.

Hupp, C., Osterkamp, W. and Howard, A., 1995 Biogeomorphology, terrestrial and freshwater systems. In: C. Hupp, W. Osterkamp and A. Howard (Editors), 26th Binghamton Symposium in Geomorphology. Elsevier.

Hutchinson, M.F., 1997. ANUDEM. Centre for Resource and Environmental Studies. Australian National University, Canberra.

Hutchinson, M.F. and Gallant, J.C., 2000. Digital elevation models and representation of terrain shape. In: J.P. Wilson and J.C. Gallant (Editors), Terrain Analysis Principles and Applications. John Wiley and Sons, New York, pp. 29-37.

Hutley, L.B., Doley, D., Yates, D.J. and Boonsaner, A., 1997. Water balance of an Australian subtropical rainforest at altitude: The ecological and physiological significance of intercepted cloud and fog. Australian Journal of Botany, 45(2): 311-329.

Hutley, L.B., O'Grady, A.P. and Eamus, D., 2000. Evapotranspiration from eucalypt open-forest savanna of northern Australia. Functional Ecology, 14(2): 183-194.

Hutley, L.B., O'Grady, A.P. and Eamus, D., 2001. Monsoonal influences on evapotranspiration of savanna vegetation of northern Australia. Oecologia, 126(3): 434-443.

Huttl, R.F. and Bradshaw, A., 2001. Ecology of post-mining landscapes. Restoration Ecology, 9(4): 339-340.

Huttl, R.F. and Gerwin, W., 2003. Forest rehabilitation on post-mining landscapes - a German case study. RAP Publication(No.14): 85-96.

Huttl, R.F. and Gerwin, W., 2005. Disturbed landscapes - development of ecosystems. Geoderma, 129(1-2): 1-3.

Huttl, R.F. and Weber, E., 2001. Forest ecosystem development in post-mining landscapes: a case study of the Lusatian lignite district. Naturwissenschaften, 88(8): 322-329.

ICMM, 2006. Guidance Paper: Financial Assurance for Mine Closure and Reclamation, International Council on Mining & Metals, London.

Jenny, H., 1941. Factors of Soil Formation, a System of Quantitative Pedology. McGraw-Hill, New York.

Jim, C.Y., 2001. Ecological and landscape rehabilitation of a quarry site in Hong Kong. Restoration Ecology, 9(1): 85-94.

Jochimsen, M.E., 2001. Vegetation development and species assemblages in a long-term reclamation project on mine spoil. Ecological Engineering, 17(2/3): 187-198.

Johnson, E.A. and Miyanishi, K., 2008. Creating new landscapes and ecosystems - The Alberta Oil Sands, Year In Ecology And Conservation Biology 2008. Annals Of The New York Academy Of Sciences. Blackwell Publishing, Oxford, pp. 120-145.

Johnson, T., 2002. Design of Erosion Protection for Long-Term Stabilization. NUREG-1623, United States Nuclear Regulatory Authority, Washington, DC.

Johnston, A., 2002. Appendix 1: Ranger Environmental Requirements explanatory material relating to Section 3.3 background values for key variables in water quality (amended November 2001). 2001-2002, Supervising Scientist.

Kelley, G., 2002. Tree water use and soil water dynamics in a savanna of northern Australia. PhD Thesis.

Kelley, G., O'Grady, A.P., Hutley, L.B. and Eamus, D., 2007. A comparison of tree water use in two contiguous vegetation communities of the seasonally dry tropics of northern Australia: the importance of site water budget to tree hydraulics. Australian Journal of Botany, 55: 700-708.

Kew, G.A., Mengler, F.C. and Gilkes, R.J., 2007. Regolith strength, water retention, and implications for ripping and plant root growth in bauxite mine restoration. Restoration Ecology, 15(4): S54-S64.

Kienzle, S., 2004. The effect of DEM raster resolution on first order, second order and compound terrain derivatives. Transactions in GIS, 8(1): 83-111.

Kiesel, G., Mahn, E.G. and Tauchnitz, J.G., 1985. Effect of dump site on the vegetation structure and course of secondary (plant) succession. Part 1. Municipal refuse tips. Hercynia, 22(1): 72-102.

Kirmer, A. and Mahn, E.G., 2001. Spontaneous and initiated succession on unvegetated slopes in the abandoned lignite-mining area of Goitsche, Germany. Applied Vegetation Science, 4(1): 19-27.

Klessa, D.A. (Editor), 2000. The rehabilitation of Nabarlek uranium mine. supervising scientist report, 160. Environmental Research Institute of the Supervising Scientist, Darwin.

Knoche, D., Embacher, A. and Katzur, J., 2002. Water and element fluxes of red oak ecosystems during stand development on post-mining sites (Lusatian lignite district). Water, Air, and Soil Pollution, 141(1/4): 219-231.

Koch, J.M., 2007. Restoring a jarrah forest understorey vegetation after bauxite mining in Western Australia. Restoration Ecology, 15(4): S26-S39.

Koch, J.M. and Hobbs, R.J., 2007. Synthesis: is Alcoa successfully restoring a jarrah forest ecosystem after bauxite mining in Western Australia? Restoration Ecology, 15(4): S137-S144.

Koch, J.M., Ward, S.C., Grant, C.D. and Ainsworth, G.L., 1996. Effects of bauxite mine restoration operations on topsoil seed reserves in the jarrah forest of Western Australia. Restoration Ecology, 4(4): 368-376.

Koerselman, W. and Meuleman, A.F.M., 1996. The vegetation N:P ratio: A new tool to detect the nature of nutrient limitation. Journal of Applied Ecology, 33(6): 1441-1450.

Kramer, P.A., Zabowski, D., Scherer, G. and Everett, R.L., 2000. Native plant restoration of copper mine tailings: II. Field survival, growth, and nutrient uptake. Journal of Environmental Quality, 29(6): 1770-1777.

Kuo, E.Y., Garvie, A.M. and Plotnikoff, W.W., 1999. Modeling the response of lysimeters, Tailings and Mine Waste '99. Balkema: Rotterdam, pp. 225 - 234.

Kuo, E.Y., Ritchie, A.I.M. and Timms, G.P., 2003. Long-term monitoring of water infiltration through covers at White's waste rock dump at the Rum Jungle site using lysimeters, 6th ICARD Conference, Cairns, Qld.

Laffan, S.W. and Lees, B.G., 2004. Predicting regolith properties using environmental correlation: a comparison of spatially global and spatially local approaches. Geoderma, 120(3-4): 241-258.

