MONITORING REHABILITATION SUCCESS ON NAMAKWA SANDS HEAVY MINERALS MINING OPERATION, NAMAQUALAND, SOUTH AFRICA

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Submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

IN CONSERVATION ECOLOGY AND ENTOMOLOGY

at the University of Stellenbosch

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May 2006

DECLARATION

I, the undersigned, hereby declare that the content contained in this thesis is my own original work and that I have not previously submitted it, in its entirety or in part, at any university for a degree.

…….. Date: ……

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ABSTRACT

Anglo American Corporation's Namakwa Sands heavy minerals mining and beneficiation operation has been strip-mining a heavy mineral deposit, rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994. The mine is located in the vicinity of Brand-se-Baai on the west coast of South Africa, approximately 385 km north of Cape Town. Strip-mining causes total destruction of natural ecosystems through the removal of vegetation and soil in the area where mining is being undertaken. Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward. Due to the difficulty of rehabilitating mined out areas as a result of harsh environmental factors, Namakwa Sands has initiated various research projects to gain an understanding of the baseline conditions and ecosystem function in order to increase plant cover and biodiversity on post-mined areas. This on-going research and the development of rehabilitation and mining techniques have resulted in the implementation of four rehabilitation techniques varying in investment of topsoil replacement, seeding and plant translocation. This study assesses the success and effectiveness of these techniques in terms of various vegetation and soil parameters. In addition, those parameters that are considered useful for monitoring are identified.

This study indicated that topsoil replacement and plant translocation facilitate the return of similarity, species richness, species diversity and vegetation cover to post-mined areas. The rehabilitation site that had the greatest amount of biological input (topsoil replacement and plant translocation) appeared to be the most successful technique in facilitating vegetation recovery similar to reference sites. In comparison, the site that had the least amount of biological input performed the worst and requires adaptive management, e.g. reseeding and / or plant translocation. Namakwa Sands should continue to replace topsoil in all future rehabilitation efforts and, when possible (e.g. after sufficient winter rain), continue to translocate species in multi-species clumps.

In terms of species selected for translocation, *Othonna cylindrica, Ruschia versicolor* and *Lampranthus suavissimus* should be considered for future large-scale translocation projects. *Zygophyllum morgsana* appears to be more difficult to re-establish under the current climatic conditions (below average rainfall). The long-term viability of rehabilitated *Z. morgsana* populations needs to be determined before considering this species for any future large-scale translocation purposes. No translocated *Asparagus* spp. individuals survived and should therefore not be considered for any further translocation purposes. The grass *Ehrharta calycina*, which is dominant in the site seeded, should continue to be considered for future seeding.

Species and functional diversity appear to be the most limiting factors within all the rehabilitation sites and Namakwa Sands will not be able to meet their long-term objective of small-stock farming if diversity and the number of palatable species do not increase significantly. Adaptive management should seriously be considered in order to speed up this process. Alternatively, an appropriate grazing strategy, which is related to the *Tetragonia fruticosa* dominated vegetation within rehabilitation sites, would need to be determined and adopted.

More time is needed to ameliorate the rehabilitated soil profiles to the same level as in reference sites, especially with regard to carbon, pH and sodium levels. In order to increase organic matter within rehabilitation areas, Namakwa Sands should consider creating clumps with cleared vegetation from the mining front.

Since the long-term rehabilitation goal has not been achieved, Namakwa Sands will need to continue to monitor plant and soil changes until it has been achieved. The objectives of the current rehabilitation programme are limited and Namakwa Sands should develop additional objectives relating to the structure and function of the natural vegetation. This will give a better indication of whether rehabilitation sites are progressing towards the desired end point and if adaptive management is required. In addition, the current monitoring programme (vegetation survey) implemented at Namakwa Sands could be improved by increasing the vegetation parameters to be monitored. It is recommended that the following vegetation parameters be monitored as part of the long-term monitoring programme: species composition and similarity, species richness, species diversity, vegetation cover, species dominance, vertical structure and functional diversity of the vegetation (clumps and inter-clumps). It is also recommended that carbon, pH and sodium of soil profiles be monitored as part of the long-term monitoring programme. These parameters should not be seen as exhaustive as this study only considered various vegetation parameters and soil chemistry between rehabilitation and reference sites. The results of other studies on the fauna, mycorrhiza, insects, etc. should also be taken into consideration and the monitoring parameters expanded accordingly.

OPSOMMING

Anglo American Corporation se Namakwa Sands beoefen vlak mynbou van ‘n swaar mineraal-afsetting, ryk in die kommersieel kosbare minerale ilmeniet, rutiel en sirkoon, sedert September 1994. Die myn is geleë in die omgewing van Brand-se-Baai aan die weskus van Suid-Afrika, ongeveer 385 km noord van Kaapstad. Vlak mynbou veroorsaak totale vernietiging van natuurlike ekosisteme deur die verwydering van plantegroei en grond in die area waar gemyn word. Namakwa Sands het onderneem om gemynde areas te rehabiliteer soos die mynwerksaamhede aanbeweeg. As gevolg van ekstreme omgewingsfaktore is dit moeilik om klaar gemynde areas te rehabiliteer. Namakwa Sands het dus talle navorsingsprojekte geloods om begrip van die basiese toestande en ekosisteemfunksies uit te brei en sodoende plantbedekkng en biodiversiteit in klaar gemynde gebiede te bevorder. Hierdie voortgesette navorsing en ontwikkeling van rehabilitasie en myntegnieke het gelei tot die implementering van vier rehabilitasietegnieke wat wissel in die biologiese inset van bogrondvervanging, saai en planttranslokasie. Hierdie studie evalueer die sukses en effektiwiteit van hierdie tegnieke in terme van verskeie plantegroei- en grondparameters. Die parameters wat as bruikbaar vir monitering geag word, word ook geïdentifiseer.

Hierdie studie het gevind dat bogrondvervanging en planttranslokasie die terugkeer van eendersheid, spesierykheid, spesiediversiteit en plantbedekking na klaar gemynde areas, fasiliteer. Die rehabilitasieperseel wat die grootste hoeveelheid biologiese inset (bogrondvervanging en planttranslokasie) gehad het, het geblyk om die mees suksesvolle tegniek in die fasilitering van plantegroeiherstel tot toestande eenders as verwysingspersele, te wees. In vergelyking het die perseel wat die minste biologiese inset gehad het, die slegste gevaar en benodig aanpassingsbestuur (bv. hersaai en/of planttranslokasie). Namakwa Sands behoort aan te hou om bogrond in alle toekomstige rehabilitasiepogings te vervang en waar moontlik (bv. na voldoende winterreëns) spesies te translokeer in multi-spesie groepe.

In terme van die spesies wat vir translokasie geselekteer is, behoort *Othonna cylindrica*, *Ruschia versicolor* en *Lampranthus suavissimus* vir toekomstige grootskaalse translokasieprojekte oorweeg te word. Dit wil voorkom of *Zygophyllum morgsana* moeiliker is om te hervestig onder huidige klimaatstoestande (onder-gemiddelde reënval). Die langtermyn lewensvatbaarheid van *Z. Morgsana* populasies behoort vasgestel te word voordat hierdie spesie vir toekomstige grootskaalse translokasie doeleindes oorweeg kan word. Geen translokeerde *Asparagus* spp. individue het oorleef nie en behoort dus nie vir verdere transloksie doeleindes oorweeg te word nie. Die grassoort *Ehrharta calycina*, wat dominant in die gesaaide perseel is, behoort steeds vir toekomstige saaigeleenthede oorweeg te word.

Spesies- en funksionele diversiteit blyk die mees beperkende faktore binne al die rehabilitasiepersele te wees en Namakwa Sands sal nie in staat wees om hul lantermyn doelwit van kleinveeboerdery te bereik as diversiteit en die aantal eetbare spesies nie drasties vermeerder nie. Aanpassingsbestuur behoort ernstig oorweeg te word om sodoende die proses te bespoedig. As alternatief behoort ’n gepaste weidingstrategie met betrekking tot die rehabilitasie persele waar *Tetragonia fruticosa* dominant is, vasgestel en aanvaar te word.

Meer tyd word benodig om die gerehabiliteerde grondprofiele (veral met betrekking tot koolstof, pH en natrium vlakke) tot dieselfde vlakke as die verwysingspersele te verbeter. Om organiese materiaal binne die rehabilitasie areas te vermeerder, behoort Namakwa Sands dit te oorweeg om van die plantegroei wat van die mynarea verwyder is, in hope in die rehabilitasie areas te plaas.

Aangesien die langtermyn rehabilitasie doelwit nie bereik is nie, sal Namakwa Sands moet aanhou om plant- en grondveranderinge te monitor totdat al die rehabilitasiedoelwitte bereik is. Die doelwitte van die huidige rehabilitasieprogram is beperk en Namakwa Sands behoort addisionele doelwitte met betrekking tot die struktuur en funksie van die natuurlike plantegroei te ontwikkel. Dit sal ‘n beter aanduiding gee of rehabilitasiepersele na die gewensde eindpunt beweeg en of aanpassingsbestuur nodig is. Verder kan die huidige moniteringsprogram (of plantegroei opname) wat deur Namakwa Sands geïmplementeer word, verbeter word deur die hoeveelheid plantegroeiparameters wat gemonitor word te vermeerder. In hierdie opsig word dit aanbeveel dat die volgende plantegroeiparameters gemonitor word as deel van die langtermyn moniteringsprogram: spesiesamestelling en eendersheid, spesierykheid, spesiediversiteit, plantdekking, spesiedominansie, vertikale struktuur en funksionele diversiteit van die plantegroei (binne plantgroepe en tussen groepe). Dit word ook aanbeveel dat koolstof, pH en natrium van die grond profiele gemonitor word as deel van die langtermyn moniteringsprogram. Hierdie parameters moet nie as alomvattend beskou word nie aangesien hierdie studie slegs verskeie plantegroeiparameters en die chemiese grondsamestelling tussen rehabilitasie en verwysingspersele ondersoek het. Die resultate van ander studies op fauna, mychorrhiza, insekte, ens. moet ook inaggeneem word en die moniteringsparameters derhalwe uitgebrei word.

TABLE OF CONTENTS

DECLARATION i

ABSTRACT ii

OPSOMMING iv

TABLE OF CONTENTS vi

ACKNOWLEDGEMENTS x

CHAPTER 1: GENERAL INTRODUCTION

1.1 BACKGROUND 1-1
1.2 RATIONALE FOR RESEARCH 1-2
1.3 RESEARCH OBJECTIVES 1-2
1.4 THESIS STRUCTURE 1-3
1.5 REFERENCES 1-4

CHAPTER 2: LITERATURE REVIEW: REHABILITATION PLANNING AND MONITORING

ABSTRACT 2-1
2.1 INTRODUCTION 2-2
2.2 TERMINOLOGY 2-2
2.3 REHABILITATION PLANNING 2-3
2.3.1 Rehabilitation goals 2-4
2.3.2 Rehabilitation objectives 2-5
2.3.3 Indicators 2-6
2.3.4 Monitoring and rehabilitation success 2-10
2.3.5 Adaptive management 2-16
2.4 REFERENCES 2-17

CHAPTER 3: DESCRIPTION OF STUDY AREA

3.1 LOCATION 3-1
3.2 CLIMATE 3-2
3.2.1 Precipitation 3-2
3.2.2 Temperature 3-2
3.2.3 Wind 3-3
3.2.4 Relative humidity 3-4

3.3 GEOLOGY AND SOILS 3-5
3.4 SURFACE AND GROUNDWATER 3-6
3.4.1 Surface water 3-6
3.4.2 Groundwater 3-6
3.5 VEGETATION 3-9
3.5.1 Vegetation communities 3-9
3.5.2 Vegetation structure and pattern 3-14
3.5.3 Seed bank 3-14
3.6 FAUNA 3-15
3.6.1 Insects 3-15
3.6.2 Herpetofauna (amphibians and reptiles) 3-16
3.6.3 Birds 3-16
3.6.4 Mammals 3-17
3.7 LAND USE 3-17
3.8 MINING AND REHABILITATION 3-18
3.9 REFERENCES 3-21

CHAPTER 4: VEGETATION RECOVERY IN MINED OUT AREAS AT NAMAKWA SANDS

ABSTRACT 4-1
4.1 INTRODUCTION 4-2
4.2 METHODS 4-3
4.2.1 Study area 4-3
4.2.2 Study sites 4-5
4.2.3 Vegetation sampling 4-6
4.2.4 Data analysis 4-10
4.3 RESULTS 4-12
4.3.1 Species composition and similarity 4-12
4.3.2 Species diversity and evenness 4-15
4.3.3 Species richness 4-17
4.3.4 Growth forms and functional guilds 4-17
4.3.5 Vegetation cover 4-20
4.3.6 Abundance 4-21
4.3.7 Clumping 4-23
4.3.8 Vegetation structure 4-26
4.4 DISCUSSION 4-29
4.5 CONCLUSIONS 4-41
4.6 REFERENCES 4-43
APPENDIX A 4-48

CHAPTER 5: ASSESSMENT OF AMELIORATION OF MINED SOILS AT NAMAKWA SANDS

ABSTRACT 5-1
5.1 INTRODUCTION 5-2
5.2 METHODS 5-4
5.2.1 Study area 5-4
5.2.2 Study sites 5-5
5.2.3 Soil sampling 5-6
5.2.4 Vegetation sampling 5-7
5.2.5 Data analysis 5-8
5.3 RESULTS 5-8
5.3.1 Microsite effect 5-8
5.3.2 Site effect 5-9
5.3.3 Vegetation 5-10
5.3.4 Correlations 5-15
5.4 DISCUSSION 5-16
5.5 CONCLUSIONS 5-22
5.6 REFERENCES 5-23

CHAPTER 6: THE EFFECT OF TRANSLOCATION ON THE POPULATION STRUCTURE (SIZE CLASS DISTRIBUTION) OF FIVE SEMI-ARID PLANT SPECIES AT NAMAKWA SANDS

ABSTRACT 6-1
6.1 INTRODUCTION 6-2
6.2 METHODS 6-5
6.2.1 Study area 6-5
6.2.2 Study sites 6-6
6.2.3 Vegetation sampling 6-8
6.2.4 Data analysis 6-10
6.3 RESULTS 6-11
6.3.1 Number of individuals and recruitment 6-11
6.3.2 Cover 6-15
6.3.3 Population structure / size class distributions 6-15
6.4 DISCUSSION 6-24
6.5 CONCLUSIONS 6-30
6.6 REFERENCES 6-31

CHAPTER 7: CONCLUSIONS AND RECOMMENDATIONS: MONITORING AND FUTURE REHABILITATION

7.1 INTRODUCTION 7-1
7.2 REHABILITATION PLANNING AND MONITORING AT NAMAKWA SANDS 7-1
7.2.1 Rehabilitation objectives 7-1
7.2.2 Monitoring parameters 7-2
7.2.3 Reference and rehabilitation sites 7-2
7.2.4 Monitoring frequency 7-3
7.2.5 Monitoring success 7-3
7.3 REHABILITATION SUCCESS, ADAPTIVE MANAGEMENT AND FURTHER RESEARCH OPPORTUNITIES 7-4
7.4 REHABILITATION MANAGEMENT 7-5
7.5 REFERENCES 7-6

ACKNOWLEDGMENTS

There are a number of people and organisations that require mention because if it were not for them this study would not have been possible. I would like to express my sincere gratitude to the following:

Namakwa Sands (Pty) Ltd, a subsidiary of Anglo American Corporation, for providing the opportunity and funding to undertake this study. Namakwa Sands's continued funding of research projects at the mine site is commendable.

My supervisors, Associate Professor Karen J. Esler and Professor Sue J. Milton of the University of Stellenbosch, for all their guidance, advice and valuable input throughout this study. Their knowledge of the Karoo and Namaqualand vegetation is truly amazing!

Torsten Hälbich of Anglo American Corporation for his interest and input into this study. In addition, the many Namakwa Sands and rehabilitation employees for all their logistical support and data provision.

Martin Kidd of the Centre for Statistical Consultation (University of Stellenbosch) for his tireless assistance with the statistical analysis.

Edwina Marinus of Compton Herbarium (Kirstenbosch Research Centre, South African National Biodiversity Institute) for all her help in facilitating the identification of my plant specimens.

Jonathan Crowther of CCA Environmental (Pty) Ltd for allowing me to undertake this study part time.

Natasha Curry for proof-reading and editing.

Eloise Costandius for the translation of the abstract into Afrikaans.

And lastly to my loving wife, Bronwyn, for all her support, encouragement and patience. Without her I would not have been able to "tame this beast"!



CHAPTER 1

GENERAL INTRODUCTION

1.1 BACKGROUND

Heavy mineral mining has been carried out in South Africa for many years and Anglo American Corporation's Namakwa Sands heavy minerals mining and beneficiation operation (hereafter referred to as Namakwa Sands) is the second major mineral sands operation in South Africa, after Richards Bay (Suttill 1995). The Namakwa Sands mining and beneficiation operation is situated at three sites along the west coast of South Africa (Figure 1.1):

1. The mine and concentration plants (both primary and secondary) are situated in the vicinity of Brand-se-Baai, approximately 385 km north of Cape Town;
2. The mineral separation plant is located approximately 7 km from Koekenaap (near Lutzville); and
3. The smelter is located near Saldanha Bay (Namakwa Sands 2002).

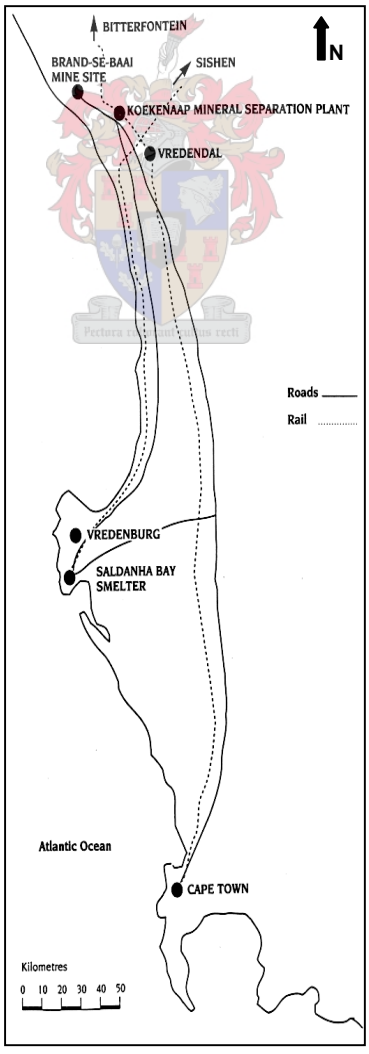

Figure 1.1: Locality of the Anglo American Corporation's Namakwa Sands mine, mineral separation plant and smelter (after Namakwa Sands 2002).

Namakwa Sands has been mining the heavy mineral deposit, rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994. Mining is a continuous process of forward movement from mined to un-mined areas. Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward with approximately 200 to 335 ha of mined out land requiring rehabilitation per annum (Namakwa Sands 2001).

Due to the difficulty of rehabilitating mined out areas at Namakwa Sands as a result of harsh environmental factors (e.g. aridity, wind, and saline, nutrient-poor soils), Namakwa Sands has initiated various research projects to gain an understanding of the baseline conditions and ecosystem function in order to increase plant cover and biodiversity on post-mined areas. This on-going research and the development of rehabilitation and mining techniques have resulted in the implementation of four rehabilitation techniques varying in the investment of topsoil replacement, seeding and plant translocation.

1.2 RATIONALE FOR RESEARCH

Successful rehabilitation projects have been carried out following dune mining in many different countries. The establishment of vegetation and the rehabilitation of an ecosystem will vary according to site-specific environmental conditions, the nature of the soil and various other biotic influences characteristic to the region. Therefore, it is important to consider rehabilitation independently for each site (Lubke & Avis 1998). Monitoring forms an important component in the rehabilitation process (Cooke & Johnson 2002) and will ultimately determine whether rehabilitation is proceeding as predicted and has been successful (Holl & Cairns 2002). Monitoring will also establish which parameters are useful in determining whether rehabilitation has been successful (Lubke & Avis 1998).

To date no attempts have been made to assess the effectiveness or success of the four rehabilitation techniques implemented at Namakwa Sands nor have attempts been made to determine which parameters are useful for monitoring rehabilitation success. Knowing whether rehabilitation has been successful or is proceeding in the right direction has various cost implications and will ultimately determine whether or not a closure certificate is issued to Namakwa Sands by the Department of Minerals and Energy.

1.3 RESEARCH OBJECTIVES

Namakwa Sands commissioned this research project to investigate the effectiveness / success of the four rehabilitation techniques implemented at the mine in order to determine whether specific end points have been reached and, if not, whether rehabilitation is proceeding as predicted. The specific objectives of this study were:

1. to determine which rehabilitation technique was the most successful in facilitating the recreation of a vegetation composition, structure and function similar to the natural vegetation (Chapter 4);
2. to determine if soil parameters are heterogeneously distributed within the natural vegetation and, if so, to what extent the rehabilitation sites show this characteristic (Chapter 5);
3. to determine which rehabilitation technique was the most successful in ameliorating the soil chemistry of mined soils (Chapter 5);
4. to determine whether the translocation of plants has resulted in the development of self-perpetuating populations of the translocated species (Chapter 6); and
5. to refine the current monitoring programme (vegetation survey) at Namakwa Sands based on the results of this study (Chapter 7).

1.4 THESIS STRUCTURE

The remainder of this thesis is structured as follows:

- Chapter 2 reviews the various stages in the conceptual planning of rehabilitation and emphasises the importance of having an understanding of the pre-mining ecosystem, its structure, diversity, dynamics and ecological processes prior to mining.
- Chapter 3 provides a detailed account of the study area (including climate, geology, soils, surface and groundwater, vegetation and fauna) and then briefly describes mining, rehabilitation and monitoring at Namakwa Sands.
- Chapter 4 investigates which of the four rehabilitation techniques is most successful in facilitating the recreation of a vegetation composition, structure and function similar to that found within two reference sites.
- Chapter 5 investigates soil heterogeneity within reference and rehabilitation sites and which of the four rehabilitation techniques has been most successful in ameliorating the soil chemistry of mined soils.
- Chapter 6 investigates the effect of translocation on the population structure (or size class distribution) of five semi-arid plant species.
- Chapter 7 discusses the findings of this study and makes various recommendations for the refinement of the current monitoring programme, adaptive management, future research opportunities and management considerations.

It should be noted that Chapters 4 to 6 have been written in scientific paper format and as a result have some overlap with Chapters 2 and 3, as well as with each other.

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CHAPTER 2

LITERATURE REVIEW:
REHABILITATION PLANNING AND MONITORING

ABSTRACT

Rehabilitation success depends on effective planning and the identification of key ecological processes within the ecosystem of interest. An understanding of the pre-disturbance ecosystem provides the necessary basic information on the ecosystem that requires rehabilitation and is of fundamental importance for the development of sound rehabilitation, management and conservation strategies.

Monitoring and assessing rehabilitation success is an important component of rehabilitation planning. In order to achieve a successful monitoring programme, it is important to clearly define the rehabilitation goals before rehabilitation commences. Rehabilitation goals must be possible and sustainable and must meet local needs at a reasonable cost. Once the rehabilitation goal has been defined it should be translated into specific rehabilitation objectives, which should be formulated from a detailed knowledge of the natural system. Rehabilitation objectives are evaluated on the basis of performance standards known as success criteria. To do this it is necessary to choose indicators that can be used as criteria of performance. The selection of effective indicators is the key to the overall success of any monitoring programme. Indicators should ideally be easily measured, sensitive to stresses on the system, respond to stresses or disturbances in a predictable manner and have a low variability in response. Ecosystems are inherently complex and as a result no single indicator can be expected to measure ecological integrity of an ecosystem, but rather a suite of indicators should be selected.

Monitoring will ultimately determine whether the rehabilitation goals or objectives have been achieved. When designing a monitoring programme a number of criteria should be considered, including: reference site(s) selection; spatial scale; temporal scale (duration and frequency); action thresholds (i.e. the stage where corrective action should be taken); sampling methods and data collection; and data analysis. Monitoring will not only help determine whether specific end points have been reached, but it will also determine if rehabilitation is not proceeding as predicted. If rehabilitation is not proceeding as predicted, corrective measures can be implemented (referred to as adaptive management). Adaptive management is essentially the incorporation of “experimental” or monitored results in a rehabilitation programme to rectify problems.

Key words: rehabilitation; monitoring; success; planning; goals; objectives; indicators; adaptive management.

2.1 INTRODUCTION

The impacts of mining on natural ecosystems are usually significant through the removal of vegetation and soil or their burial beneath mining waste. Strip-mining, as is practised at Namakwa Sands, in particular causes total destruction of natural ecosystems through the removal of vegetation and soil in the area where mining is being undertaken (Cooke & Johnson 2002). Strip-mining is expanding in the arid, winter-rainfall areas of South Africa and, although is economically important, it is having a detrimental effect on biologically diverse environments where vegetation growth is restricted by saline soils, aridity, wind and nutrient-poor soils (Milton 2001; Blignaut & Milton 2005).

Mining is a temporary land-use as the mineral resource, in any one place, is finite and eventually exhausted (Cooke & Johnson 2002). In South Africa, in terms Section 38(1d) of the Minerals and Petroleum Resources Development Act (No. 28 of 2002), mining companies are compelled, as far as it is reasonably practicable, to rehabilitate mined areas to its natural or predetermined state or to a land use which conforms to the generally accepted principle of sustainable development.

Rehabilitation should be viewed as an ongoing process that results in self-sustaining systems (Holl & Cairns 2002). Effective planning and monitoring is important for rehabilitation success and many rehabilitation projects fail because project planning or monitoring has been limited (Pastorok *et al.* 1997). This chapter provides a general overview of rehabilitation planning and monitoring from the literature.

2.2 TERMINOLOGY

In the literature there appears to be no agreed terminology in restoration / rehabilitation ecology. Therefore, terminology was proposed that reflects the goals of the process (Cooke & Johnson 2002):

- "*Reclamation*" is used when some new form of land use will result. The main objectives of reclamation include stabilisation, public safety, aesthetic improvement and usually to a land use with a useful purpose.
- "*Reallocation*" is also used to describe the process when the new land use does not bear an intrinsic relationship with the pre-disturbance ecosystem's structure and functioning. In contrast to rehabilitation and restoration (see below), reallocation assumes a permanent managerial role for people and normally requires inputs in the form of energy, water and fertiliser.
- "*Revegetation*" is the establishment of vegetation in a degraded environment. This goal is simple and easy to achieve. Revegetation is normally a component of land reclamation.
- "*Replacement*" is referred to when the established vegetation is not the same type as was there pre-disturbance. This may be the desired goal of the reclamation process.
- "*Rehabilitation*" refers to the process of restoring a degraded habitat to its original state. Therefore, it shares with restoration (see below) a fundamental focus on historical or pre-existing ecosystems as references, but may settle on one of many possible alternative steady states or a synthetic "simplified ecosystem".

- "*Restoration*" implies returning a degraded habitat to its original state or historic trajectory. The goal of restoration is to emulate structure, functioning, diversity and dynamics of the specified ecosystem. It is unlikely that complete restoration will be achieved, and thus it may be more appropriate to use the term "rehabilitation" (Bradshaw & Chadwick 1980; Aronson *et al*. 1993; Lubke & Avis 1998; SER 2002). This definition of "restoration" is contradictory to the definition presented in the SER primer on Ecological Restoration (SER 2002). The SER primer (SER 2002) states that an ecosystem is considered to have been restored when it contains sufficient biotic and abiotic resources to continue its development towards the intended goals or reference without further assistance. The full expression of all ecosystem attributes is not considered essential to demonstrate restoration.

Hobbs & Norton (1996) felt that it is more important to emphasise the idea that restoration occurs along a continuum and that different activities are simply different forms of restoration. Nevertheless, due to the fact that the possibility of returning the degraded habitat back to its original state (i.e. restoration) is unlikely and that it will be a progression towards its original state, the term "rehabilitation" has been used in this thesis. However, for the purposes of comparison to other studies, the terms reclamation, rehabilitation and restoration are used interchangeably.

2.3 REHABILITATION PLANNING

The stages in conceptual rehabilitation planning are presented in Figure 2.1. Rehabilitation should take place as soon as possible after mining and should ideally move along at the same pace as the mining. Rehabilitation success depends on identifying the key ecological processes within the ecosystem of concern and understanding those processes in relation to the objectives of the project (Pastorok *et al.* 1997). Therefore, an understanding of the pre-mining ecosystem, its structure, diversity, dynamics and ecological processes (e.g. dispersal, colonisation and persistence of local populations), is vital prior to mining as this provides the necessary basic information on the ecosystem that requires rehabilitation (Lubke *et al.* 1996; Lindborg & Eriksson 2004) and is of fundamental importance for the development of sound rehabilitation, management and conservation strategies (de Villiers *et al.* 1999).

Restoration ecology has largely progressed on an ad hoc, site- and situation-specific basis that has not allowed the transfer of information and methodologies from one situation to another (Hobbs & Norton 1996). At present there are no guidelines or performance standards for re-establishing self-sustaining vegetation in degraded South Africa west coast environments where conditions are harsh (Milton 2001). However, the Institute for Plant Conservation at the University of Cape Town has initiated the Namaqualand Restoration Initiative, which seeks to, amongst other things, develop 'best practice' restoration guidelines to restore mixed species communities and improve ecological functioning of previously mined areas. The initiative also seeks to develop a monitoring and evaluation system for restored sites, which will set out criteria for establishing the success of restoration projects at set time-scales after project implementation (Carrick *pers. comm.*).

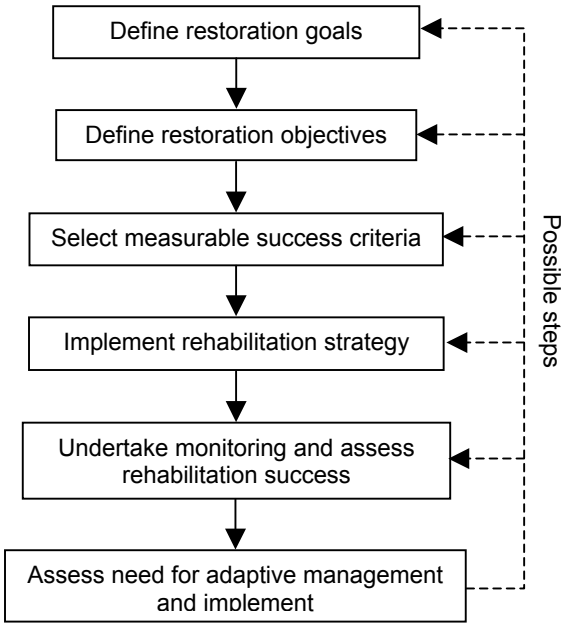

Figure 2.1: Stages in the conceptual planning of the ecological restoration of mined land (adapted from Cooke & Johnson 2002).

2.3.1 REHABILITATION GOALS

In order to achieve a successful monitoring programme, it is important to clearly define the rehabilitation goals before rehabilitation commences (Pastorok *et al.* 1997; Lubke & Avis 1998; Holl & Cairns 2002). However, these rehabilitation goals must be possible and sustainable and must meet local needs at a reasonable cost (Cooke & Johnson 2002). If goals are defined too ambiguously, subsequent performance monitoring may be difficult and this may lead to further discussion on the goals to be achieved. For example, the rehabilitation of a site back to its historical condition may be difficult due to the lack of quantitative data on the site's historical condition (Westman 1991).

Rehabilitation goals may differ according to the existing environmental conditions, the nature of the disturbance and the required end use of the rehabilitated land (Grindley & Barbour 1990). Goals should be determined with the input from various stakeholders (e.g. mining company, authorities, ecologists and members of the public). The weighing up of alternatives (including ecological and socio-economic) and selecting various trade-offs is fundamental to the process of goal setting (Wyant *et al.* 1995). It is important to note that rehabilitation or management goals are not static and they may need to be refined or changed as society's view of management changes. Changes to the rehabilitation goals will require that the monitoring programme and indicators are subject to ongoing review to determine their capability of assessing whether or not the rehabilitation goals have been attained (Cairns *et al*. 1993).

Different rehabilitation goals sometimes require different indicators to be measured in order to assess rehabilitation success. Therefore, goals should be specific enough to allow the evaluation of success (Holl & Cairns 2002). Rehabilitation goals should also recognise alternative states that a natural system may

adopt, as a result of certain natural perturbations (e.g. fire). The range of variance between different states can be determined through studies on the baseline conditions (Westman 1991; Hobbs & Norton 1996). It is not simply the return of the degraded habitat to its original state (or similar) that is a criterion for rehabilitation success. Rehabilitation goals can also be split up into short-term and long-term goals. The initial goal may be to achieve soil stabilisation and self-sustaining vegetation cover. However, the long-term goal may be to rehabilitate the site so that it is similar to the pre-disturbance state in species composition and structure (Westman 1991; Lubke & Avis 1998).

A common goal of rehabilitation programmes is to generate an ecosystem with a similar structure and function to the natural ecosystem (Tomlinson 1984; Chambers *et al.* 1994). However, depending on the goal, rehabilitated ecosystems may not necessarily have the same dominant species, species diversity, production, nutrient cycling, etc. to the undisturbed state (Wyant *et al.* 1995).

2.3.2 REHABILITATION OBJECTIVES

Once the rehabilitation goal has been defined it should be translated into specific process attributes (i.e. rehabilitation objectives or performance standards), which should be formulated from a detailed knowledge of the basic structural and functional characteristics of the natural system and may be derived from descriptions of reference sites or best attainable situation (Tomlinson 1984; Cairns *et al*. 1993; Pastorok *et al.* 1997; Thom 1997; Cooke & Johnson 2002), for example:

- composition (species present and their relative abundance);
- structure (vertical arrangement of vegetation and soil components);
- physical organisation or pattern (horizontal arrangement of system components);
- heterogeneity (a variable composing of composition, structure and pattern);
- function (performance of basic ecosystem processes, e.g. energy flow, water retention, nutrient cycling, etc.);
- species interactions (clumping, pollination, seed dispersal, etc.); and
- dynamics and resilience (succession and the ability to recover after disturbance) (Hobbs & Norton 1996; Cooke & Johnson 2002).

When defining objectives, reference area characterisation is important as it allows one to define the current status of the site, the potentially attainable conditions for the site to be rehabilitated and a point of reference for evaluating rehabilitation success (Pastorok *et al.* 1997). The rehabilitation goal may implicitly want to achieve all ecosystem attributes (i.e. achieve a state that closely corresponds to the pre-disturbance state) but practically the speed of attainment, economics (cost-benefit), achievability and the long-term stability should be considered. Practical measures need to be considered otherwise unrealistic ecological and economic objectives may be set (Cooke & Johnson 2002). Choosing 100% similarity between the reference sites and the rehabilitation sites as the acceptable standard of performance is ecologically unrealistic, as it fails to account for the variation among baseline / reference sites. More

relevant is “replicate similarity”, which is the mean percentage similarity among sites within the regional baseline sample plus or minus some margin of error (e.g. 5 to 10%) (Westman 1991).

Rehabilitation objectives should be specific in order to maximise rehabilitation success and they should define a suite of habitat conditions beneficial to the desired rehabilitation goal and the time period over which they should be achieved (Westman 1991; Pastorok *et al.* 1997).

2.3.3 INDICATORS

Once the rehabilitation goal/s and objective/s have been defined, it is important to translate these into a working definition. Rehabilitation objectives are evaluated on the basis of performance standards known as success criteria (SER 2002). To do this it is necessary to choose indicators (commonly referred to as “parameters” in the literature) that can be used as criteria of performance (Westman 1991). The selected indicators should allow a successful monitoring programme, rather than the collection of endless data that are never used to evaluate the rehabilitation success. However, the exact indicators to be monitored will be specific to the system of interest (Holl & Cairns 2002) and will depend on the reason for monitoring / data collection (Cairns *et al*. 1993).

Indicators should be determined before project implementation. However, in many instances monitoring is an afterthought and is only considered after project implementation. There may also be a lack of funds and trained personnel available to undertake the required monitoring (Holl & Cairns 2002). The purpose of monitoring influences the choice of indicator (Dale & Beyeler 2001). Monitoring is usually undertaken for one of five reasons and depends on the reason different types of indicators are selected:

1. Assessing the current condition of the environment or attainment of goals and objectives (i.e. compliance indicators);
2. Assessing trends in the condition of the environment over time (i.e. compliance indicators or early warning indicators);
3. Anticipation of deleterious conditions before adverse impact occurs (i.e. early warning indicators);
4. Identifying the causative agents causing an impact (or non-compliance) so appropriate management actions can be implemented (i.e. diagnostic indicators); and
5. Demonstrate the interdependence between indicators to make the assessment process more cost-effective (i.e. correlations between various indicators) (Cairns *et al*. 1993).

A challenge in developing and using ecological indicators is determining which of the numerous measures of ecological systems characterise the entire system but are simple enough to be efficiently monitored. The selection of effective indicators is the key to the overall success of any monitoring programme (Dale & Beyeler 2001). The selection of monitoring indicators normally requires an understanding of the pre-mining vegetation and ecological processes (Tomlinson 1984; Cairns *et al*. 1993; Lubke *et al.* 1996; Pastorok *et al.* 1997; Thom 1997; Cooke & Johnson 2002).

In general those indicators that are useful in judging compliance with an objective (i.e. compliance indicator) are not necessarily the best for determining why objectives are not being met (i.e. diagnostic indicator). The selection of indicators is not an easy task and the importance of indicators selection cannot be overemphasised. Any long-term monitoring programme will only be as effective as the indicators chosen (Cairns *et al*. 1993). As most ecosystems are very complex it is useful to select multiple indicators at different levels of biological organisation, as no single indicators will possess all the desirable properties required to assess rehabilitation success (Noss 1990; Holl & Cairns 2002). On the other hand one does not want to select too many indicators if a smaller set will suffice, as management decisions need to be made in a cost effective and timely manner and the monitoring of too many indicators may exceed available resources (Westman 1991; Cairns *et al*. 1993; Thom 2000). The selection of multiple indicators also provides good assurance against false positives (Type I error[1]) and false negatives (Type II error[2]) (Holl & Cairns 2002).

Indicators should be selected to maximise relevant information and minimise redundant information. Westman (1991) suggests that, ideally, indicators that have relatively low levels of inter-correlation should be chosen to maximise information content. The size of the area to be rehabilitated, its borders and surrounding habitat type will influence its functioning. Therefore, indicators that are comparable and compatible from the site to be rehabilitated and the reference site should be chosen (Westman 1991). Indicators that are useful in judging the degree to which specified environmental conditions or objectives have been achieved or maintained must be selected (Cairns *et al*. 1993). Dale & Beyeler (2001) suggest that indicators should be easily measured, sensitive to stresses on the system, respond to stresses or disturbances in a predictable manner and have a low variability in response.

Indicators can be split into physiochemical, socio-economic and biological. Ecosystem objectives will most likely require the identification of biological indicators that can serve as key compliance indicators. Management efforts will be greatly enhanced by the identification of other biological indicators that function as diagnostic and early warning indicators (Cairns *et al*. 1993). Biological indicators can be considered at four levels of biological organisation: genetic; individual / population; community / ecosystem; and landscape (Cairns *et al*. 1993; Dale & Beyeler 2001; Holl & Cairns 2002) (Figure 2.2). Measurements of populations, communities and ecosystems are generally more appropriate compliance indicators for assessing the achievement of goals and objectives. Measurements of individuals on the other hand will tend to be better diagnostic and early warning indicators (Cairns *et al*. 1993).

Within each level of biological organisation different ecosystem attributes can be monitored, these include composition, structure and function. Ideally the suite of indicators selected should represent key information about the structure, composition and function of the system under consideration (Dale & Beyeler 2001). Noss (1990) developed a framework showing the three ecosystem attributes (compositional, structural and functional) at four levels of biological organisation (Figure 2.2). Figure 2.2

[1] False positives (Type I error) are when there are indications that some undesirable effect has occurred, when in fact it has not (Holl & Cairns 2002).
[2] False negatives (Type II error) are when there are indications that no deleterious effect has occurred, when in fact it has (Holl & Cairns 2002).

suggests that knowledge of one part of the triangle may provide information about other aspects of the system. Sometimes measures from one scale can provide information relevant to another scale (Dale & Beyeler 2001).

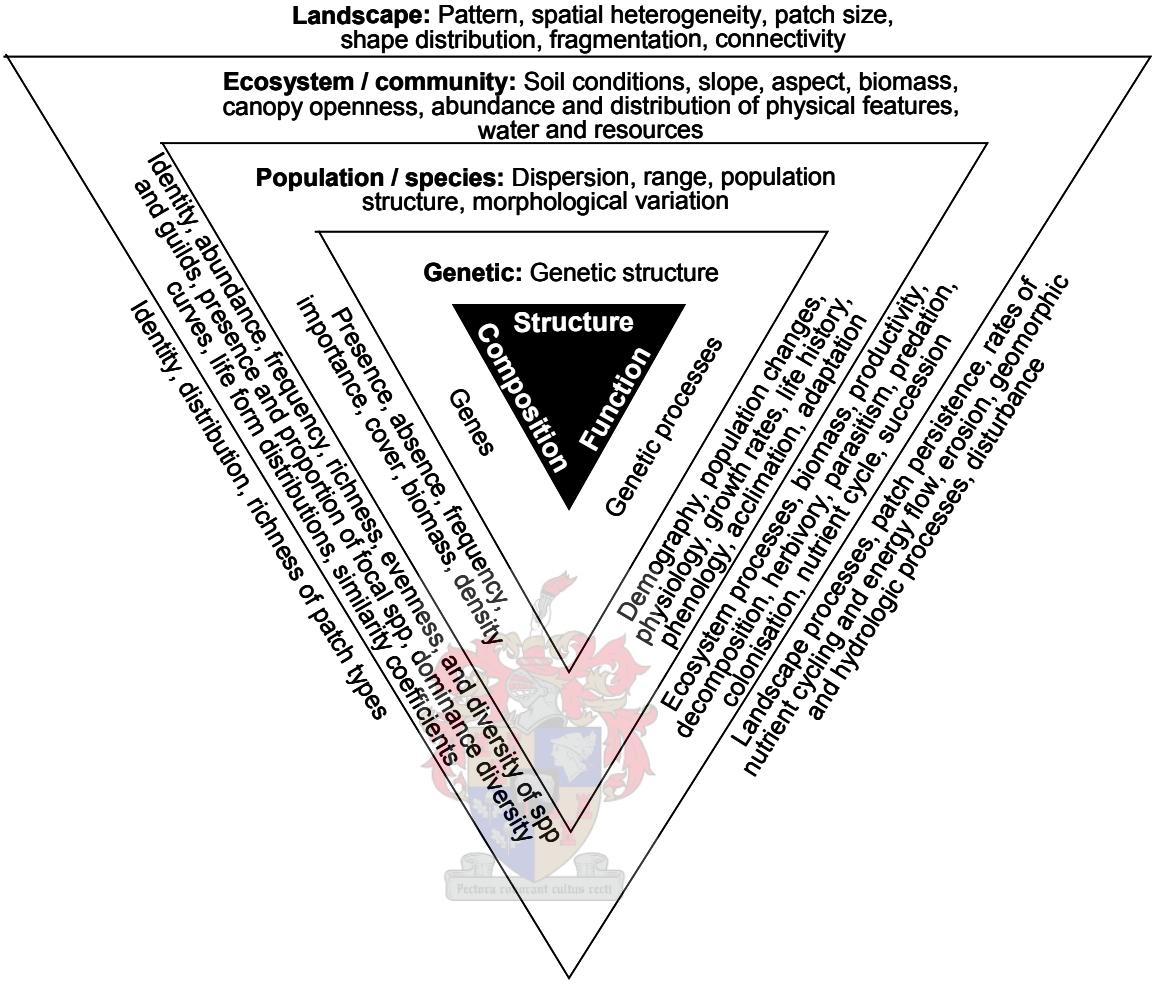

Figure 2.2: Biological indicators (compositional, structural and functional) at four levels of biological organisation (genetic; individual / population; community / ecosystem; and landscape) (after Noss 1990; Dale & Beyeler 2001).

Evaluation criteria most commonly monitored and compared relate to the composition, structure and pattern of the vegetation (Cooke & Johnson 2002; Holl & Cairns 2002). Compositional indicators (e.g. focal species, indicator species, keystone species and exotics) selected will vary from region to region and ecosystem to ecosystem. Structure attributes to consider could include landscape fragmentation and connectivity and patch / inter-patch size and distances. However, in the long-term nothing is more crucial to ecosystem health and integrity than natural processes or function (e.g. competition, predation, herbivory, weathering, decomposition, succession, etc.) (Andreasen *et al.* 2001). Recreating structure and composition without function does not constitute complete rehabilitation (Reay & Norton 1999).

Where the rehabilitation goal is to restore entire ecosystems both structural (i.e. the physical organisation or pattern of a system) and functional (i.e. the ecological and evolutionary processes) indicators should be measured (Holl & Cairns 2002). Aronson *et al*. (1993) list 18 vital structural and functional ecosystem attributes for evaluating degradation and rehabilitation experiments. Structural ecosystem attributes include: perennial and annual species richness, total plant cover, aboveground phytomass, beta diversity, life form spectrum, keystone species, microbial biomass and soil biota diversity. Functional ecosystem attributes include: biomass productivity, soil organic matter, maximum available soil water reserves, coefficient of rainfall efficiency, rain use efficiency, length of water availability period, nitrogen use efficiency, microsymbiont effectiveness, cycling indices.

It is frequently assumed that the rehabilitation of ecosystem structure will result in the rehabilitation of ecosystem function, but this is not always a valid assumption as structural and functional attributes of an ecosystem will not always develop at the same rate. Restoration goals may therefore be defined to some extent in terms of structural and functional indicators. For example, where the rehabilitation goal is to achieve a state similar to that prior to disturbance, rehabilitation will have been achieved once all the required species have been demonstrated to persist for a particular period of time. However, to achieve levels of absolute and relative abundance similar to pre-disturbance levels may require a longer period of time (Westman 1991). Monitoring soil is important because of its importance to the establishment and survival of plant and animal species. If efforts are made to ameliorate the soil conditions, soil indicators must be monitored (Holl & Cairns 2002).

"Indicator species" can also be used to assess ecosystem responses to environmental perturbations as well as rehabilitation success. Numerous species have been used in the past, for example fish, ants and butterflies (Andersen 1997; Holl & Cairns 2002). The basis for selecting an "indicator species" is to select a species that provides interpretable indications of changing environmental conditions (Cairns *et al*. 1993). "Indicator species" should have a well-known life history, they should be easily surveyed and identified, and should be sensitive to changing environmental conditions (Holl & Cairns 2002). Ant fauna has been used to assess rehabilitation success in Western Australian rehabilitated bauxite mines in the long-term and short-term. Ants make good indicators of ecological condition because they are diverse, abundant, functionally important and sensitive to disturbance (Andersen *et al*. 2002). Majer & Nichols (1998) have shown that the degree of ant recolonisation provides some indication of the degree of re-establishment of ecosystem functioning. Netshilaphala *et al.* (2005) found that ants could be potentially used as indicators of ecological recovery of mined land at Namakwa Sands and that a comprehensive sampling is not essential for revealing ant assemblage responses to land use.

The use of "indicator species" can be problematic for a number of reasons: (1) "indicator species" do not indicate the whole problem; (2) the extrapolation from a single species to an entire ecosystem cannot be done with confidence; (3) few species act similarly across a range of ecosystems and scales; (4) different indicators respond to change at different scales; and (5) "indicator species" may lead to Type I and Type II errors (Holl & Cairns 2002). It is for these reasons that monitoring multiple indicators is recommended (Noss 1990; Holl & Cairns 2002).

2.3.4 MONITORING AND REHABILITATION SUCCESS

Monitoring and assessing rehabilitation success is an important component of rehabilitation planning (Figure 2.1) and should be gauged against the goals and objectives defined for the rehabilitation programme (Ludwig *et al.* 2003). Monitoring rehabilitation success helps make informed decisions and will ultimately determine whether the rehabilitation goals or specific end points have been reached (Holl & Cairns 2002; Cummings *et al*. 2005). Comparisons between rehabilitation and reference sites are normally made after relatively short periods of time (e.g. 3 to 10 years). Even though rehabilitation sites are seldom similar to reference sites after this short period, such comparisons provide valuable information on the effectiveness of the rehabilitation methods used to achieve a composition, structure and function similar to the reference area (Chambers *et al.* 1994).

Rehabilitation success is not only important from an ecological viewpoint but also from a mining organisations' viewpoint, as in South Africa mine operators are responsible for rehabilitation until a closure certificate has been issued by the Department of Minerals & Energy. Monitoring will allow the authorities to make an informed assessment of whether or not the mine operator has fulfilled its rehabilitation obligations (Tomlinson 1984). Monitoring will not only help determine whether specific end points have been reached, but it will also determine if rehabilitation is not proceeding as predicted. If rehabilitation is not proceeding as predicted, corrective measures can be implemented, which may save money in the long run, because if problems are detected early, then corrective actions may be less costly. On the other hand, monitoring when the rehabilitation goals and objectives have been achieved is unnecessary and will lead to increased rehabilitation costs (Holl & Cairns 2002). Monitoring will also weigh up the success of a rehabilitation technique against the cost of implementation (Espelta *et al.* 2003). Therefore, knowing whether or not rehabilitation has been successful or is proceeding in the right direction has various cost implications and will ultimately determine if a closure certificate is issued.

Rehabilitation success will depend on whether or not the rehabilitated site and all its elements fulfil the function for which it was designed. However, it should be noted that rehabilitation might be perceived to be successful by the designer, but not from a future land user (Harris *et al*. 1996). Rehabilitation may fail due to:

1. Unrealistic objectives or standards of performance (Thom 2000);
2. Inadequate site surveys (Harris *et al*. 1996);
3. Poor planning at the various rehabilitation stages and vague or ambiguous objectives (Tomlinson 1984; Pastorok *et al.* 1997);
4. Poor definition of initial problem or inadequate understanding of the undesirable ecosystem characteristics (Pastorok *et al.* 1997);
5. Poor understanding of the ecosystem under consideration (e.g. processes, nutritional requirements, unpredicted species interaction or natural disturbances, undesirable species, successional processes, etc.) (Pastorok *et al.* 1997);

6. Anthropogenic disturbances (Pastorok *et al.* 1997) (e.g. grazing rehabilitated sites to soon or too heavily after rehabilitation);
7. Vegetation regression, which can result from a number of causes (e.g. decline in soil pH, lack of nutrient cycling, overgrazing, decline in legumes, loss of soil structure, deficient drainage, etc.) (Harris *et al*. 1996); and
8. Insufficient time spent on aftercare management and maintenance (Tomlinson 1984).

When designing a monitoring programme to assess the success of rehabilitation a number of criteria should be considered with regards to sample design:

1. Selecting a reference site / reference data

Depending on the rehabilitation goals, reference sites or reference data are required against which to compare rehabilitation sites. These sites or data define the desired or "target" state and are used in determining whether rehabilitation has been successful (Tomlinson 1984; Andreasen *et al.* 2001). Reference sites or data should represent the desired habitat quality and functions (Pastorok *et al.* 1997) and should be based on a similar landform, soil, biotic and climatic conditions as the rehabilitation site (Hobbs & Norton 1996). The reference data can be obtained from historic data or baseline data taken prior to disturbance. However, a number of problems exist with all these alternatives (Holl & Cairns 2002). In many cases, historic data prior to human interference in not available as human disturbance extends back many thousands of years and ecosystem conditions have also changed over this period. Therefore, the selection of a historical reference state is subjective (Westman 1991; Holl & Cairns 2002). Since sites vary in structure and function with climatic variation and disturbance, the variance from mean condition is difficult to define due to the lack of quantitative data on historical condition. Therefore, the historic condition of a habitat is frequently only possible in general terms (Westman 1991). In terms of the collection of baseline data, often extensive damage has already occurred to the area before the reference data can be collected (Holl & Cairns 2002).

In terms of selecting a nearby reference site there are three main problems. Firstly, it is impossible to find an identical system, as sites may differ in terms of soil type, land use, slope, aspect, etc. A way to counter this is to select a range of reference sites. Secondly, as is the case with historical and baseline data, one has to decide with which successional stage to compare the rehabilitated site. It may be necessary to select sites with a range of successional stages. Thirdly, biotic communities are naturally spatially and temporally distributed. It is therefore necessary to sample reference sites across a sufficient spatial scale to incorporate the natural variation within systems (Holl & Cairns 2002).

2. Spatial scale of monitoring

Ideally a restoration project should be monitored over large areas with the minimum area being the scale at which ecosystem processes occur (Westman 1991; SER 2002), but due to budget and personnel constraints the opposite is usually true. A trade-off usually occurs between frequently sampling a number

of indicators at a few points; infrequently sampling many indicators at relatively few locations; and infrequently sampling a few indicators at many locations (Holl & Cairns 2002)

How best to allocate monitoring efforts over time and space will depend on the goals and objectives as well as the ecosystem being monitored. Ecosystem monitoring must often be conducted at large spatial scales as often the project actions affect ecosystems over large areas or the indicators being monitored need to be monitored over a large area (e.g. nutrients in surface water may require the monitoring of the entire watershed). Remote sensing can be used to monitor ecosystems at large spatial scales (Westman 1991; Holl & Cairns 2002). If monitoring takes place at too small a scale, it must be recognised that some processes may be compromised by habitat fragmentation (e.g. animal migration, seed arrival, etc.).

3. Temporal scale

Timing is important for two aspects, *viz*. for how long should the rehabilitated site be monitored and at what frequency should the indicators be monitored (Holl & Cairns 2002). A system should be monitored long enough until reasonable assurances are provided that the system has either met its performance criteria or it will not likely meet the criteria (Thom 2000). The frequency will depend on ecosystem type and the indicator being monitored. For example, vegetation cover may only need to be monitored annually, whereas to identify all species the monitoring may have to be more frequent (e.g. seasonal). In selecting the frequency of monitoring, the cost and labour should be considered. Even so, it is important to consider the initial goals of rehabilitation to determine whether sufficient data will be collected in order to assess rehabilitation success. It may be possible to start off monitoring more frequently and then reduce monitoring with time (Holl & Cairns 2002). A single survey (or “snapshot”) does not provide an accurate assessment of whether or not the rehabilitation objectives have been met (Tomlinson 1984). Areas may need to be monitored over a long time period just to determine the indicators that may be useful in determining whether or not rehabilitation has been successful (Lubke & Avis 1998).

4. Action thresholds

One of the purposes of monitoring is to provide an early warning system that recovery is not proceeding as predicted. Action thresholds refer to the stage or level where corrective action should be taken because rehabilitation is not proceeding as predicted. It is important to decide during the planning stage as to how much variability is acceptable for a particular indicators or the degree of closeness to the reference site. Westman (1991) suggests the mean percentage similarity among sites within a regional baseline sample plus or minus some margin of error (e.g. 5 to 10%). If sample size is increased the variation is reduced, which increases ones ability of detecting an effect. The number of samples needed depends on the amount of natural variance and the predetermined level of difference from the reference condition. Some preliminary or baseline sampling is a good way to assess the amount of natural variability (Holl & Cairns 2002). Others have suggested an appropriate criterion of success may be ±1 standard deviation of the natural condition (Hackney 2000).

If action threshold values are set too low it can result in false positive readings (Type I error) and if the corrective action value is set too high it can result in false negative readings (Type II error). Taking action when there is no need or not taking action when it is necessary leads to increased costs. If it is identified early on that rehabilitation is not going to reach the rehabilitation goal or it is acknowledged that there are problems with the rehabilitation plan and one takes corrective actions (adaptive management), the process of rehabilitation is likely to be more cost effective in the long-run compared to ignoring the problem (Holl & Cairns 2002).

5. Data collection / sampling methods

Once the objectives and the indicators for monitoring have been selected the next step is to design and implement a sampling programme. There are a number of sampling or monitoring techniques that can be used to monitor rehabilitation success. The techniques used will depend on what ecosystem attribute (compositional, structural or functional) and at what organisational level is being monitored (Noss 1990) (Figure 2.2).

If monitoring over large areas at a *landscape level*, achievement of performance can be monitored through aerial photography and satellite imagery (remote sensing) and the data can be displayed using Geographical Information System (GIS) (Noss 1990; Westman 1991). Monitoring the position of ecotones at various spatial scales can be a useful tool to track rehabilitation success or regression. However, monitoring landscape composition will require more intensive ground-truthing than monitoring structure, as the dominant species composition must be identified. The monitoring of landscape function can be monitored through disturbance-recovery processes and rates of energy flows. Other monitoring tools include: time series analysis; spatial statistics; and mathematical indices (of pattern, heterogeneity, connectivity, layering, edge, morphology, diversity, autocorrelation and fractal dimension) (Noss 1990).

Techniques for monitoring at a *community-ecosystem level* of organisation must rely more upon ground-level surveys and measurement than on remote sensing, as the community-ecosystem level of organisation is relatively homogeneous when viewed at the scale of a conventional aerial photograph. Tools and techniques for monitoring at a community-ecosystem level include: aerial photography; remote sensing; time series analysis; physical habitat measures and resource inventories; habitat suitability indices; mathematical indices (of diversity, heterogeneity, biotic integration and layering dispersion); and censuses, observations and inventories (Noss 1990).

Monitoring at a *population–species level* is often directed at habitat variables assumed to be important to the species, rather than the population itself. However, the presence of a suitable habitat does not guarantee that the species of interest are present. Therefore, monitoring both habitat and population variables should be undertaken in most cases. Tools and techniques for monitoring at a population–species level include: censuses, observations and inventories; remote sensing; habitat suitability index; species-habitat modelling; and population viability analysis (Noss 1990).

It is advisable to use standard methods of sampling, particularly when measurements are made by people with less formal training. However, the disadvantage of standard methods is their extreme rigidity. The use of non-standard techniques or new and improved techniques may improve accuracy and time efficiency over older techniques (Holl & Cairns 2002).

6. Data analysis

The data should be analysed and synthesised in a way that is useful to decision or policy-makers (Noss 1990; Holl & Cairns 2002). Quantitative approaches to determine if objectives have been met include performance curves, resemblance functions, modelling and resilience indices (Westman 1991; Pastorok *et al.* 1997).

Indices are needed to determine the development and sustainability of rehabilitated ecosystems and a knowledge of those indicators by which ecosystem development and sustainability can be measured is important (Bell 2001). Indices of integrity can function both in assessment (i.e. "snapshot") and in monitoring (i.e. periodic measurements over time) (Andreasen *et al.* 2001). There are a few commonly used criteria for determining ecosystem rehabilitation success (e.g. plant species diversity, density and succession; soil development; nutrient cycling; faunal colonization; and habitat development) and many mines attempt to measure as many components of the developing ecosystem as possible in order to make an assessment, which can be very costly (Bell 2001).

Andreasen *et al.* (2001) defined a rigorous set of criteria for the development of a terrestrial index that could summarise the condition of ecosystems so that changes can be tracked over time and on which decision could be made. However, because ecosystems are inherently complex, no single indicator can be expected to measure ecological integrity of an ecosystem, but rather a suite of indicators should be selected. An index should be flexible so as to take account of the dynamic and changing nature of ecosystems. A general index of ecological composition, structure and function that could be monitored at large scales would be valuable. However, such an index would be so general that it would not be able to address specific problems. There is no single way of reducing the complexity of terrestrial ecosystems and an index of integrity is best thought of as background information (Andreasen *et al.* 2001).

With integrated indicators of site performance (e.g. above-ground biomass), a comparison of trends over time in relation to regional baseline levels is an adequate measure of performance (i.e. trajectory analysis), bearing in mind the need to average performance over a suitable period to smooth temporal variations (Westman 1991; SER 2002). However, with other indicators (e.g. composition and relative abundance) data from individual species on a site can be integrated by resemblance functions of various types. Resemblance functions provide a quantitative approach for comparing current site conditions to the baseline goal. These functions (e.g. "coefficient of community" or "indices of percentage similarity") are indices that compare the composition of pairs of sites (Westman 1991).

Depending on the rehabilitation goal and what one wants to achieve, the analysis of collected data may be different. “Coefficient of community” indices (e.g. Jaccard, Czekanowski and Sorensen’s indices) use only qualitative (presence/absence) data and are preferred when gross differences in composition are of interest. “Indices of percentage similarity” indices (e.g. Euclidian and Czekanowski indices) use quantitative data and are preferred when subtle changes in composition and abundance over time are of interest. Weighting of particular species can refine the emphasis of each index. For example, if the rehabilitation goal were to rehabilitate habitat for an endangered species, the Czekanowski percentage similarity index would be most appropriate, by weighting the species of interest. If the restoration goal were to create a vegetation cover similar in composition and relative abundance to the reference site, the Euclidean percentage similarity index would be more preferable. However, if the rehabilitation goal were to restore levels of biodiversity (which emphasises composition rather than abundance) a “coefficient of community” would be more appropriate (Westman 1991).

Since achieving rehabilitation goals may take a long time (e.g. 100 years) it is necessary to determine whether rehabilitation is proceeding as predicted (i.e. in the direction of the rehabilitation goal) and to predict on the basis of current performance how long it will take to meet the rehabilitation goal (Westman 1991). Westman (1991) suggests two approaches in predicting long-term performance:

1. Markov modelling: This approach uses remote sensing or intensive field survey to track rates of recovery across the site. A Markov matrix is constructed to express probabilities of transition from one recovery class to the next. One limitation is that this approach assumes that future rates of recovery will be the same as those of the past and since succession rates are not linear this is not an ideal assumption.
2. Resilience indices: The use of standard measures of resilience could build up an understanding of ecosystem recovery behaviour that would enhance one’s ability to interpret rehabilitation performance. Ecologists have recognised five distinct components of resilience, namely elasticity, amplitude, hysteresis, malleability and damping.

Out of a need for commonly agreed indicators and a quick but rigorous method for monitoring ecosystem success Tongway *et al.* (1997) developed and evaluated a monitoring procedure termed “Ecosystem Function Analysis (EFA)”. They showed that the EFA evaluation procedure could potentially be used for a wide range of mine types (including bauxite, mineral sands, coal, gold, uranium, nickel and iron ore) across a wide range of climatic conditions. EFA essentially uses simple ecological indicators that relate spatial patterns to complex ecological processes (Dale & Beyeler 2001; Ludwig *et al.* 2003). The EFA approach is comprised of three modules used together to make the assessment, namely:

1. Landscape Function: Landscape Function involves the calculation of three major soil habitat quality indices based on source / sink patches and soil surface condition and texture within these patches;
2. Vegetation composition and dynamics: using species composition, species similarity to reference sites, presence of framework species (i.e. those species that provide shade and shelter and facilitate nutrient cycling) and physical development of target species; and

3. Habitat complexity: Habitat complexity is based on canopy cover, shrub cover, amount of litter, fallen logs and rocks and free available water.

Tongway & Hindley (2003) tested the EFA indices and found that the indicators correlated well with measured properties in the soil surface and had a broad application across a wide range of climatic conditions. Most monitoring systems focus on showing that the proposed indicator suitably reflects the system under examination. EFA not only tracks the system status over time but also specifies a target region for EFA index values for self-sustaining ecosystems and assesses the rate at which the system is approaching the target region. In addition, EFA uses simple, rapidly assessed, visual indicators of surface processes mediated by both physical and biological components of the system, so repeated assessment is more cost-effective. It is this focus on ecosystem processes that allows it to be used across a wide range of landscapes and climates (Bell 2001; Tongway & Hindley 2003).

2.3.5 ADAPTIVE MANAGEMENT

Adaptive management may be implemented whether rehabilitation is deemed to have been successful or not. When the desired trajectory of a rehabilitated ecosystem is achieved, it may no longer require external assistance to ensure its future health and integrity. However, rehabilitated ecosystems often require continued management to counteract the invasion of opportunist species, climate change and other unforeseeable events (SER 2002). If rehabilitation is not proceeding as predicted, corrective measures can be implemented (Holl & Cairns 2002). Monitoring will not necessarily explain the underlying causes of the result and it may be necessary to determine the reason (e.g. lack of nutrients, water stress, competition, wind, etc.) through experimentation. However, monitoring will definitely provide some insight into the factors influencing rehabilitation (Holl & Cairns 2002). Once the underlying cause(s) of rehabilitation failure has been identified it is important to consider whether the cause(s) can be rectified so that the rehabilitation goal can be achieved or whether an alternative rehabilitation goal is more appropriate (Harris *et al*. 1996).

Due to uncertainties as a result of incomplete knowledge about ecosystems and / or the dynamic and changing nature of ecosystems, the principles of adaptive management are particularly useful in both the planning and management of rehabilitation projects to ensure a greater probability of success (Thom 1997). Adaptive management is essentially the incorporation of “experimental” or monitored results in a rehabilitation programme to rectify problems (i.e. learning from mistakes, failures and successes) (Pastorok *et al*. 1997; Thom 1997; Cummings *et al.* 2005). For example, if rehabilitation appears to be failing for either biophysical or social reasons actions should be implemented to improve the performance of a rehabilitated system (Thom 1997; 2000). Adaptive management may also include the redefining of more detailed success criteria, as an understanding of ecosystem processes is improved during the monitoring process (Hackney 2000). Adaptive management may involve an iteration of any of the planning steps shown in Figure 2.1.

Adaptive management requires monitoring of the system using selected indicators towards specified goals and objectives in order to consider whether or not to make selective manipulations of a rehabilitation technique to improve the outcome relative to the stated goals and objectives (Pastorok *et al*. 1997; Thom 1997; 2000). Manipulations may include translocation of indigenous species, seeding, (de Villiers 2000), fertiliser application, mulching, etc. Mulching enhances soil fertility and structure, and may restrict the establishment of competitive species (Cummings *et al.* 2005). A contingency plan represents a set of alternative actions if monitoring indicates that rehabilitation is not proceeding as anticipated. The contingency plan should be based on the rehabilitation conceptual model (i.e. a list of the controlling factors and the desired structure and function) established for the project (Thom 2000). The contingency actions should ideally be specified in the planning stages when the goals and objectives are established. Monitoring should continue after making selective manipulations in order to evaluate and assess the effectiveness of the manipulations. If it becomes clear that the system cannot predictably reach the original goals, even with the manipulations, modification of the rehabilitation goals may be required (Thom 1997).

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CHAPTER 3

DESCRIPTION OF STUDY AREA

3.1 LOCATION

Anglo American Corporation's Namakwa Sands heavy minerals mining operation is located in the vicinity of Brand-se-Baai (31° 15.889' S 17° 56.003' E), approximately 385 km north of Cape Town. Namakwa Sands has been mining the heavy mineral deposit, which is rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994. Namakwa Sands is currently mining an area of approximately 4 700 ha. This mining area is divided into two sectors, Graauwduinen East or East Mine (approximately 3 370 ha) and Graauwduinen West or West Mine (approximately 1 400 ha) (EEU 1990) (Figure 3.1). This study was located within the East Mine.

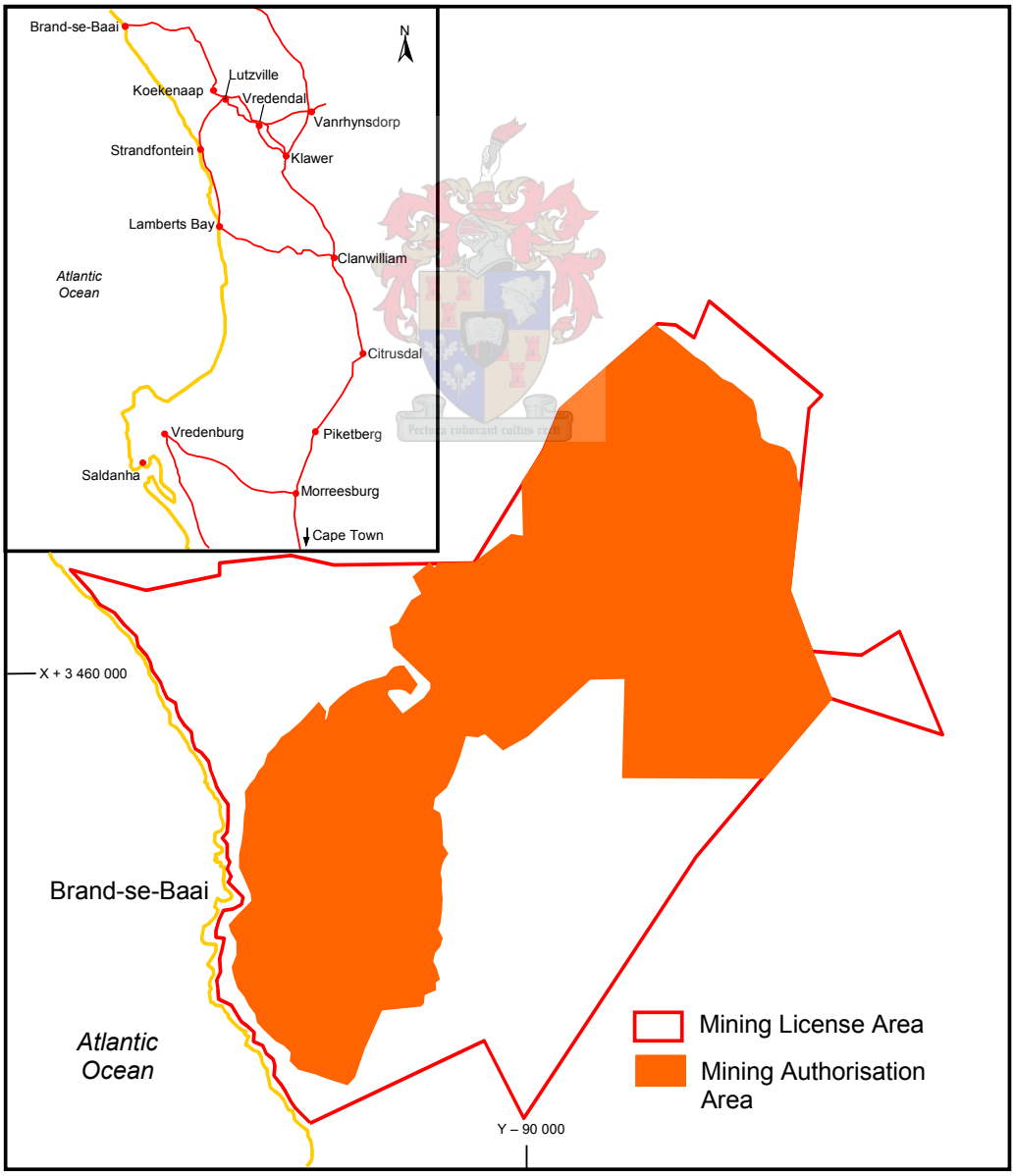

Figure 3.1: Namakwa Sands is located in the vicinity of Brand-se-Baai, South Africa.

3.2 CLIMATE

Climate is one of the most important factors affecting the success of a rehabilitation plan and having a detailed knowledge and understanding of the local climatic conditions, both macro and micro, is an essential element to the success of the rehabilitation plan (Grindley & Barbour 1990).

3.2.1 PRECIPITATION

Namaqualand is characterised by hot, dry summers and sporadic winter rainfall, falling mainly in the months from May to July (EEU 1990; le Roux & Schelpe 1997). The germination response of seeds of desert plants to sporadic rainfall (Higgins *et al.* 2000) results in stochastic (i.e. non-deterministic or variable) and episodic recruitment events (Milton *et al.* 1999).

Annual rainfall increases from the north to the south, with an average of 160 mm per annum occurring in the mining area (Mahood 2003a). Interestingly, this low rainfall is highly predictable and prolonged droughts are very rare (Cowling *et al.* 1999). The mining area received an annual rainfall between 82.9 mm and 207 mm from 2001 to 2005 with an annual average of 141.4 mm (Table 3.1). Compared to the average of 160 mm per annum (Mahood 2003a), it would appear that the study area has experienced a drought over the last few years with an annual rainfall of 100.5 mm, 137.5 mm and 82.9 mm being recorded in 2003, 2004 and 2005, respectively. Over the last five years the maximum average monthly rainfall is 27.1 mm in winter (August) and the minimum monthly average rainfall is 1.5 mm in summer (December). The highest rainfall in any given month between 2001 and 2005 was 53 mm in July 2001.

Rainfall is augmented by heavy dew falls and sea fogs (approximately 100 days per annum in the area of the mine site). Fog is characteristic of the Namaqualand coast and occurs throughout the year (EEU 1990). Rainfall, sea fog and dew fall measured at Brand-se-Baai between March 1993 and February 1997 amounted to a cumulative average annual precipitation of 282 mm per annum measured (de Villiers *et al.* 1999).

3.2.2 TEMPERATURE

The study area has relatively moderate temperatures through the year with an average annual temperature in the mining area of 15.9 ^{o}C from 2003 to 2004. Between September 2002 and August 2005, February had the highest average temperature (18 ^{o}C) and August had the lowest average temperature (12.8 ^{o}C) (Table 3.2). The highest average monthly maximum temperature was 37 ºC in March (summer) and the lowest average monthly minimum temperature was 4.7 ºC in July (winter) (CSIR 2003; 2004a; 2004b; 2005a; 2005b).

Table 3.1: Monthly and annual rainfall (mm) in the study area from 2001 to 2005 (source: Namakwa Sands).

Month	Rainfall (mm)					
	2001	2002	2003	2004	2005	Average monthly
January	2.0	8.0	7.0	30.5	1.5	9.8
February	12.0	25.0	1.0	2.5	1.0	8.3
March	19.0	5.0	13.0	4.0	1.5	8.5
April	31.0	7.0	8.0	8.0	10.0	12.8
May	18.0	35.0	8.0	2.5	13.3	15.4
June	14.0	19.0	0.0	14.0	7.0	10.8
July	53.0	48.0	5.5	11.5	4.0	24.4
August	23.0	35.0	43.0	5.0	29.6	27.1
September	0.0	14.0	8.0	13.0	10.0	9
October	0.0	7.0	6.0	42.0	0.0	11
November	4.0	2.0	0.0	3.0	5.0	2.8
December	3.0	2.0	1.0	1.5	0.0	1.5
Total (Annual)	179	207	100.5	137.5	82.9	Average annual 141.4

Table 3.2: Average monthly temperature (ºC) in the study area between September 2002 and August 2005 (CSIR 2003; 2004a; 2004b; 2005a; 2005b).

Month	Average temperature (ºC)						
	2002	2003	2004	2005	Average monthly	Average monthly maximum	Average monthly minimum
January	-	17.4	17.3	18.9	17.9	32.6	10.6
February	-	18.4	17.6	18.1	18.0	33.1	10.4
March	-	17.8	16.4	18.1	17.4	37.0	9.9
April	-	17.1	17.2	16.3	16.9	35.4	7.6
May	-	15.8	15.3	14.8	15.3	32.3	7.0
June	-	14.5	14.4	13.1	14.0	31.1	5.4
July	-	13.6	14.3	14.8	14.2	28.0	4.7
August	-	12.2	13.5	12.7	12.8	29.7	5.6
September	14.8	14.0	14.5	-	14.4	33.7	5.9
October	14.9	16.7	14.9	-	15.5	35.0	7.1
November	15.7	16.8	16.8	-	16.4	33.7	7.7
December	17.8	16.8	18.3	-	17.6	29.4	10.6

3.2.3 WIND

Over the period from 2003 to 2005, the dominant winds during the spring and summer (September to March) were from the south-southeast and south but winds from the north-northwest and northwest were not uncommon. The dominant winds during the winter months (June to August) were from the north-northwest, northeast, northwest and east-northeast but winds from the south and southeast were not uncommon (CSIR 2003; 2004a; 2004b; 2005a; 2005b). Easterly berg winds blow from the interior, bringing hot, dry conditions to the coast (Washington 1990).

The average annual wind speed in the mining area from 2003 to 2004 was 4.1 m/s. The highest average monthly wind speed was 5.8 m/s in winter (July) and the lowest average monthly wind speed was 3.7 m/s in summer (February) between September 2002 and August 2005 (Table 3.3). July also had the highest average wind speeds exceeded at 5% and 50% of 10.7 m/s and 5.5 m/s, respectively. Wind speeds reach a peak velocity between 16h00 and sunset and are at a minimum at 06h00 in summer and 10h00 in winter (Washington 1990).

Table 3.3: Average monthly wind speed (m/s) in the study area between September 2002 and August 2005 (CSIR 2003; 2004a; 2004b; 2005a; 2005b).

Month	Wind velocity (m/s)						
	2002	2003	2004	2005	Average monthly	Average wind speed exceeded	
						50%	5%
January	-	3.7	4.0	4.4	4.0	3.6	8.8
February	-	3.6	3.3	4.2	3.7	3.2	8.5
March	-	3.5	4.0	4.2	3.9	3.2	8.9
April	-	3.3	3.7	4.7	3.9	3.3	9.0
May	-	4.0	3.4	4.5	4.0	3.4	8.6
June	-	3.8	3.5	4.6	4.0	3.5	8.4
July	-	4.3	5.0	8.0	5.8	5.5	10.7
August	-	4.6	4.5	4.7	4.6	4.2	9.2
September	4.9	4.7	4.7	-	4.8	4.3	9.8
October	4.7	4.3	4.9	-	4.6	4.2	9.2
November	4.9	4.3	3.9	-	4.4	3.8	9.4
December	4.2	4.5	4.0	-	4.2	3.9	8.8

3.2.4 RELATIVE HUMIDITY

The average annual relative humidity in the mining area was 76.1% from 2003 to 2004. The highest average monthly relative humidity between September 2002 and August 2005 was 82.7% in summer (February) and the lowest monthly average relative humidity was 60.7% in winter (July) (Table 3.4).

Table 3.4: Average monthly and annual temperature (ºC) in the study area between September 2002 and August 2005 (CSIR 2003; 2004a; 2004b; 2005a; 2005b).

Month	Relative humidity (%)				
	2002	2003	2004	2005	Average monthly
January	-	83	82	82	82.3
February	-	84	83	81	82.7
March	-	81	76	77	78.0
April	-	80	71	76	75.7
May	-	70	77	80	75.7
June	-	57	no data	78	67.5
July	-	62	65	55	60.7
August	-	73	77	79	76.3
September	80	77	76	-	77.7
October	78	73	81	-	77.3
November	72	74	85	-	77.0
December	84	77	84	-	81.7

3.3 GEOLOGY AND SOILS

The mining area is situated in a geomorphological subdivision of the Namib Desert, which is referred to as the Namaqualand Sand Namib. A thick overburden of marine and aeolian sediments overlie older basement rocks of the Namaqualand granite-gneiss suite and metamorphosed Vanrhynsdorp Group rocks (EEU 1990).

The mining area has been split into two distinct ore bodies, i.e. the deeper and more complex west orebody (Graauwduinen West) and the relatively simpler east orebody (Graauwduinen East). The depth to which the mineral deposits are mined varies from Graauwduinen East (East Mine) to Graauwduinen West (Wine Mine). At the West Mine excavation is to depths of between 2 and 35 meters (Namakwa Sands 2002). A hard layer of pedocrete (cemented silica, iron oxide and calcium carbonate), known as the Dorbank layer lies between 1 and 5 m below the surface. This layer is between 1 and 6 m thick and is excavated to gain access to lower lying mineral deposits (EEU 1990). At the East Mine the bottom of the ore body is the contact with a hard Dorbank layer and excavation is therefore very shallow (between 1 and 3 m) (Namakwa Sands 2002). However, this section of the mine covers a greater area than that at the West Mine (EEU 1990).

A generalised cross-section through the orebody is presented in Figure 3.2 and is discussed below (EEU 1990; Namakwa Sands 2001):

1. *Bedrock:* The bottom layer of the profile is Pre-Cambrian bedrock.
2. *Other sands:* This is a collective term used for older, barren sands underlying the west orebody. These sands are generally fine-grained, yellow-white. The heavy mineral content in this layer is low and the sands may represent an old barren dunefield. The thickness of layer is variable but generally increases in thickness to the east, reaching a thickness of over 60 m.
3. *Eastern Strandline:* This layer often has a pebble bed at it base, which is overlain by fine grained yellowish-grey sand. This layer has an average total heavy mineral content of 18%. The thickness of this layer is estimated to be between 2 and 6 m.
4. *Orange Feldspathic Sand:* This layer is a fine to medium grained, dark yellowish-brown to greenish, clayey layer, which is comprised mainly of quartz sand with a significant proportion of feldspar and other silicates. This layer forms the bulk of the ore reserves at the west orebody.
5. *Western Strandline:* This layer lies on the bedrock at an elevation ranging between 20 to 25 mamsl. This layer is generally fine to medium-grained with a heavy mineral content of 28%.
6. *Dorbank layer:* A hard layer of pedocrete (cemented silica, iron oxide and calcium carbonate), known as the Dorbank layer lies between 1 and 5 m below the surface. This layer is between 1 and 6 m thick.
7. *Red Aeolian Sand:* This layer is a dark reddish-brown, medium grained sand, which covers the whole of the area inland of the rocky shore and younger dunes. This sand is derived from the underlying Orange feldspathic Sand where iron oxides have been weathered from the ferro-magnesium silicates and / or ilmenite to give it its distinctive colour. The east orebody consists mainly of Red Aeolian Sand.

Soil investigations in the study area have shown that there is variation in the chemical, physical and morphological characteristics of the soil under natural conditions (Lanz 2003; Prinsloo 2005). Prinsloo (2005) found that the soils in the mining area could be classified in seven soil forms (including Bloemdal, Clovelly, Garies, Namib, Oudstoorn, Pinedene and Tukulu) and eleven soil horizons (six of which are considered transitional in character). Pedological features such as surface water repellency, permeable subsurface apedal horizons, subsurface impediments such as cemented (calcrete or dorbank) hardpans and significantly more clayey (cutanic, luvic) hozions have been identified in the study area (Prinsloo 2005).

Soils tend to be saline and alkaline (with a pH exceeding 8) due to the deposition of salts at the soil surface through wind-blown salt spray (EEU 1990; Lanz 2003; Blignaut & Milton 2005). Soils closer to the coast generally have a higher pH level compared to soils further inland (Lanz 2003). Textural properties vary significantly between soil horizons ranging from 0.1 to 20.9% silt plus clay content. Most of the surface and middle horizons in the soil profiles of the study area are single grained and weakly structured or structureless. However, firm more clayey subsurface horizons above the dorbank show some signs of structural development (Prinsloo 2005).

Water repellency and microtopographic mounds cause infiltration fingering. The permeable subsurface apedal horizons allow for deep percolation of water while subsurface impediments such as cemented (calcrete or dorbank) hardpans and significantly more clayey (cutanic, luvic) horizons result in the accumulation of groundwater. These cemented hardpan or clayey horizons act as impermeable barriers and can result in the formation of perched aquifers (Prinsloo 2005). Prinsloo (2005) speculates that the water repellent surface horizon could reduce the upward movement of capillary water, which may be an important mechanism of natural water conservation in an arid environment. He also speculates that the nocturnal cooling of the soil surface and subsequent subsurface dew formation could be an indirect method for the utilisation of deeper groundwater by the shallow rooted vegetation (Prinsloo 2005).

3.4 SURFACE AND GROUNDWATER

3.4.1 SURFACE WATER

The area is extremely dry with no visible surface water within the study area. The catchments of the Groot Goerap River and Salt River, which flow episodically (i.e. at regular intervals), are the only drainage systems near the mining area (EEU 1990).

3.4.2 GROUNDWATER

Groundwater flow in the areas of the mine closely follows the topography and generally flows in a westerly direction toward the sea (Figure 3.3). Monitored groundwater levels at the mine show little seasonal or long-term fluctuation. Borehole yields are reportedly low, thereby limiting groundwater potential use for mining purposes. Groundwater quality is naturally poor, limiting its suitability for use as a domestic water supply. Groundwater within the mining area ranges between 10 metres above mean sea level (mamsl) closest to the coast and 70 mamsl further inland (Figure 3.3) (Parsons and Associates Specialist Groundwater Consultants 2003).

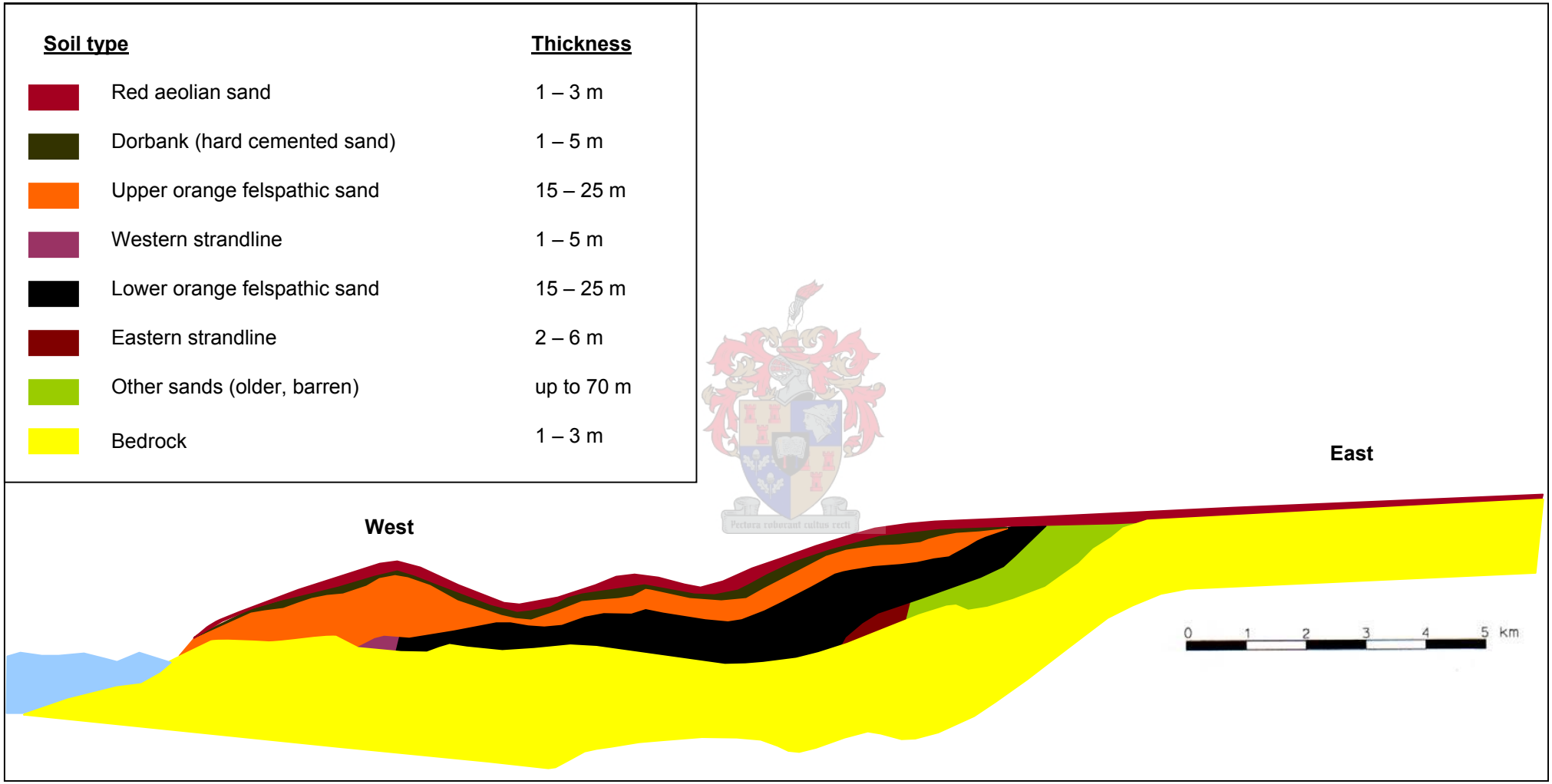

Figure 3.2: Geology and soil profile of the West and East mining areas (after Namakwa Sands 2001).

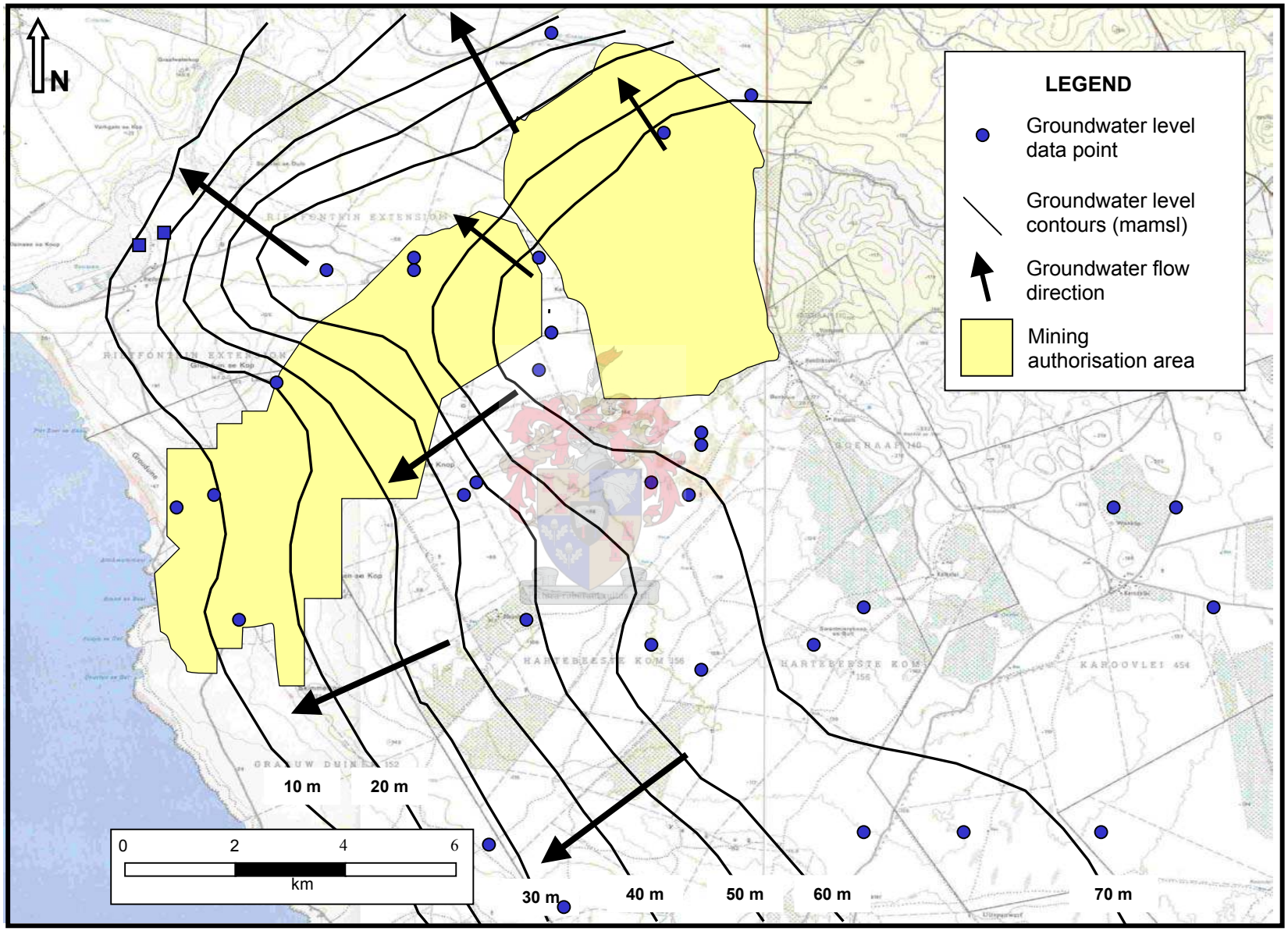

Figure 3.3: Groundwater contour map of the mine showing the depth and direction of groundwater flow (after Parsons and Associates Specialist Groundwater Consultants 2003).

3.5 VEGETATION

According to Acocks (1988) the vegetation in the study area is classified as Strandveld proper (Veld Type 34b) with the Namaqualand Coastal Belt Succulent Karoo (Veld Type 31a) in the north-eastern part of the study area (de Villiers *et al.* 1999). The Strandveld proper is an open, semi-arid succulent scrub veld type, forming an intermediate between Coastal Fynbos and the Succulent Karoo. The Namaqualand Coastal Belt Succulent Karoo is dominated by members of the Mesembryanthemaceae. A feature of these two vegetation types is the presence of "heuweltjies" or termite termitaria (Acocks 1988). Low and Rebelo (1996) classified the vegetation in the study area as consisting of Strandveld Succulent Karoo and Lowland Succulent Karoo, both of which are classified under the Succulent Karoo Biome. The Strandveld Succulent Karoo is associated with areas of calcareous sand confined to the sandy coastal plains and contains many drought deciduous and succulent species. The smaller patches of Lowland Succulent Karoo are characterised by a sparse cover of dwarf succulent-leafed shrubs that is sensitive to disturbance (Low and Rebelo 1996). More recently Mucina *et al.* (2005) classified the vegetation in the study area as Namaqualand Strandveld and Namaqualand Sand Fynbos, both of which are considered to be "least threatened".

Boucher and Le Roux (1989) undertook a preliminary vegetation pre-mining vegetation survey and classified the Strandveld vegetation of the mine site into three variants according to vegetation height (i.e. tall, medium and short Strandveld). These three generalised categories were based on a combination of vegetation structure and floristic content. The vegetation varies in height according to the depth of the sand with the shortest vegetation growing on exposed calcrete and coastal rocks and the tallest vegetation found growing in areas where deep calcareous sand occurs. The Tall Strandveld occurs on relatively deep calcareous sand, with a canopy cover of 60 – 75%, under a light grazing regime. The Medium Strandveld is characterised by plants that are in the region of 50 cm tall and has a projected canopy cover of perennial species of between 50 and 60%. The Short Strandveld varies in average height from 10 – 35 cm. This community occurs in shallow soils where there is very little storage of moisture. The projected vegetation canopy cover of perennial species is usually less than 50%.

De Villiers *et al.* (1999) undertook a pre-mining benchmark survey at Namakwa Sands for rehabilitation purposes. They identified six vegetation communities, some of which included several variants (Figure 3.4).

3.5.1 VEGETATION COMMUNITIES

De Villiers *et al*. (1999) identified 230 plant species within the six plant communities. Certain species had narrow ecological amplitudes and were correlated with particular environmental factors and as a result were restricted to a particular community. They speculated that these species would be most difficult to re-establish after mining and thus did not recommended these species for use in the initial revegetation programme. Other species were adapted to varying environmental conditions and were found throughout

the area. Many of these species were perennials with high cover-abundance values and are typical of Strandveld vegetation as a whole. They speculated that revegetation using these species would help to stabilise the mined sand during the windy, dry and hot summer months and should largely restore the former appearance and structure of the vegetation. They also speculated that the usefulness of annual species in the rehabilitation programme is restricted to the wet and cool winter months.