Lagacherie, P., 2008. Digital Soil Mapping: A State of the Art, Digital Soil Mapping With Limited Data, pp. 3-14.

Lagacherie, P. and Holmes, S., 1997. Addressing geographical data errors in a classification tree for soil unit prediction. International Journal of Geographical Information Systems, 11(2): 183-198.

Lane, P.W., 2002. Generalized linear models in soil science. European Journal of Soil Science, 53: 241-251.

Lawler, J.J. and Edwards, T.C., 2002. Landscape patterns as habitat predictors: building and testing models for cavity-nesting birds in the Uinta Mountains of Utah, USA. Landscape Ecology, 17(3): 233-245.
Leathwick, J.R. and Austin, M.P., 2001. Competitive interactions between tree species in New Zealand's old-growth indigenous forests. Ecology, 82(9): 2560-2573.
Lehmann, A., 2005. GRASP v.3.2 user's manual. Swiss Center For Faunal Cartography, Neuchatel, Switzerland.
Lehmann, A., Overton, J.M. and Leathwick, J.R., 2003. GRASP: generalized regression analysis and spatial prediction. Ecological Modelling, 160(1-2): 165-183.
Leoni, G.L.M., Almeida, M.d.S.S. and Fernandes, H.M., 2004. Computational modelling of final covers for uranium mill tailings impoundments. Journal of Hazardous Materials, 110(1-3): 139-149.
Leps, J. and Smilauer, P., 2003. Multivariate Analysis of Ecological Data Using CANOCO. Cambridge University Press, Cambridge 267 pp.
Leuning, R., Cleugh, H.A., Zegelin, S.J. and Hughes, D., 2005. Carbon and water fluxes over a temperate Eucalyptus forest and a tropical wet/dry savanna in Australia: measurements and comparison with MODIS remote sensing estimates. Agricultural and Forest Meteorology, 129(3-4): 151-173.
Levin, S.A., 1992. The problem of pattern and scale in ecology. Ecology, 73(6): 1943-1967.
Li, B.-L., 2002. A Theoretical Framework of Ecological Phase Transitions for Characterizing Tree-Grass Dynamics. Acta Biotheoretica, 50(3): 141-154.
Li, H. and Wu, J., 2004. Use and misuse of landscape indices. Landscape Ecology, 19(4): 389-399.
Li, M.S., 2006. Ecological restoration of mineland with particular reference to the metalliferous mine wasteland in China: a review of research and practice. Science of the Total Environment, 357(1/3): 38-53.
Li, X., Lu, L., Cheng, G. and Xiao, H., 2001. Quantifying landscape structure of the Heihe River Basin, north-west China using FRAGSTATS. Journal of Arid Environments, 48(4): 521-535.
Lichstein, J.W., Simons, T.R., Shriner, S.A. and Franzreb, K.E., 2002. Spatial autocorrelation and autoregressive models in ecology. Ecological Monographs, 72(3): 445-463.
Lindenmayer, D. et al., 2008. A checklist for ecological management of landscapes for conservation. Ecology Letters, 11(1): 78-91.
Lipsey, M.K. and Child, M.F., 2007. Combining the fields of reintroduction biology and restoration ecology. Conservation Biology, 21(6): 1387-1388.
Liu, C.R., Berry, P.M., Dawson, T.P. and Pearson, R.G., 2005. Selecting thresholds of occurrence in the prediction of species distributions. Ecography, 28(3): 385-393.
Loch, R., 1997. Landform design - better outcomes and reduced costs applying science to above ground and below ground issues, 22nd Annual Environmental Workshop. Minerals Council of Australia, Adelaide, Australia, pp. 550-563.
Loch, R., 1998. Slope research related to the final landform at ERA Ranger Mine, LANDLOCH Pty Ltd.

Lottermoser, B.G. and Ashley, P.M., 2006. Physical dispersion of radioactive mine waste at the rehabilitated Radium Hill uranium mine site, South Australia. Australian Journal of Earth Sciences, 53(3): 485.

Lowry, J., Evans, K., Moliere, D. and Hollingsworth, I., 2006. Assessing landscape reconstruction at Ranger Mine using landform evolution modelling. In: A. Fourie and M. Tippett (Editors), 1st International Workshop on Mine Closure. Australian Centre for Geomechanics.

Ludwig, J.A. et al., 2000a. Monitoring Australian rangeland sites using landscape function indicators and robust ground and remote techniques. Environmental Monitoring and Assessment, 64: 167-178.

Ludwig, J.A., Eager, R.W., Bastin, G.N., Chewings, V.H. and Liedloff, A.C., 2002. A leakiness index for assessing landscape function using remote sensing. Landscape Ecology, 17(2): 157-171.

Ludwig, J.A., Hindley, N. and Barnett, G., 2003. Indicators for monitoring minesite rehabilitation: trends on waste-rock dumps, northern Australia. Ecological Indicators, 3(3): 143-153.

Ludwig, J.A. and Tongway, D.J., 1995. Spatial-organization of landscapes and its function in semiarid woodlands, Australia. Landscape Ecology, 10(1): 51-63.

Ludwig, J.A. and Tongway, D.J., 2000. Viewing rangelands as landscape systems, Rangeland Desertification: Advances in Vegetation Sciences. Kluwer Academic Publishers, Dordrecht, pp. 39-52.

Ludwig, J.A., Tongway, D.J., Bastin, G.N. and James, C.D., 2004. Monitoring ecological indicators of rangeland functional integrity and their relation to biodiversity at local to regional scales. Austral Ecology, 29(1): 108-120.

Ludwig, J.A., Tongway, D.J., Freudenberger, D., Noble, J.C. and Hodgkinson, K.C., 1997. Landscape ecology, function and management: principles from Australia' s rangelands. CSIRO Publishing, Melbourne, Australia.

Ludwig, J.A., Tongway, D.J. and Marsden, S.G., 1999. Stripes, strands or stipples: modelling the influence of three landscape banding patterns on resource capture and productivity in semi-arid woodlands, Australia. Catena, 37(1-2): 257-273.

Ludwig, J.A., Wiens, J.A. and Tongway, D.J., 2000b. A scaling rule for landscape patches and how it applies to conserving soil resources in savannas. Ecosystems, 3(1): 84-97.

Mackey, B.G., Berry, S.L. and Brown, T., 2008. Reconciling approaches to biogeographical regionalization: a systematic and generic framework examined with a case study of the Australian continent. Journal Of Biogeography, 35(2): 213-229.