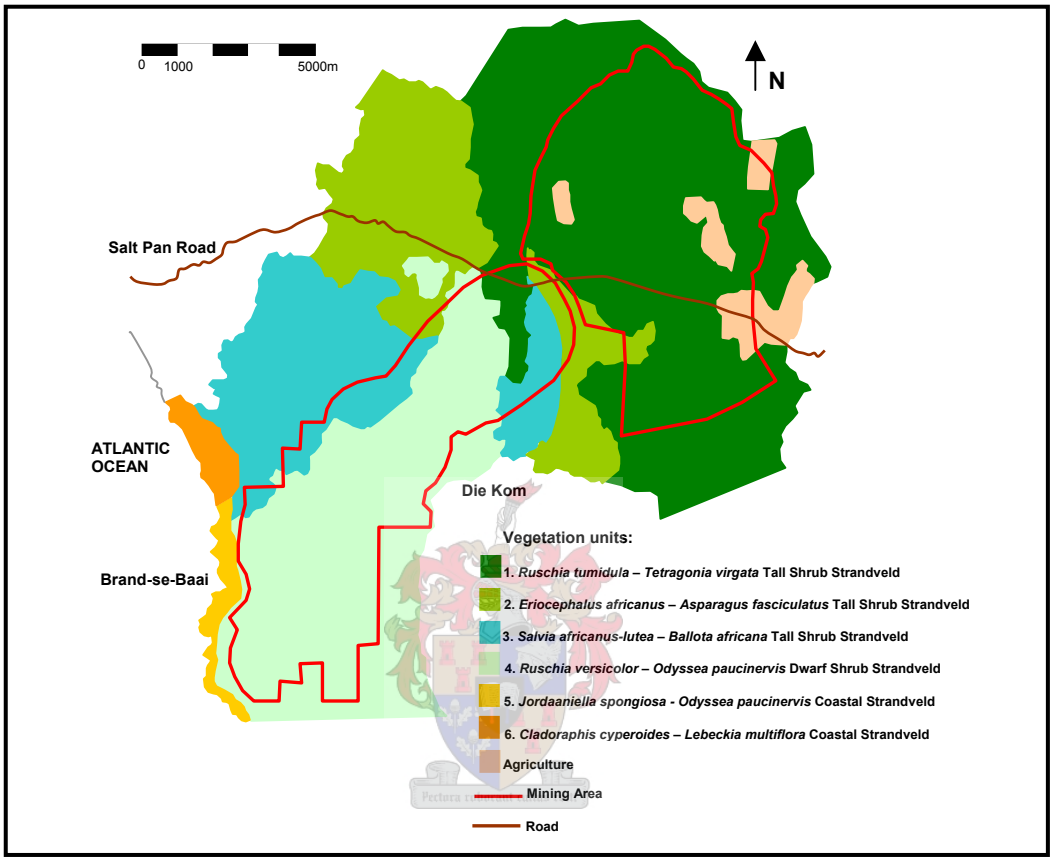

Figure 3.4: Vegetation map of the mining area and its surrounds (after de Villiers *et al.* 1999).

The six vegetation communities classified by de Villiers *et al.* (1999) included the following:

- ***Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld (Vegetation Unit 1)**
  This vegetation unit is situated farthest inland, and as a result receives the least amount of fog and salt spray and is the driest of the communities in the study area. The area consists mainly of small dune systems, which give rise to four community variants (described below). The community occurs on a range of soil depths, but is restricted to the more yellow sands. 132 species were recorded in this vegetation unit and it had a shrub and herbaceous canopy cover of 22.2 and 8%, respectively. Diagnostic species are *Ruschia tumidula*, *Galenia africana*, *Leysera gnaphalodes*, *Pharnaceum lanatum, Oncosiphon suffruticosum* and several *Pteronia* species (de Villiers *et al.* 1999). The four variants found in this community include:

i. *Stipagrostis zeyheri – Lapeirousia* spp. Variant: This variant is found in the dunes valleys in the eastern part of the mining. Diagnostic species include: *Lapeirousia* spp. and *Sarcocaulon* sp. Other abundant species include: *Wahlenbergia paniculata*, *Stipagrostis zeyheri* and *Hermannia modesta*.

ii. *Scirpoides dioecus – Stoebe nervigera* Variant: This variant is found on small dunes and the diagnostic species include *Scirpoides dioecus*, *Tripteris sinuata, Stoebe nervigera, Monilaria chrysoleuca, Gymnodiscus capillaris* and *Wahlenbergia sonderi*. Other conspicuous species include: *Salvia africana-lutea, Amellus tenuifolius, Conicosia pugioniformis* and *Ursinia speciosa*.

iii. *Pentaschistis patula – Chenopodium opulifolium* Variant: This variant is found in disturbed areas in the *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld. Diagnostic species in this variant include: *Hermannia cuneifolia* and *Chenopodium opulifolium*. Other conspicuous species include: *Ruschia tumidula, Galenia africana, Conicosia pugioniformis* and *Eriocephalus africanus*.

iv. *Eriocephalus africanus – Ferraria densepunctulata* Variant: Small patches of Lowland Succulent Karoo vegetation occur within this variant. Diagnostic species include: *Ferraria densepunctulata* and *Crassula dichotoma*. Other conspicuous species include: *Pharnaceum lanatum, Felicia merxmuelleri, Salvia africana-lutea, Asparagus aethiopicus, Amellus tenuifolius, Manulea altissima, Eriocephalus africanus* and *Hermannia amoena* (de Villiers *et al.* 1999).

A large part of this community will be destroyed by future mining activities (Figure 3.4).

- ***Eriocephalus africanus – Asparagus fasciculatus*** **Tall Shrub Strandveld (Vegetation Unit 2)**

This vegetation unit represents a transition between vegetation units one and three, and is found on small dune systems. Sea fog and salt spray intensity are less than that of the communities closer to the coast. 109 species were recorded in this vegetation unit and had a shrub stratum and herbaceous stratum canopy cover of 14.7 and 4.9%, respectively. Conspicuous shrubs within this community include *Asparagus aethiopicus*, *Nestlera biennis, Eriocephalus africanus*, *Asparagus capensis* and *Pharnaceum aurantium*. Abundant species included in the herbaceous stratum are *Manulea altissima* and *Oxalis* species (de Villiers *et al.* 1999). This community is comprised of two variants:

i. *Othonna floribunda – Lebeckia lotonoides* Variant: This variant is situated in the dune valleys and the dominant species are smaller in stature than the ones found on the dunes. There are no diagnostic species in this variant, but the following species are conspicuous: *Euphorbia caput-medusae, Pelargonium senecioides, Nestlera biennis, Cotula thunbergii, Eriocephalus africanus, Oxalis* sp., *Ruschia bolusiae* and *Pharnaceum aurantium*.

ii. *Zygophyllum morgsana – Coelanthum semiquinquefidum* Variant: This variant is located on small dunes and is taller than the variant described above, mainly due to a greater soil depth. This variant also has no diagnostic species, and is differentiated from the variant above by the presence of *Salvia africana-lutea, Amellus tenuifolius, Conicosia pugioniformis, Hermannia scordifolia, Ornithoglossum* sp. and *Hermannia cernua* (de Villiers *et al.* 1999).

Only a small part of this community is included in the area to be mined (Figure 3.4).

- ***Salvia africanus-lutea – Ballota africana*** **Tall Shrub Strandveld (Vegetation Unit 3)**
  This vegetation unit is associated with loose, yellow sand and due to the deep soil on which it occurs is taller than that of the surrounding communities. Although this community is closer to the sea than vegetation unit one it still received relatively little salt spray and fog. 140 species were recorded in this vegetation unit and had a shrub stratum and herbaceous stratum canopy cover of 26 and 6%, respectively. Abundant and conspicuous species in this community included *Salvia africana-lutea, Eriocephalus africanus*, *Helichrysum hebelepis*, *Conicosia pugioniformis, Dimorphotheca pluvialis* and *Nemesia bicornis.* Approximately 40% of this community will be destroyed by mining (de Villiers *et al.* 1999).

- ***Ruschia versicolor – Odyssea paucinervis*** **Dwarf Shrub Strandveld (Vegetation Unit 4)**
  This vegetation unit is found in the largest part of the western areas to be mined (Figure 3.4). Soils vary from compact, dark red in the west to loose yellowish sand in the east. This community receives more sea spray and fog compared to vegetation units 1, 2 and 3, but less sea spray and fog than vegetation units 5 and 6. A total of 171 species were identified in this community, which is the highest value for all the communities. This community has a shrub stratum and herbaceous stratum canopy cover of 16.8 and 15.5%, respectively. Diagnostic species include: *Ruschia versicolor, Tripteris clandestina, Pelargonium senecioides, Thesium spinosum, Ficinia argyropa, Euphorbia caput-medusae, Senecio bulbinifolius, Chaetobromus dregeanus, Chrysocoma longifolia, Indigofera amoena* and *Albuca* sp. *Ruschia caroli* and *Asparagus capensis* dominate the shrub stratum. Ephemeral species and *Odyssea paucinervis*, a perennial creeping grass dominate the herbaceous stratum (de Villiers *et al.* 1999). This community is comprised of three variant communities:

  i. *Ruschia caroli – Aspalathus divaricata* Variant: This variant is located in the central part of the vegetation unit, and is found on dark red sandy soils. This variant is dominated by *Asparagus divaricata*. *Ruschia cymosa* and *Trichogyne ambigua*. The shrub stratum includes the following conspicuous species: *Ruschia versicolor, Ruschia caroli, Asparagus capensis* and *Hermannia cernua*. Conspicuous species of the herbaceous layer include: *Adenogramma littoralis, Ursinia speciosa* and *Odyssea paucinervis*. Most of this variant will be mined.

ii. *Tripteris oppositifolia – Cissampelos capensis* Variant: This variant is situated in the eastern part of the vegetation unit, and is found on yellowish sandy soils. Diagnostic species include: *Phyllobolus* sp. and *Cissampelos capensis*. Conspicuous shrub species include: *Ruschia versicolor, Asparagus capensis* and *Hermannia cernua*. Conspicuous herbaceous species include: *Tripteris clandestina, Adenogramma littoralis* and *Arctotheca calendula*. Most of this variant will be mined.

iii. *Ehrharta calycina – Crassula expansa* Variant: This variant is located in the southern part of the vegetation unit, and is found on compact, reddish soil. Diagnostic species include: *Ruschia* sp., *Aloe framesii* and *Crassula expansa*. Conspicuous shrubs include: *Ruschia versicolor, Eriocephalus africanus, Arctotis scullyi, Vanzijlia annulata, Asparagus capensis, Helichrysum hebelepis* and *Hermannia cernua*. Conspicuous herbaceous species include: *Didelta carnosa* and *Odyssea paucinervis*. The area to be mined does not include much of this variant (de Villiers *et al.* 1999).

A large part of this community has already been destroyed by the mining activities.

- ***Jordaaniella spongiosa* (previously *Cephalophyllum spongiosum*) - *Odyssea paucinervis* Coastal Strandveld (Vegetation Unit 5)**

  This vegetation unit is found in a narrow strip along the coast and is not included in the area to be mined (Figure 3.4). This vegetation unit is found on yellowish sand to the south and vegetation is dominated by *Jordaaniella spongiosa*, *Drosanthemum calycinum, Helichrysum incarnatum* and *Hypertelis salsoloides*. Conspicuous dwarf shrubs include *Galenia sarcophylla, Arctotis scullyi, Vanzijlia annulata, Pharnaceum aurantium* and *Cladoraphis cyperoides*. The ephemeral *Didelta carnosa,* and *Mesembryanthemum crystallinum* and the perennial grass *Odyssea paucinervis* are conspicuous species in the herbaceous stratum. A total of 83 species were identified in this community, and has a shrub stratum and herbaceous stratum canopy cover of 13 and 26%, respectively (de Villiers *et al.* 1999).

- ***Cladoraphis cyperoides – Lebeckia multiflora* Coastal Strandveld (Vegetation Unit 6)**

  This vegetation unit is also found in a narrow strip along the coast and is not included in the area to be mined (Figure 3.4). This vegetation unit is found predominantly on white sand dunes on the northern coast of the study area. *Leipoldtia jacobseniana* is the only diagnostic species in this community. A total of 23 species were identified in this community, which is the lowest value for all the communities. This community has a shrub stratum and herbaceous stratum canopy cover of 4 and 4.3%, respectively (de Villiers *et al.* 1999; de Villiers 2000).

3.5.2 VEGETATION STRUCTURE AND PATTERN

The structure of vegetation is a function of the interactions between the biota and environmental conditions in the area. Vegetation structure in arid ecosystems is commonly arranged with high plant cover patches interspersed in a low-cover matrix and the vegetation is characterised by the size, shape and spatial distribution of the high plant-cover patches. In arid ecosystems there are two main patterns of clumping, namely “banded” (where dense patches of vegetation form bands or stripes mainly perpendicular to a slope) or “spotted” (where dense vegetation patches are irregular in shape). Both patterns generally originate from the same mechanisms (i.e. facilitation and competition), but each are dominated by different drivers. “Banded” vegetation occurs when water is the dominant driver of redistribution of materials and propagules, whereas “spotted” vegetation results when wind is the major distribution driver (Aguiar & Sala 1999). The vegetation at the mine site is dominated by perennial plants that occur in mixed species clumps, interspersed with open areas that are covered by annual plants in the winter (Eccles *et al.* 1999; van Rooyen 2001; Mahood 2003b), i.e. a “spotted” pattern of clumping. Although clumping is an unchanging feature in the vegetation at the mine site, the amount, height, area, composition and percentage cover of the clumps does fluctuate (van Rooyen 2001). Eccles (2000) found that vegetation clumps in the medium Strandveld are irregularly shaped and are generally separated by between 1 and 2m of bare sand. An average clump contains 10 to 25 perennial plants of 7 to 11 species. Succulent and non-succulent species are generally found clumped together within the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (van Rooyen 2001).

Vegetation patchiness or clumping is thought to optimise the capture and storage of limited resources (water and nutrients) from source areas into sinks or patches (Ludwig & Tongway 1996). These areas of increased fertility are known as “fertile islands” and they are an integral part of desert landscapes and play a critical part in the structuring and functioning of desert systems (Titus *et al*. 2002). The spatial heterogeneity resulting from clumping increases in total production and enhances alpha and beta diversity (Aguiar & Sala 1999). The loss of landscape patchiness during mining could result in a reduction of available water, nutrient availability, soil fertility, protection from wind, grazing and heat, facilitation of seedling establishment, etc. Thus, in the context of rehabilitation at Namakwa Sands it is essential to re-establish a patchy vegetation structure similar to the pre-mining vegetation (Mahood 2003a).

3.5.3 SEED BANK

The replacement of topsoil after mining not only helps to improve certain soil properties but it also contains the seed bank, which will be a vital source of seed for annual and, to a less extent, perennial species recruitment. The replacement of topsoil will ultimately dictate the future vegetation (de Villiers *et al.* 2003).

The soil seed bank of the Strandveld Succulent Karoo yielded a mean emerged seedling density of 2 725 m^{-2}. In all vegetation units to be mined the species richness of standing vegetation was higher than in the seed bank. A total of 109 species were recorded in the soil seed bank, which is markedly lower than the 230 spices recorded in the standing vegetation in the study area. The similarity in total species composition between the standing vegetation and the soil seed bank was 54.3% (de Villiers 2000; de Villiers *et al.* 2001). The natural vegetation (which is dominated by perennial species) at Namakwa Sands was not well represented in the seed bank (de Villiers 2000).

3.6 FAUNA

3.6.1 INSECTS

Picker (1989) undertook a pre-mining survey of the insect fauna associated with the mining area using two well-studied insect groups, namely the Neurotera and the Lycaenidae, as indicators of general insect endemicity of the area. Picker (1989) did not find the presence of any rare or threatened insect species in the immediate vicinity of the mine area. However, at least four Lycaenid (butterfly) species, three listed as indeterminate and one as rare in the Red Data Book for butterflies, are likely to occur in the area.

There are a large number of termite termitaria or “heuweltjies” in and around the mining area. Termites (*Microhodotermes viator*), which form “heuweltjies”, play a major role in creating patches of nutrient-enriched soil through the collection and breakdown of litter. Termite activity makes the soils of “heuweltjies” finer, moister and more alkaline than their surrounds (Midgley and Musil 1990). Many animal species utilise these “heuweltjies” and further contribute to the nutrient enrichment of these areas. The transportation of material to burrows within “heuweltjies” create small patches of increased “fertility” that encourages plant colonisation in an otherwise edaphically harsh environment (Milton and Dean 1990; Desmet & Cowling 1999). Mining would destroy these large and well established “heuweltjies” resulting in a major disturbance to the ecological cycle of the mining area. The time taken for these colonies to reach their present size may have taken hundreds of years. Therefore, the loss of these “heuweltjies” could have an impact on ecological cycle for a long time (Picker 1989).

Invertebrates make good indicators of ecological change or condition and ants have been used as indicators of biodiversity and of rehabilitation success (Andersen *et al*. 2002). A total of 14 ant species have been identified at three sites at Namakwa Sands (i.e. natural vegetation and two rehabilitation sites of differing ages) (Netshilaphala *et al.* 2005). Netshilaphala *et al.* (2005) found that ant species richness and diversity was greater in autumn compared to winter. In addition, they found that ant abundance and species richness was greater in the natural vegetation compare to the two-year old and the one-year old rehabilitation sites sampled. The ant assemblages in the rehabilitation sites were more similar to one another than to those of the natural vegetation (Netshilaphala *et al.* 2005).

3.6.2 HERPETOFAUNA (AMPHIBIANS AND REPTILES)

De Villiers (1990) undertook a pre-mining survey of the amphibians and reptiles populations associated with the mining area. Mouton & Alblas (2003) undertook a further study on terrestrial fauna in the study area. The mining area contains no rocky habitats, nor does it support any wetland systems. However, these habitats do occur in the near vicinity (de Villiers 1990).

To date, no frog species have been recorded in the Namakwa Sands mining area (Mouton & Alblas 2003). However, two amphibian species are expected to occur in the mining area. The Namaqua rain frog (*Breviceps namaquensis*) and Namaqua caco (*Cacosternum namaquense*) may potentially occur in the mining area. The Namaqua rain frog is a totally non-aquatic burrowing species and requires no water for breeding (de Villiers 1990; Mouton & Alblas 2003). The Namaqua caco, on the other hand, needs at least a temporary water body for breeding. It should be noted that Mouton & Alblas (2003) also found tadpoles of the Clicking stream frog (*Strongylopus grayi*) approximately 10 km inland from the mining area along the Goeraap River. None of these three species are classified as Red Data species (Mouton & Alblas 2003). However, the Namaqua rain frog is endemic to the western Cape region (de Villiers 1990). The mining area lies on the extreme of the distribution ranges of these three species and is therefore probably transitional and does not represent an optimum habitat (Mouton & Alblas 2003).

Only 26 of the 44 reptile species (i.e. 3 tortoises, 30 lizards and 11 snakes) expected to occur in the mining area have been recorded in the mining area. Fifty percent of the reptile species that are expected to occur in the mining area are endemic to the western coastal region of South Africa and southern Namibia. Endemism is greatest among the lizards (63%) and lowest among the snakes (18%). One of the three tortoise species occurring in the area is endemic to the West Coast (Mouton & Alblas 2003). Nine of the reptile species are listed as Red Data species:

- Vulnerable: Namaqua dwarf adder (*Bitis schneideri*), Armadillo lizard (*Cordylus cataphractus*) and the Lomi's blind legless skink (*Typhlosaurus lomii*).
- Lower Risk: Large-scaled girdled lizard (*Cordylus macropholis*) and Namaqua plated lizard (*Gerrhosaurus typicus*).
- Data Deficient (considered to be Threatened): Cuvier's blind legless skink (*Typhlosaurus caecus*), Austen's thick-toed gecko (*Pachydactylus austeni*), the Rough thick-toed gecko (*Pachydactylus rugosus*) and the Speckled padloper (*Homopus signatus cafer*) (Mouton & Alblas 2003).

3.6.3 BIRDS

Bird fauna occurring in the mining area was surveyed by Allan & Jenkins (1990). Eighty-three species of birds were recorded within and around the mine site, four of which were incidental vagrants, and a further 66 species can be expected to occur within the area. Of the 145 species that are expected to occur there (i.e. all except the vagrants), 108 inhabit the inland areas, 32 are restricted to the coast and 5 inhabit both inland and coastal areas. Most of the species are resident or visit the site at any time of the year, but a few (mainly coastal species) are seasonal migrants (Allan & Jenkins 1990).

Fifty-seven (i.e. 39%) species are endemic to southern Africa. Only one Red Data species was confirmed on site and that was *Neotis ludwigii* (Ludwig's Bustard) (Vulnerable). However, four additional Red Data species are expected to occur on site. These include *Polemaetus bellicosus* (Martial Eagle) (Vulnerable), *Hydroprogne caspia* (Caspian Tern) (Rare), *Sterna vittata* (Antarctic Tern) (Rare) and *S. balaenarum* (Damara Tern) (Rare). Five other species are considered "near threatened". These include *Phalacrocorax carbo* (Crowned Cormorant), *P. neglectus* (Bank Cormorant), *Hieraaetus pinnatus* (Booted Eagle), *Circus maurus* (Black Harrier) and *Eremomela icteropygialis* (Karoo Eremomela) (Allan & Jenkins 1990).

3.6.4 MAMMALS

Brand-se-Baai has an exceptionally low mammal species diversity (Rautenbach 1990). Rautenbach (1990) is of the opinion that the low mammal diversity is a consequence of low ecological diversity preventing the occurrence of certain habitat-specific species (i.e. there is a lack of habitat diversity).

Only 36 mammal species are expected to occur within and around the study area, 20 of which have been confirmed to occur on site. These 36 species include seven insectivores, four bats, two hare/rabbit species, 10 rodents, one felid, three canids, one mustelid, five viverrids, dassie, and two antelope species (Rautenbach 1990; Mouton & Alblas 2003). Two of the 36 species occurring or expected to occur in the mining area are endemic to the arid western coastal region. Three species are listed as Vulnerable species. These include the African wild cat (*Felis lybica*), Grant's golden mole (*Erimitalpa granti*) and Round-eared Elephant-shrew (*Macroscelides proboscideus*) (Mouton & Alblas 2003).

Rautenbach (1990) identified three major habitat types in the mining area, i.e. rocky outcrops, white coastal sand dunes and the inland Succulent Karoo area. Rautenbach (1990) considered the inland rocky outcrops in the mining area too small and too isolated to support viable populations of rock-dwelling mammals, e.g. the Cape rock elephant shrew (*Elephantulus edwardii*), the Namaqua rock rat (*Aethomys namaquensis*), the Rock dormouse (*Graphiurus platyops*), the spectacled dormouse (*G. ocularis*) and Smith's red rock rabit (*Pronolagus rupestris*). However, Mouton & Alblas (2003) found that many mammal species used the rock outcrops extensively for shelter and as foraging sites. Mining would result in the loss of these habitats in the mining area. However, most mammal species present could, theoretically, relocate to similar habitats adjacent to the mining area (Mouton & Alblas 2003).

3.7 LAND USE

Most land in the area is used for agricultural purposes including dry cultivation, irrigated lands and grazing area. Irrigation is carried out in the Olifants River floodplain for vegetable and vineyards. Farms further away from the river practise dry methods for growing cereal crops (e.g. wheat and rye) and small stock farming. Stocking rates area generally low, varying 10 to 20 hectares per small stock unit (EEU 1990).

Most mining to date has taken place within the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (Figure 3.4) and a large part of this community has already been destroyed by the mining activities. Most of the remaining unmined land within the Mining Authorisation Area (Figure 3.1) is located within the *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld (Figure 3.4). This area is generally uncultivated and is used for periodic small stock grazing (sheep) at a grazing intensity of one small stock unit per 10 ha (Pool *pers. comm.*). However, there are a few isolated cultivated / fallow lands in the north-eastern portion of the mining area (Figure 3.4).

3.8 MINING AND REHABILITATION

Namakwa Sands has been mining the heavy mineral deposit, rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994. Before any area is strip-mined, all vegetation and a minimum of 5 cm of topsoil are removed by bulldozer leaving the sub-soil exposed, resulting in a total loss of vegetation from the site. Topsoil removed prior to mining is either stored temporarily (no longer than 3 months) or transported directly to mined out areas for rehabilitation purposes (Mahood 2003a; Namakwa Sands 2001; 2002). The mineral enriched sand is extracted from the west and east mine and then processed on site at the primary and secondary concentration plants to produce a magnetic and non-magnetic stream. These streams are then transported to the mineral separation plant near Koekenaap where the magnetic stream is processed to produce an ilmenite stream and the non-magnetic stream is processed to produce rutile and zircon for export. The ilmenite stream is railed to the smelter for further processing to produce titania slag (which is further separated into chloride and sulphate slag) and pig iron, which are then exported (Namakwa Sands 2002).

Mining is a continuous process of forward movement from mined to un-mined areas whereby the subsoil is removed from the advancing front and the processed sand (tailings) is returned to the mined out area, a process known as “backfilling” (Figure 3.5). Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward with approximately 200 to 335 ha of mined out land requiring rehabilitation per annum (Namakwa Sands 2001).

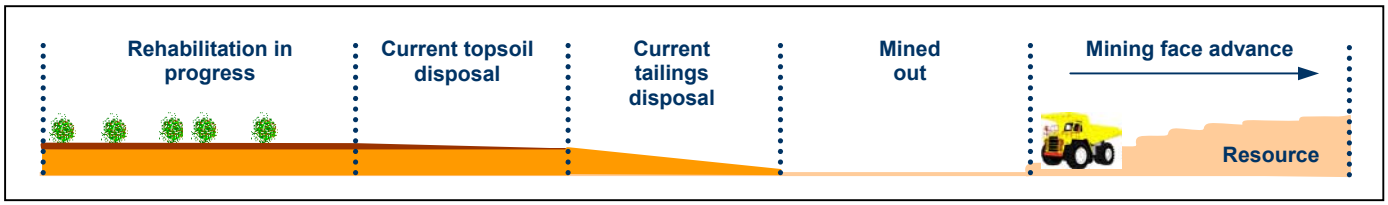

Figure 3.5: Mining at Namakwa Sands is a continuous process from un-mined to rehabilitated land.

The long-term rehabilitation goal of Namakwa Sands is to restore the area to, or as close as possible to, its natural state and achieve a vegetation cover and productivity similar to the pre-mining land-use (i.e. small-stock farming) (Grindley & Barbour 1990). This goal has been translated into the following rehabilitation objectives:

1. The number of indigenous plant species is to be at least 30% of the average number of species in reference areas within three years and 60%[1] by the end of rehabilitation (EEU 1990);
2. To achieve an average vegetation cover of rehabilitated areas of 50% and 80% within three and five years, respectively, compared to the average cover for reference areas (EEU 1990);
3. The criteria mentioned in (1) and (2) must be achieved for two successive seasons with no artificial inputs (EEU 1990); and
4. Ensure the post-mined areas have a carrying capacity of between 10 and 20 hectares per small stock unit (Hälbich *pers. comm.*).

Namakwa Sands has initiated various research projects to gain an understanding of the baseline conditions and ecosystem function in order to increase plant cover and biodiversity on post-mined areas. This on-going research and the development of rehabilitation and mining techniques have resulted in the implementation of four rehabilitation techniques:

1. Rehabilitation Scenario 1 (or Site S1): In this scenario the tailings from the primary and secondary concentration plants were transported back to mined out areas and deposited as backfill and shaped to fit contours of the surrounding landscape (EEU 1990; Washington 1990). Stockpiled topsoil was then spread over the backfilled area to a minimum thickness of approximately 5 cm. Once the topsoil had been profiled and topsoiled, the area was stabilised with windbreaks (Plate 3.1). Windbreaks were installed to control wind erosion, prevent the loss of the soil's seed bank and prevent damage to plants (Namakwa Sands 2001). Windbreaks consisted of polyethylene shade cloth (40% shade) with a height of approximately 1 m, with vertical pockets sown into it to metal droppers to anchor the windbreaks. Windbreaks were placed at 4 to 5 m intervals perpendicular to the dominant wind directions (Mahood 2003a). Therefore, this scenario relied entirely on the germination of the seed bank in the topsoil and the importation of seed from areas outside the area of disturbance.

[1] De Villiers *et al.* (1999) identified 230 species within all six vegetation communities, which would mean that 168 species should be introduced to the area. However, the list of species that de Villiers *et al.* (1999) suggested could be used due to their adaptation to varying environmental conditions amounted to only 65 species (i.e. 28.3%). Consequently, de Villiers *et al.* (1999) suggested that the "species richness" objective of reintroducing 60% of the natural vegetation species was unrealistic and recommended that a more obtainable goal would be 30% of the total species present prior to mining. It should be pointed out that de Villiers *et al.* (1999) based their calculation on the total number of species identified within all six vegetation communities (even though two vegetation communities are not included in the Mining Authorisation Area) and should rather have been calculated per vegetation community. This would have resulted in a lower number of species to be introduced into mined out areas.

Plate 3.1: Windbreaks were erected perpendicular to the major wind direction in Site S1 in order to control wind erosion, prevent the loss of the soil's seed bank and prevent damage to plants.

2. Rehabilitation Scenario 2 (or Site S2): This rehabilitation technique was originally implemented by Namakwa Sands when topsoil was processed together with the subsoil. However, this technique is no longer implemented, as the topsoil is no longer processed as per the requirements of the mining license (Hälbich *pers. comm.*).

   In this scenario the tailings from the primary and secondary concentration plants were transported back to mined out areas and deposited as backfill and shaped to fit contours of the surrounding landscape (EEU 1990; Washington 1990). Windbreaks were then erected to control wind erosion and prevent damage to plants (as described for Rehabilitation Scenario 1). Once windbreaks had been established areas were seeded with seed collected by vacuum harvesting of Strandveld species in areas not affected by mining, together with the addition of *Eragrostis curvula* and *Sorghum* sp. seeds. *Sorghum* sp. and *Eragrostis curvula*, although not indigenous to the mining area, were selected for seeding as they were considered to be suitable for the stabilisation of post-mined areas without competing with the natural vegetation. It was anticipated that both these species would be replaced by indigenous species when they died (Namakwa Sands 2001; Hälbich *pers. comm.*). Therefore, this rehabilitation scenario essentially involved the seeding of indigenous species and the importation of seed from areas outside the area of disturbance.

3. Rehabilitation Scenario 3 (or Site S3): This rehabilitation scenario is similar to Rehabilitation Scenario 1 but included the translocation of five locally common perennial plant species from pre-mined areas in multi-species clumps (each clump consisted of one of each of the 5 species) as undertaken by Mahood (2003a). The five species used in the translocation trials included: *Othonna cylindrica, Zygophyllum morgsana, Ruschia versicolor, Lampranthus suavissimus* and *Asparagus spp*. Therefore, this scenario relied on the germination of the seed bank in the topsoil and the importation of seed from areas outside the area of disturbance and the facilitation or acceleration of successional processes due to the translocation of species in clumps.

4. Rehabilitation Scenario 4 (or Site S4): This rehabilitation scenario is similar to Rehabilitation Scenario 2 (except that *Ehrharta calycina* was seeded in place of *Eragrostis curvula* and *Sorghum sp.*) but also included the translocation of three locally common perennial plant species (*Othonna cylindrica, Ruschia versicolor and Lampranthus suavissimus*) in multi-species clumps based on the results on Mahood (2003a). Whole plants (as apposed to damaged plants) of these species are collected from undisturbed vegetation ahead of the mining front. Multi-species clumps were irrigated for a week after translocation in order to help plant establishment (Hälbich *pers. comm.*). Therefore, this rehabilitation scenario essentially involved the seeding of indigenous species, the importation of seed from areas outside the area of disturbance and the facilitation or acceleration of successional processes due to the translocation of species in clumps.

Namakwa Sands currently undertakes an annual vegetation survey to measure species richness and cover of selected reference and rehabilitation transects. Each transect consists of 4 000 step points using the point survey method. Rehabilitation success is currently determined by cover and species richness, both being expressed as a percentage of the reference site (Grobler 2003).

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CHAPTER 4

VEGETATION RECOVERY IN MINED OUT AREAS AT NAMAKWA SANDS

ABSTRACT

Namakwa Sands has been strip-mining a heavy mineral deposit since September 1994 and has been rehabilitating mined out areas as the mining front moves forward. The mining company has implemented four rehabilitation techniques, varying in the investment of topsoil replacement, seeding and plant translocation, in order to increase plant cover and biodiversity on post-mined areas. The main aims of this study were to determine whether the four rehabilitation techniques have achieved any of the rehabilitation objectives and to determine which of the four rehabilitation techniques has been the most successful in facilitating the recreation of a vegetation composition, structure and function similar to that found within the natural vegetation. Sampling was undertaken along line transects at six sites (i.e. two reference and four rehabilitation sites) in the east mine over four sampling periods between February 2004 and September 2005. The four rehabilitation sites corresponded to the four rehabilitation techniques implemented at the mine, namely site S1 (topsoil only), site S2 (seeding only), site S3 (plant translocation with topsoil) and site S4 (seeding and plant translocation). Various vegetation parameters were measured along each line transect in order to determine species richness and similarity, species diversity and evenness, cover, abundance, species dominance, growth forms, persistence (annual or perennial), functional guilds (pollination and dispersal) and vegetation structure. The results show that Site S3 appears to be the most successful technique in facilitating vegetation recovery, followed by sites S4 and S1. Areas where only seeding was implemented (site S2) performed the worst. All rehabilitation sites met the three-year cover objective. Although sites S1, S3 and S4 met the three-year species richness objective, all rehabilitation sites had a significantly lower species diversity than the reference sites and were dominated by one or two species (generally *Tetragonia fruticosa*). This dominance by *Tetragonia fruticosa* is of concern due to its palatability. The dominance by a single palatable species may falsely enhance the grazing capacity, which could result in overgrazing if current grazing practices are implemented. Species and functional diversity appears to be the most limiting factors within the rehabilitation sites. Namakwa Sands will not be able to meet their long-term rehabilitation goal if species diversity does not improve significantly and they may need to consider adaptive management. Namakwa Sands will need to continue to monitor plant and soil changes until the rehabilitation goal and objectives have been achieved. It is recommended that species composition and similarity, species richness, species diversity and evenness, vegetation cover, species dominance, vertical structure and functional diversity of the vegetation (clumps and inter-clumps) within reference and rehabilitation sites be monitored as part of the long-term monitoring programme. Only when all rehabilitation objectives have been met, species diversity is not significantly different to reference sites and when the number of palatable species increases should rehabilitation areas be considered for grazing. Alternatively, an appropriate grazing strategy, which is related to the *Tetragonia fruticosa* dominated vegetation within rehabilitation sites, would need to be determined and adopted.

Key words: strip-mining; heavy minerals; Strandveld; rehabilitation techniques; topsoil; seeding; translocation; rehabilitation success; monitoring; fertile islands; clumping.

4.1 INTRODUCTION

Strip-mining is expanding in the arid, winter-rainfall areas of South Africa and although economically important, it is having a detrimental effect on biologically diverse environments where vegetation growth is restricted by aridity, wind and saline, nutrient-poor soils (Milton 2001; Blignaut & Milton 2005). Strip-mining, as is practised at Namakwa Sands (Figure 3.1, Chapter 3), causes total destruction of natural ecosystems through the removal of vegetation and soil in the area where mining is being undertaken (Cooke & Johnson 2002).

Namakwa Sands has been mining a heavy mineral deposit, rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994. Before any area is mined, all vegetation and topsoil (a minimum of 5 cm) is removed by bulldozer leaving the sub-soil exposed (Namakwa Sands 2001). During mining and processing of soil at Namakwa Sands, the chemical and physical properties of the soil are destroyed or altered due to the removal of heavy minerals (Lubke *et al*. 1996) and fines or slimes (material <45 μm) (Prinsloo 2005), as well as the use of seawater in the extraction process (de Villiers *et al*. 1999).

In terms of South African legislation (Minerals and Petroleum Resources Development Act No. 28 of 2002), mining companies are compelled to rehabilitate mined areas and they are responsible for rehabilitation until a closure certificate has been issued by the Department of Minerals and Energy. Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward with approximately 200 to 335 ha of mined out land requiring rehabilitation per annum (Namakwa Sands 2001).

A common goal of rehabilitation programmes is to generate a sustainable ecosystem with a similar structure and function to the natural ecosystem (Chambers *et al.* 1994; Lubke *et al.* 1996). In order to rehabilitate resilient ecosystems in arid areas, it is desirable to restore as many aspects of the natural vegetation as possible (Blignaut & Milton 2005). To do this, however, it is essential to know what ecosystem processes and landscape functions have been disturbed so that rehabilitation procedures can be designed to restore these processes and functions (Ludwig & Tongway 1996). Therefore, an understanding of the pre-mining ecosystem, its structure, diversity, dynamics and ecological processes (e.g. dispersal, colonisation and persistence of local populations) is vital prior to mining (Lubke *et al.* 1996; Lindborg & Eriksson 2004). In this regard, De Villiers *et al.* (1999) undertook a pre-mining vegetation benchmark survey at Namakwa Sands for rehabilitation purposes.

The rehabilitation goal, the type of vegetation and the characteristics of the site will ultimately determine the nature of establishment, monitoring and management practices that have to be adopted (Harris *et al*. 1996). The long-term rehabilitation goal at Namakwa Sands is to restore the area to, or as close as possible to, its natural state and achieve a vegetation cover and productivity similar to the pre-mining land-use (i.e. small-stock farming) (Grindley & Barbour 1990; Mahood 2003b). Based on the rehabilitation goal and objectives, it is important to design a unique rehabilitation management plan for each mining site

and to encourage research that complements the rehabilitation process. Pilot studies should be used to develop cost-effective rehabilitation techniques that best suite the environment in which the mine is found (Lubke & Avis 1998). Due to the difficulty of rehabilitating mined out areas at Namakwa Sands as a result of harsh environmental factors, the mining company implemented various rehabilitation techniques in order to increase plant cover and biodiversity on post-mined areas (Mahood 2003b). As a result of on-going research and the development of rehabilitation and mining techniques, four rehabilitation techniques varying in the investment of topsoil replacement, seeding and plant translocation have been implemented (i.e. (1) topsoil only; (2) seeding without topsoil; (3) plant translocation with topsoil; and (4) seeding and plant translocation without topsoil). These techniques are presented in more detail in Chapter 3. This variation in rehabilitation design increases the probability that at least one rehabilitation technique is successful, and provides information that might be key to adaptive management considerations (Pastorok *et al.* 1997).

Rehabilitation should be viewed as an ongoing process that results in self-sustaining systems and monitoring rehabilitation success will help determine whether the rehabilitation goals or specific end points have been reached (Holl & Cairns 2002). To date no attempts have been made to assess the effectiveness or success of the four rehabilitation techniques implemented at Namakwa Sands. Therefore, the aim of this study was to determine whether the techniques have been successful in achieving any of the rehabilitation objectives and are on a successional trajectory towards the desired endpoint after approximately four years (2001 – 2005). In addition, this study aimed to determine which of the four rehabilitation techniques has been the most successful in facilitating the recreation of a vegetation composition, structure and function similar to that found within reference sites.

4.2 METHODS

4.2.1 STUDY AREA

The study was undertaken at the Namakwa Sands east mine, which is situated in the vicinity of Brand-se-Baai on the west coast of South Africa, approximately 385 km north of Cape Town (Figure 3.1; Chapter 3).

The study area is characterised by hot, dry summers and sporadic winter rainfall, falling mainly in the months from May to July (EEU 1990; le Roux & Schelpe 1997). The rainfall increases from the north to the south, with an average of 160 mm per annum occurring in the mining area (Mahood 2003a). Rainfall is augmented by heavy dew falls and sea fogs. Rainfall, sea fog and dew fall amount to a cumulative average annual precipitation of 282 mm per annum measured over a four year period (de Villiers *et al.* 1999). Compared to the average of 160 mm per annum (Mahood 2003a), the study area has experienced a below average rainfall over the last few years with an annual rainfall of 100.5 mm, 137.5 mm and 82.9 mm being recorded in 2003, 2004 and 2005, respectively.

Climatically the Namaqualand-Namib region has relatively moderate temperatures through the year with an average annual temperature of 15.9 ^{o}C in the mining area from 2003 to 2004. The highest average monthly maximum temperature over this period was 37 ºC in summer (March) and the lowest average monthly minimum temperature was 4.7 ºC in winter (July).

The strong wind regime in the area is a major cause of erosion at Namakwa Sands (Washington 1990). The dominant winds during the spring and summer (September to March) between 2003 and 2005 were from the south-southeast and south but winds from the north-northwest and northwest were not uncommon. The dominant winds during the winter months (June to August) were from the north-northwest, northeast, northwest and east-northeast but winds from the south and southeast were not uncommon. The average annual wind speed in the mining area between 2003 and 2004 was 4.1 m/s. The highest average monthly wind speed was 5.8 m/s in winter (July) and the lowest monthly average wind speed was 3.7 m/s in summer (February).

The vegetation in the study area is classified by Low and Rebelo (1996) as consisting of Strandveld Succulent Karoo and Lowland Succulent Karoo, both of which are classified under the Succulent Karoo Biome. The Strandveld Succulent Karoo occupies the sandy coastal plain throughout Namaqualand (Cowling *et al.* 1999). It is associated with areas of calcareous sand and contains many drought deciduous and succulent species (Low and Rebelo 1996). The Lowland Succulent Karoo is dominated by members of the Mesembryanthemaceae, especially the species of *Ruschia, Drosanthemum, Malephora* and *Delosperma.* Boucher & Le Roux (1989) classified the Strandveld vegetation of the mine site into three variants according to vegetation height, i.e. tall, medium and short Strandveld. These three generalised categories were based on a combination of vegetation structure and floristic content. The three variants vary in height according to the depth of the soil profile. Tall Strandveld, dominated by shrubs between 1 to 2 m tall, occurs on relatively deep calcareous sand, with a canopy cover of 60 – 75%, under a light grazing regime. Medium Strandveld is characterised by plants that are in the region of 50 cm tall and it has a projected canopy cover of perennial species of between 50 and 60%. Short Strandveld varies in average height from 10 – 35 cm. This community occurs in shallow soils where there is very little storage of moisture. The projected vegetation canopy cover of perennial species is usually less than 50%. De Villiers *et al.* (1999) undertook a detailed survey of the mine site and divided the mine area into six vegetation communities or associations:

1. *Ruschia tumidula - Tetragonia virgata* Tall Shrub Strandveld;
2. *Eriocephalus africanus - Asparagus fasciculatus* Tall Shrub Strandveld;
3. *Salvia africanus-lutea - Ballota africana* Tall Shrub Strandveld;
4. *Ruschia versicolor - Odyssea paucinervis* Dwarf Shrub Strandveld;
5. *Jordaaniella spongiosa - Odyssea paucinervis* Coastal Strandveld; and
6. *Cladoraphis cyperoides - Lebeckia multiflora* Coastal Strandveld.

These vegetation communities are described in more detail in Chapter 3.

4.2.2 STUDY SITES

The vegetation was sampled at six sites (i.e. four rehabilitation and two reference sites) in the east mine. Due to the diversity of the vegetation on the mine site, reference transects were located in two of the six surrounding vegetation communities (Figure 3.4, Chapter 3), namely *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld and *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld (as classified by de Villiers *et al.* 1999). These two vegetation communities were selected as most of mining and rehabilitation to date has been undertaken within the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld, and most future mining will be located within the *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld.

The four rehabilitation sites relate to the four rehabilitation techniques implemented at Namakwa Sands. The need to find nearby sites of a similar age and differing in rehabilitation technique was a constraint that resulted in pseudo-replicated sampling with only one large area per treatment type sampled. Age and rehabilitation technique also determined the size and shape of the selected sites. The six sites sampled were as follows:

1. Reference site R1 (31° 15.853' S 17° 58.415' E): Four 50 m x 50 m plots were located within the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (Plate 4.1). The four plots were arranged in series from west to east with a 10 m interval between each plot.
2. Reference site R2 (31° 16.451' S 17° 56.181' E): Four 50 m x 50 m plots were located within the *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld (Plate 4.2). This site is subject to periodic grazing by sheep at a grazing intensity of one small-stock unit per 10 ha. The four plots were arranged in series from north to south with a 10 m interval between each plot.
3. Rehabilitation site S1 (31° 15.840' S 17° 56.143' E): This rehabilitation scenario involved the spreading of topsoil over tailings with no seeding and no plant translocation (Plate 4.3). The precise source of the topsoil from within the east mine and the length of topsoil stockpiling (up to three months) are unknown. This scenario relies entirely on the germination of the seed bank in the topsoil and the importation of seed from areas outside the area of disturbance. The site selected was rehabilitated in June 2001. Four 50 m x 50 m plots, arranged in series from north to south with a 10 m interval between each plot, were located within this site.
4. Rehabilitation site S2 (31° 15.461' S 17° 55.544' E): This rehabilitation scenario involved the seeding of indigenous species (including the addition of *Eragrostis curvula* and *Sorghum sp.*) onto tailings (no topsoil) (Plate 4.4). Seeds were collected by vacuum harvesting of Strandveld species in areas not affected by mining and as a result the species mix is unknown. The site selected was seeded in 2001 (month unknown). Four 50 m x 50 m plots, arranged in series from north to south with a 10 m interval between each plot, were located within this site.
5. Rehabilitation site S3: This rehabilitation scenario involved the spreading of topsoil over tailings with the translocation of five indigenous species (*Ruschia versicolor, Lampranthus suavissimus, Othonna cylindrica, Zygophyllum morgsana* and *Asparagus spp.*) into multi-species clumps. The precise source of the topsoil from within the east mine and the length of topsoil stockpiling (up to three months) are unknown. This site is the non-irrigated translocation trials of Mahood (2003a)

undertaken in June 2001 (Plate 4.5). Since there were only three 50 m x 50 m non-irrigated plots (31º 15.889' S 17º 56.003' E; 31º 16' 007' S 17º 56 129' E; 31º 16' 165' S 17º 56 303' E) the middle plot was sampled twice.

6. Rehabilitation site S4 (31° 15.267' S 17° 55.433' E): This rehabilitation scenario involved the seeding of indigenous species (including the addition of *Ehrharta calycina*) directly onto tailings (no topsoil) with the translocation of three indigenous species (*Ruschia versicolor, Lampranthus suavissimus* and *Othonna cylindrica*) into multi-species clumps (Plate 4.6). Seeds were collected by vacuum harvesting of Strandveld species in areas not affected by mining and as a result the species mix is unknown. The site selected was seeded in 2001 and translocation took place between July and August 2002. Four 50 m x 50 m plots, arranged in series from north to south with a 10 m interval between each plot, were located within this site.

The locality of these sites is presented in Figure 4.1.

4.2.3 VEGETATION SAMPLING

Sampling was undertaken four times between February 2004 and September 2005. The sampling periods were as follows:

1. Summer 2004 (18 to 26 February 2004);
2. Winter 2004 (28 August 2004 to 3 September 2004);
3. Summer 2005 (20 to 25 February 2005); and
4. Winter 2005 (28 August 2005 to 1 September 2005).

The vegetation was sampled using the Line Intercept Method (Sutherland 1997). This sampling method was selected as the natural vegetation is patchy and this method is suitable for use in sparse, low vegetation with a projected canopy cover of less than 50%. Within each of the 24 plots (i.e. four plots per site), 50 m line transects were orientated in a "W" pattern (Figure 4.2). The reason for the "W" orientation was to ensure that the line transects were not orientated parallel to rows of translocated clumps (as undertaken in rehabilitation sites S3 and S4), thereby ensuring that the translocated clumps were sampled. Three line transects were located within each of the two reference sites and five line transects were located within each of the four rehabilitation sites, resulting in a total of 26 line transects per sampling.

For all line transects, a 50 m tape measure was laid across each transect. Species, canopy height (accuracy of ± 0.5 cm) and cover of individual plants intercepted (above or below) along the 50 m line transect was determined. In addition, it was noted whether intercepted plants were solitary or part of a clump (i.e. where the canopy of two individuals overlapped). Wherever the tape measure intercepted (above or below) a clump, the height of the clump canopy and cover was determined. The calculation of total vegetation cover for each transect took the overlap of individuals into account to ensure that the total vegetation cover was not overestimated, but this was not the case for the calculation of individual species cover and annual / perennial cover.

Plate 4.1: Reference site R1 was located within the *Ruschia versicolor - Odyssea paucinervis* Dwarf Shrub Strandveld.

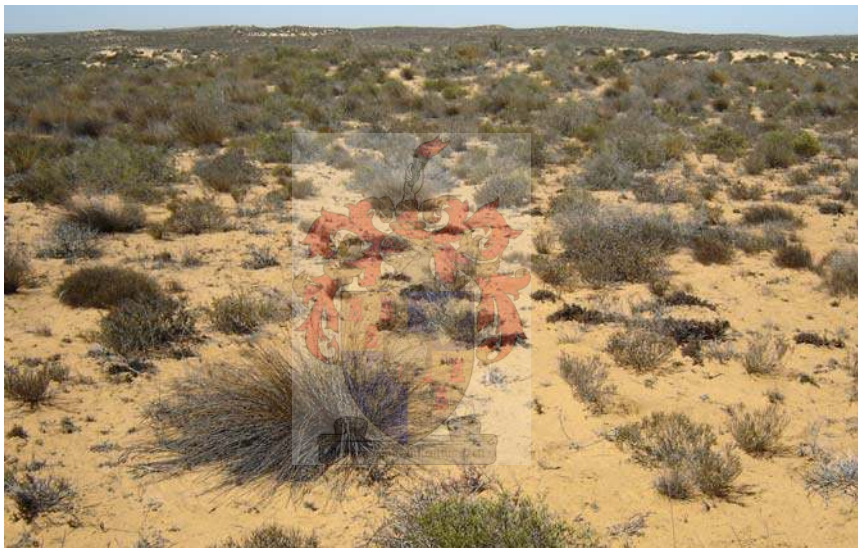

Plate 4.2: Reference site R2 was located within the *Ruschia tumidula - Tetragonia virgata* Tall Shrub Strandveld.

Plate 4.3: Rehabilitation site S1 was located in an area where topsoil was spread over tailings. Windbreaks were erected in all rehabilitation sites to control wind erosion.

Plate 4.4: Rehabilitation site S2 showing the dominance by *Tetragonia fruticosa* (Slaaibos), which grew very large and formed large clumps (up to 7.8 m meters long and 4.5 m wide).

Plate 4.5: Rehabilitation site S3 was located within the three non-irrigated translocation sites of Mahood (2003a). In the foreground is a clump that was translocated in June 2001.

Plate 4.6: Rehabilitation site S4 involved the seeding of indigenous species directly onto tailings with the translocation of three indigenous species into multi-species clumps.

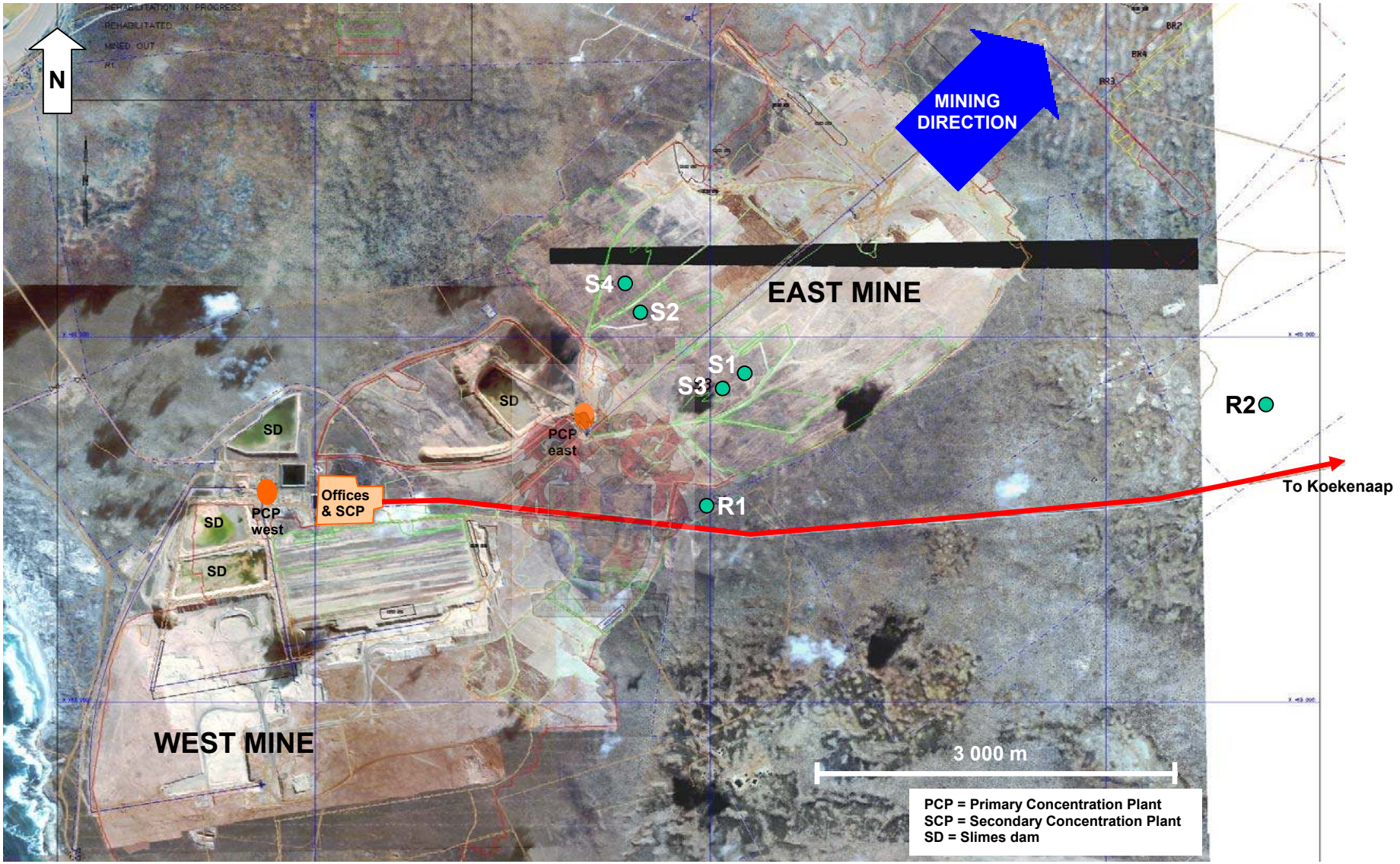

Figure 4.1: Approximate localities of the six sampling sites at Namakwa Sands. Reference site R1 = *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld; Reference site R2 = *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld; Rehabilitation site S1 = topsoil only; Rehabilitation site S2 = seeding without topsoil; Rehabilitation site S3 = plant translocation with topsoil; and Rehabilitation site S4 = seeding and plant translocation without topsoil.

The size (or cover) of individual clumps intercepted along the line transects was estimated by measuring two perpendicular diameters of the clump canopy and assuming that the projection of the canopy onto the soil was a rectangle (i.e. cover = length x width, with length being the longer axis). Perpendicular diameters of the canopy were measured to an accuracy of ± 0.5 cm.

All species associated with each clump intercepted along the line transect were recorded. An 80 cm diameter ring was placed over each clump and the cover (%) of those species occurring within the ring was estimated. Cover values greater than 100% indicate multi-layered vegetation. For every clump encountered along the line transect, an inter-clump microsite was sampled. For inter-clump microsites the 80 cm diameter ring was placed 2 m away from the clump in a northerly direction and all species within the ring were recorded and cover (%) estimated. If another clump was intercepted the ring was placed a further 2 m away. This continued until an inter-clump microsite was encountered. The average species cover within clump and inter-clump microsites was calculated over the four sampling periods.

Voucher specimens were collected for all species recorded in the field. Species encountered were identified in the field using Le Roux & Schelpe (1997) or voucher specimens were identified by Professors Sue Milton and Karen Esler or at the Compton Herbarium at Kirstenbosch Botanical Garden in Cape Town, South Africa. Nomenclature follows Germishuizen & Meyer (2003).

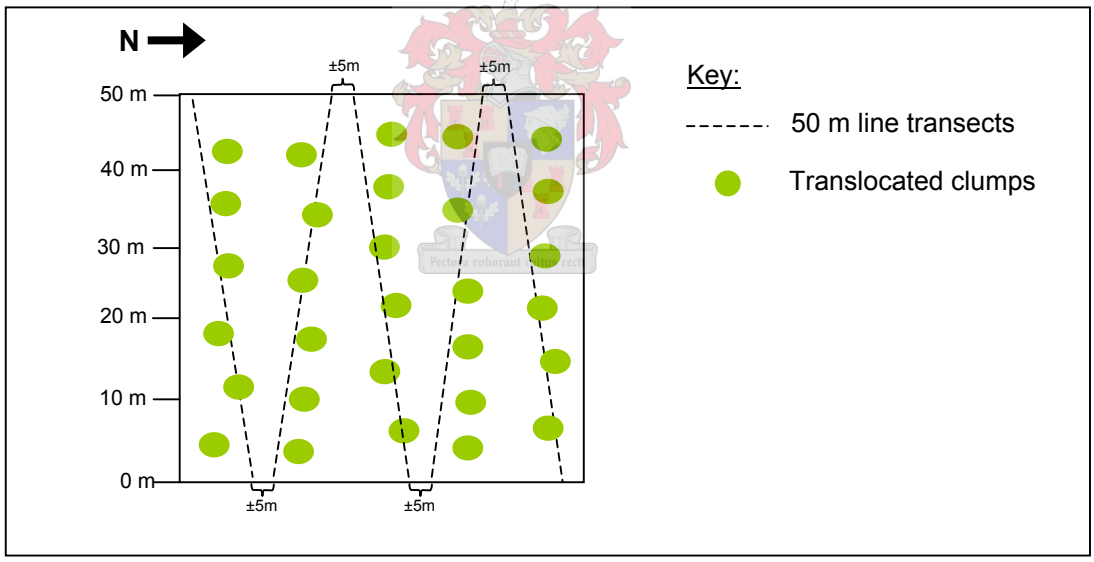

Figure 4.2: Layout of the five line transects within the rehabilitation sites showing the "W" pattern of transect orientation.

4.2.4 DATA ANALYSIS

Repeated Measures Analysis of Variance

Within each plot, species richness, vegetation cover, annual cover, perennial cover, abundance, number of clumps, clump cover, clump size, clump height, inter-clump distance, clump cover within 80 cm ring and inter-clump cover within 80 cm ring recorded in each transects were grouped for analysis. Differences between sites and over time (two years) were evaluated using Repeated Measures Analysis

of Variance (ANOVA) using STATISTICA 7.0 (StatSoft, Inc. 2004). Normal probability plots of the residuals were inspected for deviations from normality that might have affected the results. Where the data were normally distributed, *Post Hoc* tests were conducted by Fisher LSD using STATISTICA 7.0. Where the data were not normally distributed (i.e. abundance, inter-clump distance, annuals cover, clump cover within 80 cm ring and inter-clump cover within 80 cm ring), individual differences ($p<0.05$) between sites and over time were evaluated by doing pairwise comparisons using a bootstrap technique (Efron & Tibshirani 1998) and then applying Bonferroni multiple testing.

Similarity index

Species similarity between the six sites was calculated using the Sorensen's Similarity Index (Mueller-Dombois & Ellenberg 1974).

Sorensen's quotient of similarity = [2c/(a+b)]*100

where (a) and (b) are the number of species between sites A and B, and (c) is the number of species in common.

Species diversity and Evenness

Shannon's Diversity Index (H), which is a widely employed species diversity index to characterize diversity in a community, was used in order to calculate the diversity for each of the six sites over four sampling periods. Shannon's diversity index accounts for both abundance and evenness of the species present. The proportion of species *i* relative to the total number of species (p_i) is calculated, and then multiplied by the natural logarithm of this proportion ($\ln p_i$). The resulting product is summed across species, and multiplied by -1:

$$H = -\sum_{i=1}^{s} p_i \ln p_i$$

Shannon's Diversity Index was calculated using the software package Species Diversity & Richness Version 2.65 (Pisces Conservation 2001). A randomization test for a significant difference in diversity between two samples was undertaken using the software package. This test resamples 10 000 times from a distribution of species abundances produced by a summation of the two samples.

Equitability or evenness (i.e. the pattern of distribution of individuals between the species) for each site was also calculated using the software package Species Diversity & Richness Version 2.65.

Cluster analysis

Cluster analyses for each sampling using species cover of all species occurring within each site were undertaken using STATISTICA 7.0. The purpose of this algorithm is to join together transects into successively larger clusters, using a measure of similarity or distance of species composition and cover among sites. In order to cluster sites with similar species cover, Euclidean distances were calculated

using the single linkage (or "nearest neighbour") rule. In the single linkage rule, as larger and larger clusters are formed of less and less similar objects, the distance between any two clusters is determined by the closest objects in those two clusters.

Importance Values

Average Importance Values of all species occurring in each site were calculated using relative cover, relative frequency and relative density as parameters of importance, summed to give a maximum importance value of 300 (Mueller-Dombois & Ellenberg 1974).

Importance Value of species *i* = relative cover of species *i* + relative frequency of species *i* + relative density of species *i.*

Species cover

Within each plot, species cover occurring along line transects as well as the species cover within clump and inter-clump areas were grouped for analysis. The average species cover was calculated for each site over the four sampling periods (n=4).

Growth form, persistence (annual versus perennial) and pollination and dispersal guilds

Growth form and persistence of species identified in the study area is based on Germishuizen & Meyer (2003). The allocation of species to pollination and seed dispersal guilds is based on personal observations (Milton personal observation).

Vegetation structure

Since the age of individuals sampled is not known, the analysis of vegetation structure was based on the plant height. Height (cm) class frequency histograms were produced for the number of individuals and cover for each site over the four sampling periods. The number of individuals and cover were grouped into ten even-sized height classes.

The distribution of individuals and cover across size classes between sites were compared using contingency tables. The expected values were calculated based on the null hypothesis of independence and tested using the chi-squared statistic (STATISTICA 7.0).

4.3 RESULTS

4.3.1 SPECIES COMPOSITION AND SIMILARITY

The Sorensen's quotient of similarity calculated for each site (Table 4.1) shows that the rehabilitation sites are more similar to one another in terms of species composition than they are to the reference sites. Sites S1 and S3 (both rehabilitation scenarios including topsoil) were the most similar in all four sampling periods. The similarity between the rehabilitation sites was variable, but all had a quotient above 50%.

Site R1 had a greater similarity to the rehabilitation sites than to site R2. The sites where topsoil was replaced (sites S1 and S3) had a greater similarity to site R1 than those where no topsoil was replaced (sites S2 and S4). Site R2 was most similar to site R1 (although these sites are located within different vegetation units) and was the least similar to the rehabilitation sites. A list of all species sampled during this study and the sites in which they occurred is presented in Appendix A.

Cluster analyses (based on vegetation species richness and cover) confirmed that, despite seasonal differences, the rehabilitation sites were more similar to one another than to reference sites. The reference sites are separated from the rehabilitation sites within the first two levels of division (Figure 4.3). Generally, site R2 was separated off first, showing that site R1 is more similar in terms of species richness and cover to the rehabilitation sites than site R2. There appears to be no clear trend of clustering within the rehabilitation sites.

Table 4.1: Sorensen's quotient of similarity between sites over four sampling periods. Bold values indicate percentages greater than 50%. The six sites included: Reference site R1 (*Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld); Reference site R2 (*Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld); Rehabilitation site S1 (topsoil only); Rehabilitation site S2 (seeding only); Rehabilitation site S3 (topsoil and plant translocation); and Rehabilitation site S4 (seeding and plant translocation).

		R2	S1	S2	S3	S4
February 2004	R1	**56.14**	41.86	33.33	**50.00**	32.56
	R2		33.33	21.28	31.37	25.93
	S1			**60.61**	**81.08**	**70.00**
	S2				**66.67**	**60.61**
	S3					**75.68**
September 2004	R1	40.91	46.43	29.63	39.22	32.79
	R2		30.77	28.95	24.66	33.73
	S1			**59.09**	**82.93**	**54.90**
	S2				**61.54**	**73.47**
	S3					**52.17**
February 2005	R1	40.00	**54.90**	38.30	**52.00**	35.71
	R2		29.09	23.53	33.33	30.00
	S1			**66.67**	**88.89**	**66.67**
	S2				**63.41**	**63.83**
	S3					**64.00**
September 2005	R1	31.03	46.43	31.37	40.00	36.07
	R2		20.83	23.26	23.81	18.87
	S1			**68.29**	**75.00**	**54.90**
	S2				**51.43**	**65.22**
	S3					**53.33**

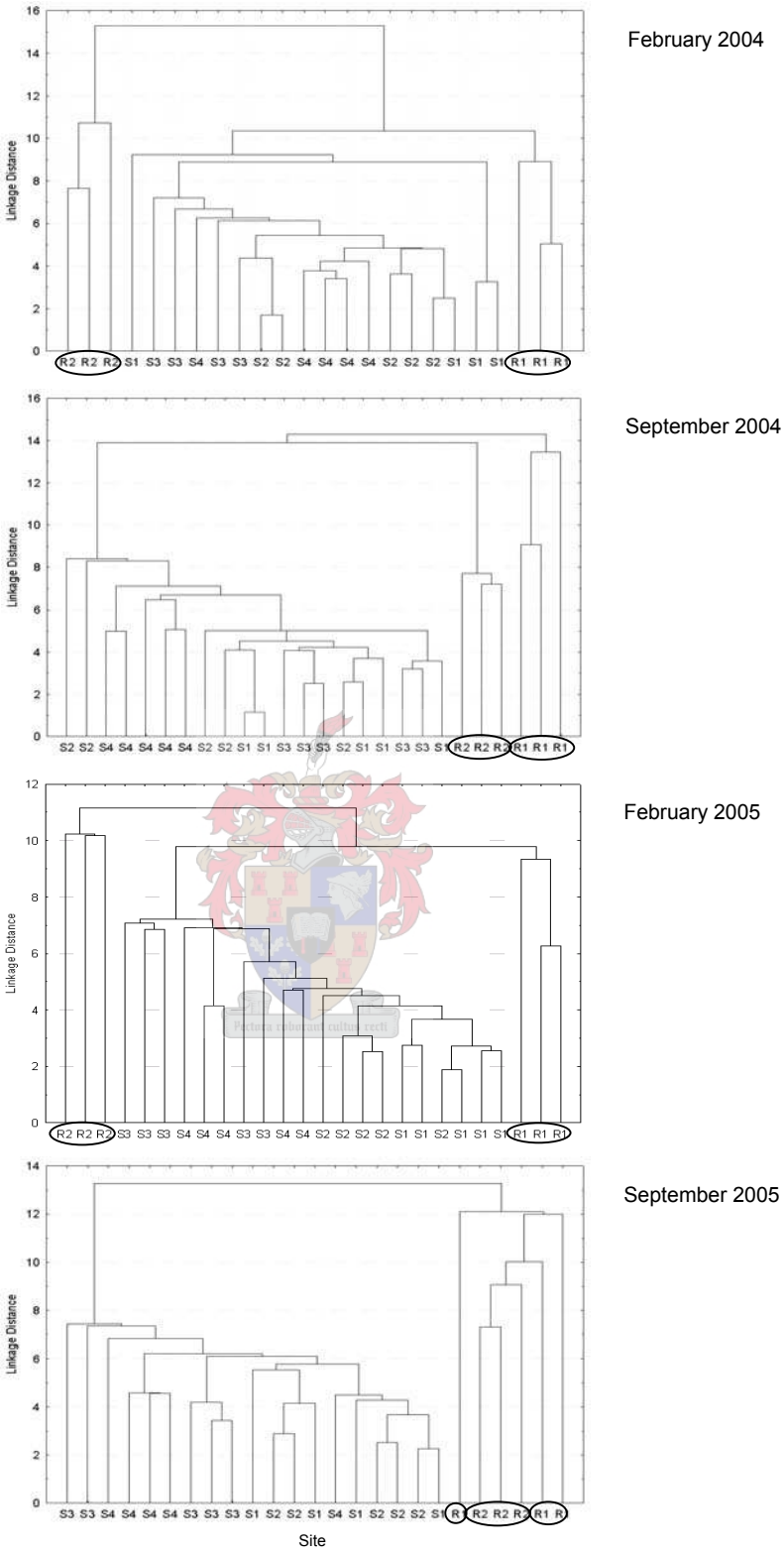

Figure 4.3: Cluster analyses of six Namakwa Sands sites over four sampling periods (n=3 for reference sites; n=5 for rehabilitation sites). The six sites are described in Table 4.1. The reference sites are circled.

4.3.2 SPECIES DIVERSITY AND EVENNESS

In all four sampling periods the two reference sites had a significantly greater species diversity and evenness than the rehabilitation sites, with site R2 generally having the greatest species diversity and evenness (Table 4.2). There was no clear trend of species diversity and evenness within the rehabilitation sites. Site S3 had a significantly greater species diversity and evenness in February 2004 and February 2005 than the other three rehabilitation sites. However, site S4 had the greatest species diversity and evenness in September 2004 and September 2005, but these differences were not always significant between sites (Table 4.2).

Table 4.2: Shannon Wiener diversity index (H) & Equitability or Evenness (J) at six Namakwa Sands sites over four sampling periods (n=3 for reference sites; n=5 for rehabilitation sites). Values with different superscripts are different at $p<0.05$. The six sites are described in Table 4.1.

Site	February 2004		September 2004		February 2005		September 2005	
	H	J	H	J	H	J	H	J
R1	2.42^{a}	0.63^{a}	2.62^{a}	0.61^{a}	2.57^{a}	0.65^{a}	2.76^{a}	0.64^{a}
R2	2.69^{b}	0.7^{b}	3.167^{b}	0.74^{b}	2.52^{a}	0.64^{a}	2.82^{a}	0.65^{a}
S1	1.68^{c}	0.44^{c}	2.07^{c}	0.48^{c}	1.9^{b}	0.48^{b}	2.37^{b}	0.55^{b}
S2	1.74^{c}	0.45^{c}	2.18^{cd}	0.51^{cd}	1.85^{b}	0.47^{b}	2.07^{c}	0.48^{c}
S3	1.98^{d}	0.52^{d}	2.08^{c}	0.48^{c}	2.3^{c}	0.59^{c}	2.09^{c}	0.48^{c}
S4	1.65^{c}	0.43^{c}	2.31^{d}	0.54^{d}	2.06^{b}	0.52^{b}	2.39^{b}	0.55^{b}

The lower evenness found within the rehabilitation sites represents a more uneven distribution of species found in these sites and greater dominance by a single or a few species. This finding is more apparent when comparing the Importance Values calculated for each species in each site (Table 4.3). The rehabilitation sites were dominated by one or two species with *Tetragonia fruticosa* generally being the most dominant species. Only site S4 had a species more dominant than *Tetragonia fruticosa* and that was the grass *Ehrharta calycina*, which was one of the species seeded at that site. *Ehrharta calycina* was also dominant in site S2, even though it was not seeded in that site. *Othonna cylindrica*, which was translocated in site S3, was the second most dominant species in that site. The reference sites, on the other hand, were characterised by a more even spread of Importance Values with a greater number of dominant species. Site R1 was dominated by the grass, *Chaetobromus involucratus*, followed by *Ruschia versicolor, Tetragonia fruticosa, Tripteris oppositifolia, Asparagus capensis* var. *litoralis* and *Zygophyllum morgsana.* Site R2 was dominated by *Lampranthus suavissimus, Willdenowia incurvata, Chrysocoma ciliata* and *Tripteris oppositifolia*. From this it is apparent that the two reference sites, which are located within different vegetation units, are dominated by a different assembly of species. A few of these dominant species were also found to occur in the rehabilitation sites, e.g. *Tetragonia* fruticosa (all sites), *Lampranthus suavissimus* (all sites), *Ruschia versicolor* (sites S1, S3 & S4) and *Tripteris oppositifolia* (site S1).

Table 4.3: Average Importance Value (IV) per species per site (n=4). Only those species with an Importance Value greater than 5 have been included in the table. The six sites are described in Table 4.1.

Site R1		Site R2		Site S1		Site S2		Site S3		Site S4	
Species	IV	Species	IV	Species	IV	Species	IV	Species	IV	Species	IV
Chaetobromus involucratus	40.35	*Lampranthus suavissimus*	35.33	*Tetragonia fruticosa*	132.29	*Tetragonia fruticosa*	111.84	*Tetragonia fruticosa*	83.41	*Ehrharta calycina*	74.74
Ruschia versicolor	31.06	*Willdenowia incurvata*	29.94	*Conicosia elongata*	24.44	*Ehrharta calycina*	38.29	*Othonna cylindrica*	31.99	*Tetragonia fruticosa*	59.26
Tetragonia fruticosa	29.35	*Chrysocoma ciliata*	24.76	*Lampranthus suavissimus*	17.19	*Eragrostis curvula*	25.35	*Conicosia elongata*	28.57	*Conicosia elongata*	17.23
Tripteris oppositifolia	24.99	*Tripteris oppositifolia*	25.12	*Mesembryanthemum guerichianum*	15.84	*Conicosia elongata*	24.80	*Mesembryanthemum guerichianum*	25.89	*Lampranthus suavissimus*	14.82
Asparagus capensis var. litoralis	23.13	*Trichogyne repens*	15.52	*Manochlamys albicans*	12.57	*Galenia africana*	16.33	*Lampranthus suavissimus*	18.84	*Oncosiphon grandiflorum*	14.73
Zygophyllum morgsana	22.99	*Asparagus capensis* var. *litoralis*	15.11	*Galenia africana*	12.38	*Salsola kali*	12.14	*Salsola kali*	15.17	*Eragrostis curvula*	12.89
Hermannia subgen. *Mahernia*	12.31	*Hermannia cuneifolia*	12.26	*Ruschia versicolor*	10.28	*Oncosiphon grandiflorum*	9.17	*Ruschia versicolor*	13.12	*Othonna cylindrica*	12.19
Ruschia paripetala	12.08	*Othonna cylindrica*	12.02	*Lebeckia spinescens*	9.22	*Atriplex lindleyi* spp. *Inflata*	8.53	*Oncosiphon grandiflorum*	11.15	*Galenia africana*	11.37
Mesembryanthemum guerichianum	11.70	*Stipagrostis namaquensis*	11.09	*Amellus flosculosus*	7.95	*Mesembryanthemum guerichianum*	8.33	*Atriplex lindleyi* spp. *Inflata*	10.86	*Erucastrum strigosum*	11.07
Eriocephalus racemosus var. *affinis*	11.02	*Ruschia versicolor*	9.82	*Atriplex lindleyi* spp. *Inflata*	7.79	*Lampranthus suavissimus*	7.80	*Manochlamys albicans*	10.05	*Hermannia cuneifolia*	10.38
Hermannia disermifolia	8.78	*Stipagrostis obtusa*	9.27	*Oncosiphon grandiflorum*	7.34	*Manochlamys albicans*	6.69	*Exomis microphylla* var. *axyrioides*	9.21	*Wahlenbergia adpressa*	10.05
Ficinia argyropa	8.12	*Stoebe nervigera*	8.77	*Exomis microphylla* var. *axyrioides*	6.91	*Lebeckia spinescens*	6.08	*Galenia africana*	8.24	*Atriplex lindleyi* spp. *Inflata*	6.28
Lampranthus suavissimus	7.36	*Tetragonia fruticosa*	7.47	*Tripteris oppositifolia*	6.63			*Zygophyllum morgsana*	6.87	*Ruschia versicolor*	5.59
Othonna cylindrica	6.98	*Ficinia argyropa*	6.13	*Othonna cylindrica*	5.25			*Atriplex semibaccata*	5.77	*Mesembryanthemum guerichianum*	5.39
Didelta carnosa	5.28	*Euphorbia mauritanica*	5.85								
		Arctotis auriculata	5.12								

4.3.3 SPECIES RICHNESS

In all four sampling periods, the two reference sites had a greater total and average species richness than the rehabilitation sites (Table 4.4 & 4.5), with site R2 having the greatest total and average species richness (but not always significant). Similarly, the reference sites also had a greater total species richness in clumps and inter-clump areas (Table 4.4). Within the rehabilitation sites, site S4 generally had the greatest total and average species richness followed by site S1 (Table 4.4 & 4.5). Sites S2 had a slightly greater species richness than site S3 (Table 4.4), but site S3 had a greater average species richness in all sampling periods (Table 4.5). Site S4 also had the greatest total species richness in clump and inter-clump areas followed by site S1 and S3. Site S2 had the lowest total species richness in clump and inter-clump areas (Table 4.4).

Within site R1 and all rehabilitation sites more species were found to occur within clumps than inter-clump areas (Table 4.4). However, site R2 had a slightly greater species richness in inter-clump areas. Within all six sites there was a greater total number of perennial species than annual species. The reference sites generally had a similar number of annual species to the rehabilitation sites. However, the reference sites had a greater number of perennial species than the rehabilitation sites (Table 4.4).

Sites S1, S3 (except September 2005) and S4 achieved at least 30% of the number of species occurring in site R1 within approximately four years, thereby achieving the three-year objective of at least 30% indigenous vegetation species in common with the reference site (EEU 1990) (Table 4.6). However, this was not true for site R2. Site S2 performed the worst in terms of the total number of species present and the number of species in common with the reference sites. It should be noted that the final rehabilitation objective (in 2006) of at least 60% of the average number of indigenous vegetation species (EEU 1990) has not yet been met by any of the rehabilitation sites.

There were differences in species richness between summer and winter with a greater number of species present during winter, but these findings were not always significant and were not true for all summer and winter samplings (Table 4.4).

4.3.4 GROWTH FORMS AND FUNCTIONAL GUILDS

The growth forms of all species encountered in the study area are presented in Appendix A and summarised in Table 4.4. Most notably is the absence of geophytes (e.g. *Albuca* spp., B*abiana brachystachys, Bulbine praemorsa, Chlorophytum rangei, Eriospermum* sp., *Ornithogalum* sp., *Pelargonium triste, Trachyandra* spp.), Restionaceae (e.g. *Ischyrolepis gaudichaudianus, Willdenowia incurvata*), *Crassula* spp., *Asparagus* spp. and *Euphorbia* spp. in rehabilitation sites. Few grass species, other than those seeded (*Eragrostis curvula*, *Ehrharta calycina* and *Sorghum* sp.), were found within rehabilitation sites.

In an area dominated by strong winds, the reference and rehabilitation sites were dominated by vegetation species that are insect pollinated rather than relying on wind pollination (Table 4.4). However, the majority of the species found within the reference and rehabilitation sites were wind dispersed. The second most common dispersal guild within reference sites was bird dispersal, whereas in rehabilitation sites it was mammal dispersal.

Table 4.4: Total, clump and inter-clump species richness, number of annual and perennial species, growth forms and the breakdown of pollination and dispersal guilds for those species found in each site over all four sampling periods (i.e. over a line transect distance of 600 m and 1 000 m for reference and rehabilitation sites, respectively). Unknowns relate to those species that could not be identified due to quality of specimen or sterile material. The six sites are described in Table 4.1. A list of all species sampled during this study and in which sites they occurred is presented in Appendix A.

		Site					
		R1	R2	S1	S2	S3	S4
Species richness	Total	56	83	36	33	31	49
	Clumps	48	69	27	22	27	38
	Inter-clumps	44	70	23	19	22	31
Persistence	Perennial	46	64	25	22	23	32
	Annual	8	12	9	8	8	13
	Unknown	2	7	2	3	-	4
Growth form	Shrub	18	27	11	6	9	8
	Dwarf shrub	9	13	4	7	5	6
	Geophyte	7	8	-	-	-	1
	Herb	12	18	11	9	7	18
	Succulent	6	4	6	6	7	7
	Graminoid	2	4	3	3	3	4
	Scrambler / climber	2	2	-	-	-	-
	Unknown	-	7	1	2	-	5
Pollination guild	Insect	47	68	26	23	21	37
	Wind	7	11	9	9	9	10
	Bird	2	2	1	-	1	1
	Unknown	1	2	1	1	-	1
Dispersal guild	Wind	28	44	15	14	13	17
	Hygrochastic	5	4	4	5	5	6
	Explosive	4	4	5	1	3	6
	Bird	8	10	4	2	3	2
	Gravity	4	10	1	3	1	7
	Mammal	6	8	6	7	6	10
	Unknown	1	3	1	1	-	1

Table 4.5: Average (± standard deviation) species richness, vegetation cover, perennial cover, annual cover, clump cover and abundance at six Namakwa Sands sites (n=3 for reference sites; n=5 for rehabilitation sites) over four sampling periods. Values with different superscripts (letters: differences between sites within a sampling; roman numerals: differences between samplings within a site) are different at $p<0.05$. The six sites are described in Table 4.1.

Site	Sampling	Species richness	Cover (%)	Perennials cover (%)	Annuals cover (%)	Clump cover (%)	Abundance
R1	February 2004	13.67 ±1.53$^{a(i)}$	34.75 ±3.81$^{a(i/ii)}$	37.17 ±6.72$^{ab(i/ii)}$	1.63 ±1.37$^{ab(i/ii)}$	22.34 ±4.62$^{a(i)}$	57.67 ±3.79$^{a(i)}$
	September 2004	16.67 ±1.63$^{a(i/ii)}$	38.66 ±6.03$^{a(i)}$	42.17 ±7.73$^{a(i)}$	0.02 ±0.03$^{a(iii)}$	27.39 ±6.60$^{a(i/ii)}$	75.33 ±5.86$^{a(i)}$
	February 2005	14.33 ±2.08$^{a(i)}$	26.66 ±3.72$^{a(ii)}$	27.83 ±5.13$^{ac(ii)}$	0.39 ±0.46$^{ab(i)}$	22.73 ±5.06$^{a(i/ii)}$	56.33 ±5.13$^{a(i)}$
	September 2005	19.67 ±3.51$^{a(ii)}$	32.48 ±2.60$^{ac(i/ii)}$	34.82 ±5.59$^{a(i/ii)}$	1.02 ±0.96$^{ab(ii)}$	29.75 ±5.58$^{a(ii)}$	73.33 ±16.50$^{a(i)}$
R2	February 2004	17.00 ±1.73$^{b(i)}$	34.28 ±4.89$^{a(i)}$	38.39 ±7.92$^{b(i)}$	0.11 ±0.20$^{ab(i)}$	23.81 ±5.22$^{a(i)}$	56.00 ±5.20$^{a(i)}$
	September 2004	23.00 ±4$^{b(ii)}$	31.13 ±2.17$^{a(i/ii)}$	34.23 ±1.89$^{a(i/ii)}$	0.23 ±0.13$^{a(ii)}$	15.95 ±4.29$^{b(ii)}$	60.00 ±8.54$^{ac(i)}$
	February 2005	14.67 ±1.53$^{a(i)}$	28.81 ±5.42$^{a(i/ii)}$	31.69 ±5.65$^{a(i/ii)}$	0.21 ±0.26$^{ab(i/ii)}$	20.43 ±4.67$^{ac(i/ii)}$	63.33 ±4.16$^{a(i)}$
	September 2005	16.67 ±0.58$^{a(i)}$	25.75 ±1.85$^{ab(ii)}$	28.44 ±3.69$^{ab(ii)}$	0.11 ±0.03$^{a(i/ii)}$	18.77 ±4.06$^{b(i/ii)}$	46.67 ±8.14$^{b(i)}$
S1	February 2004	8.40 ±2.02$^{c(i/iii)}$	30.74 ±7.82$^{a(i)}$	29.78 ±7.94$^{a(i)}$	1.40 ±0.68$^{a(i)}$	11.45 ±4.85$^{b(i)}$	33.60 ±7.02$^{b(i)}$
	September 2004	10.60 ±2.30$^{c(i/ii)}$	24.21 ±4.71$^{b(ii)}$	24.24 ±5.14$^{b(i)}$	0.52 ±0.57$^{a(i)}$	9.29 ±5.06$^{b(i)}$	35.00 ±3.94$^{b(i)}$
	February 2005	8.00 ±1.73$^{b(iii)}$	17.34 ±0.19$^{b(iii)}$	16.89 ±3.59$^{bd(ii)}$	0.42 ±0.52$^{ab(i)}$	9.15 ±3.62$^{b(i)}$	24.00 ±2.24$^{b(i)}$
	September 2005	11.60 ±2.30$^{bd(ii)}$	24.14 ±7.80$^{bd(ii)}$	23.28 ±6.81$^{bc(i/ii)}$	0.97 ±1.20$^{ab(i)}$	13.93 ±6.00$^{bc(i)}$	32.00 ±10.27$^{bc(i)}$
S2	February 2004	6.60 ±1.14$^{c(i)}$	19.44 ±5.40$^{b(i)}$	16.92 ±7.62$^{c(i)}$	2.84 ±1.91$^{ab(i)}$	8.15 ±6.25$^{b(i)}$	23.00 ±3.74$^{b(i)}$
	September 2004	9.60 ±1.67$^{c(ii)}$	22.89 ±7.79$^{b(i)}$	22.48 ±7.63$^{b(i)}$	0.64 ±0.46$^{a(ii)}$	8.96 ±5.39$^{b(i)}$	32.20 ±1.79$^{b(i)}$
	February 2005	7.60 ±0.89$^{b(i/ii)}$	18.38 ±1.95$^{b(i)}$	18.36 ±2.20$^{b(i)}$	0.25 ±0.24$^{a(ii)}$	8.90 ±2.70$^{b(i)}$	30.40 ±5.13$^{b(i)}$
	September 2005	8.40 ±1.52$^{c(i/ii)}$	19.63 ±4.84$^{b(i)}$	18.98 ±4.90$^{c(i)}$	0.68 ±0.42$^{ab(ii)}$	8.62 ±4.21$^{c(i)}$	26.60 ±4.10$^{c(i)}$
S3	February 2004	7.80 ±1.10$^{c(i)}$	21.00 ±5.13$^{b(i)}$	20.13 ±5.88$^{c(i)}$	3.37 ±3.32ab$^{(i/ii/iii)}$	10.52 ±4.20$^{b(i)}$	25.40 ±5.22$^{b(i)}$
	September 2004	10.00 ±2.12$^{c(i/ii)}$	26.29 ±3.07$^{b(i)}$	24.37 ±3.34$^{b(i)}$	2.40 ±1.25$^{a(ii)}$	12.15 ±2.95$^{b(i)}$	45.60 ±5.90$^{bc(ii)}$
	February 2005	12.20 ±2.39$^{a(ii)}$	25.70 ±4.30$^{a(i)}$	20.66 ±4.54$^{cd(i)}$	5.22 ±2.07$^{ab(i)}$	13.49 ±3.12$^{bc(i)}$	62.20 ±15.22$^{a(ii)}$
	September 2005	10.40 ±2.30$^{bc(ii)}$	34.40 ±2.31$^{c(ii)}$	34.18 ±2.05$^{a(ii)}$	1.39 ±0.15$^{b(iii)}$	20.19 ±3.66$^{b(ii)}$	46.80 ±6.30$^{b(ii)}$
S4	February 2004	7.40 ±0.89$^{c(i)}$	20.88 ±6.41$^{b(i)}$	21.74 ±7.24$^{c(i)}$	0.05 ±0.11$^{b(i)}$	10.98 ±4.26$^{b(i)}$	27.40 ±7.70$^{b(i)}$
	September 2004	14.00 ±1.00$^{a(ii)}$	26.15 ±3.39$^{b(i/ii)}$	25.12 ±4.71$^{b(i)}$	2.57 ±2.19$^{a(ii)}$	12.98 ±3.02$^{b(i)}$	48.40 ±7.09$^{bc(ii)}$
	February 2005	12.00 ±2.55$^{a(ii)}$	22.45 ±5.57$^{ab(i)}$	22.02 ±6.20$^{cd(i)}$	1.27 ±0.45$^{b(ii)}$	11.61 ±5.28$^{b(i)}$	50.80 ±8.17$^{a(ii)}$
	September 2005	13.20 ±1.30$^{d(ii)}$	28.96 ±4.87$^{acd(ii)}$	27.74 ±5.49$^{ab(i)}$	1.54 ±1.20$^{ab(ii)}$	15.60 ±2.20$^{b(i)}$	48.40 ±7.13$^{b(ii)}$

Table 4.6: Proportional cover and species richness within rehabilitation sites compared to reference sites. Bold values indicate percentages greater than 30% for species richness and greater than 50% for vegetation cover, which are both three-year objectives at Namakwa Sands. For species richness calculations, only those species in common with the reference sites were used. The six sites are described in Table 4.1.

Site	Sampling	Species richness		Cover	
		% of R1 spp	% of R2 spp	% of R1 cover	% of R2 cover
S1	February 2004	**39.13**	26.47	**88.47**	**89.67**
	September 2004	**39.39**	21.82	**62.62**	**77.77**
	February 2005	**42.86**	25.00	**65.04**	**60.19**
	September 2005	**56.25**	22.92	**74.33**	**93.75**
S2	February 2004	26.09	14.71	**55.94**	**56.7**
	September 2004	27.27	20.00	**59.20**	**73.53**
	February 2005	28.57	18.75	**68.93**	**63.79**
	September 2005	26.83	18.75	**60.44**	**76.23**
S3	February 2004	**43.48**	23.53	**60.43**	**61.25**
	September 2004	**30.30**	16.36	**68.01**	**84.47**
	February 2005	**46.43**	28.13	**96.41**	**89.23**
	September 2005	29.27	20.83	**105.92**	**133.59**
S4	February 2004	**34.78**	20.59	**60.09**	**60.91**
	September 2004	**30.30**	25.45	**67.65**	**84.02**
	February 2005	**32.14**	28.13	**84.20**	**77.93**
	September 2005	**31.71**	27.08	**89.15**	**112.44**

4.3.5 VEGETATION COVER

The reference sites generally had a greater vegetation cover, perennial cover and clump cover than the rehabilitation sites (except September 2005 where site S3 had the greatest vegetation cover and site S4 had a greater cover than site R2), but these differences were not always significant (Table 4.5). Site R1 generally had a greater vegetation and clump cover compared to site R2, but these differences were mostly not significant. Within the reference sites, sites S3 and S4 (i.e. translocation sites) had a greater vegetation, perennial and clump cover compared to sites S1 (except February 2004) and S2, but these differences were not always significant. Site S2 generally had the poorest vegetation, perennial and clump cover.

The cover of annuals was far less than that for perennial species (Table 4.5). There was no obvious trend between reference and rehabilitation sites. Sites S3 and S4 generally had the greatest cover of annuals, but these differences were mostly not significant.

All rehabilitation sites achieved a vegetation cover of greater than 55% of the reference sites, thereby achieving the three-year objective of an average cover of 50% of the reference sites (EEU 1990) (Table 4.6). In certain sampling periods, sites S1, S3 and S4 met the five-year objective of an average vegetation cover of at least 80% relative to the cover of the references sites. In September 2005, sites S3 and S4 had a greater vegetation cover than the reference sites. Site S2 appears to have performed the worst in terms of average vegetation cover compared to the reference sites (Table 4.6).

Cover within the reference sites was dominated by a different assemblage of species (Table 4.7). Site R1 was dominated by *Tetragonia fruticosa, Chaetobromus involucratus, Tripteris oppositifolia, Ruschia versicolor, Zygophyllum morgsana* and *Asparagus capensis* var. *litoralis*. Whereas site R2 was dominated by *Lampranthus suavissimus, Tripteris oppositifolia, Willdenowia incurvata, Chrysocoma ciliata* and *Asparagus capensis* var. *litoralis*. Cover within the rehabilitation sites was dominated almost entirely by *Tetragonia fruticosa*. Although *Tetragonia fruticosa* also had the greatest cover within site R1, the rehabilitation sites had values between 2 and 3.5 times greater than in site R1 (Table 4.7), which indicates a far greater dominance in terms of cover by this species in the rehabilitation sites.

There were differences in vegetation, perennial and clump cover between summer and winter. In general, there was greater cover during winter in all sites (except site R2), but this was not true between all summer and winter samplings (Table 4.5).

4.3.6 ABUNDANCE

The reference sites generally had a significantly greater number of individuals than the rehabilitation sites in all four sampling periods, except in February and September 2005 where site S3 had the second largest abundance (Table 4.5). Within the rehabilitation sites, sites S3 and S4 (i.e. translocation sites) generally had a greater abundance (not always significant) than sites S1 (except February 2004) and S2. Site S2 generally had the lowest abundance. In general, there was also an inter-seasonal variation in abundance between February 2004 (summer) and September 2004 (winter) with more occurring in the winter sampling periods, but these differences were not significant for sites S3 and S4 (Table 4.5). In 2005 only sites R1 and S2 showed a greater winter abundance.

Table 4.7: Average species cover occurring along line transects per site over the four sampling periods (n=12 for reference sites; n=20 for rehabilitation sites). Only those species with a cover of greater than 0.5% have been included in the table. The six sites are described in Table 4.1.

Site R1		Site R2		Site S1		Site S2		Site S3		Site S4	
Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)
Tetragonia fruticosa	4.91	*Lampranthus suavissimus*	4.76	*Tetragonia fruticosa*	18.38	*Tetragonia fruticosa*	13.33	*Tetragonia fruticosa*	14.65	*Tetragonia fruticosa*	9.93
Chaetobromus involucratus	4.86	*Tripteris oppositifolia*	4.19	*Galenia africana*	0.95	*Eragrostis curvula*	1.38	*Othonna cylindrica*	2.68	*Ehrharta calycina*	5.73
Tripteris oppositifolia	4.48	*Willdenowia incurvata*	3.95	*Manochlamys albicans*	0.87	*Ehrharta calycina*	1.38	*Conicosia elongata*	1.60	*Galenia africana*	1.48
Ruschia versicolor	4.10	*Chrysocoma ciliata*	2.32	*Conicosia elongata*	0.86	*Galenia africana*	1.22	*Lampranthus suavissimus*	1.25	*Conicosia elongata*	1.32
Zygophyllum morgsana	4.07	*Asparagus capensis var. litoralis*	2.01	*Lampranthus suavissimus*	0.77	*Conicosia elongata*	0.86	*Mesembryanthemum guerichianum*	1.12	*Lampranthus suavissimus*	1.05
Asparagus capensis var. litoralis	3.39	*Othonna cylindrica*	1.24			*Salsola kali*	0.64	*Salsola kali*	1.05	*Othonna cylindrica*	1.00
Eriocephalus racemosus var. *affinis*	1.73	*Ruschia versicolor*	1.03					*Exomis microphylla* var. *axyrioides*	0.86	*Eragrostis curvula*	0.92
Hermannia disermifolia	1.28	*Trichogyne repens*	0.88					*Manochlamys albicans*	0.8	*Hermannia cuneifolia*	0.75
Ruschia paripetala	1.12	*Stoebe nervigera*	0.67					*Galenia africana*	0.71	*Wahlenbergia adpressa*	0.61
Othonna cylindrica	0.98	*Stipagrostis namaquensis*	0.64					*Ruschia versicolor*	0.65	*Oncosiphon grandiflorum*	0.55
Lycium ferocissimum	0.70	*Tetragonia fruticosa*	0.59					*Atriplex lindleyi spp. Inflata*	0.55		
Mesembryanthemum guerichianum	0.66	*Eriocephalus racemosus* var. *affinis*	0.57								
Hermannia subgen. *Mahernia*	0.55	*Hermannia cuneifolia*	0.55								
		Pteronia paniculata	0.55								

4.3.7 CLUMPING

Clumping appears to be an important feature of the vegetation in both reference and rehabilitation sites. However, the reference sites had a significantly greater number of clumps than the rehabilitation sites in all sampling periods (Table 4.8). The number of clumps between the two reference sites was generally not significantly different (except in September 2005 where site R1 had a significantly greater number of clumps). Within the rehabilitation sites, site S3 generally had the greatest number of clumps with site S2 having the least, but these differences were generally not significant. Related to the number of clumps is the inter-clump distance. Clumps within rehabilitation sites were found to be further apart (i.e. a greater inter-clump distance) than the reference sites (Table 4.8). However, these differences were not found to be significant due to the large standard deviations found within all sites. These large standard deviations show a large amount of plot-to-plot variation. Within the rehabilitation sites, clumps within sites S3 and S4 (i.e. translocation sites) tended to be closer together than in sites S1 and S2, with site S2 generally having the greatest inter-clump distances.

In general, rehabilitation sites S4, S3 and S2 had a greater average clump size than the reference sites with site S4 having the largest average clump size (Table 4.8). Within the reference sites, site R1 had a greater average clump size than site R2, but these differences were only significant in February and September 2005.

Site R1 generally had the greatest clump height followed by sites R2 and S4, but these differences were not always significant (Table 4.8). There were no obvious differences in clump height between the remaining rehabilitation sites.

Clumped areas had a greater cover than inter-clump areas in all sites and across all sampling periods (Table 4.8). The rehabilitation sites generally had a greater cover within clumps (i.e. cover within the 80 cm ring) than the reference sites but these differences were generally not significant. The reference sites generally had a greater inter-clump cover than rehabilitation sites, but these differences were not always significant (Table 4.8).