Mackey, B.G. et al., 2000. Towards a Spatial Model of Boreal Forest Ecosystems: The Role of Digital Terrain Analysis. In: J.P. Wilson and J.C. Gallant (Editors), Terrain Analysis Principles and Applications. John Wiley & Sons, Brisbane, pp. 391-422.

Mackey, B.G., Nix, H.A., Hutchinson, M.F., Macmahon, J.P. and Fleming, P.M., 1988. Assessing representativeness of places for conservation reservation and heritage listing. Environmental Management, 12(4): 501-514.

Maggini, R., Lehmann, A., Zimmermann, N.E. and Guisan, A., 2006. Improving generalized regression analysis for the spatial prediction of forest communities. Journal of Biogeography, 33(10): 1729-1749.

Maiti, S.K. and Ghose, M.K., 2005. Ecological restoration of acidic coal-mine overburden dumps - an Indian case study. Land Contamination & Reclamation, 13(4): 361-369.

Mando, A., 1997. Effect of termites and mulch on the physical rehabilitation of structurally crusted soils in the Sahel. Land Degradation and Development, 8(3): 269-278.

Margules, C. and Austin, M. (Editors), 1991. Nature Conservation: Cost Effective Biological Surveys and Data Analysis. CSIRO, Australia, 203 pp.

Margules, C.R., Nicholls, A.O. and Austin, M.P., 1987. Diversity of Eucalyptus Species Predicted by a Multivariable Environmental Gradient. Oecologia, 71(2): 229-232.

Marsden, J., 2003. Firewatch. Department of Land Information, Government of Western Australia.

Marsden, S.J., Fielding, A.H., Mead, C. and Hussin, M.Z., 2002. A technique for measuring the density and complexity of understorey vegetation in tropical forests. Forest Ecology and Management, 165(1-3): 117-123.

Marshall, T.J., Holmes, J.W. and Rose, C.W., 1996. Soil Physics. Cambridge University Press, Cambridge.

Martinez-Sanchez, M.J. et al., 2008. Assessment of the mobility of metals in a mining-impacted coastal area (Spain, Western Mediterranean). Journal Of Geochemical Exploration, 96(2-3): 171-182.

McBratney, A., 2004. Digital Soil-Terrain Modelling: The Predictive Potential & Uncertainty.

McBratney, A.B., Mendonca Santos, M.L. and Minasny, B., 2003. On digital soil mapping. Geoderma, 117(1-2): 3-52.

McBratney, A.B., Minasny, B., Cattle, S.R. and Vervoort, R.W., 2002. From pedotransfer functions to soil inference systems. Geoderma, 109(1-2): 41-73.

McBratney, A.B., Odeh, I.O.A., Bishop, T.F.A., Dunbar, M.S. and Shatar, T.M., 2000. An overview of pedometric techniques for use in soil survey. Geoderma, 97: 293-327.

McCune, B. and Grace, J., 2002. Analysis of Ecological Communities. MJM, Gleneden Beach, Oregon 97388 USA, 284 pp.

McDonald, R.C., Isbell, R.F., Speight, J.G., Walker, J. and Hopkins, M.S., 2009. Australian Soil and Land Survey Field Handbook. CSIRO Publishing, Melbourne, 250 pp.

McGarigal, K. and Marks, B.J., 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. General Technical Report - Pacific Northwest Research Station, USDA Forest Service(No. PNW-GTR-351): 122 pp.

McGarigal, K., Tagil, S. and Cushman, S., 2009. Surface metrics: an alternative to patch metrics for the quantification of landscape structure. Landscape Ecology, 24(3): 433-450.

McIntyre, D.S. and Loveday, J., 1974. Bulk Density. In: J. Loveday (Editor), Methods for Analysis of Irrigated Soils. Technical Communication. Commonwealth Agricultural Bureaux, pp. 38-42.

McKenzie, N. and Ryan, P., 1999. Spatial prediction of soil properties using environmental correlation. Geoderma, 89: 67-94.

McKenzie, N.J. and Austin, M.P., 1993. A quantitative Australian approach to medium and small scale surveys based on soil stratigraphy and environmental correlation. Geoderma, 57(4): 329-355.

McKenzie, N.J., Gessler, P.E., Ryan, P.J. and O'Connell, D.A., 2000. The role of terrain analysis in soil mapping. In: J.P. Wilson and J.C. Gallant (Editors), Terrain Analysis Principles and Applications. John Wiley & Sons, New York, pp. 245-265.

McKenzie, N.L., Belbin, L., Margules, C.R. and Keighery, G.J., 1989. Selecting representative reserve systems in remote areas: A case study in the Nullarbor region, Australia. Biological Conservation, 50(1-4): 239.

Meurk, C.D. and Swaffield, S.R., 2000. A landscape ecological framework for indigenous regeneration in rural New Zealand-Aotearoa*1. Landscape and Urban Planning, 50(1-3): 129-144.

Miao, Z., Bai, Z. and Gao, L., 2000. Ecological rebuilding and land reclamation in surface mines in Shanxi Province, China. Journal of Environmental Sciences, 12(4): 486-497.

Miller, G., 2006. Guidance paper: financial assurance for mine closure and reclamation. International Council on Mining & Metals.

Miller, J. and Franklin, J., 2002. Modeling the distribution of four vegetation alliances using generalized linear models and classification trees with spatial dependence. Ecological Modelling, 157(2-3): 227-247.

Miller, J., Franklin, J. and Aspinall, R., 2007. Incorporating spatial dependence in predictive vegetation models. Ecological Modelling, 202(3-4): 225-242.

Miller, J.R. and Hobbs, R.J., 2007. Habitat restoration - Do we know what we're doing? Restoration Ecology, 15(3): 382-390.

Milnes, A.R., 1989. Waste rock dumps, Ranger No. 1 Uranium mine, Jabiru, N.T. Comparison of minesoils with stockpiled and undisturbed natural soils: physical, chemical, microbiological and vegetation characteristics, CSIRO Division of Soils.

Milnes, A.R. and Fazey, P.G., 1988. Acid leaching of uranium from ore stockpiles and waste dumps in the Ranger project area, East Jabiru. 2, CSIRO Division of Soils, Adelaide.

Minasny, B. and McBratney, A.B., 2001. A rudimentary mechanistic model for soil formation and landscape development II. A two-dimensional model incorporating chemical weathering. Geoderma, 103(1-2): 161-179.

Minasny, B. and McBratney, A.B., 2002. The neuro-m method for fitting neural network parametric pedotransfer functions. Soil Science Society of America Journal, 66(4): 1407-1407.

Minasny, B., McBratney, A.B. and Salvador-Blanes, S., 2008. Quantitative models for pedogenesis - A review. Geoderma, 144(1-2): 140-157.

Minchin, P.R., 1987a. An evaluation of the relative robustness of techniques for ecological ordination. Vegetatio, 69(1-3): 89-107.

Minchin, P.R., 1987b. Simulation of multidimensional community patterns - towards a comprehensive model. Vegetatio, 71(3): 145-156.

Minty, B., Franklin, R., Milligan, P., Richardson, L. and Wilford, J., 2009. The Radiometric Map of Australia, 20th International Geophysical Conference and Exhibition. Australian Society of Exploration Geophysicists, Adelaide.

Miranda, M. et al., 2004. Mining and Critical Ecosystems: Mapping the Risks, World Resources Institute, Washington DC.

Moliere, D.R., Evans, K.G., Willgoose, G.R. and Saynor, M.J., 2002. Temporal trends in erosion and hydrology for a post-mining landform at Ranger Mine, Northern Territory. 165, Supervising Scientist.

Moore, I.D., Grayson, R.B. and Ladson, A.R., 1991. Digital terrain modeling: a review of hydrological, geomorphological, and biological applications. Hydrological Processes, 5: 3-30.
Moore, I.D., Norton, T.W. and Williams, J.E., 1993. Modeling environmental heterogeneity in forested landscapes. Journal of Hydrology, 150: 717-747.
Moore, J.N. and Luoma, S.N., 1990. Hazardous Wastes From Large-Scale Metal Extraction - A Case-Study. Environmental Science & Technology, 24(9): 1278-1285.
Morin, K.A. and Hutt, N.M., 2001. Prediction of minesite-drainage chemistry through closure using operational monitoring data. Journal of Geochemical Exploration, 73(2): 123-130.
Morley, A., Brown, T. and Koontz, D., 1985. The limnology of a naturally acidic tropical water system in Australia. Verb. Internaitonal Verein. Limnology, 22: 2125-2130.
Mudd, G.M., 2007. An analysis of historic production trends in Australian base metal mining. Ore Geology Reviews, 32(1-2): 227-261.
Murphy, S.R. and Lodge, G.M., 2006. Root depth of native and sown perennial grass-based pastures, North-West Slopes, New South Wales. 2. Estimates from changes in soil water content. Australian Journal of Experimental Agriculture, 46(3): 347-359.
Nanson, G.C., East, T.J. and Roberts, R.G., 1993. Quaternary stratigraphy, geochronology and evolution of the Magela Creek catchment in the monsoon tropics of northern Australia. Sedimentary Geology, 83(3-4): 277-302.
Needham, R.S., 1988. Geology of the Alligator Rivers uranium field, Northern Territory. Bureau of Mineral Resources Bulletin, 224.
Neel, M.C., McGarigal, K. and Cushman, S.A., 2004. Behavior of class-level landscape metrics across gradients of class aggregation and area. Landscape Ecology, 19(4): 435-455.
Nelson, A., Oberth, T. and Cook, S., 2007. Multi-scale correlations between topography and vegetation in a hillside catchment of Honduras. International Journal of Geographical Information Science, 21(2): 145-174.
Nichols, O.G., Carbon, B.A., Colquhoun, I.J., Croton, J.T. and Murray, N.J., 1985. Rehabilitation after Bauxite Mining in Southwestern Australia. Landscape Planning, 12(1): 75-92.
Nicolau, J.M., 2003. Trends in relief design and construction in opencast mining reclamation. Land Degradation & Development, 14(2): 215-226.
Nicolau, J.M. and Asensio, E., 2000. Rainfall erosion on opencast coal-mine lands: Ecological perspective, Reclaimed Land: Erosion Control, Soils and Ecology. Land Reconstruction and Management Series, pp. 51-73.
Nott, J., 1994a. The Influence of Deep Weathering on Coastal Landscape and Landform Development in the Monsoonal Tropics of Northern Australia. Journal of Geology, 102(5): 509-522.
Nott, J., 1994b. Long-Term Landscape Evolution in the Darwin Region and Its Implications for the Origin of Landsurfaces in the North of the Northern-Territory. Australian Journal of Earth Sciences, 41(5): 407-415.
Nriagu, J.O., 1996. A history of global metal pollution. Science, 272(5259): 223-224.
O'Callaghan, J. and Mark, D., 1984. The extraction of drainage networks from digital elevation data. Computer Vision, Graphics and Image Processing, 28(3): 323-344.

O'Grady, A.P., Eamus, D. and Hutley, L.B., 1999. Transpiration increases during the dry season: patterns of tree water use in eucalypt open-forests of northern Australia. Tree Physiology, 19: 591-597.

O'Kane, M. et al., 2003. Evaluation of the Long-Term Performance of Dry Cover Systems - A Summary of the INAP Project on Dry Cover System Longevity, Heap Leach Closure Workshop. The International Network for Acid Prevention (INAP) and Mining Life-Cycle Center (MLC) Mackay School of Mines, Elko, Nevada.

O'Kane, M. and Wels, C., 2003. Mine waste cover system design – linking predicted performance to groundwater and surface water impacts, 6th ICARD Conference Cairns, Queensland, pp. 341-349.

Odeh, I.O.A., 1997. Predicting soil properties from ancillary information: non-spatial models compared with geostatistical and combined methods, 5th International Geostatistics Congress, Woollongong, pp. 22-27.

Odeh, I.O.A., Chittleborough, D.J. and McBratney, A.B., 1991. Elucidation of soil-landform interrelationships by canonical ordination analysis. Geoderma, 49(1-2): 1-32.

Odeh, I.O.A., McBratney, A.B. and Chittleborough, D.J., 1994. Spatial prediction of soil properties from landform attributes derived from a digital elevation model. Geoderma, 63(3-4): 197-214.

Odeh, I.O.A., McBratney, A.B. and Chittleborough, D.J., 1995. Further results on prediction of soil properties from terrain attributes: heterotopic cokriging and regression-kriging. Geoderma, 67(3-4): 215-226.

Odum, E.P., 1971. Fundamentals of Ecology. W. B. Saunders Company, Philadelphia.

Odum, H.T. and Odum, B., 2003. Concepts and methods of ecological engineering. Ecological Engineering, 20: 339-361.

Oksanen, J. and Minchin, P.R., 2002. Continuum theory revisited: what shape are species responses along ecological gradients? Ecological Modelling, 157(2-3): 119.