Clumps and inter-clump areas within reference and rehabilitation sites were dominated in terms of cover by a different assemblage of species (Table 4.9). Clumps within rehabilitation sites were dominated almost entirely by *Tetragonia fruticosa*, which grew very large (up to 7.8 m meters long and 4.5 m wide) along windbreaks within rehabilitation areas. Clumps within reference sites, on the other hand, were characterised by a more even spread of cover with a greater number of dominant species. Clumps within site R1 were generally dominated by *Zygophyllum morgsana, Tetragonia fruticosa, Tripteris oppositifolia, Ruschia versicolor, Asparagus capensis* var. *litoralis, Chaetobromus involucratus* and *Eriocephalus racemosus* var. *affinis.* Clumps within site R2 were generally dominated by *Tripteris oppositifolia, Willdenowia incurvata, Lampranthus suavissimus, Tetragonia fruticosa, Othonna cylindrica, Asparagus capensis* var. *litoralis, Chrysocoma ciliata* and *Stoebe nervigera* (Table 4.9). Species cover within inter-clump areas of rehabilitation sites (except site S4) were also dominated by *Tetragonia fruticosa.* However, site S4 was dominated by the grass *Ehrharta calycina*, which was seeded in that site. Inter-clumps within site R1 were generally dominated by *Chaetobromus involucratus* and *Ruschia versicolor,* while site R2 was dominated by *Chrysocoma ciliata* and *Lampranthus suavissimus.*

Table 4.8: Average (± standard deviation) number of clumps, clump size, clump height, inter-clump distance, clump cover within 80 cm ring and inter-clump cover within 80 cm ring at six Namakwa Sands sites (n=3 for reference sites; n=5 for rehabilitation sites) over four sampling periods. Values with different superscripts (letters: differences between sites within a sampling; roman numerals: differences between samplings within a site) are different at $p<0.05$. The six sites are described in Table 4.1.

Site	Sampling	Number of clumps	Clump size (m^2)	Clump height (cm)	Inter-clump distance (cm)	Clump cover (%) in 80 cm ring	Inter-clump cover (%) in 80 cm ring
R1	February 2004	13.67 ±2.52$^{a(i)}$	1.79 ±0.57$^{a(i)}$	58.77 ±3.71$^{a(i)}$	259.21 ±40.27$^{a(i)}$	53.27 9.72$^{a(i/iii)}$	16.17 3.91$^{a(i)}$
	September 2004	19.33 ±4.04$^{a(ii)}$	1.67 ±0.58$^{a(i)}$	44.25 ±0.59$^{a(ii)}$	184.87 ±54.65$^{a(ii)}$	54.61 1.71$^{a(i)}$	11.15 3.81$^{a(ii)}$
	February 2005	14.30 ±3.21$^{a(i)}$	2.73 ±1.01$^{a(i)}$	66.72 ±9.59$^{a(i)}$	266.47 ±53.48$^{a(i)}$	61.07 4.96$^{a(iii)}$	12.63 3.56$^{a(ii)}$
	September 2005	14.67 ±3.06$^{a(i)}$	3.40 ±2.02$^{a(i)}$	85.72 ±16.14$^{a(iii)}$	248.54 ±87.08$^{a(i)}$	79.46 12.99$^{a(ii)}$	14.37 5.16$^{a(i/ii)}$
R2	February 2004	13.67 ±1.15$^{a(i/ii)}$	1.58 ±0.36$^{a(i)}$	44.70 ±2.80$^{bd(i)}$	271.10 ±51.88$^{a(i)}$	64.28 2.94$^{a(i)}$	10.76 2.54$^{ab(i)}$
	September 2004	11.33 ±3.21$^{b(i)}$	1.75 ±0.24$^{a(i)}$	50.58 ±13.74$^{a(i/ii)}$	382.09 ±94.93$^{a(ii)}$	51.65 6.06$^{a(ii)}$	12.54 0.59$^{a(ii)}$
	February 2005	17.33 ±5.13$^{a(ii)}$	1.00 ±0.23$^{b(i)}$	43.50 ±5.16$^{b(i)}$	237.33 ±84.60$^{a(i)}$	52.22 4.55$^{a(ii)}$	11.05 9.64$^{a(i/ii)}$
	September 2005	15.00 ±1.73$^{a(i/ii)}$	1.81 ±0.15$^{b(i)}$	60.61 ±14.22$^{b(ii)}$	272.86 ±51.05$^{a(i/ii)}$	59.12 10.54$^{a(i/ii)}$	2.26 1.57$^{a(i)}$
S1	February 2004	7.40 ±2.51$^{b(i)}$	1.48 ±0.44$^{a(i)}$	32.90 ±3.80$^{c(i/ii)}$	685.34 ±444.18$^{a(i)}$	63.27 5.86$^{a(i)}$	12.49 5.66$^{ab(i)}$
	September 2004	6.80 ±3.03$^{c(i)}$	1.49 ±0.53$^{a(i)}$	30.87 ±3.98$^{b(i)}$	687.59 ±408.30$^{a(i)}$	71.17 10.34$^{ab(ii/iii)}$	7.34 3.23$^{a(i)}$
	February 2005	6.40 ±2.41$^{b(i)}$	2.23 ±0.55$^{ab(i)}$	39.98 ±4.16$^{b(ii)}$	787.84 ±379.46$^{a(i)}$	72.19 7.00$^{a(i/iii)}$	6.85 3.37$^{a(i)}$
	September 2005	7.40 ±2.30$^{bc(i)}$	2.57 ±0.76$^{ab(i)}$	49.84 ±6.32$^{c(iii)}$	633.65 ±309.28$^{a(i)}$	82.54 7.88$^{a(ii)}$	8.26 3.09$^{a(i)}$
S2	February 2004	4.40 ±2.88$^{b(i)}$	2.19 ±1.80$^{ab(i)}$	36.83 ±7.97$^{bc(i)}$	1203.18 ±1250.56$^{a(i)}$	63.88 33.25$^{a(i)}$	3.81 2.43$^{b(i/ii)}$
	September 2004	5.20 ±1.79$^{c(i)}$	2.49 ±5.39$^{a(i/ii)}$	41.85 ±9.80$^{ac(i)}$	739.23 ±271.24$^{a(i)}$	85.52 9.54$^{ab(i)}$	3.60 1.72$^{a(i)}$
	February 2005	5.80 ±2.28$^{b(i)}$	2.23 ±0.87$^{ab(i)}$	39.56 ±3.92$^{b(i)}$	912.10 ±819.33$^{a(i)}$	79.86 8.54$^{a(i)}$	5.86 2.25$^{a(ii)}$
	September 2005	4.20 ±1.48$^{b(i)}$	3.70 ±1.45$^{a(ii)}$	39.83 ±6.42$^{d(i)}$	870.99 ±284.00$^{a(i)}$	86.88 15.55$^{a(i)}$	10.07 6.83$^{a(ii)}$
S3	February 2004	4.60 ±1.14$^{b(i)}$	2.02 ±0.67$^{a(i)}$	40.04 ±9.35$^{bc(i/ii)}$	869.61 ±609.49$^{a(i)}$	71.07 6.94$^{a(i)}$	8.27 6.71$^{ab(i/ii)}$
	September 2004	7.60 ±1.67$^{bc(ii)}$	2.56 ±1.23$^{a(i/ii)}$	32.98 ±6.14$^{bc(i)}$	556.16 ±131.67$^{a(i)}$	75.59 13.10$^{ab(i/ii)}$	8.75 2.67$^{a(i)}$
	February 2005	8.00 ±2.55$^{b(ii)}$	1.98 ±0.55$^{ab(i)}$	44.52 ±5.14$^{b(ii)}$	565.36 ±140.95$^{a(i)}$	78.64 4.83$^{a(i/ii)}$	12.94 2.22$^{a(ii)}$
	September 2005	9.60 ±1.67$^{c(ii)}$	3.77 ±0.91$^{a(ii)}$	46.44 ±3.52$^{cd(ii)}$	422.96 ±116.09$^{a(i)}$	87.48 4.54$^{a(ii)}$	13.97 7.37$^{a(ii)}$
S4	February 2004	7.00 ±2.92$^{b(i/ii)}$	3.43 ±1.09$^{b(i)}$	54.36 ±8.82$^{ad(i)}$	777.22 ±411.86$^{a(i)}$	85.09 7.84$^{a(i)}$	8.57 3.66$^{ab(i)}$
	September 2004	4.20 ±1.30$^{c(i)}$	5.15 ±2.32$^{b(ii)}$	45.83 ±5.44$^{a(i)}$	889.38 ±385.72$^{a(i)}$	94.75 10.13$^{b(ii)}$	11.78 8.75$^{a(i)}$
	February 2005	7.20 ±2.28$^{b(ii)}$	2.57 ±0.71$^{a(i)}$	48.03 ±2.11$^{b(i)}$	578.30 ±224.11$^{a(i)}$	71.62 3.03$^{a(iii)}$	10.09 1.76$^{a(i)}$
	September 2005	6.60 ±1.14$^{bc(ii)}$	3.70 ±0.64$^{a(i)}$	53.32 ±2.05$^{bc(i)}$	627.85 ±159.23$^{a(i)}$	86.43 5.69$^{a(i/iii)}$	8.37 3.59$^{a(i)}$

Table 4.9: Average species cover occurring within clumps and inter-clump areas over the four sampling periods (n=12 for reference sites; n=20 for rehabilitation sites). Only those species with a cover of greater than 0.5% have been included in the table. The six sites are described in Table 4.1.

	Site R1		Site R2		Site S1		Site S2		Site S3		Site S4	
	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)	Species	Ave cover (%)
Clumps	*Zygophyllum morgsana*	9.77	*Tripteris oppositifolia*	11.13	*Tetragonia fruticosa*	61.76	*Tetragonia fruticosa*	69.00	*Tetragonia fruticosa*	51.72	*Tetragonia fruticosa*	52.39
	Tetragonia fruticosa	9.50	*Willdenowia incurvata*	11.10	*Galenia africana*	2.73	*Galenia africana*	5.06	*Othonna cylindrica*	7.22	*Galenia africana*	7.07
	Tripteris oppositifolia	8.04	*Lampranthus suavissimus*	8.69	*Manochlamys albicans*	2.05	*Conicosia elongata*	1.97	*Lampranthus suavissimus*	3.68	*Othonna cylindrica*	6.03
	Ruschia versicolor	7.05	*Tetragonia fruticosa*	5.63	*Exomis microphylla* var. *axyrioides*	1.10	*Salsola kali*	1.36	*Manochlamys albicans*	3.40	*Conicosia elongata*	3.76
	Asparagus capensis	6.84	*Othonna cylindrica*	3.01	*Conicosia elongata*	0.96	*Manochlamys albicans*	1.19	*Exomis microphylla* var. *axyrioides*	2.39	*Lampranthus suavissimus*	3.72
	Chaetobromus involucratus	3.67	*Asparagus capensis*	2.96	*Lampranthus suavissimus*	0.89	*Ehrharta calycina*	0.79	*Galenia africana*	1.92	*Ehrharta calycina*	2.84
	Eriocephalus racemosus var. *affinis*	3.60	*Chrysocoma ciliata*	2.55	*Ruschia versicolor*	0.86	*Psilocaulon subnodosum*	0.58	*Conicosia elongata*	1.54	*Atriplex semibaccata*	1.24
	Hermannia disermifolia	2.83	*Stoebe nervigera*	2.45	*Tripteris oppositifolia*	0.77	*Eragrostis curvula*	0.50	*Atriplex lindleyi spp. inflata*	1.39	*Wahlenbergia adpressa*	0.98
	Othonna cylindrica	2.18	*Ruschia versicolor*	1.87					*Ruschia versicolor*	1.26	*Eriocephalus racemosus* var. *affinis*	0.88
	Ruschia paripetala	1.36	*Trichogyne repens*	1.30					*Zygophyllum morgsana*	1.25	*Ruschia versicolor*	0.72
	Lampranthus suavissimus	1.10	*Eriocephalus racemosus* var. *affinis*	1.27					*Atriplex semibaccata*	0.61	*Salsola kali*	0.63
	Lycium ferocissimum	1.02	*Euphorbia mauritanica*	1.15					*Tripteris oppositifolia*	0.60	*Galenia* sp.	0.61
	Lebeckia spinescens	0.97	*Pteronia paniculata*	0.94							*Eragrostis curvula*	0.50
			Stipagrostis namaquensis	0.73								
			Tripteris sinuata	0.68								
			Stipagrostis obtusa	0.63								
			Tetragonia sarcophylla	0.59								
Inter-clump areas	*Chaetobromus involucratus*	4.02	*Chrysocoma ciliata*	2.15	*Tetragonia fruticosa*	4.30	*Tetragonia fruticosa*	2.04	*Tetragonia fruticosa*	2.63	*Ehrharta calycina*	3.87
	Ruschia versicolor	2.53	*Lampranthus suavissimus*	1.84	*Conicosia elongata*	0.78	*Ehrharta calycina*	0.96	*Mesembryanthemum guerichianum*	1.48	*Conicosia elongata*	1.15
	Zygophyllum morgsana	1.58	*Trichogyne repens*	0.73	*Mesembryanthemum guerichianum*	0.61	*Conicosia elongata*	0.95	*Salsola kali*	1.17	*Oncosiphon grandiflorum*	0.65
	Ruschia paripetala	0.69	*Stipagrostis namaquensis*	0.67	*Lampranthus suavissimus*	0.57	*Eragrostis curvula*	0.59	*Othonna cylindrica*	0.93	*Tetragonia fruticosa*	0.62
	Mesembryanthemum guerichianum	0.62	*Ruschia versicolor*	0.62					*Conicosia elongata*	0.88	*Lebeckia spinescens*	0.58
	Hermannia subgen. *Mahernia*	0.60	*Ficinia argyropa*	0.54					*Lampranthus suavissimus*	0.79	*Wahlenbergia adpressa*	0.55
	Ficinia argyropa	0.52							*Oncosiphon grandiflorum*	0.75		
									Exomis microphylla var. *axyrioides*	0.61		
									Ruschia versicolor	0.57		

4.3.8 VEGETATION STRUCTURE

Inter-site and inter-seasonal differences in vegetation structure are presented in Figures 4.4 and 4.5. The distribution of individuals (except in February 2005) and cover over the ten height classes within the six sites was not significantly different ($p<0.05$) (Table 4.10).

Vegetation cover within reference sites was generally distributed over the entire range of height classes compared to the rehabilitation sites where plants were limited to the lower and middle height classes (Figure 4.4).

The reference sites generally had a relatively high density of individuals in the first two size classes, which resulted in these populations showing a "reverse-J" population distribution. The rehabilitation sites were represented by both the "reverse-J" and "bell-shaped" population distributions (Figure 4.5). The initial relatively steeper slope of the smaller size classes (i.e. the "reverse-J" population distribution) indicates a larger increment of change in the smaller size classes. The "reverse-J" population distribution can be attributed to self-thinning within each site (Goodburn & Lorimer 1999) or to age-specific mortality (Milton *pers. comm.*).

The sporadic rainfall and the germination response of desert plant seeds to rainfall (Higgins *et al.* 2000) results in stochastic (i.e. non-deterministic or variable) recruitment events. The majority of the sites showed seasonal variation in abundance and cover. Both abundance and cover generally increased in size class one from summer sampling (i.e. February) to the winter sampling (i.e. September) (Figure 4.4).

Table 4.10: The distribution of individuals and cover across size classes between sites were compared using contingency tables based on the null hypothesis of independence and tested using the chi-squared statistic (χ^2). χ^2 and P-values evaluating the distribution of individuals and cover across size classes between six sites (n=3 for reference sites; n=5 for rehabilitation sites) at Namakwa Sands are presented below. Significant values ($p<0.05$) are in bold.

Parameter	Sampling	χ^2	P
Abundance	Feb 04	56.61	**0.02**
	Sept 04	41.40	0.21
	Feb 05	60.04	0.22
	Sept 05	30.38	0.86
Cover (%)	Feb 04	50.02	0.13
	Sept 04	49.72	0.05
	Feb 05	35.56	0.67
	Sept 05	57.93	0.09

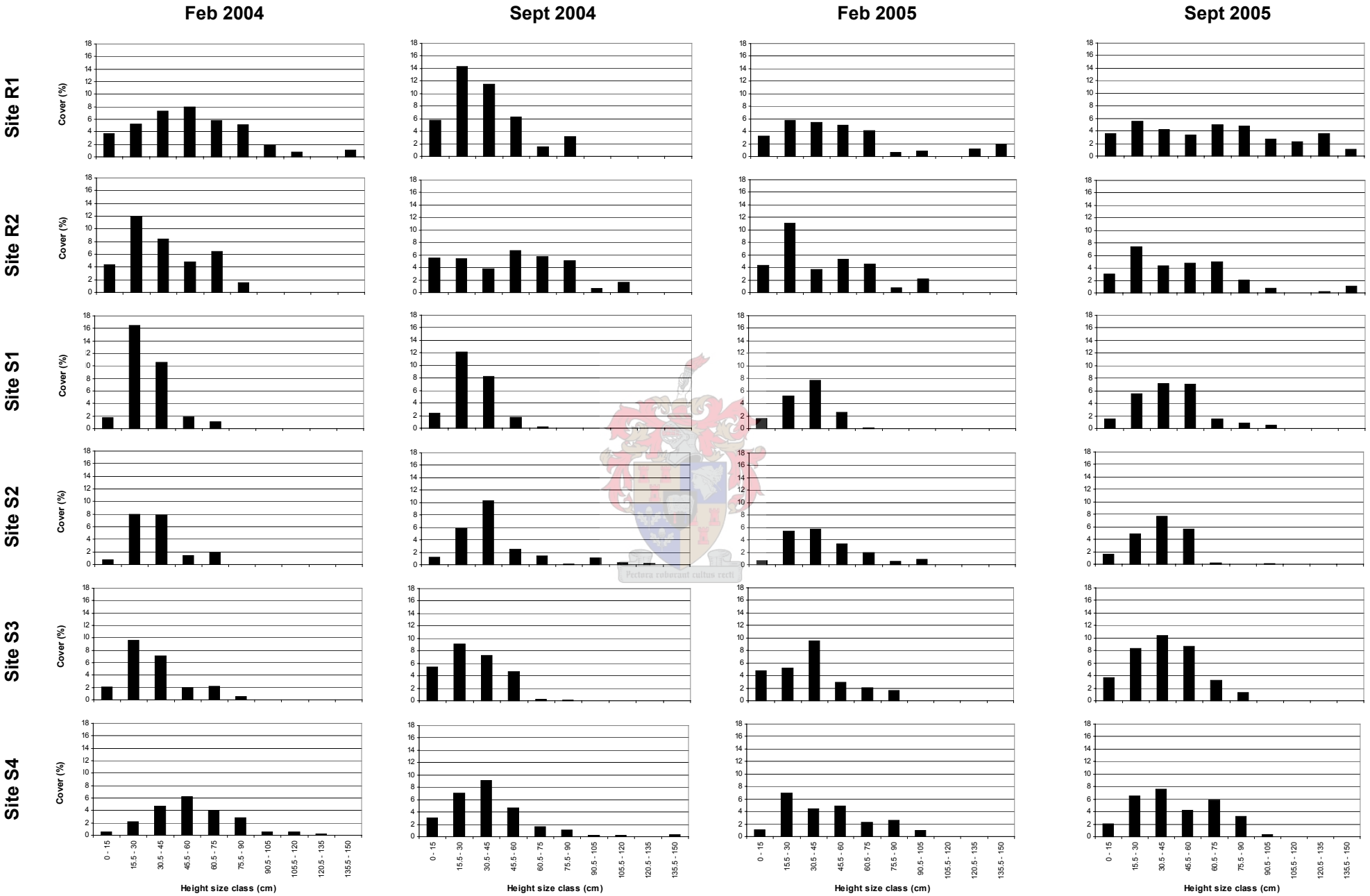

Figure 4.4: Cover by height class at six Namakwa Sands sites over four seasons. The six sites are as described in Table 4.1.

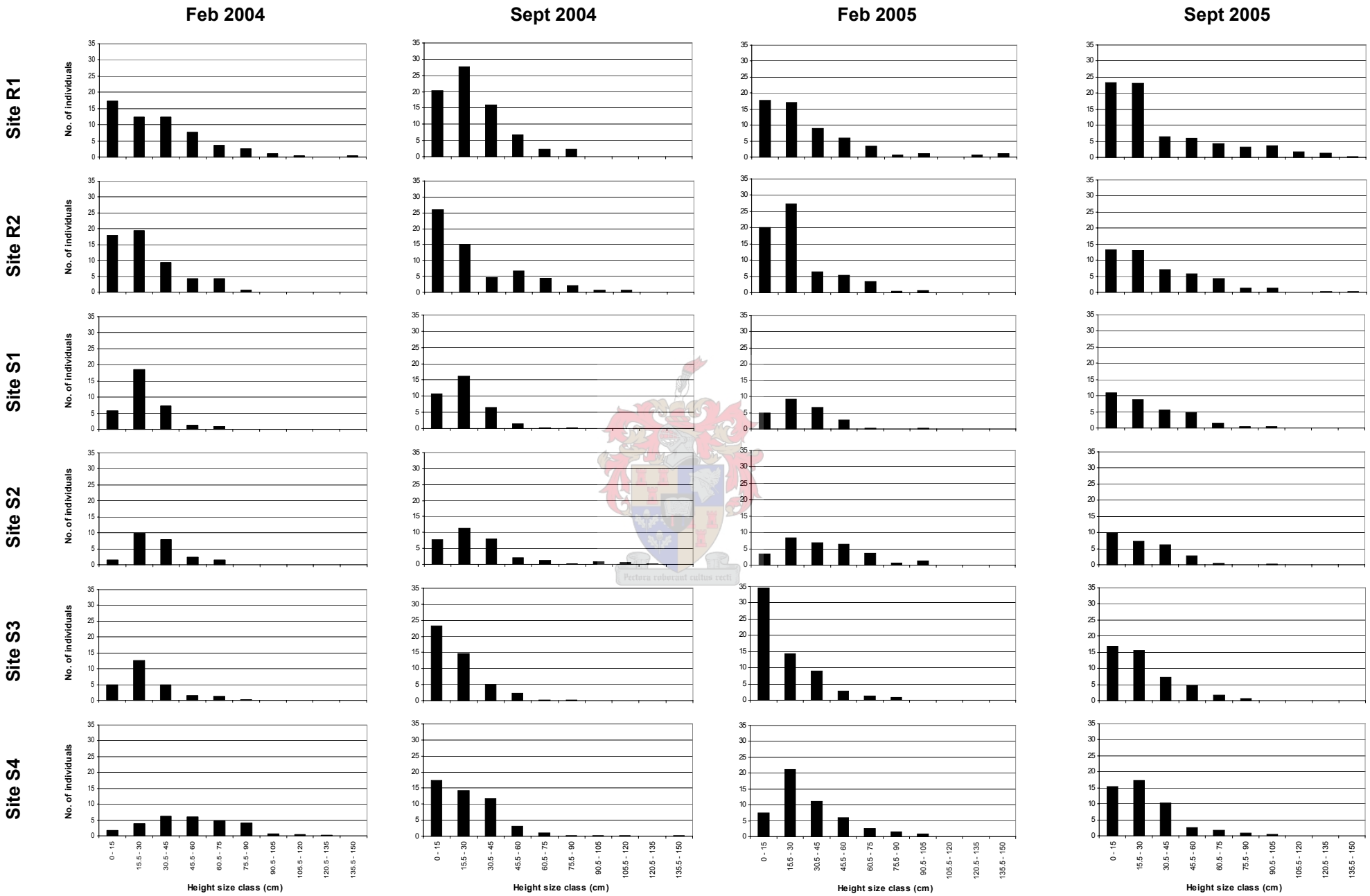

Figure 4.5: Numbers of individuals by height class at six Namakwa Sands sites over four seasons. The six sites are as described in Table 4.1.

4.4 DISCUSSION

Namakwa Sands has implemented four rehabilitation techniques to achieve the long-term rehabilitation goal of a similar cover and productivity to the pre-mining land-use (i.e. small-stock farming). This study investigates the effectiveness and success of these four rehabilitation techniques in facilitating vegetation recovery and in meeting the rehabilitation goal and objectives.

Reference site selection

Reference sites are generally required to compare with rehabilitation sites as they define the desired or "target" state and are used in determining whether rehabilitation has been successful (Tomlinson 1984; Andreasen *et al.* 2001). Reference sites should represent the desired habitat quality and functions (Pastorok *et al.* 1997) and should be based on similar landform, soil, biotic and climatic conditions as the rehabilitation site (Hobbs & Norton 1996). However, the selection of reference sites in this study proved problematic as the vegetation within the study area varies both spatially and temporarily. De Villiers *et al.* (1999) divided the mine area into six vegetation communities (including various variants), two of which were selected as reference sites for this study. Since the two reference sites were located in different communities, the differences found between them in terms of composition and structure were to be expected. The selection of reference sites in different communities may result in different desired or "target" states, which may have implications in determining whether rehabilitation has been successful. Failure to take account of naturally occurring spatial distribution of plant communities during monitoring could lead to false interpretations of the data (Charley & West 1975) and may seriously affect the outcome of a monitoring programme. Since most mining to date has taken place within the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (site R1) and all rehabilitation sites in this study occurred within this vegetation community, the success of the four rehabilitation techniques (sites S1, S2, S3 & S4) should ideally be determined in relation to site R1 rather than site R2 (*Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld). Support for this is provided if one compares the species richness of rehabilitation sites to the reference sites. Rehabilitation sites S1, S3 and S4 are considered to have met the three-year species richness objective when comparisons are made in relation to site R1, but not if the comparisons are made in relation to site R2 (Table 4.6).

Due to the spatial variation of the vegetation within the mining area, reference sites must be carefully selected for the long-term monitoring programme at Namakwa Sands, especially once mining moves out of the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (site R1) and into the *Ruschia tumidula – Tetragonia virgata* Tall Shrub (site R2). It is suggested that comparisons should be made against a single vegetation community and preferably the community that existed prior to mining or alternatively the community that Namakwa Sands has identified as the end point.

Species similarity

Monitoring species composition provides valuable information regarding community change (Pyke & Archer 1991) and ecologists have become increasingly aware that species establishment at the time of rehabilitation can have a lasting effect on successional trajectories and ecosystem function. Initial species establishment can be influenced by the species seeded (e.g. sites S2 and S4) or translocated (e.g. sites

S3 and S4), the method of seeding and the addition of topsoil (e.g. sites S1 and S3), fertilisers and organic matter. If the reference site species are not established initially, the similarity of the rehabilitation and reference sites can remain low for long periods (Chambers *et al.* 1994). De Villiers (2000) found an approximate 42% similarity between the vegetation and the seed bank within an equivalent vegetation community to site R1, and suggested that topsoil replacement will be essential for the rehabilitation of mined areas in the Strandveld Succulent Karoo. However, he speculated that since the natural vegetation at Namakwa Sands, which is dominated by perennial species, is not well represented in the topsoil seed bank, the recruitment from the seed bank alone would not be sufficient and that the large seeded perennial species would probably not be recruited in sufficient numbers. For this reason, and because strip-mining destroys the standing vegetation and associated aerial seed banks, de Villiers (2000) speculated that perennial plants would need to be reintroduced by seeding and / or plant translocation.

The finding that the rehabilitation sites had a greater similarity in terms of species composition to reference site R1 than to site R2 (Table 4.1) was not totally unexpected as all rehabilitation sites originally formed part of site R1 prior to mining (i.e. *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld) and topsoil (with the associated seed bank) from site R1 was used to rehabilitate sites S1 and S3. Similarly, sites S2 and S4 were seeded with indigenous species collected from within site R1. Site R1 had a greater similarity to sites where topsoil was replaced (sites S1 and S3) than sites where no topsoil was replaced (sites S2 and S4). This would suggest that the replacement of topsoil increased the similarity between the rehabilitation sites and reference site R1. It is speculated that this increased similarity could be attributed to the germination of the topsoil seed bank, as well as factors such as increased fertility and cation exchange capacity (Chapter 5). There was no definitive evidence that the translocation of species into multi-species clumps resulted in an increased similarity between rehabilitation and reference sites, as the results between translocation and non-translocation sites (comparing S1 to S3 and S2 to S4 to eliminate the topsoil and seeding variables) were inconclusive (Table 4.1). The use of topsoil is considered to be the most efficient and cost-effective means of returning biodiversity to rehabilitation areas (de Villiers 2000) and would appear to be important in order to achieve the long-term rehabilitation goal at Namakwa Sands.

Since the long-term rehabilitation goal at Namakwa Sands is to restore the area to, or as close as possible to, its natural state (Grindley & Barbour 1990; Mahood 2003b), it is recommended that site similarity be monitored as part of the long-term monitoring programme. This would enable Namakwa Sands to consider adaptive management (e.g. reseeding and / or translocations of selected species before the winter rains) should the similarity between reference and rehabilitation sites decrease over the long-term.

Species richness

The two reference sites had a greater total and average species richness (not always significant) than the rehabilitation sites (Table 4.4 & 4.5). This is to be expected since species richness in young rehabilitation sites seldom equals that of reference sites (Chambers *et al.* 1994). As previously documented, sites S1, S3 (except September 2005) and S4 are considered to have met the three-year objective of comprising at least 30% of the species occurring within reference site R1 (Table 4.6). Site S2 did not meet this short-

term objective in any of the four sampling periods and requires adaptive management (e.g. reseeding and / or translocations of selected species before the winter rains) in order to increase the establishment of more indigenous species and to facilitate natural successional processes.

Mahood (2003a) speculated that the spreading of topsoil might facilitate the return of species to post-mined areas. This did not seem to hold true in this study as site S4 (which had no topsoil) generally had the greatest total and average species richness. The fact that site S4 included both seeding and translocation could explain its generally superior species richness despite the lack of topsoil. However, since sites S1 and S3 had a greater similarity to site R1, it would appear that topsoiling did facilitate the return of R1 species. Sites S2 and S4 seem to have more r-selected species (or "weedy" species) present that are able to tolerate disturbance. The initial dominance of "weedy" species is not uncommon after disturbance (Milton *et al.* 1995), as these species are generally good colonisers (generalists); reach sexual maturity quickly; have a high fecundity; and the ability to disperse widely (Grime 1977). A possible explanation for site S3 having a lower species richness than site S1 could be the competitive exclusion of indigenous species. High-intensity rehabilitation inputs that result in competitive exclusion of native species often exhibit the lowest species number (Chambers *et al.* 1994). In general, it appears as if plant translocation and topsoil replacement has facilitated the return of species to post-mined areas.

In September 2002, fifteen months after Mahood (2003a) initiated the translocation trials the greatest number of species found within the non-irrigated, topsoiled treatments was 24 species. This is only slightly less than the 31 species found within the same treatments (site S3) in this study three years later. Therefore, although topsoiling and plant translocation appear to facilitate the return of species there is little optimism that any of rehabilitation sites will meet the final species richness objective of at least 60% of the number of species occurring within reference sites in the near future. The slow rate of colonisation may be due to unfavourable conditions (e.g. below average rainfall), lack of suitable pioneer species (Jochimsen 2001) and / or safe sites or slow dispersal into rehabilitation areas due to the lack of perches (Esler *pers. comm.*). The germination response of seeds of desert plants to sporadic rainfall (Higgins *et al.* 2000) results in stochastic (i.e. non-deterministic or variable) and episodic recruitment events (Milton *et al.* 1999). Therefore, the below average rainfall the study area has experienced over the last few years may be affecting the rate of colonisation and species richness. Species richness, especially of rarer species, is expected to increase over time as well as in response to more favourable germination conditions. Mahood (2003a) recommended that irrigation not be used as a rehabilitation mechanism, as she found that in fact winter and spring irrigation of post-mined areas might negatively affect the biodiversity in rehabilitation areas. Since species richness is one of the short-term objectives at Namakwa Sands, it is considered to be a good parameter to monitor as part of the long-term monitoring programme.

Species diversity and evenness

Although sites S1 and S3 had quotients of similarity above 50% compared to site R1, all rehabilitation sites had a significantly lower species diversity and evenness (Table 4.2). The lower species diversity and evenness found within the rehabilitation sites is due to an uneven distribution of species found in these sites and the greater dominance by one or two species. *Tetragonia fruticosa* was by far the most dominant species within sites S1, S2 (Plate 4.4) and S3, and the second most dominant species within

site S4. The most dominant species in site S4 was the seeded grass, *Ehrharta calycina* (Table 4.3). Competitive species can arrest successional processes and slow the rate of recovery in terms of species diversity for a long time by forming dense stands and are, therefore, not desirable from a rehabilitation point of view (Prach & Pyšek 2001). Whether *Tetragonia fruticosa* is a competitive species is not known, and it is recommended that further research be undertaken in order to determine if the dominance of this species is in fact slowing the rate of vegetation recovery. Nevertheless, one would expect species diversity to increase over time (Reay & Norton 1999). Species diversity and richness in pioneer communities, such as the rehabilitation sites, may decrease after initial seedling or topsoiling due to the exchanging of pioneer species with secondary species. Thus, alpha diversity recorded for only one vegetation period does not represent a valuable parameter for assessing the state of vegetation (Jochimsen 2001). Since the rehabilitation sites are relatively young in rehabilitation terms, it is recommended that species diversity and evenness be monitored over the long-term. Namakwa Sands will not be able to meet their long-term goal of restoring the area to, or as close as possible to, its natural state and achieving a vegetation cover and productivity similar to the pre-mining land-use (i.e. small-stock farming) if species diversity does not improve significantly and they may need to consider adaptive management (e.g. reseeding and / or translocations of selected species before the winter rains).

Vegetation patchiness or clumping is thought to optimise the capture and storage of limited resources (water and nutrients) from source areas into sinks or patches (Ludwig & Tongway 1996). These areas of increased fertility are known as “fertile islands” and play a critical part in the structuring and functioning of desert systems (Titus *et al*. 2002). Clumps in the vegetation of the study area are associated with increased fertility and support the concept of “fertile islands” (Chapter 5). The establishment of seedlings below canopies could have a number of other advantages, e.g. protection from high irradiance, high temperatures, high rates of transpiration, herbivory and wind and increased infiltration (de Villiers *et al*. 2001; Su & Zhao 2003). Several studies in arid ecosystems have demonstrated that seedling germination, establishment and survival are facilitated by the presence of shrub species (“nurse-plant” effect) compared to open inter-clump areas resulting from the amelioration of various environmental factors (Franco-Pizaña *et al.* 1996; Ludwig & Tongway 1996; Reay & Norton 1999). However, other studies have found no evidence of the “nurse-plant” effect (Franco-Pizaña *et al.* 1996; de Villiers *et al.* 2001; Blignaut & Milton 2005). There are varying theories regarding competition among desert plants. Some authors are of the opinion that competition is infrequent or absent and may become relatively less important and facilitation relatively more important as abiotic conditions become more limiting (Riginos *et al.* 2005). Others suggest that competition among plants is a common factor influencing plant populations (Miller & Huenneke 2000; Carrick 2003). A simple gradient of abiotic harshness is not sufficient to predict the outcome of plant-plant interactions and the process may be more complex involving both facilitative (e.g. increase seed germination) and competitive effects (e.g. inhibit establishment and growth) of neighbouring adult “nurse” plants (Franco-Pizaña *et al.* 1996; Riginos *et al.* 2005). Eccles *et al.* (1999) argue that if the net interaction were not positive then plants would not occur in clumps.

Based on this explanation, de Villiers *et al.* (2001) speculated that several advantages could be obtained by translocating perennial shrubs in mined out areas. De Villiers *et al.* (2001) also speculated that the replacement of the topsoil seed bank would be a vital source of seed for species recruitment. Within

rehabilitation sites there was no clear trend between topsoil replacement or plant translocation and species diversity or evenness, as no one site had the greatest species diversity and evenness over all sampling periods. However, the translocation sites (sites S3 and S4) tended to have the greatest diversity and evenness over all four sampling periods. Therefore, it would appear that plant translocation generally improved species diversity and evenness, but not always to a significant degree. Mahood (2003a) recommended that, although certain species could potentially be used for large-scale translocation, other rehabilitation techniques (e.g. seeding with perennial species) might be needed to facilitate the return of perennial biodiversity.

Vegetation composition

The reference sites were characterised by a more even spread of Importance Values with a greater number of dominant species, whereas the rehabilitation sites were dominated by only one or two species (Table 4.3). Even though a few of the dominant reference site species were found within the rehabilitation sites, the rarer species had not returned. It is speculated that these species will require more time or intervention (e.g. seeding and / or plant translocation). The absence of geophytes (e.g. *Albuca* spp., B*abiana brachystachys, Bulbine praemorsa, Chlorophytum rangei, Eriospermum* sp., *Ornithogalum* sp., *Pelargonium triste, Trachyandra* spp.), Restionaceae (e.g. *Ischyrolepis gaudichaudianus, Willdenowia incurvata*), *Crassula* spp., *Asparagus* spp. and *Euphorbia* spp. in rehabilitation sites is clearly evident (Appendix A). There were also very few grass species, other than those seeded (*Eragrostis curvula*, *Ehrharta calycina* and *Sorghum* sp.), found within rehabilitation sites.

Four of the five species translocated by Mahood (2003a) in multi-species clumps still survived after approximately four years with *Othonna cylindrica* appearing to be the most successful, followed by *Lampranthus suavissimus* and *Ruschia versicolor* (Table 4.3). All translocated *Asparagus* spp. had died by the time this study commenced in February 2004, supporting the conclusion of Mahood (2003a) that *Asparagus* spp. should not be considered for any future large-scale translocation purposes. Mahood (2003a) speculated that the poor survival of *Asparagus* spp. could be due to storage organs, damaged during clearing, not carrying sufficient water and nutrient reserves to allow this species to cope with relocation stress. Mahood's (2003a) recommendation that *Othonna cylindrica, Ruschia versicolor* and *Lampranthus suavissimus* should be considered for future large-scale translocation projects is supported. However, the long-term viability of rehabilitated *Zygophyllum morgsana* populations is in doubt due to the almost complete lack of recruitment and is likely to remain difficult to re-establish during periods of below average rainfall (Chapter 6). The efforts of translocating *Asparagus* spp. should not be considered to have been a waste of time or money, as these dead individuals still contribute to vegetation recovery by stabilising soil, breaking down to form humus and reducing wind speed (Mahood 2003a). Ludwig & Tongway (1996) found that clumps created with branches were effective in capturing soil sediments and litter to form "fertile islands" and vegetation responded favourably within these patches.

The ability to recolonise is affected by longevity of the seed bank or by dispersal from source populations (Lindborg & Eriksson 2004). Even though *Lampranthus suavissimus* and *Ruschia versicolor* were not seeded or translocated in site S1, these species were two of the more dominant species within that site.

Therefore, it would appear as if the replacement of topsoil has facilitated the establishment of these species. As mentioned earlier, geophytes are poorly represented in the rehabilitation sites, which would seem to indicate that the replacement of topsoil in sites S1 and S3 has not facilitated the return of these species. This could be due to, amongst other things, the bulbs being damaged during topsoil removal or bulbs rotting once topsoil has been stockpiled.

The grasses *Eragrostis curvula* and *Ehrharta calycina* were dominant in the sites in which they were seeded (i.e. site S2 and site S4, respectively). In addition, *Ehrharta calycina* was the second most dominant species in site S2, which would indicate that this species is providing a source of seed and is spreading into other rehabilitation areas. These two species could potentially be considered for future seeding. However, since *Eragrostis curvula* is not indigenous to the area, it is recommended that only *Ehrharta calycina* continue to be considered for future seeding. No live *Sorghum* sp. individuals, seeded in site S2, were found during this study. *Sorghum* sp. was planted to initially stabilise the soil rather than dominate the vegetation. It was anticipated that *Sorghum* sp. would be replaced by indigenous species when it died (Hälbich *pers. comm*.).

Changes in species composition of each rehabilitation site and their relative importance in the community can be related to site age and maturity, and may be an indication of successional change in rehabilitation communities (Lubke *et al*. 1996). Certain theories suggest that it is impossible for a community to tend towards its original state from its constituent species, as communities may have obtained their state through an assembly of species that no longer exist. A direct way of determining if a community is returning to its former composition is to evaluate the trajectories of species assemblages in a chronosequence relative to a reference site. This would also allow one to determine if rehabilitation sites achieve an equilibrium state and whether convergence is possible within a reasonable period (Wassenaar *et al*. 2005). Monitoring changes in species Importance Values would provide valuable information regarding succession and community change and it is therefore recommended that species Importance Values are monitored as part of the long-term monitoring programme at Namakwa Sands.

Vegetation persistence

The vegetation at the mine site is dominated by perennial plants that occur in mixed species clumps (Eccles *et al.* 1999; de Villiers 2000). This concurs with the findings of this study as perennial plants dominated all sites. However, it is in contrast to the results of Mahood (2003a) who found that annual species tended to dominate the returning vegetation in site S3 in terms of species richness and abundance. It is interesting to note that 10 of the 20 species identified 15 months after translocation by Mahood (2003a) were not encountered in site S3 during any of the four sampling periods. These species were all annuals or geophytes, namely: *Amellus tenuifolius, Chaetobromus involucratus, Conicosia pugioniformis, Crassula expansa, Drosanthemum hispidum, Oxalis obtusa, Arctotheca calendula, Mollugo cerviana, Wahlenbergia paniculata* and *Zaluzianskya benthamiana*. The replacement of annuals with perennials (cover and abundance) is a typical successional pattern in the southern Karoo and Namaqualand (Milton *pers. comm*.). Areas disturbed by grazing are initially colonised by annuals, which ordinarily cannot compete with perennial shrubs for nutrients and water (Steinschen *et. al.* 1996). Annuals

generally appear to be poor competitors, relying on soil-stored, dormant seeds and rapid growth (Milton *et al.* 1995) and as a result can be considered "opportunist" species (Connell & Slatyer 1977). The recovery of perennial shrubs in the Karoo after a major disturbance appears to be extremely slow (Milton *et al.* 1995).

In addition to competitive or facilitative interactions between plants, interactions with other factors (e.g. grazing and climate) are also of critical importance to the course of succession (Connell & Slatyer 1977). Therefore, the loss of these annual species could also be partly related to the below average rainfall the study area has experienced between 2003 and 2005. Species composition is seldom constant and is often determined by stochastic changes (e.g. climate) (Wassenaar *et al*. 2005). In Namaqualand, the species composition of ephemeral populations varies considerably between localities and from year to year. The most limiting factor in arid environments is moisture availability and it is not surprising that seedling establishment is directly related to rainfall timing and amount (Rösch *et al.* 1997; Esler 1999). In the southern Karoo and Namaqualand, seedlings generally emerge in autumn or early winter in response to rain (Milton 1995). In addition, temperatures at the time of the first rainfall event determine which species will germinate optimally (Rösch *et al.* 1997).

Although all sites were dominated by perennials, the reference sites had a greater number of perennial species than the rehabilitation sites. Interestingly the rehabilitation sites had a similar, although slightly lower (except in site S4), number of annual species to the reference sites (Table 4.4). The low number of annual species found in all sites (ranging from 8 to 13 species) during this study could also be attributed to the below average rainfall the area has experienced over the past few years.

Although the rainfall during the study was limited, it had a positive effect on the vegetation in all sites. All sites generally showed inter-seasonal variation in species richness, abundance, vegetation cover, clump cover and perennial cover with the winter samplings generally having the greater values, but these differences were not always significant.

Functional guilds

In an area dominated by strong winds, the reference and rehabilitation sites were surprisingly dominated by plant species that are insect pollinated rather than relying on wind pollination (Table 4.4). In Namaqualand insects tend to visit more than one host plant and most plant species attract a variety of floral visitors. The dominance of generalist pollinators is reflected by the predominance of plants species with large, open flowers with a variety of petal colours that are accessible to a range of insects (Esler 1999). Pollination by birds is less common than insect pollination in the Karoo (Esler 1999). This seems to hold true for the study area where bird pollinated species had the lowest numbers (Table 4.4).

Although the majority of the plant species were insect pollinated, the majority of the species encountered within both the reference and rehabilitation sites were wind dispersed. The distances colonisers are able to travel depends on the seed size and means of dispersal (Jochimsen 2001). Wind is an important element in the area (Washington 1990) and many seeds have adapted to wind dispersal with

appendages like wings and thin hairs (Le Roux & Schelpe 1997). Most large-seeded, wind dispersed species with wing-like appendages are ultimately trapped and establish under “nurse” plants (Esler 1999). Studies investigating microsites (closed-canopy versus open) showed that closed-canopy microsites contained higher seed densities (two- to four-fold greater) than open microsites (Jones & Esler 2004). Long-distance seed dispersal can increase seedling survival due to increasing the possibility of finding a suitable germination site and reduced competition from the parent plant and other seedlings. It is speculated that since the majority of the species encountered within both the reference and rehabilitation sites are wind dispersed, it will facilitate the dispersal of seed over greater distances within rehabilitation areas. The second most common dispersal guild within reference sites (bird) and rehabilitation sites (mammal) are also considered to be long-distance dispersal guilds (Table 4.4.). Bird dispersed species (e.g. *Asparagus* spp., *Chrysanthemoides incana, Cissampelos capensis, Diospyros* sp., *Rhus undulata*) within the rehabilitation sites are noticeably lacking (Appendix A). Bird dispersed species are generally the taller thicket element shrubs (e.g. *Euclea* sp*., Rhus* sp*., Lycium* sp. and *Maytenus* sp.) that give the vegetation vertical structure (Milton *pers. comm.*). The low height of the vegetation within rehabilitation sites (Figures 4.4 & 4.5) does not provide ideal perching sites for birds. Thus, it is speculated that the lack of perching sites will decrease the rate of dispersal of these species throughout the rehabilitation areas. It is recommended that the functional diversity of the vegetation within rehabilitation sites be monitored as part of the long-term monitoring programme.

A characteristic of many desert plants is the relatively short seed dispersal distances (de Villiers 2000). It has been suggested that short-distance dispersal and / or seed retention (e.g. hygrochastic dispersal) has a selective advantage over long-distance dispersal because seedlings have a higher probability of survival close to where the parent plant has survived to reproduction (Esler 1999). The Mesembryanthemaceae (e.g. *Lampranthus suavissimus, Dorotheanthus rourkei*, *Drosanthemum* c.f. *luederitzii*, *Psilocaulon subnodosum* and *Ruschia* spp.) have hygrochastic capsules, which open during rainfall to release seeds when moisture conditions are favourable for germination and establishment (Esler & Cowling 1995). The distance of seed dispersal associated with hygrochastic dispersal (up to 1.7 m) is thought to be significantly increased by sheet flow (Esler 1999). This suggests that dispersal of species with short distance dispersal guilds (e.g. hygrochastic, gravity and explosive) may be limited within large disturbed areas (Mahood 2003a). Thus, the distribution of these species will initially depend on the rehabilitation scenario (e.g. seeding, plant translocation and / or germination of topsoil seed bank) implemented.

Vegetation cover

Achieving an adequate vegetation cover in rehabilitation sites reduces wind erosion, increases infiltration (de Villiers *et al*. 2001; Su & Zhao 2003) and has major effects on soil resources and physical properties (Schlesinger *et al.* 1996). The two reference sites generally had a greater vegetation and clump cover than the rehabilitation sites (Table 4.5), except in September 2005 where site S3 had the greatest vegetation cover and site S4 had a greater cover than site R2. This finding was expected as the rehabilitation sites are relatively young (approximately four years old) and a low plant cover is the most striking feature of a pioneer community (Jochimsen 2001). However, all rehabilitation sites met the three-

year cover objective (Table 4.6). In addition, sites S1, S3 and S4 also met the five-year cover objective, but this was not true across all sampling periods or across both reference sites (Table 4.6). Since site S3 (topsoil and plant translocation) generally had the greatest cover among rehabilitation sites and site S1 (topsoil only) generally had a greater cover than site S2 (seeding only), it would appear that plant translocation and topsoil replacement facilitated the return of vegetation cover. Site S2 appeared to perform the worst in terms of vegetation cover. These findings are supported by Prach & Pyšek (2001) who found that vegetation cover generally increased faster in areas with secondary succession, with soil seed bank and organic topsoil present at the onset of succession.

Although all rehabilitation sites met the three-year cover objective, a concern is that the cover within all rehabilitation sites was dominated almost entirely by one species, namely *Tetragonia fruticosa* (Table 4.7; Plate 4.4). *Tetragonia fruticosa* grew very large and formed large clumps (up to 7.8 m meters long and 4.5 m wide) along windbreaks within rehabilitation sites. The almost total dominance by *Tetragonia fruticosa* is of concern as this species is not considered to be very stable and is known to die off (Milton *pers. comm*.). Species diversity and evenness is expected to increase over time and with it the cover of other species. However, adaptive management (e.g. seeding and / or plant translocation) should seriously be considered in order to speed up this process.