Olsen, C.J., 1980. A Report on the Land Resources of South East Bathurst Island. LC 80/2, Land Conservation Unit, Conservation Commission of the Northern Territory, Darwin, N.T.

Palik, B.J., Goebel, P.C., Kirkman, L.K. and West, L., 2000. Using landscape hierarchies to guide restoration of disturbed ecosystems. Ecological Applications, 10(1): 189-202.

Palmer, M.A., Ambrose, R.F. and Poff, N.L., 1997. Ecological theory and community restoration ecology. Restoration Ecology, 5(4): 291-300.

Parker, K.C. and Bendix, J., 1996. Landscape-scale geomorphic influences on vegetation patterns in four environments. Physical Geography, 17(2): 113-141.

Parrotta, J.A. and Knowles, O.H., 2001. Restoring tropical forests on lands mined for bauxite: Examples from the Brazilian Amazon. Ecological Engineering, 17(2-3): 219.

Paschke, M.W., Redente, E.F. and Brown, S.L., 2003. Biology and establishment of mountain shrubs on mining disturbances in the Rocky Mountains, USA. Land Degradation & Development, 14(5): 459-480.

Pearson, D.M., 2002. The application of local measures of spatial autocorrelation for describing pattern in north Australian landscapes. Journal of Environmental Management, 64(1): 85-95.

Pelka-Gosciniak, J., 2006. Restoring nature in mining areas of the Silesian Upland (Poland). Earth Surface Processes and Landforms, 31(13): 1685-1691.

Pensa, M., Sellin, A., Luud, A. and Valgma, I., 2004. An analysis of vegetation restoration on opencast oil shale mines in Estonia. Restoration Ecology, 12(2): 200-206.

Pfeffer, K., Pebesma, E. and Burrough, P., 2003. Mapping alpine vegetation using vegetation observations and topographic attributes. Landscape Ecology, 18(8): 759.

Phillips, J.D., 1989. An evaluation of the state factor model of soil ecosystems. Ecological Modelling, 45(3): 165-177.

Phillips, J.D., 2001. Divergent evolution and the spatial structure of soil landscape variability. CATENA, 43(2): 101-113.

Pickup, G., Wasson, R.J., Warner, R.F., Tongway, D. and Clark, R.L., 1987. A feasibility study of geomorphic research for the long term management of uranium mill tailings. 87/2, CSIRO Division of Water Resources Research, Canberra.

Pidsley, S., 2002. Rum Jungle Rehabilitation Project Monitoring Report 1993-1998, 1/2002. Department of Infrastructure Planning & Environment.

Pierce, L.L. et al., 1993. Ecohydrological Changes in the Murray-Darling Basin .3. A Simulation of Regional Hydrological Changes. Journal of Applied Ecology, 30(2): 283-294.

Rayfield, B., Madhur, A. and Laurence, S., 2005. Assessing simple versus complex restoration strategies for industrially disturbed forests. Restoration Ecology, 13(4): 639-650.

Rayment, G.E. and Higginson, F.R., 1992. Australian Laboratory Handbook of soil and water chemical methods. Inkata Press, Sydney.

Rempel, R. and Kushneriuk, R., 2003. The influence of sampling scheme and interpolation method on the power to detect spatial effects of forest birds in Ontario (Canada). Landscape Ecology, 18(8): 741.

Rempel, R.S. and Kaufmann, C.K., 2003. Spatial modeling of harvest constraints on wood supply versus wildlife habitat objectives. Environmental Management, 32(5): 646-659.

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K. and Yoder, D.C., 1997. Predicting Soil Erosion by Water: A Guide to Conservation Planning With the Revised Universal Soil Loss Equation (RUSLE). Agriculture Handbook No. 703. U.S. Department of Agriculture, 404 pp.

Richards, B.G., 1987. Settlement under rock dumps. No. 1, CSIRO Division of Soils, Adelaide.

Riley, S.J., 1995a. Geomorphic estimates of the stability of a uranium mill tailings containment cover: Nabarlek, Northern Territory, Australia. Land Degradation & Rehabilitation, 6: 1-16.

Riley, S.J., 1995b. Issues in assessing the long-term stability of engineered landforms at Ranger uranium mine, Northern Territory, Australia. Journal and Proceedings, Royal Society of New South Wales, 128: 67-78.

Riley, S.J. and Rich, J.F., 1998. Geochemical assessment of an analogue site for an engineered landform at Ranger uranium mine, Northern Territory, Australia. Environmental Geology, 34(2/3): 203-213.

Ripley, E.A., Redmann, R.E. and Crowder, A.A., 1996. Environmental effects of mining. St. Lucie Press, Delray Beach, FL.

Rodriguez-Iturbe, I., D'Odorico, P., Laio, F., Ridolfi, L. and Tamea, S., 2007. Challenges in humid land ecohydrology: Interactions of water table and

unsaturated zone with climate, soil, and vegetation. Water Resources Research, 43(9).
Rose, C. and Stern, W., 1967. Determination of withdrawal of water from soil by crop roots as a function of depth and time. Australian Journal of Soil Research, 5(1): 11-19.
Rose, C.W. and Krishnan, A., 1967. A method of determining hydraulic conductivity characteristics for non-swelling soils in situ and of calculating evaporation from bare soil. Soil Science, 103(6): 369-373.
Ross, K.A., Taylor, J.E., Fox, M.D. and Fox, B.J., 2004. Interaction of multiple disturbances: importance of disturbance interval in the effects of fire on rehabilitating mined areas. Austral Ecol, 29(5): 508-529.
Roux, C.l., 2002. Rehabilitating open-cast mines and quarries. Bois et Forets des Tropiques(No.272): 5-19.
RUEI, 1977. Fox Report, Second Report 1, Australian Government Publishing Service, Canberra.
Russell-Smith, J., 1995. Flora. In: T. Press, D. Lea, A. Webb and A. Graham (Editors), Kakadu Natural and Cultural Heritage and Management. Australian nature Conservation Agency & North Australia Research Unit, ANU, Darwin, pp. 127-166.
Russell-Smith, J., Whitehead, P.J., Williams, R.J. and Flannigan, M., 2003. Fire and savanna landscapes in northern Australia: regional lessons and global challenges - Preface. International Journal of Wildland Fire, 12: 5-9.
Ryan, C. and Boyd, M., 2002. CatchmentSIM: A new GIS tool for topographic geo-computation and hydrologic modelling, 28th International Hydrology and Water Resources Symposium. The Institution of Engineers, Australia, Wollongong, NSW.
Ryan, P.J. et al., 2000. Integrating forest soils information across scales: spatial prediction of soil properties under Australian forests. Forest Ecology and Management, 138(1-3): 139-157.
Sahr, K., White, D. and Jon Kimerling, A., 2003. Geodesic Discrete Global Grid Systems. Cartography and Geographic Information Science, 30(2): 121-134
Sarrailh, J.M., 2002. The replantation of mining sites: the New Caledonian example. Bois et Forets des Tropiques(272): 21-31.
Schaap, M.G., Leij, F.J. and van Genuchten, M.T., 2001. Rosetta: a computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. Journal of Hydrology, 251(3-4): 163-176.
Schaap, M.G., Nemes, A. and van Genuchten, M.T., 2004. Comparison of models for indirect estimation of water retention and available water in surface soils. Vadose Zone Journal, 3(4): 1455-1463.
Schenk, H.J. and Jackson, R.B., 2005. Mapping the global distribution of deep roots in relation to climate and soil characteristics. Geoderma, 126(1-2): 129-140.
Schodde, R., Hedley, A.B., Mason, I.J. and Martensz, P.N., 1987. Vegetation habitats Kakadu National Park, Alligator Rivers Region, Northern Territory, Australia (Final Phase), Division of Wildlife and Rangelands Research, CSIRO, Canberra.
Schwenke, G.D., Ayre, L., Mulligan, D.R. and Bell, L.C., 2000a. Soil stripping and replacement for the rehabilitation of bauxite-mined land at Weipa. II. Soil organic matter dynamics in mine soil chronosequences. Australian Journal of Soil Research, 38(2): 371-393.

Schwenke, G.D., Mulligan, D.R. and Bell, L.C., 2000b. Soil stripping and replacement for the rehabilitation of bauxite-mined land at Weipa. III. Simulated long-term soil organic matter development. Australian Journal of Soil Research, 38(2): 395-410.
Sesnie, S.E., Gessler, P.E., Finegan, B. and Thessler, S., 2008. Integrating Landsat TM and SRTM-DEM derived variables with decision trees for habitat classification and change detection in complex neotropical environments. Remote Sensing Of Environment, 112(5): 2145-2159.
Shinneman, D.J., Baker, W.L. and Lyon, P., 2008. Ecological restoration needs derived from reference conditions for a semi-arid landscape in Western Colorado, USA. Journal Of Arid Environments, 72(3): 207-227.
Short, T.A., Menzies, N.W. and Mulligan, D.R., 2000. Mining disturbance alters phosphorus fractions in northern Australian soils. Australian Journal of Soil Research, 38(2): 411-21.
Simon-Coinason, R., Spain, A.V. and Milnes, A.R., 2003. Landform Processes in the Post Coal-Mining Landscape, Bowen Basin, Australia. A Geomorphological Approach. International Journal of Mining, Reclamation and Environment, 17(1): 20 - 50.
Sklenicka, P. and Lhota, T., 2002. Landscape heterogeneity--a quantitative criterion for landscape reconstruction. Landscape and Urban Planning, 58(2-4): 147-156.
Smith, F.C., Johnson, A.H., Dranoff, M. and Wibiralske, A., 1997. Biomass and nutrient accumulation during natural afforestation of iron-smelting slag. Restoration Ecology, 5(1): 56-65.
Sneath, P. and Sokal, R., 1973. Numerical Taxonomy: The Principles and Practice of Numerical Classification. W.H. Freeman & Co., San Francisco, 573p pp.
Snow, V.O. et al., 1999. Modelling the water balance of effluent-irrigated trees. Agricultural Water Management, 39(1): 47-67.
Soares, J.V. and Almeida, A.C., 2001. Modeling the water balance and soil water fluxes in a fast growing Eucalyptus plantation in Brazil. Journal of Hydrology, 253(1-4): 130-147.
Soloman, F., Schiave, P., Horowitz, L., Rouse, A. and Rae, M., 2006. Mining certification evaluation project final report, CSIRO Canberra.
Sorooshian, B.I.S., Mayr, T., Schaap, M., Wösten, H. and Scholes, B., 1999. Comparison of pedotransfer functions to compute water holding capacity using the van Genuchten model in inorganic soils, IGBP-DIS Working Paper #22.
Speight, J., 2009. Landform, Australian soil and land survey field handbook. CSIRO Publishing, Melbourne, pp. 15-55.
Steiner, N.C. and Kohler, W., 2003. Effects of landscape patterns on species richness - A modelling approach. Agriculture Ecosystems & Environment, 98(1-3): 353-361.
Stockwell, D.R.B. and Peterson, A.T., 2002. Effects of sample size on accuracy of species distribution models. Ecological Modelling, 148(1): 1-13.
Suksi, J., Ruskeeniemi, T. and Saarinen, L., 1996. Selective extractions in uranium migration studies - Findings from a natural analogue study at Palmottu, southern Finland. Journal Of Contaminant Hydrology, 21(1-4): 47-58.
Swets, J.A., 1988. Measuring The Accuracy Of Diagnostic Systems. Science, 240(4857): 1285-1293.

Szota, C., Veneklaas, E.J., Koch, J.M. and Lambers, H., 2007. Root architecture of jarrah (Eucalyptus marginata) trees in relation to post-mining deep ripping in Western Australia. Restoration Ecology, 15(4): S65-S73.

Talsma, T., 1987. Re-evaluation of the well permeameter as a field method for measuring hydraulic conductivity. Australian Journal of Soil Research, 25: 361-368.

Taylor, G., Spain, A., Timms, G., Kuznetsov, V. and Bennett, J., 2003. The medium-term performance of waste rock covers - Rum Jungle as a case study, 6th ICARD Conference, Cairns, Qld, pp. 383-397.

Taylor, J.A. and Tulloch, D., 1985. Rainfall in the wet-dry tropics: extreme events at Darwin and similarities between years during the period 1870-1983 inclusive. Australian Journal of Ecology, 10: 281-295.

Temperton, V.M., 2007. The recent double paradigm shift in restoration ecology. Restoration Ecology, 15(2): 344-347.

ter Braak, C., 1988. CANOCO - a FORTRAN program for canonical community ordination by [partial] [detrended] [canonical] correspondence analysis, principal components analysis and redundancy analysis (verison 2.1), Agricultural Mathmatics Group, Wageningen.

ter Braak, C., 1996. Unimodal models to relate species to environment. DLO-Agricultural Mathematics Group, Wageningen, The Netherlands, 150 pp.

ter Braak, C. and Smilauer, P., 2002. CANOCO Reference Manual and CanoDraw for Windows User's Guide : Software for Canonical Community Ordination Version 4.5. Microcomputer Power, Ithaca, NY, USA, 500 pp.