Since vegetation cover is included as one of the short-term objectives of the Namakwa Sands rehabilitation programme, it should continue to be measured as part of the long-term monitoring programme. If the objectives are not achieved then adaptive management should be considered.

Palatability

Although the long-term rehabilitation goal at Namakwa Sands is to return the land to small-stock farming (EEU 1990), the dominance of rehabilitation sites by *Tetragonia fruticosa* is of concern due to its palatability and the limited protection offered by non-palatable species. *Tetragonia fruticosa* has a grazing value of 3.62 and a four star palatability score and is thus considered to be a very palatable species for sheep (Esler *et al.* 2005). The dominance by a single palatable species may falsely enhance the grazing capacity (ha per small-stock unit) within rehabilitation areas (Milton *pers. comm*.) and if current grazing practices (i.e. one small-stock unit per 10 ha) are implement in these areas overgrazing could decimate the vegetation (Mahood 2003a). Overgrazing and associated trampling may also disrupt plant-soil relations (e.g. positive feedbacks between plant cover and nutrient cycling and infiltration) and lead to the redistribution of soil moisture, erodible material and nutrients (Schlesinger *et al.* 1996; Rietkerk, *et al.* 2000). The translocation of non-palatable species in multi-species clumps as undertaken in site S3 and S4 provides some protection from grazing (de Villiers *et al*. 2001; Su & Zhao 2003).

In contrast to the almost total dominance of *Tetragonia fruticosa* within rehabilitation sites, it is interesting to note that nearby lands, which have been fallow for approximately three years, were dominated almost entirely by *Galenia africana* (personal observation) (Plate 4.7). These fallow lands are located outside the mining area and had not been rehabilitated, so revegetation relied on spontaneous succession from the topsoil and the importation of seed from areas outside the area of disturbance. Although the farmer

grazes sheep in the fallow lands and surrounding vegetation (Pool *pers. comm*.), *Galenia africana* is generally unpalatable to sheep. Since the rehabilitation sites are not grazed, the dominance of a palatable shrub such as *Tetragonia fruticosa* may be due to the release from herbivory (Milton *pers. comm*.).

Precisely when rehabilitation areas are opened up to grazing and estimation of grazing capacity is crucial in order to ensure that the rehabilitation areas are not overgrazed, as overgrazing could result in the regression of the vegetation. It is recommended that rehabilitation sites only be considered for grazing when all rehabilitation objectives have been met, species diversity of both the vegetation and clumps is not significantly different to reference sites and the number of palatable species has increased considerably. If the number of palatable species does not increase, Namakwa Sands may need to consider adaptive management (e.g. seeding and / or translocation with palatable species). Alternatively, an appropriate grazing strategy, which is related to the *Tetragonia fruticosa* dominated vegetation within rehabilitation sites, would need to be determined and adopted.

Plate 4.7: A fallow wheat field (approximately three years old) which was dominated by *Galenia africana* (Kraalbos).

Vegetation structure

The structure of vegetation plays an important role in arid system functioning (Desmet & Cowling 1999) and is a function of interactions between the biota and environmental conditions in the area. The vegetation at the mine site is dominated by perennial plants that occur in mixed species clumps, interspersed with open areas that are covered by annual plants in the winter (Eccles *et al.* 1999; van Rooyen 2001; Mahood 2003b). Although clumping is an unchanging feature in the vegetation at the mine site, the amount, height, area, composition and cover of clumps does fluctuate (van Rooyen 2001).

Clumps within all rehabilitation sites were dominated by large *Tetragonia fruticosa* individuals, whereas clump cover within reference sites were dominated by a greater number of species and these species had lower and more even cover distributions (Table 4.9). As discussed earlier, Namakwa Sands will not be able to meet their long-term objective of small-stock farming if species diversity of the vegetation (clump and inter-clump areas) does not improve significantly.

The *Tetragonia fruticosa* dominated clumps within rehabilitation sites were not only compositionally different to clumps in reference sites (Table 4.9), but they also were structurally different (personal observation). The reference site clumps are multi-layered (Plate 4.8), whereas *Tetragonia fruticosa* dominated clumps are generally not multi-layered. These clumps tend to grow all the way to the ground forming thick vegetation mats, which provides limited space and light beneath for germination and seedling establishment (Plate 4.9). Although these clumps improve soil fertility (Chapter 5), as well as infiltration and seed entrapment, it is speculated that the structure of *Tetragonia fruticosa* dominated clumps would not facilitate germination and seedling establishment to the same extent as the translocated clumps or clumps in reference sites.

The dominance by *Tetragonia fruticosa* of clumps in rehabilitation sites resulted in a skewed indication of clump structure, namely clump cover, clump size, number of clumps, clump height and inter-clump distance. The majority of the clumps in rehabilitation sites (excluding the multi-species clumps translocated in sites S3 and S4) included *Tetragonia fruticosa* and the large size of these individuals resulted in the rehabilitation sites generally having a greater average clump size than the reference sites (Table 4.8). This resulted in the average clump cover within rehabilitation sites, although lower than reference sites, to be enhanced. The number of clumps within the rehabilitation sites also appears to be inflated due to the definition of “clumps” used in this study (i.e. where the canopy of any two individuals overlapped) and the sprawling growth form of *Tetragonia fruticosa* with its long tailing branches. In many instances *Tetragonia fruticosa* appears to have grown over other species rather than facilitating the germination of seedlings below its canopy, which resulted in an exaggerated number of clumps within rehabilitation sites. As a result, the inter-clump distance may be underestimated within rehabilitation sites. Clump height and vegetation height within rehabilitation sites, although generally lower than site R1, also appears to be exaggerated due to the growth of *Tetragonia fruticosa* up against the sides of the windbreaks (approximately 1 m high). Therefore, the monitoring of clump cover, clump size, number of clumps, inter-clump distance and clump height within rehabilitation sites did not provide valuable information in determining rehabilitation success. Rather, it is recommended that species composition, richness, diversity and vertical structure of clumps be monitored and compared to reference sites.

Plate 4.8: Two clumps (dominated by difference species) located within site R1 showing the vertical structure within clumps.

Plate 4.9: A *Tetragonia fruticosa* dominated clump within site S4. *Tetragonia fruticosa* dominated clumps are very large due to growth along and up the sides of windbreaks and do not have much vertical structure.

Rehabilitation success

The four rehabilitation sites have been discussed in terms of various vegetation parameters, but the question remains regarding which technique appears to facilitate the return of vegetation to the greatest extent. An equation, termed the Success Value (SV), was developed to provide an indication of success. SV of site (*i*) is calculated using selected vegetation parameters of site (*i*) relative to those parameters in a reference site (*r*), which are summed and multiplied by 100.

$$SV_i = \left[\left(\frac{\text{Parameter 1}_i}{\text{Parameter 1}_r} \right) + ... + \left(\frac{\text{Parameter n}_i}{\text{Parameter n}_r} \right) \right] \text{x } 100$$

In determining the SV of a rehabilitation site, only those parameters that are considered ideal for monitoring programme should be used. Thus, cover, species richness, species diversity, species evenness, species similarity and abundance were used in this study resulting in a value out of 600. Based on these calculations, site S3 (topsoil and plant translocation), although the most expensive to implement (Hälbich *pers. comm.*), appears to be the most successful technique in facilitating vegetation recovery similar to reference site R1, followed by site S4 (seeding and plant translocation), site S1 (topsoil) and S2 (seeding) (Table 4.11).

Table 4.11: Average Success Value of the four rehabilitation sites (n=4) compared to site R1. The sites are described in Table 4.1.

Site	Success Value	Ranking
S1	361.39	3
S2	315.81	4
S3	398.42	1
S4	370.23	2

4.5 CONCLUSIONS

Ecosystem composition, structure and function within rehabilitation sites have been restored to a certain extent. Site S3 (topsoil and plant translocation) appears to be the most successful technique in facilitating vegetation recovery, followed by site S4 (seeding and plant translocation), site S1 (topsoil) and site S2 (seeding). However, the long-term rehabilitation goal of restoring the area to, or as close as possible to, its natural state and achieving a vegetation cover and productivity similar to the pre-mining land-use (i.e. small-stock farming) has not been achieved. Since the rehabilitation sites are very young in rehabilitation terms, it is difficult to say whether or not any of the rehabilitation techniques are on a successional trajectory towards the desired endpoint. Due to the young age of the sites and the fact that the long-term rehabilitation goal has not been achieved, Namakwa Sands will need to continue to monitor plant and soil changes occurring along a chronosequence until the rehabilitation objectives and goal have been achieved. This study could be seen as the first point in the chronosequence. Due to the spatial variation

of the vegetation within the mining area, reference sites must be carefully selected and comparisons should preferably be made against the vegetation community that existed prior to mining or alternatively the community that Namakwa Sands has identified as the end point.

This study has indicated that topsoil replacement (site S1), topsoil replacement and plant translocation (site S3), and seeding and plant translocation (site S4) appears to facilitate the return of similarity, species richness, species diversity and cover to post-mined. These sites also met the three-year objective of comprising at least 30% of the species occurring within reference site R1. Site S2 where only seeding was implemented appeared to perform the worst. Since site S2 did not meet the short-term species richness objective it would appear that this site requires adaptive management (e.g. reseeding and / or translocations of selected species before the winter rains) in order to increase the establishment of more indigenous species and to facilitate natural successional processes. All rehabilitation sites had a species diversity and evenness significantly lower than the reference sites and were dominated by one or two species. Species and functional diversity appears to be the most limiting factors within the rehabilitation sites. Namakwa Sands will not be able to meet their long-term objective of small-stock farming if species diversity does not improve significantly and thus may need to consider adaptive management.

All rehabilitation sites met the three-year objective of at least 50% of the cover occurring within reference site R1. Sites S3 & S4 (both translocation sites) and site S1 (topsoil replacement only) met the five-year cover objective of at least 80% relative to the cover of the references sites in some of the sampling periods. Although all sites met the three-year cover objective, the almost total dominance by *Tetragonia fruticosa* in rehabilitation sites is of concern for a number of reasons. Firstly, it is not considered to be very stable species and could die back resulting in a retrogression of the rehabilitation programme. Secondly, the persistence and dominance by *Tetragonia fruticosa* could arrest successional processes and prevent species diversity and evenness from improving in rehabilitation sites, but this would need to be determined through further research. Thirdly, *Tetragonia fruticosa* is a very palatable species and the dominance of this species may falsely enhance the grazing capacity. This together with the fact that there is little protection provided by non-palatable species in rehabilitation sites could result in overgrazing if current grazing practices (i.e. one small-stock unit per 10 ha) are implement in these areas. Overgrazing could decimate the vegetation, which could result in the redistribution of soil moisture, erodible material and nutrients. It is recommended that only when all rehabilitation objectives have been met, species diversity is not significantly different to reference sites and when the number of palatable species increases should these areas be considered for grazing. Alternatively, an appropriate grazing strategy, which is related to the *Tetragonia fruticosa* dominated vegetation within rehabilitation sites, would need to be determined and adopted.

It is recommended that species composition and similarity, species richness, species diversity and evenness, vegetation cover, species dominance (Importance Values), vertical structure and functional diversity of the vegetation (clumps and inter-clumps) be monitored as part of the long-term monitoring programme and compared to reference sites. It would appear that monitoring the clump cover, clump size, number of clumps, inter-clump distance and clump height within rehabilitation sites will not provide valuable information regarding rehabilitation success.

Should long-term monitoring indicate that rehabilitation is not progressing as predicted then Namakwa Sands should consider adaptive management (e.g. reseeding and or translocation of selected species before the onset of the winter rains). Based on the results of Mahood (2003a) and this study, *Othonna cylindrica, Ruschia versicolor* and *Lampranthus suavissimus* should be considered for future large-scale translocation purposes. However, other species such as *Tripteris oppositifolia, Eriocephalus racemosus* var. *affinis, Trichogyne repens, Hermannia cuneifolia, Hermannia disermifolia, Stoebe nervigera, Willdenowia incurvata* and the grasses*, Chaetobromus involucratus, Stipagrostis obtusa* and *Stipagrostis namaquensis* could also be considered for translocation trials. Namakwa Sands could also consider creating clumps with cleared vegetation from the mining front. Ludwig & Tongway (1996) found that clumps created with branches were effective in capturing soil sediments and litter to form "fertile islands" and vegetation responded favourably within these patches. In addition, plant material from the mining front could provide a potential seed source. These "clumps" should ideally be created after the windbreaks have been erected to prevent problems related to windbreak placement and to prevent damage by the vehicle/s used to lay the windbreaks.

De Villiers (2000) provided a list of species that are adapted to a wider range of environmental conditions of the study area and speculated that revegetation using these species would help to stabilise the mined sand during the windy, dry and hot summer months, and that they should largely restore the former appearance and structure of the vegetation. These species should also be considered for future plant translocation or seeding exercises. This study showed that *Eragrostis curvula* and *Ehrharta calycina* appeared to be good species for seeding. However, since *Eragrostis curvula* is not indigenous to the area, it is recommended that only *Ehrharta calycina* continue to be considered for future seeding.

Lastly, the short-term objectives of the Namakwa Sands rehabilitation plan only deal with vegetation species richness and cover. However, if rehabilitation is to be successful in restoring biodiversity, rehabilitation must go beyond the reconstruction of composition and cover and restore structure, biological interactions, processes and integrity. Recreating structure and composition without function does not constitute complete rehabilitation (Reay & Norton 1999). It is therefore recommended that Namakwa Sands develop rehabilitation objectives relating to structure and function.

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APPENDIX A: Species encountered in this study. Six species could not be identified to at least genus level and have not been included in the table below.

No.	Species	Life form	Persistence	Pollination Guild	Dispersal Guild	Site					
						R1	R2	S1	S2	S3	S4
	Aizoaceae										
1	*Galenia africana* L.	Shrub	Perennial	Insect	Mammal			x	x	x	x
2	*Galenia* c.f. *sarcophylla* Fenzl	Dwarf shrub	Perennial	Insect	Mammal						x
3	*Galenia pu*bescens (Eckl. & Zeyh.) Druce	Dwarf shrub	Perennial	Insect	Mammal				x		
4	*Galenia* sp.	?	?	Insect	Mammal						x
5	*Tetragonia fruticosa* L.	Dwarf shrub	Perennial	Insect	Wind	x	x	x	x	x	x
6	*Tetragonia sarcophylla* Fenzl var. *sarcophylla*	Dwarf shrub	Perennial	Insect	Wind		x				
7	*Tetragonia* sp.	Dwarf shrub	Perennial	Insect	Wind						x
	Amaryllidaceae										
8	*Gethyllis* c.f. *afra* L.	Geophyte	Perennial	Insect	Mammal						x
	Anacardiaceae										
9	*Rhus undulata* Jacq.	Shrub	Perennial	Insect	Bird		x				
	Anthericaceae										
10	*Chlorophytum rangei* (Engl. & K.Krause) Nordal	Herb	Perennial	Insect	Wind		x				
	Apocynaceae										
11	*Microloma sagittatum* (L.) R.Br.	Climber	Perennial	Bird	Wind	x	x				x
	Asparagaceae										
12	*Asparagus aethiopicus* L.	Climber	Perennial	Insect	Bird	x					
13	*Asparagus capensis* L. var. *litoralis* Sues. & Karl	Shrub	Perennial	Insect	Bird	x	x				
14	*Asparagus declinatus* L.	Scrambler	Perennial	Insect	Bird		x				
15	*Asparagus retrofractus* L.	Scrambler Shrub	Perennial	Insect	Bird	x	x				
	Asphodelaceae										
16	*Bulbine praemorsa* (Jacq.) Spreng.	Geophyte	Perennial	Insect	Wind	x					
17	*Trachyandra falcata* (L.f.) Kunth	Geophyte	Perennial	Insect	Wind	x	x				
18	*Trachyandra revoluta* (L.) Kunth	Geophyte	Perennial	Insect	Wind		x				
19	*Trachyandra* sp.	Geophyte	Perennial	Insect	Wind		x				
	Asteraceae										
20	*Amellus flosculosus* DC.	Herb	Perennial	Insect	Wind	x		x	x	x	x
21	*Arctotheca calendula* (L.) Levyns	Herb	Annual	Insect	Wind	x					x
22	*Arctotis auriculata* Jacq.	Dwarf shrub	Perennial	Insect	Wind	x	x				
23	*Arctotis merxmuelleri* Friedrich	Dwarf shrub	Perennial	Insect	Wind	x	x				
24	*Arctotis* sp.	Herb	?	Insect	Wind		x	x	x		x
25	*Arctotis venusta* Norl.	Herb	Annual	Insect	Wind	x	x				x
26	*Chrysanthemoides incana* (Burm.f.) Norl.	Shrub	Perennial	Insect	Bird	x	x				
27	*Chrysocoma ciliata* L.	Shrub	Perennial	Insect	Wind		x	x			
28	*Didelta carnosa* (L.f.) Aiton	Dwarf shrub	Perennial	Insect	Wind	x	x		x		
29	*Eriocephalus racemosus* L. var. *affinis* (DC.) Harv.	Shrub	Perennial	Insect	Wind	x	x		x		x
30	*Gazania* sp1	Herb	?	Insect	Wind		x				

No.	Species	Life form	Persistence	Pollination Guild	Dispersal Guild	Site					
						R1	R2	S1	S2	S3	S4
31	*Gazania* sp2	Herb	?	Insect	Wind		x				
32	*Helichrysum dasyanthum* (Willd.) Sweet	Shrub	Perennial	Insect	Wind	x					
33	*Helichrysum micropoides* DC.	Shrub	Annual	Insect	Wind		x				
34	*Oncosiphon grandiflorum* (Thunb.) Källersjö	Herb	Annual	Insect	Gravity	x		x	x	x	x
35	*Othonna cylindrica* (Lam.) DC.	Shrub, Succulent	Perennial	Insect	Wind	x	x	x	x	x	x
36	*Pteronia divaricata* (P.J.Bergius) Less.	Shrub	Perennial	Insect	Wind	x	x				
37	*Pteronia onobromoides* DC.	Shrub	Perennial	Insect	Wind	x	x			x	
38	*Pteronia paniculata* Thunb.	Shrub	Perennial	Insect	Wind		x				
39	*Senecio aloides* DC.	Shrub	Perennial	Insect	Wind	x	x				
40	*Senecio arenarius* Thunb.	Herb	Annual	Insect	Wind			x			
41	*Stoebe nervigera* (DC.) Sch.Bip.	Shrub	Perennial	Insect	Wind		x				
42	*Trichogyne repens* (L.) Anderb.	Dwarf shrub	Perennial	Insect	Wind		x		x		
43	*Tripteris clandestina* Less.	Herb, Succulent	Annual	Insect	Wind						x
44	*Tripteris oppositifolia* (Aiton) B.Nord.	Shrub	Perennial	Insect	Wind	x	x	x		x	
45	*Tripteris sinuata* DC.	Shrub	Perennial	Insect	Wind		x				
46	*Ursinia* sp.	Herb	Annual	Insect	Wind		x				
	Brassicaceae										
47	*Erucastrum strigosum* (Thunb.) O.E.Schulz	Herb	Annual	Insect	Explosive			x		x	x
48	*Heliophila lactea* Schltr.	Herb	Annual	Insect	Wind		x				x
	Campanulaceae										
49	*Wahlenbergia adpressa* (Thunb.) Sond.	Herb	Perennial	Insect	Gravity		x				x
	Celastraceae										
50	*Gloveria integrifolia* (L.f.) Jordaan	Shrub	Perennial	Insect	Bird	x	x	x			
	Chenopodiaceae										
51	*Atriplex lindleyi* Moq. spp. *inflata* (F.Muell.) P.G.Wilson	Herb	Annual	Wind	Wind			x	x	x	x
52	*Atriplex semibaccata* R.Br.	Dwarf shrub	Perennial	Wind	Bird	x		x	x	x	
53	*Chenopodium murale* L.	Herb	Annual	Wind	Wind			x	x	x	
54	*Exomis microphylla* (Thunb.) Aellen var. *axyrioides*	Shrub	Perennial	Wind	Mammal	x	x	x	x	x	x
55	*Manochlamys albicans* (Aiton) Aell.	Shrub	Perennial	Wind	Bird	x	x	x	x	x	x
56	*Salsola kali* L.	Herb	Annual	Wind	Wind	x	x	x	x	x	x
	Crassulaceae										
57	*Crassula muscosa* L. var. muscosa	Herb, Succulent	Perennial	Insect	Wind	x					
58	*Crassula subaphylla* (Eckl. & Zeyh.) Harv. var. *subaphylla*	Dwarf shrub, Succulent	Perennial	Insect	Wind		x				
59	*Crassula thunbergiana* Schult. subsp. *thunbergiana*	Herb, Succulent	Annual	Insect	Wind, Gravity	x					
	Cyperaceae										
60	*Ficinia argyropa* Nees	Herb	Perennial	Wind	Gravity	x	x				x
	Ebenaceae										
61	*Diospyros* sp.	Shrub	Perennial	Insect	Bird		x				
	Eriospermaceae										
62	*Eriospermum* sp.	Geophyte	Perennial	Insect	Wind	x	x				

No.	Species	Life form	Persistence	Pollination Guild	Dispersal Guild	Site					
						R1	R2	S1	S2	S3	S4
	Euphorbiaceae										
63	*Euphorbia caput-medusae* L.	Shrub, Succulent	Perennial	Insect	Explosive	x	x				
64	*Euphorbia mauritanica* L.	Shrub, Succulent	Perennial	Insect	Explosive		x				
	Fabaceae										
65	*Aspalathus* sp.	Shrub	Perennial	Insect	Explosive						x
66	*Crotalaria excisa* (Thunb.) Baker f.	Herb	Perennial	Insect	Explosive			x			x
67	*Lebeckia spinescens* Harv.	Shrub	Perennial	Insect	Explosive	x	x	x	x	x	x
68	*Melolobium* sp.	Shrub	Perennial	Insect	Explosive	x		x			x
	Geraniaceae										
69	*Pelargonium triste* (L.) L'Hér.	Geophyte	Perennial	Insect	Wind		x				
	Hyacinthaceae										
70	*Albuca cooperi* Baker	Geophyte	Perennial	Insect	Wind	x	x				
71	*Albuca flaccida* Jacq.	Geophyte	Perennial	Insect	Wind	x	x				
72	*Lachenalia marlothii* W.F.Barker ex G.D.Duncan	Geophyte	Perennial	Insect	Wind	x					
73	*Ornithogalum* sp.	Geophyte	Perennial	Insect	Wind	x					
	Iridaceae										
74	*Babiana brachystachys* (Baker) G.J.Lewis	Geophyte	Perennial	Insect	Gravity		x				
	Menispermaceae										
75	*Cissampelos capensis* L.f.	Shrub	Perennial	Insect	Bird		x				
	Mesembryanthemaceae										
76	*Antimima compacta* (L.Bolus) H.E.K.Harmann	Succulent	Perennial	Insect	Hygrochastic	x					
77	*Dorotheanthus rourkei* L.Bolus	Succulent	Annual	Insect	Hygrochastic		x				
78	*Drosanthemum* c.f. *luederitzii* (Engler) Schwantes	Succulent	Perennial	Insect	Hygrochastic			x	x		x
79	*Conicosia elongata* (Haw.) N.E.Br.	Succulent	Perennial	Insect	Wind			x	x	x	x
80	*Lampranthus suavissimus* (L.Bolus) L.Bolus	Succulent	Perennial	Insect	Hygrochastic	x	x	x	x	x	x
81	*Mesembryanthemum guerichianum* Pax	Succulent	Annual	Insect	Wind	x		x	x	x	x
82	*Phyllobolus* sp.	?	?	Insect	Hygrochastic				x		x
83	*Psilocaulon subnodosum* (A.Berger) N.E.Br.	Succulent	Perennial	Insect	Hygrochastic			x	x	x	x
84	*Ruschia* c.f. *brevibracteata* (L.Bolus) L.Bolus	Succulent, Shrub	Perennial	Insect	Hygrochastic	x	x			x	x
85	*Ruschia paripetala* L.Bolus	Succulent	Perennial	Insect	Hygrochastic	x				x	
86	*Ruschia versicolor* L.Bolus	Succulent	Perennial	Insect	Hygrochastic	x	x	x	x	x	x
	Molluginaceae										
87	*Limeum* sp.	?	Annual	Insect	Wind		x				
88	*Pharnaceum confertum* (DC.) Eckl. & Zeyh.	Herb	Perennial	Insect	Mammal	x	x	x	x		x
89	*Pharnaceum* sp.	Herb	?	Insect	Mammal	x	x				
90	*Psammotropha quadrangularis* (L.f.) Fenzl	Dwarf shrub	Perennial	Insect	?		x				
	Oxalidaceae										
91	*Oxalis pes-caprae* L.	Herb	Perennial	Insect	Explosive	x	x	x		x	x
	Poaceae										
92	*Chaetobromus involucratus* (Schrad.) Nees	Graminoid	Perennial	Wind	Wind	x	x				

No.	Species	Life form	Persistence	Pollination Guild	Dispersal Guild	Site					
						R1	R2	S1	S2	S3	S4
93	*Ehrharta brevifolia* Schrad. var. *brevifolia*	Graminoid	Annual	Wind	Wind			x		x	
94	*Ehrharta calycina* Sm.	Graminoid	Perennial	Wind	Wind				x		x
95	*Eragrostis curvula* (Schrad.) Nees	Graminoid	Perennial	Wind	Mammal			x	x	x	x
96	*Karroochloa schismoides* (Stapf ex Conert) Conert & Türpe	Graminoid	Annual	Wind	Wind	x		x	x	x	x
97	*Sorghum* sp.	Graminoid	Annual	Wind	Gravity						x
98	*Stipagrostis namaquensis* (Nees) De Winter	Graminoid	Perennial	Wind	Wind		x				
99	*Stipagrostis obtusa* (Delile) Nees	Graminoid	Perennial	Wind	Wind		x				
100	*Tribolium hispidum* (Thunb.) Desv.	Graminoid	Perennial	Wind	Wind		x				
	Restionaceae										
101	*Ischyrolepis gaudichaudianus* Kunth H.P.Linder	Dwarf shrub	Perennial	Wind	Gravity		x				
102	*Willdenowia incurvata* (Thunb.) H.P.Linder	Shrub	Perennial	Wind	Gravity		x				
	Scrophulariaceae										
103	*Hebenstretia dentata* L.	Herb	Annual	Insect	Gravity		x				x
104	*Hebenstretia repens* Jaroscz	Herb	Annual	Insect	Gravity	x					
105	*Lyperia tristis* (L.f.) Benth.	Herb	Annual	Insect	Gravity		x				
106	*Manulea altissima* L.f. ssp. *glabricaulis* (Hiern) Hilliard	Herb	Annual	Insect	Gravity				x		x
107	*Nemesia* c.f. *anisocarpa* E.Mey. ex Benth.	Herb	Annual	Insect	Gravity		x				
108	*Nemesia* sp.	Herb	Annual	Insect	Gravity		x				
109	*Phyllopodium pumilum* Benth.	Herb	Annual	Insect	Gravity		x		x		x
	Solanaceae										
110	*Lycium ferocissimum* Miers	Shrub	Perennial	Bird, Insect	Bird	x	x	x		x	
	Sterculiaceae										
111	*Hermannia cuneifolia* Jacq.	Dwarf shrub	Perennial	Insect	Mammal		x		x	x	x
112	*Hermannia disermifolia* Jacq.	Dwarf shrub	Perennial	Insect	Mammal	x	x				
113	*Hermannia scordifolia* Jacq.	Dwarf shrub	Perennial	Insect	Mammal		x				
114	*Hermannia* sp1	Dwarf shrub	Perennial	Insect	Mammal		x				
115	*Hermannia* sp2	Dwarf shrub	Perennial	Insect	Mammal	x		x		x	x
116	*Hermannia* subgen. *Mahernia*	Dwarf shrub	Perennial	Insect	Mammal	x	x	x	x	x	x
	Zygophyllaceae										
117	*Zygophyllum morgsana* L.	Shrub	Perennial	Insect	Wind	x	x	x		x	
118	*Zygophyllum sessilifolium* L.	Shrub	Perennial	Insect	Wind		x				
119	*Zygophyllum spinosum* L.	Shrub	Perennial	Insect	Wind		x				

CHAPTER 5

ASSESSMENT OF AMELIORATION OF MINED SOILS AT NAMAKWA SANDS

ABSTRACT

Strip-mining, as is practised at Namakwa Sands, causes total destruction of natural ecosystems through the removal of vegetation and soil in the area where mining is being undertaken. In terms of South African legislation, Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward. Namakwa Sands has implemented four rehabilitation techniques as a result of on-going research. The objectives of this study were to assess the success for the techniques in terms of restoring soil chemistry and to determine if soil parameters are heterogeneously distributed within both undisturbed and rehabilitated sites. Soil samples were collected from six sites (i.e. two reference and four rehabilitation sites) in two microsites (i.e. clump and inter-clump areas). The four rehabilitation sites corresponded to the four rehabilitation techniques implemented at the mine, namely site S1 (topsoil only), site S2 (seeding without topsoil), site S3 (plant translocation with topsoil) and site S4 (seeding and plant translocation without topsoil). Soil samples were analysed for pH, resistance (salinity), phosphorus, sodium, potassium, calcium, magnesium, percentage organic carbon and T-Values (approximation of the cation exchange capacity). Most of soil parameters (except P and Ca) within the two reference sites were found to be heterogeneously distributed with increased fertility occurring in clump microsites. After approximately three years, soil resources within rehabilitation sites were also found to be heterogeneously distributed (except P, Ca and C) with increased fertility occurring under clumps. There has been some recovery of the soils in rehabilitation areas compared to undisturbed soils with regard to salinity, P, Ca, K, Mg and T-Value. These parameters are unlikely to impair seedling establishment and plant growth. However, C and pH showed little evidence of returning to naturally occurring levels in any of the rehabilitation sites. It would also appear that Na levels have not recovered in two of the rehabilitation sites. Appropriate parameters for long-term monitoring therefore include C, pH and Na. The spreading of a 50 mm layer of topsoil over tailings did not improve levels of C, salinity, Na, pH, P or Ca levels compared to areas where no topsoil was replaced. However, it would appear that topsoiling did help to ameliorate the soil in terms of K, Mg and T-Value.

Key words: strip-mining; heavy minerals; rehabilitation; soil amelioration; Strandveld; soil heterogeneity; fertile islands; monitoring.

5.1 INTRODUCTION

Mining is expanding in the arid, winter-rainfall areas of South Africa and, although economically important, it is having a detrimental effect on biologically diverse environments where vegetation growth is restricted by aridity, wind and saline, nutrient-poor soils (Milton 2001; Blignaut & Milton 2005). Strip-mining, as is practised at Namakwa Sands (Figure 3.1; Chapter 3), causes total destruction of natural ecosystems through the removal of vegetation and soil in the area where mining is being undertaken.

Namakwa Sands has been mining a heavy mineral deposit, rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994. Before any area is strip-mined, all vegetation and topsoil (a minimum of 5 cm) is removed by bulldozer leaving the sub-soil exposed, resulting in a total loss of vegetation from the site (Namakwa Sands 2001). During mining and processing of soil at Namakwa Sands, the chemical and physical properties of the soil are destroyed or altered due to the removal of heavy minerals (Lubke *et al*. 1996) and fines or slimes (i.e. material <45 μm), as well as the use of seawater in the mineral separation process (de Villiers *et al*. 1999). Mining results in irreversible loss of pedosphere heterogeneity, alteration of surface water repellency, loss of micro-topography, and lowering of soil organic content and silt / clay content (Prinsloo 2005).

In terms of South African legislation (Minerals and Petroleum Resources Development Act No. 28 of 2002), mining companies are compelled to rehabilitate mined out areas. Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward with approximately 200 to 335 ha of mined land requiring rehabilitation per annum (Namakwa Sands 2001). The long-term rehabilitation goal at Namakwa Sands is to restore the area to, or as close as possible to, its natural state and achieve a vegetation cover and productivity similar to the pre-mining land-use (i.e. small-stock farming) (Grindley & Barbour 1990; Mahood 2003b). Due to the difficulty of rehabilitating mined out areas as a result of the various environmental factors restricting vegetation growth, Namakwa Sands has undertaken on-going research with the aim of increasing plant cover and biodiversity on post-mined areas (Mahood 2003b). This research and the subsequent evolvement of rehabilitation techniques have resulted in four rehabilitation techniques / scenarios at Namakwa Sands, namely (1) topsoil only; (2) seeding without topsoil; (3) plant translocation with topsoil; and (4) seeding and plant translocation without topsoil.

Soil investigations in the study area have shown that there is variation in both the chemistry and physical characteristics of the soil under natural conditions (Lanz 2003). The six vegetation communities identified within the study area appear to be correlated to various soil characteristics (e.g. depth of soil profile and salinity) (de Villiers *et al.* 1999). Therefore, the characteristics of the soil in mined out areas can influence levels of available resources, initial species establishment and successional trajectories (Biondini *et al*. 1985). Levels of available resources influence competitive interactions between species and can affect the composition and diversity of the vegetation (Chambers *et al.* 1994). Certain species in the study area are restricted to particular communities as they have narrow ecological amplitudes (Boucher & Le Roux 1989; de Villiers *et al.* 1999*)* making them the most difficult to re-establish after mining. The destruction of the soil's chemical and physical properties and the loss of soil variation could result in the loss of these

species from rehabilitated communities (de Villiers *et al.* 1999), thereby reducing the Alpha and Beta diversity of the vegetation (Milton *pers. comm*.).

Due to the correlation of the vegetation with various soil characteristics, the major objective of this study was to assess the success of the four rehabilitation techniques in terms of restoring soil chemistry to that found under natural conditions. Information on these changes is required for a better understanding of the rehabilitation techniques implemented, the interactions between the soil and vegetation and appropriate management of the rehabilitation areas (Su & Zhao 2003).

In order to rehabilitate once resilient ecosystems in arid areas it is desirable to restore as many aspects of the natural vegetation as possible after mining (Blignaut & Milton 2005), and it is essential to know what ecosystem processes and landscape functions have been disturbed in order to design rehabilitation procedures (Ludwig & Tongway 1996). Soil resources are often found to be heterogeneously distributed at small scales in arid regions with increased fertility occurring under shrub canopies (Charley & West 1975; Ludwig & Tongway 1996; Schlesinger *et al.* 1996; Desmet & Cowling 1999; Stock *et al.* 1999; Wezel *et al.* 2000; Titus *et al.* 2002; Su & Zhao 2003; Jones & Esler 2004). These areas of increased fertility are known as “fertile islands”. They are an integral part of desert landscapes and play a critical part in the structuring and functioning of desert systems (Titus *et al*. 2002). These “fertile islands” act as resource filters and have a function of conserving limited resources within semi-arid ecosystems (Ludwig & Tongway 1996). “Fertile islands” are also thought to play an important role in facilitating the recruitment and survival of other species (known as the “nurse-plant effect”) (de Villiers *et al.* 2001). The establishment of individuals below canopies could have one of a number of advantages such as protection from high irradiance, high temperature, high rates of transpiration, herbivory and wind; increased soil fertility; and increased infiltration (de Villiers *et al*. 2001; Su & Zhao 2003). The loss of these “fertile islands” through mining is therefore likely to affect the structuring and functioning of ecosystems of rehabilitation sites.

Since the vegetation at the mine site is dominated by perennial plants that occur in mixed species clumps interspersed with open areas covered by annual plants in the winter (Mahood 2003b; van Rooyen 2001), the second objective of this study was to determine if in fact soil parameters are heterogeneously distributed within the natural vegetation of the study area. If the soils are found to be heterogeneously distributed, the loss of these vegetated patches through mining could obstruct the movement of resources around the landscape and reduce the opportunities for nutrient cycling (Holm *et al.* 2002). This study also aims to determine to what extent the rehabilitation sites show this characteristic.

5.2 METHODS

5.2.1 STUDY AREA

The study was undertaken at the Namakwa Sands east mine, which is situated in the vicinity of Brand-se-Baai on the west coast of South Africa, approximately 385 km north of Cape Town (Figure 3.1; Chapter 3).

The study area is characterised by hot, dry summers and sporadic winter rainfall, falling mainly in the months from May to July (EEU 1990; le Roux & Schelpe 1997). The rainfall increases from the north to the south, with an average of 160 mm per annum occurring in the mining area (Mahood 2003a). Rainfall is augmented by heavy dew falls and sea fogs. Rainfall, sea fog and dew fall amount to a cumulative average annual precipitation of 282 mm per annum measured over a four year period (de Villiers *et al.* 1999). Compared to the average of 160 mm per annum (Mahood 2003a), it would appear that the study area has experienced a drought over the last few years with an annual rainfall of 100.5 mm, 137.5 mm and 82.9 mm being recorded in 2003, 2004 and 2005, respectively.

Climatically the Namaqualand-Namib region has relatively moderate temperatures through the year with an average annual temperature of 15.9 ^{o}C in the mining area from 2003 to 2004. The highest average monthly maximum temperature over this period was 37 ºC in summer (March) and the lowest average monthly minimum temperature was 4.7 ºC in winter (July).

The strong wind regime in the area is a major cause of erosion at Namakwa Sands (Washington 1990). The dominant winds during the spring and summer (September to March) between 2003 and 2005 were from the south-southeast and south but winds from the north-northwest and northwest were not uncommon. The dominant winds during the winter months (June to August) were from the north-northwest, northeast, northwest and east-northeast but winds from the south and southeast were not uncommon. The average annual wind speed in the mining area between 2003 and 2004 was 4.1 m/s. The highest average monthly wind speed was 5.8 m/s in winter (July) and the lowest monthly average wind speed was 3.7 m/s in summer (February).

The vegetation in the study area is classified by Low and Rebelo (1996) as consisting of Strandveld Succulent Karoo and Lowland Succulent Karoo, both of which are classified under the Succulent Karoo Biome. The Strandveld Succulent Karoo occupies the sandy coastal plain throughout Namaqualand (Cowling *et al.* 1999). It is associated with areas of calcareous sand and contains many drought deciduous and succulent species (Low and Rebelo 1996). The Lowland Succulent Karoo is dominated by members of the Mesembryanthemaceae, especially the species of *Ruschia, Drosanthemum, Malephora* and *Delosperma.* Boucher & Le Roux (1989) classified the Strandveld vegetation of the mine site into three variants according to vegetation height, i.e. tall, medium and short Strandveld. These three generalised categories were based on a combination of vegetation structure and floristic content. The three variants vary in height according to the depth of the soil profile. Tall Strandveld, dominated by

shrubs between 1 to 2 m tall, occurs on relatively deep calcareous sand, with a canopy cover of 60 – 75%, under a light grazing regime. Medium Strandveld is characterised by plants that are in the region of 50 cm tall and it has a projected canopy cover of perennial species of between 50 and 60%. Short Strandveld varies in average height from 10 – 35 cm. This community occurs in shallow soils where there is very little storage of moisture. The projected vegetation canopy cover of perennial species is usually less than 50%. De Villiers *et al.* (1999) undertook a detailed survey of the mine site and divided the mine area into six vegetation communities or associations:

1. *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld;
2. *Eriocephalus africanus – Asparagus fasciculatus* Tall Shrub Strandveld;
3. *Salvia africanus-lutea – Ballota africana* Tall Shrub Strandveld;
4. *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld;
5. *Jordaaniella spongiosa - Odyssea paucinervis* Coastal Strandveld; and
6. *Cladoraphis cyperoides – Lebeckia multiflora* Coastal Strandveld.

These vegetation communities are described in more detail in Chapter 3.

5.2.2 STUDY SITES

Vegetation and soil were sampled at six sites (i.e. four rehabilitation and two reference sites) in the east mine between 29 August 2004 and 3 September 2004. The locality of these sites is presented in Figure 4.1 (Chapter 4). The need to find nearby sites of a similar age and differing in rehabilitation technique was a constraint that resulted in pseudo-replicated sampling with only one large area per treatment type sampled. Age and rehabilitation technique also determined the size and shape of the selected sites. The six study sites included the following:

1. Reference site R1 (31° 15.853' S 17° 58.415' E): This site was located within a natural Strandveld community, which has been classified as *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (de Villiers *et al.* 1999).
2. Reference site R2 (31° 16.451' S 17° 56.181' E): This site was located within a natural Strandveld community, which has been classified as *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld (de Villiers et al. 1999). This site is subject to periodic grazing by sheep at a grazing intensity of 1 small stock unit per 10 ha.
3. Rehabilitation site S1 (31° 15.840' S 17° 56.143' E): This rehabilitation scenario involves the spreading of topsoil over tailings with no seeding and no plant translocation. The precise source of the topsoil from within the east mine and the length of topsoil stockpiling (up to three months) are unknown. It relies entirely on the germination of the seed bank in the topsoil and the importation of seed from areas outside the area of disturbance. The site selected was rehabilitated in June 2001.
4. Rehabilitation site S2 (31° 15.461' S 17° 55.544' E): This rehabilitation scenario involves the seeding of indigenous species (including the addition of *Eragrostis curvula* and *Sorghum sp.*) onto

tailings (i.e. no topsoil). Seeds were collected by vacuum harvesting of Strandveld species in areas not affected by mining and as a result the species mix is unknown. The site selected was seeded in 2001 (month unknown).

5. Rehabilitation site S3 (31° 16.165' S 17° 56.303' E): This rehabilitation scenario involves the spreading of topsoil over tailings with the translocation of five indigenous species (*Ruschia versicolor, Lampranthus sauvissimus, Othonna cylindrica, Zygophyllum morgsana* and *Asparagus spp.*) into multi-species clumps and no seeding. The precise source of the topsoil from within the east mine and the length of topsoil stockpiling (up to three months) are unknown. This site is the non-irrigated translocation trials of Mahood (2003a) undertaken in June 2001.
6. Rehabilitation site S4 (31° 15.267' S 17° 55.433' E): This rehabilitation scenario involves the seeding of indigenous species (including the addition of *Ehrharta calycina*) directly onto tailings (i.e. no topsoil) with the translocation of three indigenous species (*Ruschia versicolor, Lampranthus sauvissimus* and *Othonna cylindrica*) into multi-species clumps. Seeds were collected by vacuum harvesting of Strandveld species in areas not affected by mining and as a result the species mix is unknown. The site selected was seeded in 2001 and translocation took place between July and August 2002.

5.2.3 SOIL SAMPLING

Soil samples for chemical analysis were collected in a similar manner to that of Mahood (2003a). At each site, 12 samples consisting of eight randomly selected soil cores (5 cm deep and 4.7 cm in diameter) per sample, were collected from three line transects. The line transects were laid out as presented in Figure 5.1. The 12 samples were collected from two microsites (i.e. clump and inter-clump areas). The two clump and associated inter-clump microsites on each end of the three transects were selected for sampling. Soil samples from clump microsites were collected within the vegetation clumps underneath the vegetation canopy, and soil samples from inter-clump microsites were collected at least 1 m away from clumps in a northerly direction. This design yielded a total of 72 soil samples.

Soil samples were placed in labelled plastic bags, which were then sealed. Soil samples were analysed for pH, resistance (salinity), phosphorus (P), sodium (Na), potassium (K), calcium (Ca), magnesium (Mg), organic carbon (C) and T-Values[1] by BemLab (Pty) Ltd[2]. Soil samples were air dried and sieved through a 2 mm screen prior to chemical analysis. The analytical procedures used by BemLab are presented below.

1. pH: 10 g soil was placed in 25 ml 1 M KCl. pH was read after 1 hour (Mc Lean 1982).
2. Resistance: Soil samples were saturated with deionised water. Resistance was measured in a standard USDA soil cup (United States Salinity Laboratory Staff 1954).

[1] The T-Value is the sum of the exchangeable cations Na, K, Ca, Mg and H, and is a fair approximation of the cation exchange capacity for soils with a high resistance (Kotze *pers. comm.*).
[2] BemLab (Pty) Ltd. AECI Building W21, De Beers Road, Somerset West, Western Cape, South Africa.

3. Extractable cations (Na, K, Ca and Mg): 5 g soil was placed in 50 ml ammonium acetate solution (1M, pH 7), shaken for 30 minutes and then filtered. The filtrate was analysed for Na, K, Ca and Mg with a Varian-Vista MPX Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES) (Chapman 1965).
4. P (Bray 2): A Bray 2 extracting solution was prepared by placing 600 ml NH_4F solution (i.e. 186 g NH4F dissolved in 5 litres of deionised water) in a 20 litre aspirator and 10 litres of deionised water was added. 200 ml concentrated HCl (AR) was then added and diluted to 20 litres and mixed well. 6.6 g soil was placed in an extraction bottle and 50 ml Bray 2 solution was added. This was shaken for 40 seconds by hand and filtered. The filtrate was analysed for P with the ICP-OES (Bray & Kurtz 1945).
5. C (Walkley Black): 0.2 to 0.5 g soil was measured into a 500 ml Erlenmeyer flask. 10 ml 0.175 M potassium dichromate and 20 ml concentrated sulphuric acid were added and allowed to stand for 30 minutes. 150 ml deionised water and 10 ml ortho-phosphoric acid were added to the flask. The solution was tritrated with standardised iron (II) ammonium sulphate with N-phenylanthranilic acid indicator to the green end-point (Nelson & Sommers 1982).

5.2.4 VEGETATION SAMPLING

The vegetation was sampled using the Line Intercept Method (Sutherland 1997) as the natural vegetation is fairly patchy and this method is suitable for use in sparse, low vegetation with a projected canopy cover of less than 50%. Line transects (50 m) were located randomly within each site. The transects were orientated in a "W" pattern (Figure 5.1) to prevent the possibility of the transects being oriented parallel to a translocated clump row (as undertaken in rehabilitation sites S3 and S4), thereby ensuring that clumps would be sampled.

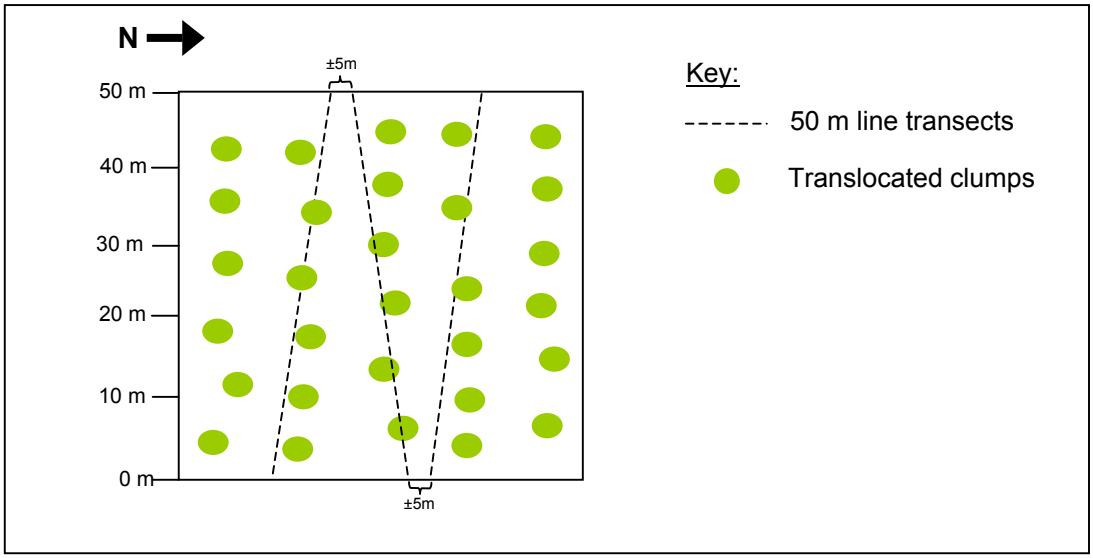


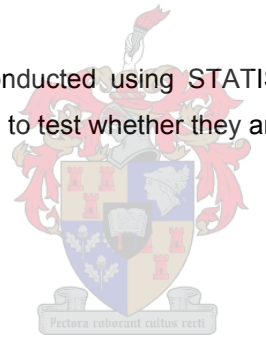
Figure 5.1: Layout of the three line transects showing the W-pattern to ensure that translocated clumps were sampled and distance between transects.