Thackway, R. and Cresswell, I., 1995. Biogeographic Regionalisation for Australia: a framework for setting priorities in the National Reserves System Cooperative Program, Australian Nature Conservation Agency, Canberra.

Thuiller, W., Brotons, L., Araujo, M.B. and Lavorel, S., 2004. Effects of restricting environmental range of data to project current and future species distributions. Ecography, 27(2): 165-172.

Tischew, S. and Kirmer, A., 2007. Implementation of basic studies in the ecological restoration of surface-mined land. Restoration Ecology, 15(2): 321-325.

Tongway, D.J. and Ludwig, J.A., 1997. The conservation of water and nutrients within landscapes. In: J. Ludwig, D. Tongway, D. Fredenberger, J. Noble and K. Hodgkinson (Editors), Landscape Ecology, Function and Management: Principles from Australia's Rangelands, Chapter 2. CSIRO Publishing, Melbourne, pp. 13-22.

Toy, T. and Black, J., 2000. Topographic Reconstruction: The Theory and Practice. In: R. Barnhisel, R. Darmody and W. Daniels (Editors), Reclamation of Drastically Disturbed Lands. Agronomy Society of America, Crop Science Society of America, Soil Science Society of America, Madison, WI, pp. 41-75.

Toy, T. and Chuse, W., 2005. Topographic reconstruction: a geomorphic approach. Ecological Engineering, 24: 29-35.

Troch, P.A., Detroch, F.P. and Brutsaert, W., 1993. Effective Water-Table Depth to Describe Initial Conditions Prior to Storm Rainfall in Humid Regions. Water Resources Research, 29(2): 427-434.

Trpak, P. et al., 2006. The use of stable cadastre maps for the identification of historical elements of landscape territorial stability as the basis for restoration of ecological stability. Ekologia (Bratislava), 25(Supplement 3): 215-231.

Urban, D., Goslee, S., Pierce, K. and Lookingbill, T., 2002. Extending community ecology to landscapes. Ecoscience, 9(2): 200-212.
Uren, C., 1992. An investigation of surface geology in the Alligator Rivers Region for possible analogues of Uranium mine rehabilitation structures. 56, Supervising Scientist for the Alligator Rivers Region.
Ustin, S.L., Roberts, D.A., Gamon, J.A., Asner, G.P. and Green, R.O., 2004. Using imaging spectroscopy to study ecosystem processes and properties. Bioscience, 54(6): 523-534.
Van Cuylenburg, H.R.M. and Dunlop, C.R., 1973. Land Units of the Seventeen Mile Plain, Melville Island. 14, Animal Industry and Agricultural Branch Department of the Northern Territory, Darwin NT.
van Diggelen, R., Grootjans, A.P. and Harris, J.A., 2001. Ecological restoration: State of the art or state of the science? Restoration Ecology, 9(2): 115-118.
van Genuchten, M.T., 1980. A closed-form equation for predicting the hydraulic condictivity of unsaturated soils. Soil Science Society of America Journal, 44: 892-898.
van Genuchten, M.T. and Nielsen, D.R., 1985. On describing and predicting the hydraulic properties of unsaturated soils. Annals of Geophysics, 3: 615-628.
Vardavas, I., 1993. A simple groundwater recharge-depletion model for the tropical Magela Creek catchment. Ecological Modelling, 68: 147-159.
Vardavas, I., 1998. A simple water balance daily rainfall-runoff model with application to the tropical Magela Creek catchment. Ecologial Modelling, 42: 245-264.
Vardavas, I.M., 1987. Modelling the seasonal variation of net all-wave radiation flux and evaporation in a tropical wet-dry region. Ecological Modelling, 39: 247-268.
Vitousek, P.M. and Farrington, H., 1997. Nutrient limitation and soil development: Experimental test of a biogeochemical theory. Biogeochemistry, 37(1): 63-75.
Wagner, H.H. and Fortin, M.-J., 2005. Spatial analysis of landscapes: concepts and statistics. Ecology, 86(8): 1975-1987.
Walker, L.R., Zarin, D.J., Fetcher, N., Myster, R.W. and Johnson, A.H., 1996. Ecosystem development and plant succession on landslides in the Caribbean. Biotropica, 28(4): 566-576.
Wang, Y. et al., 2001. Landscape ecological planning and design of degraded mining land. Land Degradation & Development, 12(5): 449-459.
Wardell-Johnson, G. and Horwitz, P., 1996. Conserving biodiversity and the recognition of heterogeneity in ancient landscapes: a case study from south-western Australia. Forest Ecology and Management, 85(1-3): 219-238.
Warhurst, A., 2002. Sustainability indicators and sustainability performance management, International Institute for Environment and Development World Business Council for Sustainable Development.
Wassenaar, T.D., Ferreira, S.M. and Aarde, R.J.v., 2007. Flagging aberrant sites and assemblages in restoration projects. Restoration Ecology, 15(1): 68-76.
Wasson, R.J., 1992. Modern sedimentation and late Quaternary evolution of the Magela Creek plain. 6, Supervising Scientist for the Alligator Rivers Region, Canberra.
Webster, R. and Oliver, M., 2001. Geostatistics for Environmental Scientists. John Wiley & Sons, New York, 265 pp.
Webster, R. and Oliver, M.A., 1990. Statistical Methods in Soil and Land Reource Survey. Oxford University Press, Oxford.

Wells, M.R. and Cuylenburg, H.R.M.v., 1978. Land Units of Areas Adjacent to the Tuyu and Yapilika Forestry Plantations, Melville Island, N.T. LC78/9, Land Conservation Unit, Territory Parks and Wildlife Commission.

Wells, M.R., Cuylenburg, H.R.M.v. and Dunlop, C.R., 1978. Land Systems of the Western Half of Melville Island, NT. LC78/10, Land Conservation Unit Report, Territory Parks and Wildlife Commission, Darwin.

Werner, P.A. and Murphy, P.G., 2001. Size-specific biomass allocation and water content of above- and below-ground components of three Eucalyptus species in a northern Australian savanna. Australian Journal of Botany, 49(2): 155-167.

Westoby, M., Walker, B. and Noy-Meir, I., 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management, 42(4): 266-274.

White, L. and Gigliotti, F., 1982. Heavy metal monitoring Alligator Rivers Region Uranium Province. LC82/1, Land Conservation Unit, Conservation Commission, Darwin.

White, P.S. and Walker, J.L., 1997. Approximating nature's variation: Selecting and using reference information in restoration ecology. Restoration Ecology, 5(4): 338-349.

Wiegleb, G. and Felinks, B., 2001. Primary succession in post-mining landscapes of Lower Lusatia - chance or necessity. Ecological Engineering, 17(2/3): 199-217.

Wilford, J., 1992. Regolith mapping using integrated Landsat TM imagery and high resolution gamma-ray spectrometric imagery - Cape York Peninsula. Record 1992/78, AGSO, Canberra.

Wilford, J., 2006. The use of airborne gamma-ray imagery for mapping soils and understanding landscape processes In: P. Lagacherie, A. McBratney and M. Voltz (Editors), Digital Soil Mapping: An Introductory Perspective. Developments in Soil Science. Elsevier, Amsterdam, pp. 207-218.

Willgoose, G. and Hancock, G., 1998. Revisiting the hypsometric curve as an indicator of form and process in transport-limited catchment. Earth Surface Processes and Landforms, 23(7): 611-623.

Willgoose, G. and Riley, S., 1998. The long-term stability of engineered landforms of the Ranger Uranium Mine, Northern Territory, Australia: Application of a catchment evolution model. Earth Surface Processes and Landforms, 23(3): 237-259.

Williams, D.J., Currey, N.A., Ritchie, P. and Wilson, G.W., 2003a. Kidston waste rock dump design and "Store and Release" cover performance seven years on, 6th ICARD Conference, Cairns, pp. 419-426.

Williams, J.W., Jackson, S.T. and Kutzbacht, J.E., 2007. Projected distributions of novel and disappearing climates by 2100 AD. Proceedings Of The National Academy Of Sciences Of The United States Of America, 104(14): 5738-5742.

Williams, R.J. et al., 2005. Carbon accounting, land management, science and policy uncertainty in Australian savanna landscapes: introduction and overview. Australian Journal of Botany, 53(7): 583-588.

Williams, R.J., Cook, G.D., Gill, A.M. and Moore, P.H.R., 1999. Fire regime, fire intensity and tree survival in a tropical savanna in northern Australia. Australian Journal of Ecology, 24(1): 50-59.

Williams, R.J., Duff, G.A., Bowman, D. and Cook, G.D., 1996. Variation in the composition and structure of tropical savannas as a function of rainfall and soil texture along a large-scale climatic gradient in the Northern Territory, Australia. Journal Of Biogeography, 23(6): 747-756.

Williams, R.J., Woinarski, J.C.Z. and Andersen, A.N., 2003b. Fire experiments in northern Australia: contributions to ecological understanding and biodiversity conservation in tropical savannas. International Journal of Wildland Fire, 12(3-4): 391-402.

Wilson, B.A., Brocklehurst, P.S., Clark, M.J. and Dickinson, K.J.M., 1991. Vegetation survey of the Northern Territory. 49, Conservation Commission of the Northern Territory.

Wilson, J.P. and Gallant, J.C., 1996. EROS: A grid-based program for estimating spatially-distributed erosion indices. Computers and Geosciences, 22: 707-712.

Wilson, J.P. and Gallant, J.C., 2000a. Secondary topographic attributes, Terrain Analysis Principles and Applications. John Wiley & Sons, New York, pp. 87-131.

Wilson, J.P. and Gallant, J.C., 2000b. Terrain Analysis Principles and Applications. John Wiley & Sons, Inc,, New York, 469 pp.

Woinarski, J.C.Z. and Fisher, A., 2003. Conservation and the maintenance of biodiversity in the rangelands. Rangeland Journal, 25(2): 157-171.

Woinarski, J.C.Z., Fisher, A. and Milne, D., 1999a. Distribution patterns of vertebrates in relation to an extensive rainfall gradient and variation in soil texture in the tropical savannas of the Northern Territory, Australia. Journal of Tropical Ecology, 15: 381-398.

Woinarski, J.C.Z. et al., 1999b. Distribution and habitat of the northern hopping-mouse, Notomys aquilo. Wildlife Research, 26(4): 495-511.

Woinarski, J.C.Z., Williams, R.J., Price, O. and Rankmore, B., 2005. Landscapes without boundaries: wildlife and their environments in northern Australia. Wildlife Research, 32(5): 377-388.

Wondzell, S.M., Cunningham, G.L. and Bachelet, D., 1996. Relationships between landforms, geomorphic processes, and plant communities on a watershed in the northern Chihuahuan Desert. Landscape Ecology, 11(6): 351-362.

Wood, B.G. and Wells, M.R., 1979. Land resource mapping and its applications in the Northern Territory, Australia. Journal of Environmental Management, 9(1): 27-32.

Woods, P.H., 1994. Likely recharge to permanent groundwater beneath future rehabilitated landforms at Ranger uranium mine, northern Australia. Australian Journal of Earth Sciences, 41(5): 505-508.

Wu, J. and Levin, S.A., 1997. A patch-based spatial modeling approach: conceptual framework and simulation scheme. Ecological Modelling, 101(2-3): 325-346.

Wu, J., Shen, W., Sun, W. and Tueller, P.T., 2002. Empirical patterns of the effects of changing scale on landscape metrics. Landscape Ecology, 17(8): 761-782.

Wu, J.G. and Hobbs, R., 2002. Key issues and research priorities in landscape ecology: An idiosyncratic synthesis. Landscape Ecology, 17(4): 355-365.

Wu, X.B. and Archer, S.R., 2005. Scale-dependent influence of topography-based hydrologic features on patterns of woody plant encroachment in savanna landscapes. Landscape Ecology, 20(6): 733-742.

Yates, C. and Russell-Smith, J., 2003. Fire regimes and vegetation sensitivity analysis: an example from Bradshaw Station, monsoonal northern Australia. International Journal of Wildland Fire, 12(3-4): 349-358.

Zhang, L.J. and Gove, J.H., 2005. Spatial assessment of model errors from four regression techniques. Forest Science, 51(4): 334-346.
Zhang, L.J., Gove, J.H. and Heath, L.S., 2005. Spatial residual analysis of six modeling techniques. Ecological Modelling, 186(2): 154-177.
Zhu, A.X., 1997. A similarity model for representing soil spatial information. Geoderma, 77: 217-242.
Zhu, M., Hastie, T.J. and Walther, G., 2005. Constrained ordination analysis with flexible response functions. Ecological Modelling, 187(4): 524-536.
Zonneveld, I.S., 1989. The Land Unit A Fundamental Concept In Landscape Ecology And Its Applications. Landscape Ecology, 3(2): 67-86.