Three transects were randomly located within each of the six sites resulting in a total of 18 transects. For all transects, a 50 m tape measure was laid across the transect and wherever the tape measure intercepted (above or below) a plant, the species was recorded and the point at which each specific plant started and stopped along the tape measure was recorded. In addition, it was noted whether the plant was solitary or part of a clump. Wherever the tape measure intercepted (above or below) a clump, the point at which each clump started and stopped was also recorded. All the intercepts for individual plants and clumps were added and then divided by the total length of the transect line (i.e. 50 m) to provide percentage cover of the vegetation (total cover and species cover) and clumps.

5.2.5 DATA ANALYSIS

For each soil parameter the mean concentration was calculated for samples taken from clump and inter-clump microsites in each of the six sites. Differences in soil parameters among sites and microsites were evaluated using Repeated Measures Analysis of Variance (ANOVA) using STATISTICA 7.0 (StatSoft, Inc. 2004). *Post Hoc* tests were conducted by Fisher LSD using STATISTICA 7.0. The data were tested for outliers using normal probability plots and in all cases the outliers did not influence the results.

Pearson's R correlation tests were conducted using STATISTICA 6.1 between soil parameters and between soil and vegetation parameters to test whether they are correlated with one another.



5.3 RESULTS

5.3.1 MICROSITE EFFECT

Independent of species composition, most of the analysed soil parameters (except P and Ca) showed that significant differences exist between soils under clumps and those in inter-clump areas (Table 5.1).

Clump microsites had significantly higher levels of soil pH, Na, K, Mg, C and T-Value than inter-clump soils (Table 5.1), but the Fisher LSD *post hoc* tests only found this to be true across all sites for pH and K (Figure 5.2 and 5.3). In fact, the significantly higher C level in clumps was only true for the reference sites where levels were increased by between two and three times in comparison to between 1.02 and 1.5 times in rehabilitation sites (Figure 5.4).

Soils from inter-clump areas had a significantly higher resistance level or lower salinity (between two and six times) than clump microsites (Table 5.1), but this finding was not significant for sites S2 and S3 (Figure 5.2).

There was no significant difference in soil P and Ca levels across microsites (Table 5.1). P levels between clumps and inter-clumps were found to be highly variable.

Table 5.1: Two-Way ANOVA F's and P's for measured soil parameters in two microsites (clump v inter-clump) at six Namakwa Sands sites (n=6 for microsite, n=12 for site; p<0.05 for significance). Significant values are in bold.

Soil parameter	Site		Microsite		Site and microsite interaction	
	F	P	F	P	F	P
pH (KCl)	19.44	**<0.001**	170.37	**<0.001**	4.01	**0.006**
Resistance (Ohm)	5.46	**<0.001**	74.48	**<0.001**	4.51	**0.003**
P (mg/kg)	30.07	**<0.001**	0.61	0.441	0.46	0.802
Na ($cmol_c/kg$)	22.05	**<0.001**	97.74	**<0.001**	5.64	**<0.001**
K ($cmol_c/kg$)	33.48	**<0.001**	92.43	**<0.001**	2.76	**0.036**
Ca ($cmol_c/kg$)	16.27	**<0.001**	4.05	0.053	1.66	0.176
Mg ($cmol_c/kg$)	77.63	**<0.001**	38.55	**<0.001**	4.60	**0.003**
C (%)	20.8	**<0.001**	9.21	**<0.005**	2.57	**<0.048**
T-Value	26.07	**<0.001**	18.31	**<0.001**	1.83	0.138

5.3.2 SITE EFFECT

Independent of species composition, all analysed soil parameters showed that significant differences exist between soils from different sites (Table 5.1).

Rehabilitation sites had higher pH levels (but not always significant) than reference sites, with sites S1 and S3 having slightly higher pH levels than sites S2 and S4 (Figure 5.2). Similarly, rehabilitation sites had higher Na levels (but not always significant) than reference sites with Site S3 having the highest Na level (Figure 5.2).

Reference sites had similar clump resistance levels to rehabilitation sites (Figure 5.2). Soil from site S3 had the lowest resistance level. Inter-clump microsites were more variable with reference sites having similar resistance levels to rehabilitation sites S1 and S4.

P, K, Ca, Mg and C levels were the highest in soils from reference site R1 (Figure 5.3 and 5.4). Reference site R1 had significantly higher P and Ca levels than all other sites. Reference site R2 had similar levels of P and Ca to the rehabilitation sites. The reference sites had significantly higher C levels than the rehabilitation sites (Figure 5.4). There was no significant difference in C levels across the rehabilitation sites. K, Mg and T-Value levels were found to be variable between sites (Figure 5.3 and 5.4). Sites R1 and S3 had the highest and second highest levels, respectively. Soil from site R2 had similar K, Mg and T-Value levels to Sites S1 and S4.

The relative proportion of Na, K, Ca and Mg contributing to the T-Value between reference and rehabilitation sites differed in both clump and inter-clump microsites. In general, the reference sites had a higher percentage of Ca contributing to the T-Value, whereas the reference sites had higher percentages of K, Na and Mg (Figure 5.5).

5.3.3 VEGETATION

The reference sites had greater vegetation and clump cover than the rehabilitation sites (Table 5.2). However, these differences were not always significant when comparing reference site R2 to the rehabilitation sites.

The rehabilitation sites were similar in terms of vegetation and clump cover. However, it should be noted that sites S3 and S4 (i.e. sites with plant translocation) generally had slightly greater vegetation and clump cover than sites S1 and S2 (i.e. sites without plant translocation).

Table 5.2: Percentage vegetation and clump cover along three transects at the six sites at Namakwa Sands (mean ± standard deviation, n=3). Vegetation parameters with different superscripts are different at $p<0.05$ by Fisher LSD *post hoc* tests. The six sites are described in Figure 5.2.

Vegetation parameter	Site					
	R1	R2	S1	S2	S3	S4
Vegetation cover (%)	38.66 ±6.03^{a}	31.13 ±2.17^{b}	23.61 ±5.94^{bc}	20.47 ±2.22^{c}	24.93 ±2.27^{bc}	25.89 ±4.55^{bc}
Clump cover (%)	27.39 ±6.60^{a}	15.95 ±4.29^{b}	10.75 ±6.564^{b}	8.62 ±4.78^{b}	11.87 ±3.07^{b}	12.47 ±3.42^{b}

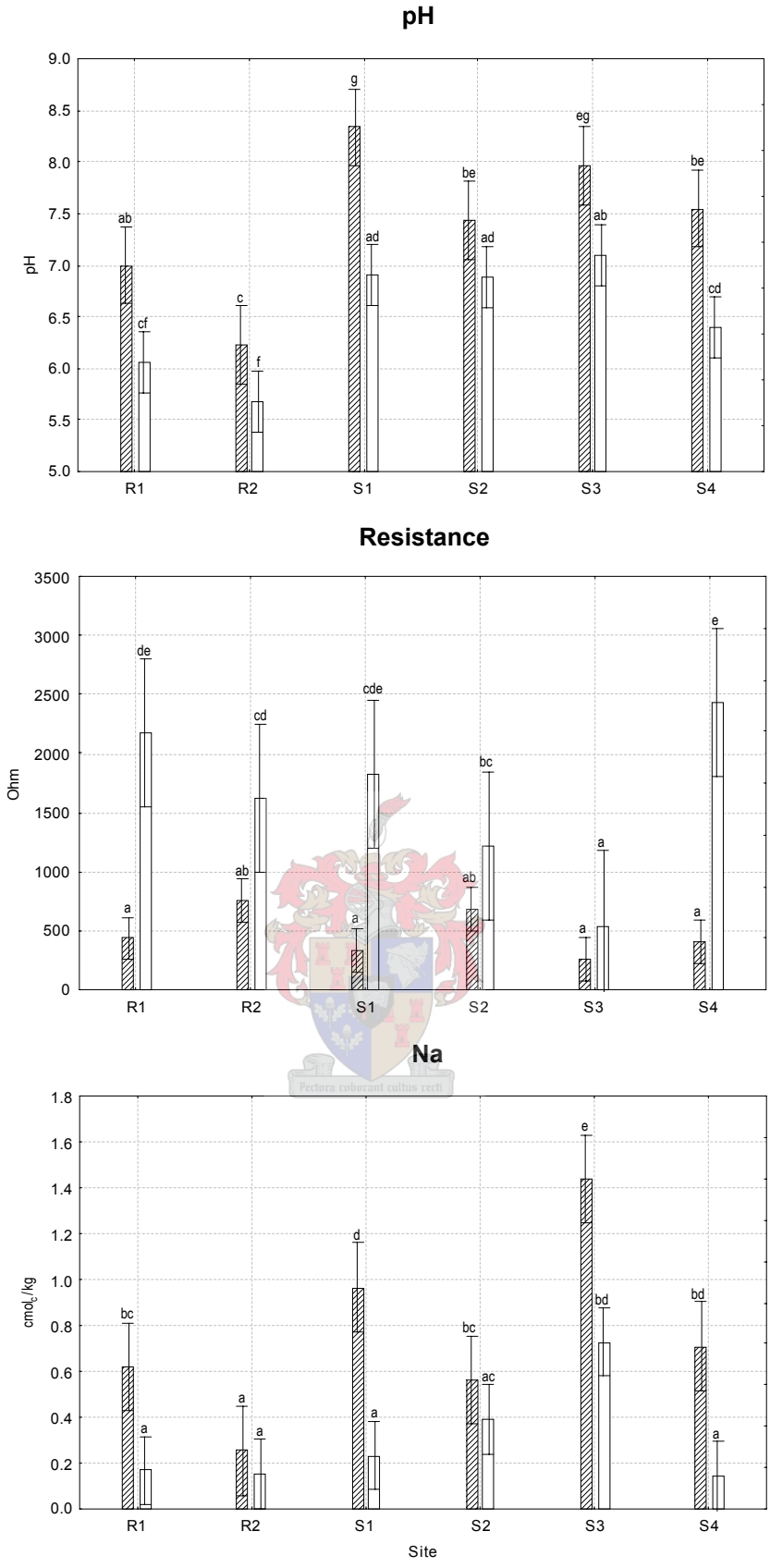


Figure 5.2: pH, Resistance and Na in clump (▨) and inter-clump (□) areas at six Namakwa Sands sites (mean ± standard deviation, n=6). Soil parameter values with different letters are different at p<0.05 by Fisher LSD *post hoc* tests. The six sites included: Reference site R1 (*Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld); Reference site R2 (*Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld); Rehabilitation site S1 (topsoil only); Rehabilitation site S2 (seeding without topsoil); Rehabilitation site S3 (plant translocation with topsoil); and Rehabilitation site S4 (seeding and plant translocation without topsoil).

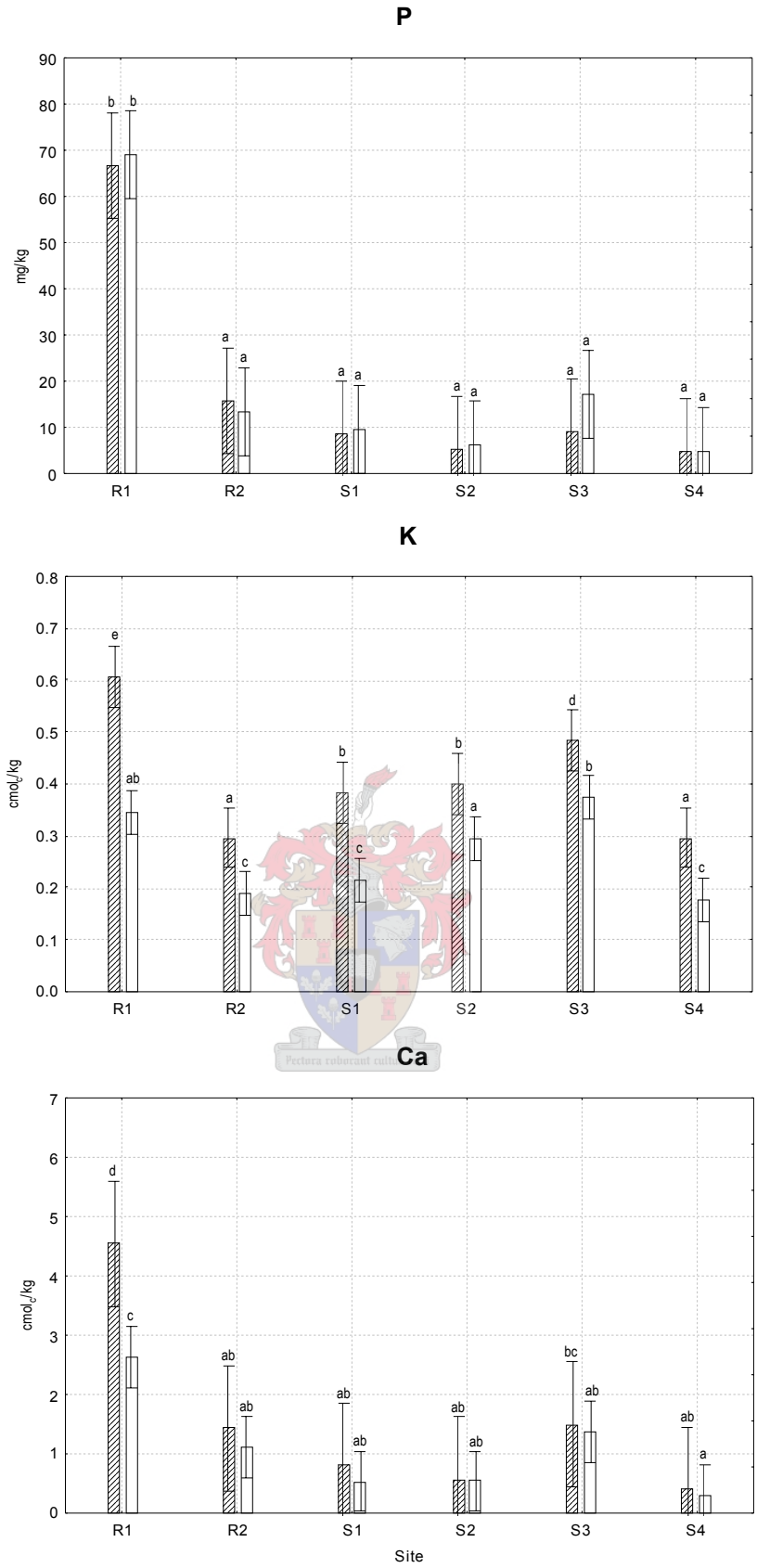


Figure 5.3: P, K and Ca in clump (▨) and inter-clump (▭) areas at six Namakwa Sands sites (mean ± standard deviation, n=6). Soil parameter values with different letters are different at $p<0.05$ by Fisher LSD *post hoc* tests. The six sites are described in Figure 5.2.

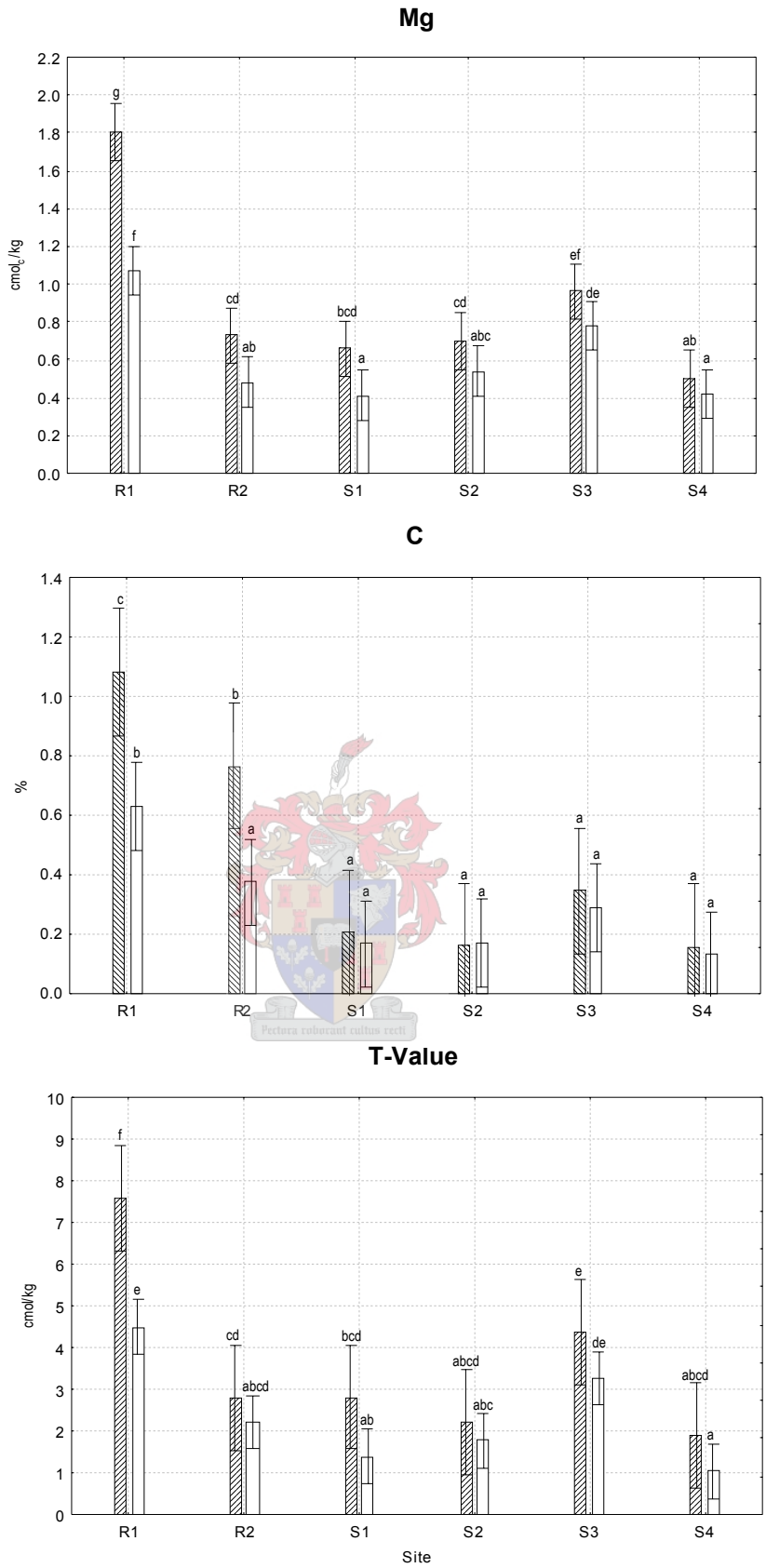


Figure 5.4: Mg, C and T-Value in clump ( ) and inter-clump ( ) areas at six Namakwa Sands sites (mean ± standard deviation, n=6). Soil parameter values with different letters are different at $p<0.05$ by Fisher LSD *post hoc* tests. The T-Value is the sum of the exchangeable cations Na, K, Ca, Mg and H. The six sites are described in Figure 5.2.

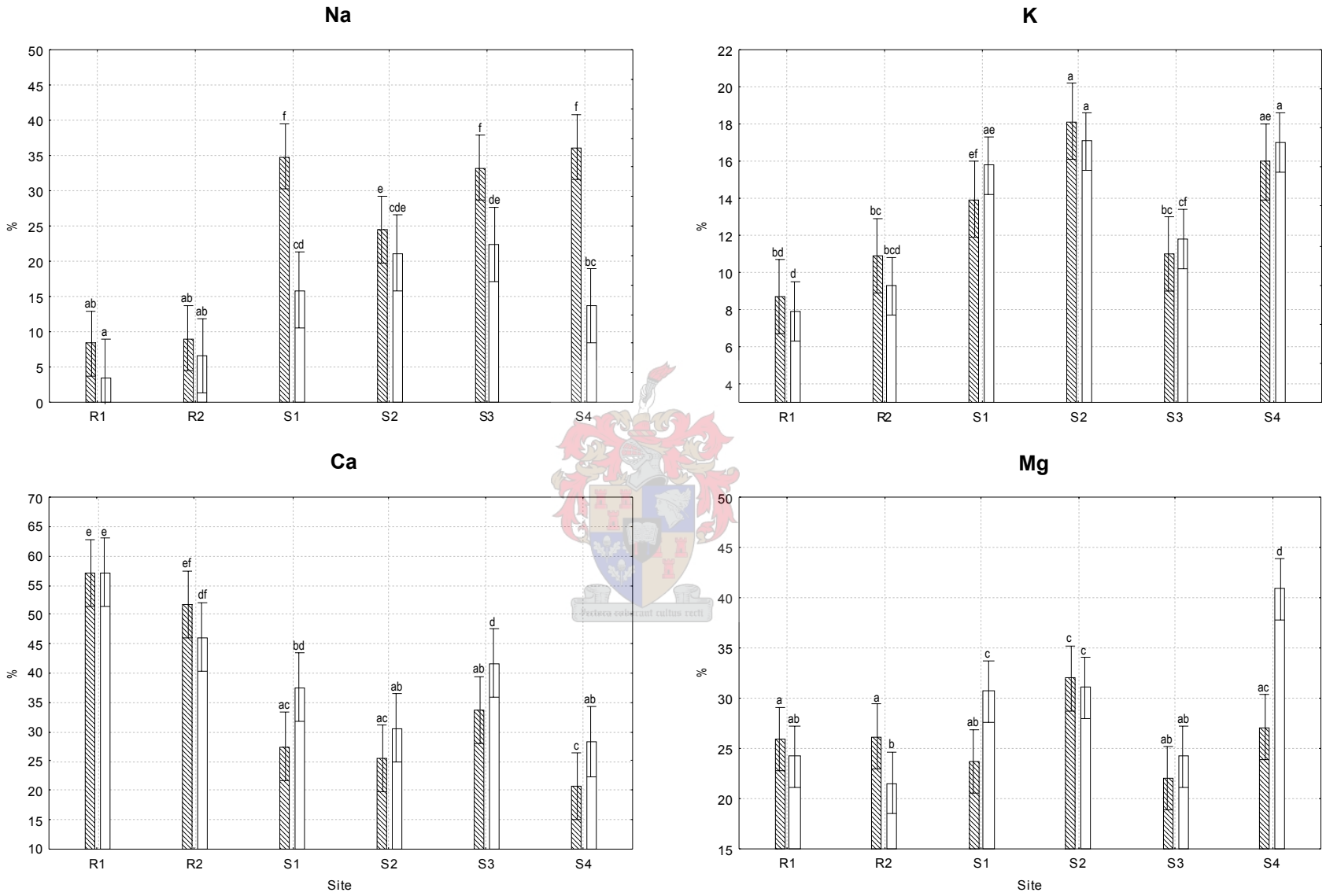


Figure 5.5: The relative proportion of Na, K, Ca and Mg contributing to the T-Value in clump (▨) and inter-clump (▭) areas at six Namakwa Sands sites (mean ± standard deviation, n=6). The T-Value is the sum of the exchangeable cations Na, K, Ca, Mg and H. Soil parameter values with different letters are different at $p<0.05$ by Fisher LSD *post hoc* tests. The six sites are described in Figure 5.2.

5.3.4 CORRELATIONS

Correlations between soil parameters and cover (total and clump) from two microsites (clump and inter-clump) are presented in Table 5.3. Many of the soil parameters are significantly (positively or negatively) correlated with each other. In both the reference and rehabilitation sites, resistance levels were negatively correlated to all other soil parameters, but not always to a significant degree. All other soil parameters (except the correlation between pH & P for rehabilitation sites) were positively correlated, but not all correlations were significant.

With the exception of K and total cover within rehabilitation sites, cover is generally positively correlated with all of the soil parameters within the reference and rehabilitation sites. However, these positive correlations are only significant for a few soil variables within the reference sites.

Table 5.3: R- and P-values of Pearson's correlation between soil parameters and cover (total and clump) from two microsites (clump and inter-clump) at the reference (n=6) and rehabilitation (n=12) sites at Namakwa Sands. Significant values (p<0.05) are in bold.

Site	Soil parameter	Resistance		P		Na		K		Ca		Mg		C		T-Value		Total cover		Clump cover	
		r	p	r	p	r	p	r	p	r	p	r	p	r	p	r	p	r	p	r	p
Reference sites	pH	-0.64	**<0.01**	0.43	**0.03**	0.81	**<0.01**	0.83	**<0.01**	0.69	**<0.01**	0.79	**<0.01**	0.68	**<0.01**	0.74	**<0.01**	0.47	0.13	0.34	0.27
	Resistance			-0.01	0.9	-0.68	**<0.01**	-0.58	**<0.01**	-0.37	0.07	-0.48	**0.02**	-0.52	**<0.01**	-0.42	**0.04**	0.04	0.90	0.03	0.91
	P					0.44	**0.03**	0.66	**<0.01**	0.67	**<0.01**	0.68	**<0.01**	0.53	**<0.01**	0.696	**<0.01**	0.83	**<0.01**	0.62	**0.03**
	Na							0.88	**<0.01**	0.84	**<0.01**	0.85	**0.01**	0.78	**<0.01**	0.89	**<0.01**	0.37	0.24	0.26	0.41
	K									0.79	**<0.01**	0.94	**<0.01**	0.72	**<0.01**	0.87	**<0.01**	0.59	**0.04**	0.57	**0.05**
	Ca											0.77	**<0.01**	0.89	**<0.01**	0.98	**<0.01**	0.69	**0.01**	0.44	0.15
	Mg													0.66	**<0.01**	0.87	**<0.01**	0.47	0.10	0.52	0.08
	C															0.87	**<0.01**	0.55	0.07	0.32	0.31
	T-Value																	0.64	**0.02**	0.48	0.12
Rehabilitation sites	pH	-0.67	**<0.01**	-0.04	0.76	0.80	**<0.01**	0.71	**<0.01**	0.299	**<0.04**	0.52	**<0.01**	0.28	**0.05**	0.62	**<0.01**	0.09	0.67	0.08	0.70
	Resistance			-0.08	0.61	-0.75	**<0.01**	-0.73	**<0.01**	-0.47	**<0.01**	-0.61	**<0.01**	-0.46	**<0.01**	-0.70	**<0.01**	0.01	0.96	-0.06	0.76
	P					0.003	0.99	0.10	0.48	0.44	**<0.01**	0.27	0.07	0.27	0.06	0.26	0.07	0.16	0.46	0.17	0.43
	Na							0.795	**<0.01**	0.54	**<0.01**	0.74	**<0.01**	0.53	**<0.01**	0.86	**<0.01**	0.17	0.43	0.18	0.41
	K									0.62	**<0.01**	0.84	**<0.01**	0.54	**<0.01**	0.85	**<0.01**	-0.01	0.97	0.02	0.89
	Ca											0.82	**<0.01**	0.85	**<0.01**	0.88	**<0.01**	0.20	0.34	0.20	0.34
	Mg													0.73	**<0.01**	0.94	**<0.01**	0.00	1.00	0.02	0.91
	C															0.79	**<0.01**	0.16	0.46	0.09	0.68
	T-Value																	0.15	0.48	0.16	0.45

5.4 DISCUSSION

Microsite effect

Shrubs have an important effect on the distribution of nutrients in the landscape with a significantly higher concentration of various soil parameters occurring under shrubs (Wezel *et al.* 2000; Su & Zhao 2003). With the exception of P and Ca, soil parameters analysed were found to be heterogeneously distributed within the reference sites, with clump microsites having significantly higher levels of soil pH, salinity, K, Na, Mg, C and T-Value than inter-clump soils. This heterogeneous distribution of cations and carbon, with increased fertility occurring under shrub canopies (know as "fertile islands"), is not uncommon in arid and semi-arid areas (Charley & West 1975; Schlesinger *et al.* 1996; Desmet & Cowling 1999; Titus *et al.* 2002; Jones & Esler 2004). These "fertile islands" appear to be an important characteristic of the Strandveld vegetation in the study area. Approximately three years after rehabilitation commenced there is some evidence of "fertile island" formation within rehabilitation sites, as clump microsites were found to have significantly higher levels of soil pH, salinity, K, Na, Mg and T-Value than inter-clump soils. Interestingly C was not found to be significantly different between clump and inter-clump microsites in rehabilitation sites.

There are a number of biotic and abiotic factors that contribute to the heterogeneous distribution of soil parameters. Most importantly, vegetated patches can preserve and concentrate soil nutrients through internal recycling or nutrient cycling (Charley & West 1975). In addition, vegetated patches are able to effectively trap material (e.g. litter and soil) from within and nearby areas by reducing wind velocities (Ludwig & Tongway 1996; Wezel *et al*. 2000; Titus *et al*. 2002). The faunal transport of material to vegetated patches and associated faunal excreta (urine and faeces) have also been found to increase soil fertility within clump microsites (Desmet & Cowling 1999; Titus *et al.* 2002). Similarly, the fluvial transport of litter, soil and nutrients under vegetated patches (Ludwig & Tongway 1996) as well as rain splash erosion where sediment in interspace areas is transported towards or beneath shrubs (Parsons *et al*. 1992) can play a role in increasing the fertility under clumps. Vegetated patches generally have greater infiltration (Ludwig & Tongway 1996; van de Koppel *et al.* 2002) due to the increased accumulation of litter and organic matter under shrubs, as well as the compaction of interspaced areas due to rain and animals (Titus *et al.* 2002). The greater the availability of water and its influence on plant growth, the greater the nutrient cycling and thus soil fertility (Holm *et al*. 2002). However, this is contrary to preliminary field investigations at Namakwa Sands that indicate that hydrophobicity was most severe under the shrub canopy (Prinsloo 2005). Prinsloo (2005) speculated that the hydrophobic soil surface surrounding shrub canopies in the study area induce and accelerate water runoff, which accumulates and infiltrates in inter-clump areas.

The significantly higher C levels in clumped microsites of reference sites is supported by the results of other studies (Charley & West 1975; Wezel *et al.* 2000; Holm *et al.* 2002; Su & Zhao 2003) and is related to the greater accumulation of plant litter or organic matter under shrubs compared to inter-patch areas (Wezel *et al.* 2000, Holm *et al.* 2002). Vegetation reduces wind velocities and results in the entrapment of sand and organic matter such as leaf litter, seeds and other plant debris underneath clumps (Wezel *et al.*

2000; Titus *et al.* 2002; personal observation), whereas the exposed inter-clump areas are subject to erosive forces of strong winds and as a result organic matter is blown away from these areas. The rehabilitated sites studied are relatively young and even though the C levels were slightly greater in clump microsites than inter-clump microsites it is speculated that more time is needed to accumulate C levels in clump microsites (Charley & West 1975; Su & Zhao 2003). The translocation of species into multi-species clumps (as in sites S3 and S4) did not result in a significantly higher C in clumps in the short-term, but this may become more apparent in the long-term. Thus, C may be a useful parameter to monitor as part of long-term monitoring programme.

The deposition of salts at the soil surface through wind-blown salt spray / mist results in undisturbed soils being naturally saline (EEU 1990; Lanz 2003; Blignaut & Milton 2005). The significantly higher salinity (i.e. lower resistance), Na, K and Mg levels in clump microsites could be attributed to the fact that vegetation clumps intercept more wind-blown sea spray / mist than inter-clump areas. The intercepted sea spray / mist and subsequent condensation thereof results in the soil directly below the clumps receiving more water, which is likely to contain sea salts (personal observation) (Plate 5.1 to 5.3). This finding is supported by the positive (although generally not significant) correlation between clump cover and Na, K and Mg (Table 5.3). The significantly higher K levels under clumps could also be linked to the higher concentration of organic matter there (Wezel *et al.* 2000). This is supported by the significant positive correlation between K and C levels in both reference and rehabilitation sites (Table 5.3). The significantly higher Na and Mg levels in clumps did not concur with the findings of Wezel *et al*. (2000), who found that Na and Mg were not significantly different under shrubs and in inter-clump areas. Since the T-Value is a fair approximation of the cation exchange capacity in soils with a low salt content (Kotze *pers. comm.*), one can assume that clump microsites have a higher cation exchange capacity than inter-clump microsites. This concurs with the result of Wezel *et al.* (2000), who found the cation exchange capacity under shrubs was slightly greater. Wezel *et al.* (2000) related this to increased soil organic matter and clay ratio under shrubs.

Soil pH values in arid regions have been found to be significantly lower (Charley & West 1975; Wezel *et al.* 2000; Titus *et al.* 2002) and similar (Su & Zhao 2003; Prinsloo 2005) in clump microsites than inter-clump microsites, whereas this study and that of Charley & West (1975) found clump microsites to have a significantly higher soil pH than inter-clump microsites. The significantly higher soil pH level of clump microsites at Namakwa Sands could be related to the fact that cations (Na, K and Mg) were also concentrated within clump microsites. This finding is supported by the significant positive correlation between these cations and pH levels (Table 5.3) and the findings of Prinsloo (2005), who found that a salt crust commonly occurs beneath shrubs throughout the study area. Prinsloo (2005) ascribed this to the interception of frequent wind-blown sea spray / mist with the shrub canopy and the subsequent increased precipitation below the canopy (Plate 5.3).

Plate 5.1: Wind-blown sea spray / mist augments the rainfall at Namakwa Sands.

Plate 5.2: Condensation of wind-blown sea spray / mist on *Tripteris oppositifolium* leaves.

Plate 5.3: The area directly below a clump canopy receives more water (which is likely to contain sea salts) due to wind-blown sea spray / mist than inter-clump areas.

Clump microsites in both reference and rehabilitation sites were generally found not to have significantly different levels of soil P and Ca to inter-clump soils (Table 5.1). However, other studies have shown that shrubs have an important effect on the distribution of P levels with a significantly higher concentration occurring under shrubs (Charley & West 1975; Wezel *et al*. 2000). Wezel *et al*. (2000) linked this to the higher concentration of organic matter under shrubs. The fact that there was no significant difference in C between clump and inter-clump microsites in rehabilitated soil profiles may explain the insignificant difference between P levels. However, this does not explain the result for reference sites as clumps did have a significantly higher C levels than inter-clump areas.

Site effect

The chemical and physical characteristics of the soil within the study area have been shown to be spatially heterogeneous (Lanz 2003). De Villiers *et al*. (1999) correlated the six vegetation communities to soil depth and salinity. Therefore, the finding that most soil parameters (except pH (inter-clump), Na (inter-clump) and salinity) were significantly different between the two reference sites was not totally unexpected. The differences in soil resources between these two vegetation communities (i.e. *Ruschia tumidula – Tetragonia virgata* Tall Shrub Strandveld and *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld) could be related to a number of environmental and anthropogenic factors such as distance from the sea, altitude, species composition, chemically distinct litter types (Charley & West 1975; Titus *et al.* 2002), clump size (Titus *et al.* 2002), wind intensity, grazing intensity (Charley & West 1975), infiltration (van de Koppel *et al.* 2002) and differences in the soil parent material (Charley & West 1975). Failure to take account of this naturally occurring spatial distribution of soil parameters during monitoring could lead to false interpretations and may seriously affect the outcome of a monitoring programme. Therefore, reference sites must be carefully selected for the long-term monitoring programme at Namakwa Sands, especially once mining moves out of the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (site R1) and into the *Ruschia tumidula – Tetragonia virgata* Tall Shrub (site R2). Comparisons should preferably be made against the vegetation community that existed prior to mining

The amelioration of the soil profile after mining takes a long time to be achieved (Lubke *et al*. 1996). Even though the rehabilitation sites studied at Namakwa Sands are relatively young in rehabilitation terms there has been some soil amelioration within the three years if one compares the rehabilitation and reference sites. However, C, pH and Na levels in the rehabilitation sites do not appear to show a tendency of returning to pre-mining levels.

Clump microsites of reference sites had significantly higher C levels than clump microsites of rehabilitation sites. This finding is consistent with that of Mahood (2003a) and Prinsloo (2005), who found that the C levels in rehabilitation sites had not recovered to levels found in undisturbed soils. Organic matter in rehabilitated profiles is likely to increase over time (Su & Zhao 2003) as a result of plant growth and the entrapment of this material underneath clumps. This is likely to be a slow process, as the vegetation in the study area is restricted by aridity, wind and nutrient-poor soils (Milton 2001). In addition, annuals that would normally provide an important pulse of soil organic carbon and nutrients (Munn &

Whitford 1998) are poorly represented in the rehabilitation sites (results are presented in Chapter 4). Mahood (2003a) suggested that topsoil should be spread over the tailings during rehabilitation in order to ameliorate C levels. The results of this study show that in the short-term (after approximately 3 years) plant translocation and the spreading of topsoil over tailings (Site S3) and the spreading of topsoil over tailings only (Site S1) did not significantly increase, although slightly higher, soil C. This concurs with the findings of Mahood (2003a) who found that after 15 months C levels in areas with and without topsoil were similar. The reduced levels of C found in the rehabilitated sites could have an impact on rehabilitation success. Thus, C appears to be a useful parameter to monitor as part of the long-term monitoring programme. Should long-term monitoring indicate that C is a limiting factor affecting the success of the rehabilitation programme, Namakwa Sands should consider placing cleared vegetation from the mining front on mined out areas. This vegetation material would decompose over time thereby increasing the organic matter in rehabilitation sites. This recommendation should be tested as part the ongoing research and rehabilitation development at Namakwa Sands.

The pH of rehabilitated soils does not appear to have recovered approximately three years after rehabilitation. In general, the reference sites (both clump and inter-clump microsites) had significantly lower pH levels than rehabilitation sites. This result is supported by the findings of Mahood (2003a), who found that after 15 months the pH levels of topsoil spread over tailings remained at slightly higher levels than topsoil from undisturbed sites. However, this finding was not supported by Prinsloo (2005), who found there to be no significant difference between the pH in natural and rehabilitated soil profiles. The vegetation occurring in the area is naturally adapted to the medium acidity of the soils. However, the significant change evident in the soil pH may negatively affect plant growth in rehabilitated areas, as pH affects nutrient availability in plants (Mahood 2003a). Lanz (2003) found that soils closer to the coast tended to have a higher pH than soils further inland. Therefore, the significantly higher pH levels in rehabilitation soils could be partly due to the fact that these sites are closer to the coast than the reference sites (Figure 4.1; Chapter 4). However, it is more likely that these sites have not recovered significantly from the use of seawater in the mineral separation process. More time is required in order for these levels to decrease to those levels evident in the reference sites. However, if over time pH is found to be one of the factors affecting the success of rehabilitation, a cost-effective amelioration measure may have to be considered for implementation by Namakwa Sands (Mahood 2003a). pH appears to be a useful parameter to monitor as part of long-term monitoring programme.

Na levels in rehabilitated soils also do not appear to have recovered approximately three years after rehabilitation. Rehabilitation sites had higher Na levels than reference sites, with sites S1 and S3 having significantly higher levels than the reference sites (Figure 5.2). Therefore, it would appear that sites S1 & S3 have not recovered significantly after mining. Mahood (2003a) also found that, after 15 months, Na levels in non-irrigated topsoil sites (site S3) were still higher than undisturbed topsoil levels. Thus, Na levels also appear to be a useful parameter to monitor as part of a long-term monitoring programme.

Various inhibitory factors may limit or prevent vegetation growth after mining and the replacement of topsoil may be important to reduce these potential inhibitory factors (Bradshaw 1997). Mahood (2003a) hypothesised that the use of stockpiled topsoil in rehabilitation was vital to achieving a suitable plant

cover on mined out areas due to the fact that the soil is processed using seawater. However, she found that after 15 months the replacement of topsoil during rehabilitation did not improve the soil pH or Na levels compared to areas where only soil tailings were replaced. Mahood (2003a) attributed the higher pH to the fact that stockpiled topsoil was found to have a significantly higher pH than processed tailings. This study found that after approximately three years, topsoiled sites (sites S1 and S3) still had higher pH and Na levels than sites where no topsoil was replaced.

Mahood (2003a) also suggested that stockpiled topsoil should be replaced over tailings in order to ameliorate K, Mg, P and Ca levels, thereby providing conditions more conducive to seedling establishment and survival. This study did not identify any clear trends with regard to K, Mg, P and Ca levels between reference and rehabilitation sites and between rehabilitation sites. Site R1 (clump microsite) had significantly higher K, Mg, P and Ca levels than all rehabilitation sites, but site R2 generally had the lowest K level and similar Mg, P and Ca levels to the rehabilitation sites. Therefore, it would appear that K, Mg, P and Ca levels within rehabilitation sites have recovered to some extent in relation to site R2 but not in relation to site R1. It is assumed that K, Mg, P and Ca levels are unlikely to have a significant effect on plant growth at the levels found in rehabilitated soils, as they fall between the levels found for the two reference sites. It would also appear that spreading of topsoil over tailings and the translocation of species into multi-species clumps did help to ameliorate the soil in terms of K and Mg levels, as site S3 (which includes both topsoil and plant translocation) had the second highest K level after site R1. However, since P and Ca levels in topsoiled areas (sites S1 and S3) was not significantly different to areas where no topsoil was replaced (sites S2 and S4), it may be concluded that the spreading of 50 mm of topsoil over tailings did not return a significant amount of P and Ca to the soil or that the small amount of P and Ca returned in the topsoil had been taken up by the vegetation. It is possible that the addition of more topsoil would have yielded different results (Milton *pers. comm.*).

Undisturbed soils are naturally saline due to the deposition of salts on the soil surface from wind-blown sea spray (EEU 1990; Lanz 2003; Blignaut & Milton 2005). High salinity, coupled with low levels of available water, can impose a limitation on plant growth and therefore rehabilitation through a high osmotic concentration resulting in a low water potential. If increased above the tolerance of a plant, it will limit water uptake by the plant (Mahood 2003a; Lanz 2003). Mahood (2003a) found that the saturation of the soil with seawater during the mineral separation process and the stockpiling of topsoil significantly increased the salinity of both tailings and topsoil. Mahood (2003a) and Lanz (2003) found that the salinity of tailings and topsoil decreased fairly rapidly in rehabilitated soils to levels similar to those in undisturbed soils. This trend was consistent with the findings of Prinsloo (2005) and this study as salinity levels in clump microsites of rehabilitation soils were not significantly different to the reference sites. Lanz (2003) and Prinsloo (2005) attribute this decreased salinity to the leaching of salts downwards in the soil profile. Thus, it can be assumed that salinity is unlikely going to be a factor limiting vegetation recovery within the rehabilitation sites.

Fines and organic matter fractions provide cation exchange sites and their absence can result in the rapid leaching of any inorganic nutrients (Tordoff *et al.* 2000). Prinsloo (2005) found that rehabilitation soils had a significantly lower silt content due to the removal of fines in the separation process, which is expected

to diminish the fertility status and cation exchange capacity of the soils. The T-Value, which can be a fair approximation of the cation exchange capacity (Kotze *pers. comm.*), shows a similar trend as found for K and Mg levels, i.e. the clump microsite of site R1 had a significantly higher T-Value than all other sites, followed by site S3. Site R2 had a similar T-Value to the other three rehabilitation sites (i.e. Sites S1, S2 & S4). Lanz (2003) also found that rehabilitated profiles had similar cation exchange capacities to undisturbed soils. Therefore, it would appear that T-Value levels of the rehabilitation sites have been ameliorated to some extent in terms of site R2 but not site R1, and that the spreading of topsoil over tailings as well as the translocation of species into multi-species clumps does help to ameliorate the soil in terms of T-Value levels.

5.5 CONCLUSIONS

The differences that exist between the soil chemical parameters of the two reference sites, which are dominated by different vegetation communities, suggests that most of soil parameters are naturally spatially distributed within the study area. With the exception of P and Ca, soil parameters measured in reference sites were also found to be heterogeneously distributed with increased fertility occurring in clump microsites, as found in other arid regions. These results support the concept of “fertile islands”, and it appears that this is an important characteristic of the Strandveld vegetation in the study area. Failure to take account of the spatial and heterogeneous distribution of soil resources during monitoring can lead to false interpretations of the data (Charley & West 1975). Therefore, reference sites must be carefully selected and comparisons should preferably be made against the vegetation community that existed prior to mining.

The soil resources within rehabilitation sites were also found to be heterogeneously distributed (except P, Ca and C) with increased fertility occurring under clumps. Therefore, it can be assumed that the rehabilitation sites are improving in terms of structure and function. However, more time is needed to ameliorate the rehabilitation soils (especially with regard to C levels) to the same level as in reference sites. The decomposition of vegetation material cleared from the mining front and placed on rehabilitation areas would increase organic matter. This proposal should be tested as part of ongoing research and rehabilitation development at Namakwa Sands.

The results suggest that there has been some recovery of the soils in rehabilitation areas with regard to salinity, P, Ca, K, Mg and T-Value. These parameters are unlikely to impair seedling establishment and plant growth. However, C and pH showed little evidence of returning to naturally occurring levels approximately three years after rehabilitation commenced. It would also appear that not all rehabilitation sites (i.e. sites S1 and S3) have been ameliorated in terms of Na levels. Therefore, C, pH and Na appear to be useful parameters to monitor as part of a long-term monitoring programme. The rehabilitated soil profiles are relatively young but should long-term monitoring show that these parameters are affecting the success of the rehabilitation, various cost-effective amelioration measures may have to be considered for implementation by Namakwa Sands.

The spreading of a 50 mm layer of topsoil over tailings did not improve levels of C, salinity, Na, pH, P and Ca levels of soils compared to areas where no topsoil was replaced. However, it would appear that topsoiling did help to ameliorate the soil in terms of K, Mg and T-Value. The replacement of topsoil, which contains the seed bank, will also be a vital source of seed for annual and perennial species recruitment and will ultimately predict the future vegetation (de Villiers *et al.* 2003), and is beneficial for mycorrhizal re-establishment in mined out areas (Ndeinoma 2005).

This study only considered the amelioration of soil chemical properties between mined and unmined areas. Since mining also alters the physical properties of the soil (Lubke *et al*. 1996), which may limit vegetation establishment and ultimately rehabilitation success, it is recommended that further research be undertaken to assess the success of the four rehabilitation techniques in terms of restoring the physical properties of the soil.

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CHAPTER 6

THE EFFECT OF TRANSLOCATION ON THE POPULATION STRUCTURE (SIZE CLASS DISTRIBUTION) OF FIVE SEMI-ARID PLANT SPECIES AT NAMAKWA SANDS

ABSTRACT

Namakwa Sands initiated trials to determine whether the translocation of selected species would facilitate successional processes in areas affected by mining. Five locally common, non-palatable, perennial plant species (namely *Othonna cylindrica*, *Zygophyllum morgsana*, *Ruschia versicolor, Lampranthus suavissimus* and *Asparagus* spp.) were translocated in multi-species clumps in June 2001. The objectives of this study were to assess the size and age structure of the five translocated species in order to determine whether translocation has resulted in the development of self-perpetuating populations and to compare seedling recruitment in translocated areas and topsoiled areas. Sampling was undertaken within relevés set out along line transects at three sites (i.e. two rehabilitation sites and one reference site) in the east mine over four sampling periods between February 2004 and September 2005. Within each relevé, height, canopy cover and sexual maturity were recorded for each individual of the translocated species encountered. No live individuals of *Asparagus* spp. were present in either rehabilitation site and as a result this species was not sampled. The population structures (or size class distributions) of the remaining four species appear to be unstable in all sites and these populations will vary over time. Under the current climatic conditions, *O. cylindrica* appeared to be the most successful species translocated, while *Asparagus* spp. was the least successful species. In the short-term (after approximately four years), the translocation of *O. cylindrica, R. versicolor* and *L. suavissimus* appears to have resulted in the recruitment of seedlings and the development of self-perpetuating populations within the translocation sites. Therefore, these three species should be considered for future large-scale plant translocation projects. However, the long-term viability of rehabilitated *Z. morgsana* populations is in doubt due to the almost complete lack of recruitment and is likely to remain difficult to re-establish during periods of below average rainfall. The long-term viability of *Z. morgsana* populations needs to be determined after the next favourable germination period in order to determine whether this species should be considered for any future large-scale translocation purposes. No translocated *Asparagus* spp. individuals survived and should therefore not be considered for any further large-scale translocation purposes. If plant translocation is to become a valuable rehabilitation tool at Namakwa Sands, all translocation efforts should be monitored over the long-term to determine if these populations are in fact self-perpetuating. This long-term monitoring will allow the techniques and species selected for translocation to be refined and improved.

Key words: population structure; size class distribution, demography; strip-mining; Strandveld; rehabilitation techniques; topsoil; translocation; *Othonna cylindrica; Ruschia versicolor; Zygophyllum morgsana; Lampranthus suavissimus; Asparagus* spp.

6.1 INTRODUCTION

Namakwa Sands has been strip-mining a heavy mineral deposit, which is rich in the commercially valuable minerals ilmenite, rutile and zircon, since September 1994 (Namakwa Sands 2001). Strip-mining causes total destruction of natural ecosystems through the removal of vegetation and soil in the area where mining is being undertaken (Cooke & Johnson 2002).

Namakwa Sands has been rehabilitating mined out areas as the mining front moves forward with approximately 200 to 335 ha of mined out land requiring rehabilitation per annum (Namakwa Sands 2001). The long-term rehabilitation goal at Namakwa Sands is to restore the area to, or as close as possible to, its natural state and achieve a vegetation cover and productivity similar to the pre-mining land-use (i.e. small-stock farming) (EEU 1990; Grindley & Barbour 1990). However, due to the unique vegetation type (Mahood 2003b) and the difficulty of rehabilitating mined out areas where vegetation growth is restricted by aridity, wind and nutrient-poor soils (Milton 2001), Namakwa Sands has facilitated ongoing research projects to incorporate this knowledge into the rehabilitation plan.

The vegetation at the mine site is dominated by perennial plants that occur in mixed species clumps, interspersed with open areas that are covered by annual plants in the winter (Eccles *et al.* 1999; van Rooyen 2001; Mahood 2003b). Several studies in arid ecosystems have demonstrated that seedling germination, establishment and survival are facilitated by the presence of an established shrub or group of shrubs (the "nurse-plant effect") due to the amelioration of various environmental factors (Franco-Pizaña *et al.* 1996; de Villiers *et al.* 2001). Vegetation patchiness or clumping is thought to optimise the capture and storage of limited resources, such as water and nutrients, from source areas into sinks or patches (Ludwig & Tongway 1996). These areas of increased fertility are known as "fertile islands". "Fertile islands" are an integral part of desert landscapes and they play a critical part in the structuring and functioning of desert systems (Titus *et al*. 2002). In addition, the establishment of seedlings below canopies could also have a number of other advantages, for example protection from high irradiance, high temperatures, high rates of transpiration, herbivory and wind as well as increased infiltration (de Villiers *et al*. 2001; Su & Zhao 2003). These factors may result in an environment more conducive to seedling establishment of certain species (Franco-Pizaña *et al.* 1996).

De Villiers *et al.* (2001) suggested that several advantages could be obtained by translocating perennial woody shrubs, for example to reduce wind speed, provide a source of seed and increase species diversity of those species that prefer conditions under the canopy. In addition, seeds formed by translocated plants facilitate the dispersal of seeds onto the mined area (Blignaut & Milton 2005). However, other studies have shown that there is no evidence to support the hypothesis that seedling germination, establishment and survival are facilitated by the proximity to established shrubs (Franco-Pizaña *et al.* 1996; de Villiers *et al.* 2001; Blignaut & Milton 2005). A simple gradient of abiotic harshness is not sufficient to predict the outcome of plant-plant interactions and the process may be more complex, involving both facilitative (e.g. increased seed germination) and competitive effects (e.g. inhibited establishment and growth) of neighbouring adult "nurse" plants (Franco-Pizaña *et al.* 1996; Riginos *et al.*

2005). Perennial shrubs may also have negative effects on seedling survival and establishment in their understorey, due to light deprivation and competition for water and nutrients (de Villiers *et al*. 2001). However, Eccles *et al.* (1999) argue that if the net interaction is not positive then plants would not occur in clumps, and speculate that clumping in the short and medium Strandveld is a product of a positive feedback between substantial physical benefits associated with mutual shading and microhabitat modification and the dominant seed dispersal strategies.

In addition to the possible facilitation by the presence of shrub species, de Villiers (2000) found that most of the perennial species dominating the pre-mining vegetation at Namakwa Sands were not well represented in the seed bank. Therefore, he speculated that the recruitment from the seed bank alone would not be sufficient and that large seeded perennials would probably not be recruited in sufficient numbers. This, together with the fact that strip-mining destroys the standing vegetation and the aerial seed banks, led him to suggest that adult perennial plants should be translocated during rehabilitation efforts.

In June 2001, Mahood (2003a) initiated trials to determine whether the translocation of selected indigenous perennial species into multi-species clumps would facilitate or accelerate the successional processes needed to create a self-sustaining vegetation in areas affected by strip-mining (Plate 6.1).

Plate 6.1: A clump translocated by Mahood (2003a) in June 2001. Species evident in this clump include (a) *Othonna cylindrica*, (b) *Zygophyllum morgsana*, and (c) *Ruschia versicolor*.

Research has shown that local species adaptation promotes higher fitness under the specific ecological conditions of a site. Locally adapted populations often represent a “genetic memory” shaped by selective events (e.g. 100-year drought). The introduction of non-local genotypes that dominate the population

initially, but cannot withstand extreme selective events over the long-term, represent a non-sustainable rehabilitation strategy (Monalvo *et al*. 1997). Mahood (2003a) selected five locally common, non-palatable, perennial plant species with storage organs for translocation (namely *Othonna cylindrica*, *Zygophyllum morgsana*, *Ruschia versicolor, Lampranthus suavissimus* and *Asparagus* spp.). Perennial species were selected as these species would help stabilise post-mined areas during the hot, dry and windy summer months when annual plants would have died. The selection of non-palatable species ensured that the translocated individuals would not be grazed to any great extent and would therefore provide protected sites for palatable species to survive and produce seed, thereby ensuring a sustained post-mined land use.

Competition, herbivory, predation, parasitism and mutualism all play a role in the development and success of restored sites (Monalvo *et al*. 1997). Population growth is a function of both the recruitment and the storage of reproductive potential over generations (Higgins *et al.* 2000). Factors that reduce the survival and reproductive success of colonising species should lead to a decrease in their populations (O'Connor 1991). Therefore, understanding and predicting seedling recruitment processes and those factors that influence colonisation, growth and distribution of populations appears to be one of the greatest challenges to determine long-term reintroduction success (Monalvo *et al*. 1997; Morgan 1999). Demographic analyses are useful tools in providing information about the growth, survival and reproduction of individuals in different size / age classes within a population, and these parameters allow the prediction of long-term population trends (Contreras & Valverde 2002). For plant translocation to be of long-term value to the rehabilitation of mined out areas at Namakwa Sands, the translocated individuals must survive and produce viable offspring. To date, no follow-up assessments of the survival and success of Mahood's (2003a) translocation trials have been undertaken. Therefore, the main objective of this study was to assess the size and age structure of the five translocated species to obtain an understanding of their dynamics and stability and to determine whether translocation has resulted in the development of self-perpetuating populations.

A second objective was to compare seedling recruitment between different rehabilitation techniques / strategies. The development and refinement of rehabilitation methods over the years at Namakwa Sands has resulted in different rehabilitation strategies, including plant translocation, topsoil replacement, seedling, and a combination of these methods. Most of the mined out area has been rehabilitated by the replacement of the top 5 cm of topsoil (Namakwa Sands 2001) and relies entirely on the germination of the topsoil seedbank and the importation of seed from outside the area of disturbance. These rehabilitation strategies all differ in their cost of implementation both financially and in time. The development of cost-effective rehabilitation techniques requires that the success of rehabilitation techniques be weighed up against the cost of implementation (Lubke & Avis 1998; Espelta *et al.* 2003). Therefore, the different rehabilitation strategies implemented at Namakwa Sands provided opportunity to assess whether translocation increased the recruitment of the translocated species compared to areas where only topsoil was replaced.

6.2 METHODS

6.2.1 STUDY AREA

The study was undertaken at the Namakwa Sands east mine, which is situated in the vicinity of Brand-se-Baai on the west coast of South Africa, approximately 385 km north of Cape Town (Figure 3.1; Chapter 3).

The study area is characterised by hot, dry summers and sporadic winter rainfall, falling mainly in the months from May to July (EEU 1990; le Roux & Schelpe 1997). The rainfall increases from the north to the south, with an average of 160 mm per annum occurring in the mining area (Mahood 2003a). Rainfall is augmented by heavy dew falls and sea fogs. Rainfall, sea fog and dew fall amount to a cumulative average annual precipitation of 282 mm per annum measured over a four year period (de Villiers *et al.* 1999). Compared to the average of 160 mm per annum (Mahood 2003a), it would appear that the study area has experienced a drought over the last few years with an annual rainfall of 100.5 mm, 137.5 mm and 82.9 mm being recorded in 2003, 2004 and 2005, respectively.

Climatically the Namaqualand-Namib region has relatively moderate temperatures through the year with an average annual temperature of 15.9 ^{o}C in the mining area from 2003 to 2004. The highest average monthly maximum temperature over this period was 37 ºC in summer (March) and the lowest average monthly minimum temperature was 4.7 ºC in winter (July).

The strong wind regime in the area is a major cause of erosion at Namakwa Sands (Washington 1990). The dominant winds during the spring and summer (September to March) between 2003 and 2005 were from the south-southeast and south but winds from the north-northwest and northwest were not uncommon. The dominant winds during the winter months (June to August) were from the north-northwest, northeast, northwest and east-northeast but winds from the south and southeast were not uncommon. The average annual wind speed in the mining area between 2003 and 2004 was 4.1 m/s. The highest average monthly wind speed was 5.8 m/s in winter (July) and the lowest monthly average wind speed was 3.7 m/s in summer (February).

The vegetation in the study area is classified by Low and Rebelo (1996) as consisting of Strandveld Succulent Karoo and Lowland Succulent Karoo, both of which are classified under the Succulent Karoo Biome. The Strandveld Succulent Karoo occupies the sandy coastal plain throughout Namaqualand (Cowling *et al.* 1999). It is associated with areas of calcareous sand and contains many drought deciduous and succulent species (Low and Rebelo 1996). The Lowland Succulent Karoo is dominated by members of the Mesembryanthemaceae, especially the species of *Ruschia, Drosanthemum, Malephora* and *Delosperma.* Boucher & le Roux (1989) classified the Strandveld vegetation of the mine site into three variants according to vegetation height, i.e. tall, medium and short Strandveld. These three generalised categories were based on a combination of vegetation structure and floristic content. The three variants vary in height according to the depth of the soil profile. Tall Strandveld, dominated by shrubs between 1 to 2 m tall, occurs on relatively deep calcareous sand, with a canopy cover of 60 –

75%, under a light grazing regime. Medium Strandveld is characterised by plants that are in the region of 50 cm tall and it has a projected canopy cover of perennial species of between 50 and 60%. Short Strandveld varies in average height from 10 – 35 cm. This community occurs in shallow soils where there is very little storage of moisture. The projected vegetation canopy cover of perennial species is usually less than 50%. De Villiers *et al.* (1999) undertook a detailed survey of the mine site and divided the mine area into six vegetation communities or associations:

1. *Ruschia tumidula - Tetragonia virgata* Tall Shrub Strandveld;
2. *Eriocephalus africanus - Asparagus fasciculatus* Tall Shrub Strandveld;
3. *Salvia africanus-lutea - Ballota africana* Tall Shrub Strandveld;
4. *Ruschia versicolor - Odyssea paucinervis* Dwarf Shrub Strandveld;
5. *Jordaaniella spongiosa - Odyssea paucinervis* Coastal Strandveld; and
6. *Cladoraphis cyperoides - Lebeckia multiflora* Coastal Strandveld.

These vegetation communities are described in more detail in Chapter 3.

6.2.2 STUDY SITES

Sampling was undertaken at three sites (i.e. two rehabilitation sites and one reference site) in the east mine. The three sites included the following:

1. Reference site R1 (31° 15.853' S 17° 58.415' E): Three plots were located within a natural Strandveld community, classified as *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld by de Villiers *et al.* (1999). This vegetation community was selected as most of the mining and rehabilitation to date has been undertaken within this community. The three 50 m x 50 m plots were arranged in series from west to east, with a 10 m interval between each plot.
2. Rehabilitation site S1: This site consisted of three plots with each plot situated in one of the three non-irrigated plots established by Mahood (2003a) in June 2001 (Figure 6.1). In these plots, topsoil was spread over tailings with the translocation of five indigenous species into multi-species clumps. The precise source of the topsoil from within the east mine and the length of topsoil stockpiling (up to three months) are unknown. The five translocated species included:
    - *Othonna cylindrica* (Asteraceae): A branched shrub growing up to 1 m high. It has a shallow rooted succulent stem with drought-deciduous succulent leaves. Flower-heads are borne in loose groups above the leaves. It is generally unpalatable except for the flowers (Mahood 2003a; le Roux & Schelpe 1997). *O. cylindrica* produces small parachute-type seeds that are wind dispersed (Milton *pers. comm.*).
    - *Zygophyllum morgsana* (Zygophyllaceae): A branched shrub growing up to 1.5 m high. It has a drought-deciduous succulent stem and the leaves are divided into two succulent leaflets. It is generally unpalatable except for new growth (Mahood 2003a; le Roux & Schelpe 1997). The fruits have four prominent membrane-like wings. Most seeds are released before fruits abscise due to wind action (van Zyl 2000), but fruits also abscise and release seeds as they roll across the soil surface (Milton *pers. comm.*).

- *Ruschia versicolor* (Mesembryanthemaceae)*:* A shallow-rooted evergreen leaf-succulent. Dry fruits are hard with five valves that open when wet to distribute the seeds and close again when fruits are dry (Mahood 2003a). Seeds are dispersed short distances (20–90 cm) during rain or fog periods (Milton *et al.* 1999; Milton *pers. comm.*).
- *Lampranthus suavissimus* (Mesembryanthemaceae): A shallow-rooted evergreen leaf-succulent up to 1 m high. Pink flowers are single or in groups of three. Dry fruits are hard with five valves that open when wet to distribute the seeds and close again when fruits are dry (Mahood 2003a; le Roux & Schelpe 1997). Seeds are dispersed short distances (20– 90 cm) during rain or fog periods (Milton *et al.* 1999; Milton *pers. comm.*).
- *Asparagus spp.* (Asparagaceae): Various *Asparagus* species, which have low ground storage organs, were used for translocation due to the difficulty in finding the same species for translocation (Mahood 2003a).

3. Rehabilitation site S2: This site consisted of three inter-plot areas (Figure 6.1). No plant translocation occurred in these plots, where topsoil was spread over tailings in June 2001. The precise source of the topsoil from within the East Mine and the length of topsoil stockpiling (up to three months) are unknown. This scenario relies entirely on the germination of the seed bank in the topsoil and the importation of seed from outside the area of disturbance.

This sample design resulted in three sites with three plots per site (i.e. a total of nine plots) with no site subject to grazing by sheep.

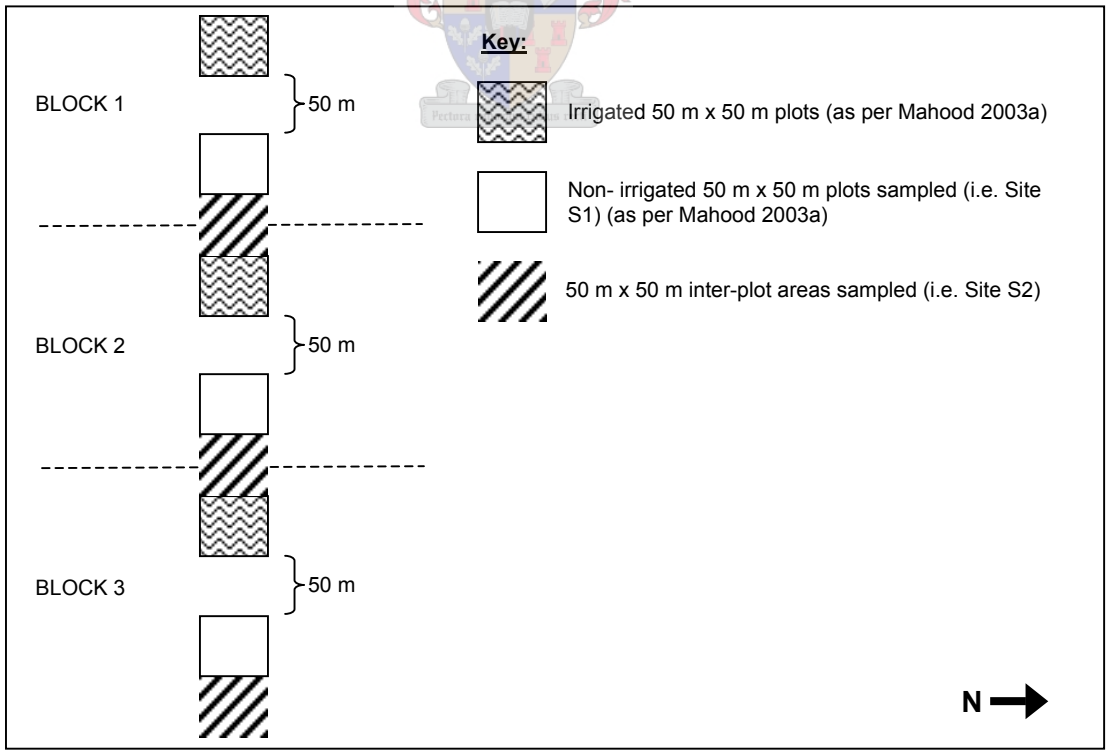

Figure 6.1: Representation of the three block experimental design of Mahood (2003a) showing the rehabilitation sites (S1 and S2) sampled in this study. Note: the reference site R1 is not depicted in this figure.

6.2.3 VEGETATION SAMPLING

The five translocated species were selected for this study (Plate 6.2), but it should be noted that *Asparagus* spp. was not sampled as no live individuals were found within the two rehabilitation sites.

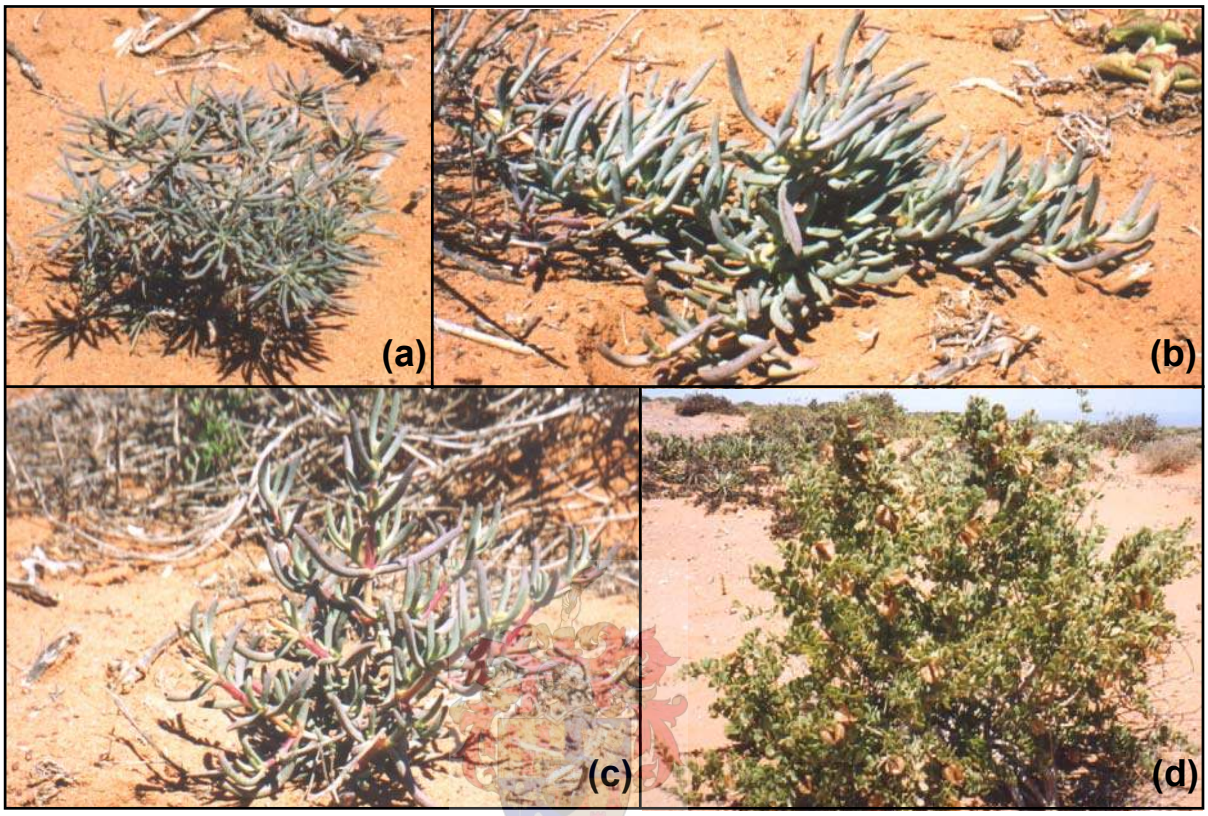

Plate 6.2: The four species sampled in this study: (a) *Othonna cylindrica*, (b) *Ruschia versicolor*, (c) *Lampranthus suavissimus* and (d) *Zygophyllum morgsana*.

Sampling was undertaken four times between February 2004 and September 2005. The sampling periods were as follows:

1. Summer 2004 (18 to 26 February 2004);
2. Winter 2004 (28 August 2004 to 3 September 2004);
3. Summer 2005 (20 to 25 February 2005); and
4. Winter 2005 (28 August 2005 to 1 September 2005).

Within each of the nine plots (i.e. three plots per site), four 50 m line transects were orientated in a "W" pattern (Figure 6.2). The reason for the "W" orientation was to ensure that the line transects were not orientated parallel to the rows of translocated clumps, thereby ensuring that the translocated clumps were sampled. A total of 12 line transects (i.e. four per plot) were sampled per site.

Along each line 50 m transect, 2 m x 2 m relevés were set out every 5 m resulting in 10 relevés per line transect (Figure 6.2 and Plate 6.3). The resulting area sampled totalled 40 m^2 per transect, 160 m^2 per plot and 480 m^2 per site. Relevés were always located on the northern side of the line transect.

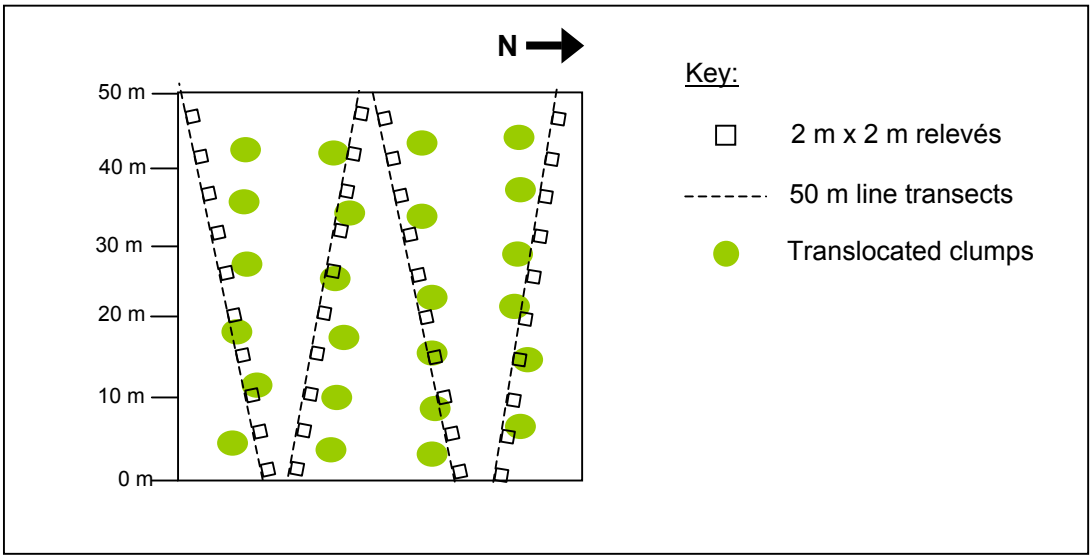

Figure 6.2: Layout of the four line transects and relevés per plot showing the "W" pattern of transect orientation.

Plate 6.3: A 2 m x 2 m relevé set out along the 50 m line transect within site R1.

Within each 2 m x 2 m relevés, height, canopy cover and sexual maturity were recorded for each *Othonna cylindrica*, *Zygophyllum morgsana*, *Ruschia versicolor* and *Lampranthus suavissimus* individual encountered. Cover was estimated by measuring two perpendicular diameters of the canopy and assuming that the projection of the canopy onto the soil was a rectangle (i.e. cover = length x width, with length being the longer axis). Height and perpendicular diameters of the canopy were measured to an accuracy of ± 0.5 cm. Individuals with flowers, old peduncles and capsules (past and present year) were considered to be mature individuals, the remainder being immature.

The identification of individuals proved to be difficult at times. Individuals growing close together within a single clump were often difficult to distinguish. Where possible, individuals that could be distinguished based on flowering, leaf / bark colour, etc. were measured separately. In addition, the multi-stemmed *O. cylindrica* together with the accumulation of sand under the canopy of larger individuals also made it difficult to distinguish between individuals. Where stems were very close together and appeared to share the same root system, they were counted as one individual.

6.2.4 DATA ANALYSIS

Within each plot, the total number of individuals, recruitment (i.e. number of individuals in size class 1) and cover (m^2) recorded in each of the four transects were grouped for analysis. Differences in the number of individuals and cover between sites and over time (two years) were evaluated using Repeated Measures Analysis of Variance (ANOVA) using STATISTICA 7.0 (StatSoft, Inc. 2004). Normal probability plots were inspected for normality. Individual differences ($p<0.05$) between sites and over time were evaluated by doing pairwise comparisons using a bootstrap technique (Efron & Tibshirani 1998), as the data were not normally distributed, and then applying Bonferroni multiple testing.

Since the age of individuals sampled is not known the analysis of population structure was based on the plant height. Height (cm) class frequency histograms were produced for the number of individuals of each species. *O. cylindrica*, *Z. morgsana*, *R. versicolor* and *L. suavissimus* individuals were grouped into ten even-sized height classes.

Pearson's R correlation tests using STATISTICA 7.0 were used to test for relationship between recruitment events (i.e. those individual located within the first height class) and both number of sexually mature individuals and total rainfall during the five months preceding sampling (Table 6.1).

Quotients between the number of individuals in successive height size classes were determined (i.e. the measure of movement from one size class to the next). The quotient between the numbers of individuals in successive size classes in stable populations should approach a constant value or be characterised by low ratios of change (Walker *et al.* 1986).

The distribution of individuals across size classes between sites were compared using contingency tables. The expected values were calculated based on the null hypothesis of independence and tested using the chi-squared statistic (STATISTICA 7.0).

Table 6.1: Rainfall (mm) measured 5 months prior to each of the four sampling periods.

Sampling period	Rainfall (mm) 5 months prior to sampling
Summer 2004	45.5
Winter 2004	40.0
Summer 2005	61.0
Winter 2005	35.8

6.3 RESULTS

6.3.1 NUMBER OF INDIVIDUALS AND RECRUITMENT

Site effect

There were no significant differences in the average number of individuals (i.e. density) or in the recruitment of seedlings (i.e. number of individuals in size class 1) of *O. cylindrica, R. versicolor, Z. morgsana* and *L. suavissimus* between sites (Figure 6.3; Tables 6.2 & 6.3). The large amount of plot-to-plot variation, illustrated by the large standard deviations, suggests a high degree of spatial heterogeneity within each site. This high degree spatial heterogeneity may make it difficult to detect a response between sites (Miller & Huenneke 2000). Even though there were few differences from a statistical perspective, there do appear to be several clear trends in the data. The number of individuals and recruitment of *R. versicolor* and *Z. morgsana* was greater in site R1 (reference site) than in either of the two rehabilitation sites (sites S1 & S2). Similarly, site S1 generally had a greater number of *R. versicolor* individuals and recruitment than site S2. The recruitment of *Z. morgsana* seedlings in all sites in all sampling periods was very poor. The number of individuals and recruitment of *O. cylindrica* was greater in site S1 than in either the reference site or in site S2 where just topsoil was replaced. Similarly, site R1 had a greater number of *O. cylindrica* individuals and recruitment than site S2. Sites S1 and S2 generally had a greater number of *L. suavissimus* individuals and seedlings than site R1 in all sampling periods, with site S1 having the greater number of individuals and seedlings.

It should be noted that no live individuals of *Asparagus* spp. were present in either rehabilitation site (i.e. sites S1 and S2). Therefore, it is evident that translocation of *Asparagus* spp. has failed.

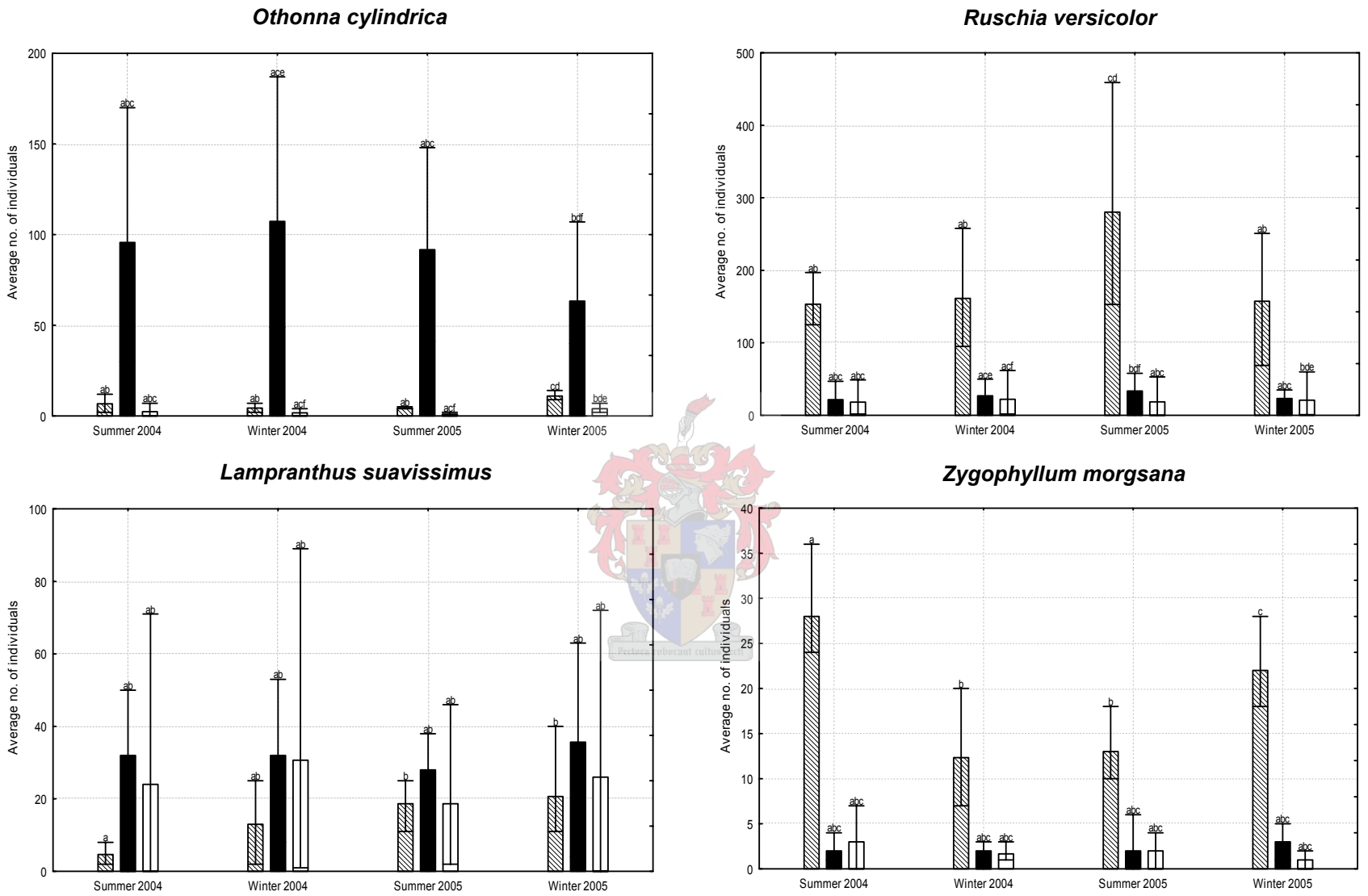

Figure 6.3: Average (± standard deviation) density of individuals per 160 m^2 of *O. cylindrica, R versicolor, Z. morgsana* and *L. suavissimus* at three Namakwa Sands sites (n=3) over four seasons. Different letters within each species are significantly different ($p<0.05$) from one another by doing pairwise comparisons using a bootstrap technique and applying Bonferroni multiple testing. The three sites included: ▨ Reference site R1 (*Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld); ■ Rehabilitation site S1 (i.e. plant translocation with topsoil); and □ Rehabilitation site S2 (i.e. topsoil only).

Table 6.2: Average (± standard deviation) density of individuals (individuals per 160 m^2) of *O. cylindrica, R versicolor, Z. morgsana* and *L. suavissimus* at three Namakwa Sands sites (n=3) over four seasons. Values with different superscripts within each species are different at p<0.05 by doing pairwise comparisons using a bootstrap technique and applying Bonferroni multiple testing. The three sites are as described in Figure 6.3.

Plant species	Site R1				Site S1				Site S2			
	Summer 2004	Winter 2004	Summer 2005	Winter 2005	Summer 2004	Winter 2004	Summer 2005	Winter 2005	Summer 2004	Winter 2004	Summer 2005	Winter 2005
Othonna cylindrica	6.67 ±5.03^{ab}	4.33 ±2.52^{ab}	4.33 ±0.58^{ab}	11.00 ±2.65^{cd}	95.67 ±70.40^{abc}	107.33 ±71.43^{ace}	91.67 ±53.72^{abc}	63.33 ±37.85^{bdf}	2.33 ±4.04^{abc}	1.67 ±2.08^{acf}	1.00 ±1.00^{acf}	4.00 ±2.65^{bde}
Ruschia versicolor	153.33 ±38.37^{ab}	161.33 ±85.63^{ab}	280.33 ±159.33^{cd}	157.33 ±91.12^{ab}	21.67 ±22.50^{abc}	27.00 ±23.52^{ace}	33.67 ±27.39^{bdf}	23.33 ±16.86^{abc}	18.33 ±26.58^{abc}	22.33 ±34.36^{acf}	18.67 ±29.77^{abc}	21.00 ±33.78^{bde}
Zygophyllum morgsana	28.00 ±6.93^{a}	12.33 ±6.81^{b}	13.00 ±4.36^{b}	22.00 ±5.29^{c}	2.00 ±1.73^{abc}	2.00 ±1.73^{abc}	2.00 ±3.46^{abc}	3.00 ±2.00^{abc}	3.00 ±3.61^{abc}	1.67 ±1.15^{abc}	2.00 ±2.00^{abc}	1.00 ±1.00^{abc}
Lampranthus suavissimus	4.67 ±3.06^{a}	13.00 ±11.53^{ab}	18.67 ±7.09^{b}	20.67 ±16.74^{b}	32.00 ±27.78^{ab}	32.00 ±22.65^{ab}	28.00 ±16.46^{ab}	35.67 ±29.74^{ab}	24.00 ±40.71^{ab}	30.67 ±50.52^{ab}	18.67 ±23.86^{ab}	26.00 ±39.95^{ab}

Table 6.3: Average (± standard deviation) density of seedling recruitment (seedlings per 160 m^2) of *O. cylindrica, R versicolor, Z. morgsana* and *L. suavissimus* at three Namakwa Sands sites (n=3) over four seasons. Values with different superscripts within each species are different at p<0.05 by doing pairwise comparisons using a bootstrap technique and applying Bonferroni multiple testing. The three sites are as described in Figure 6.3.

Plant species	Site R1				Site S1				Site S2			
	Summer 2004	Winter 2004	Summer 2005	Winter 2005	Summer 2004	Winter 2004	Summer 2005	Winter 2005	Summer 2004	Winter 2004	Summer 2005	Winter 2005
Othonna cylindrica	2.00 ±3.46^{ab}	0.67 ±1.15^{ab}	0.33 ±0.58^{ab}	0.67 ±1.15^{ab}	51.67 ±44.61^{ac}	36.33 ±30.35^{ab}	38.33 ±36.56^{bd}	12.00 ±6.24^{ab}	1.00 ±1.73^{ad}	-	-	2.00 ±1.73^{bc}
Ruschia versicolor	78.33 ±12.10^{abc}	88.00 ±58.85^{ad}	177.33 ±153.09^{be}	60.33 ±42.25^{cf}	8.67 ±9.02^{ace}	12.33 ±11.68^{bdf}	14.33 ±11.93^{bdfg}	2.33 ±1.15^{aceg}	10.00 ±16.46^{abc}	12.00 ±19.92^{abc}	6.67 ±11.55^{abc}	2.33 ±4.04^{abc}
Zygophyllum morgsana	-	-	-	0.67 ±1.15^{a}	-	-	-	0.33 ±0.58^{a}	1.00 ±1.73^{a}	-	-	0.33 ±0.58^{a}
Lampranthus suavissimus	0.67 ±0.88^{ab}	0.67 ±1.15^{ab}	0.67 ±1.15^{ab}	033 ±0.58^{ab}	11.33 ±9.87^{ab}	6.00 ±5.57^{a}	0.67 ±0.58^{b}	0.33 ±0.58^{b}	8.33 ±14.43^{ab}	4.00 ±6.93^{ab}	0.67 ±1.15^{ab}	0.33 ±0.58^{ab}

Season effect

There were differences in the number of *O. cylindrica, R. versicolor, Z. morgsana* and *L. suavissimus* individuals between seasons, but these differences were not always significant (Table 6.2). The expected increase in the number of individuals (i.e. recruitment) after the winter rains was not consistent from year to year. There were also differences in recruitment of seedlings of *O. cylindrica, R. versicolor* and *L. suavissimus* (in site S1) between seasons, but these differences were not always significant (Table 6.3). *Z. morgsana* showed very poor recruitment in all seasons. The correlations between recruitment and rainfall 5 months prior to sampling are presented in Table 6.4 and discussed below.

The number of *O. cylindrica* seedlings in site S1 decreased in the winter samplings, which is contrary to what was expected. This can be explained, however, by the fact that the winter sampling periods received less rainfall than the summer samplings (Table 6.1). This is supported by the positive correlation (non significant) between *O. cylindrica* recruitment and rainfall in site S1. The number of *O. cylindrica* seedlings in sites R1 and S2 decreased in Winter 2004, but increased in Winter 2005. This recruitment was negatively correlated (non significant) with rainfall. The number of *R. versicolor* seedlings in all sites increased from Summer 2004 to Winter 2005, but then decreased in Winter 2005. The increase and subsequent decrease in *R. versicolor* seedling recruitment is positively correlated with rainfall (but not always significantly). There was no significant correlation between the number of *Z. morgsana* and *L. suavissimus* seedlings and rainfall, but recruitment was generally negatively correlated to rainfall 5 months prior to sampling.

Table 6.4: R- and P-values of Pearson's correlation between recruitment (i.e. number of individuals in size class 1) and the number of sexually mature individuals and rainfall at three sites (n=3) at Namakwa Sands. The three sites are as described in Figure 6.3. Significant values ($p<0.05$) are in bold.

Species	Site	Sexually maturity and recruitment		Rainfall 5 months prior to sampling and recruitment	
		r	p	r	p
Othonna cylindrica	R1	-0.34	0.28	-0.21	0.79
	S1	-0.27	0.39	0.49	0.51
	S2	0.31	0.33	-0.62	0.38
Ruschia versicolor	R1	0.81	**0.001**	0.96	**0.04**
	S1	0.71	**0.01**	0.74	0.26
	S2	0.05	0.87	0.09	0.91
Zygophyllum morgsana	R1	0.24	0.45	-0.59	0.41
	S1	-0.09	0.78	-0.59	0.41
	S2	-0.03	0.48	-0.21	0.79
Lampranthus suavissimus	R1	0.10	0.76	0.59	0.41
	S1	-0.3	0.35	-0.16	0.84
	S2	-0.23	0.47	-0.13	0.87

6.3.2 COVER

Site effect

There were no significant differences in the average cover of *O. cylindrica, R. versicolor, Z. morgsana* and *L. suavissimus* between sites (Figure 6.4; Table 6.5). However, a number of trends were evident. The cover of *R. versicolor* and *Z. morgsana* was greater in site R1 (reference site) than in either site S1 or S2 (i.e. the two rehabilitation sites). Site S1 had a greater *R. versicolor* cover than site S2. The cover of *O. cylindrica* was greater in site S1 than in either the reference site or in site S2 where just topsoil was replaced. Similarly, site R1 had a greater *O. cylindrica* cover than site S2. Site S1 generally had a greater *L. suavissimus* cover than sites R1 and S2, but this was not true in Summer 2005 where site R1 had the greatest cover.

Season effect

There were differences in the cover of *O. cylindrica, R. versicolor, Z. morgsana* and *L. suavissimus* between seasons, but these differences were not always significant (Figure 6.4; Table 6.5). Cover was found to increase during the winter months, but this finding was not true for all winter samplings.

6.3.3 POPULATION STRUCTURE / SIZE CLASS DISTRIBUTIONS

The population structure (or size class distributions) of the four translocated species are presented in Figures 6.5 to 6.8. The quotients between the number of individuals in successive height size classes (i.e. population stability) are presented in Figures 6.9 & 6.10.

The populations of *O. cylindrica* (Figure 6.9), *R. versicolor* (Figure 6.9), *Z. morgsana* (Figure 6.10) and *L. suavissimus* (Figure 6.10) at all three sites show no overall pattern and do not appear stable. The quotient between the numbers of individuals in successive size classes does not appear to approach a constant value nor are they characterised by low ratios of change. However, the *L. suavissimus* population at site S1 appears relatively stable from size class three onwards (Figure 6.10). The plots of the quotients for successive size classes for *R. versicolor* shows a general decline in the number of individuals in the larger size classes.

O. cylindrica

The distribution of *O. cylindrica* individuals over the size classes was significantly different ($p<0.05$) at the three sites (Table 6.6). *O. cylindrica* individuals in site R1 were distributed across all size classes in contrast to sites S1 and S2 and, therefore, was characterised by larger *O. cylindrica* individuals (Figure 6.5). The translocation efforts of large mature individuals in site S1 was evident with site S1 having individuals distributed over larger size classes compared to site S2. Site S1 was characterised by a relatively high density of seedlings in the first two size classes (i.e. evidence of greater recruitment), in comparison to sites R1 and S2 (Figure 6.5). The greater number of seedlings evident in site S1 could be related to the site having a greater number of sexually mature individuals than sites R1 and S2. However, this assumption was not supported by the negative correlation (non significant) between recruitment and the number of sexually mature individuals in sites S1 and R1 (Table 6.4).

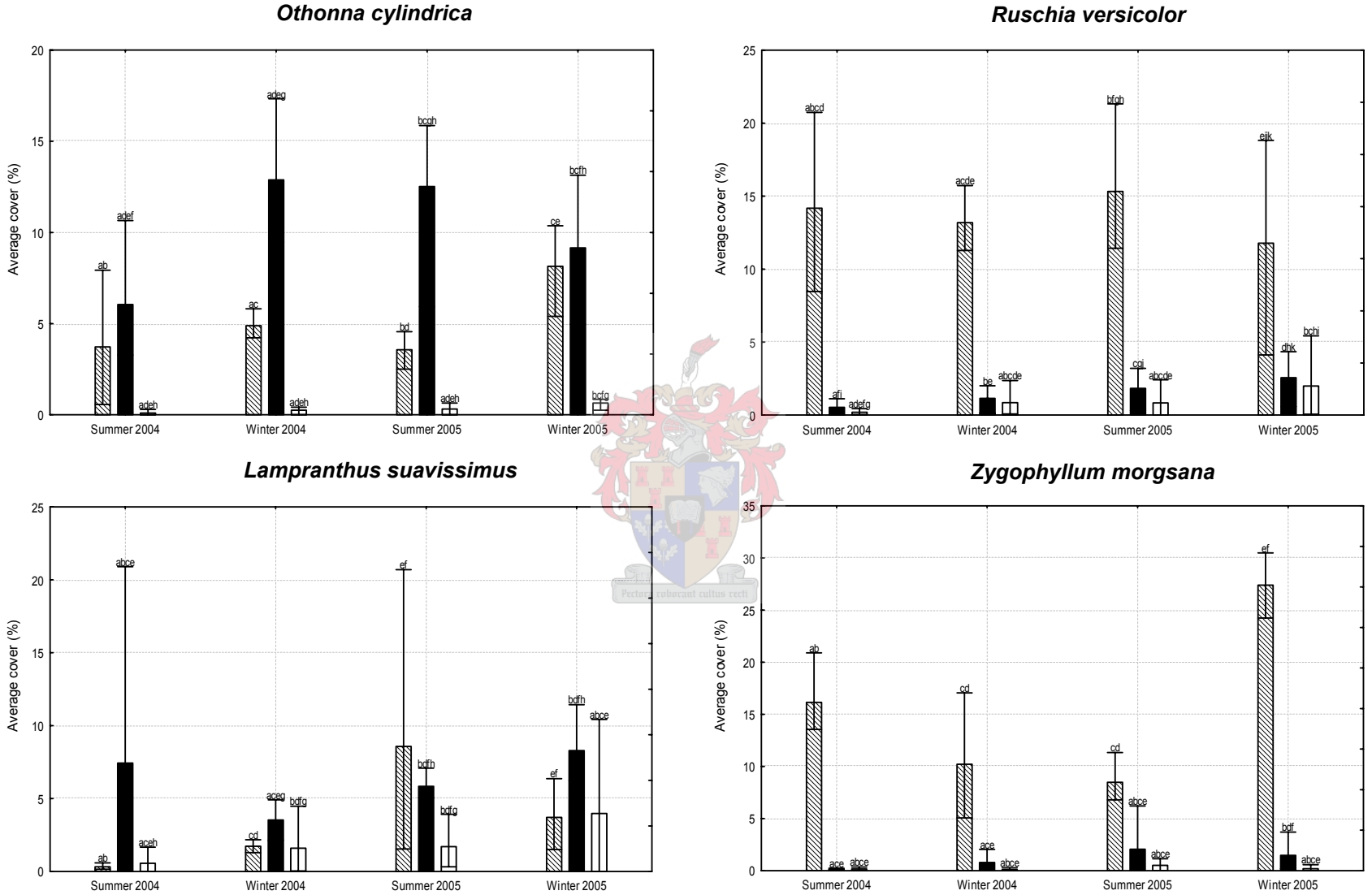

Figure 6.4: Average (± standard deviation) cover (per 160 m^2) of *O. cylindrica, R versicolor, Z. morgsana* and *L. suavissimus* at three Namakwa Sands sites (n=3) over four seasons. Different letters within each species are significantly different ($p<0.05$) from one another by doing pairwise comparisons using a bootstrap technique and applying Bonferroni multiple testing. Sites R1 (▨), S1 (■) and S2 (□) are as described in Figure 6.3.

Table 6.5: Average (± standard deviation) cover (per 160 m^2) of *O. cylindrica, R versicolor, Z. morgsana* and *L. suavissimus* at three Namakwa Sands sites (n=3) over four seasons. Values with different superscripts within each species are different at p<0.05 by doing pairwise comparisons using a bootstrap technique and applying Bonferroni multiple testing. The three sites are as described in Figure 6.3.

Plant species	Site R1				Site S1				Site S2			
	Summer 2004	Winter 2004	Summer 2005	Winter 2005	Summer 2004	Winter 2004	Summer 2005	Winter 2005	Summer 2004	Winter 2004	Summer 2005	Winter 2005
Othonna cylindrica	3.73 ±3.79^{ab}	4.88 ±0.83^{ac}	3.57 ±1.03^{bd}	8.14 ±2.52^{ce}	6.05 ±4.01^{adef}	12.89 ±4.03^{adeg}	12.52 ±5.7^{bcgh}	9.15 ±4.09^{bcfh}	0.10 ±0.17^{adeh}	0.26 ±0.23^{adeh}	0.32 ±0.33^{adeh}	0.65 ±0.35^{bcfg}
Ruschia versicolor	14.18 ±6.18^{abcd}	13.19 ±2.29^{acde}	15.32 ±5.28^{bfgh}	11.78 ±7.37^{eijk}	0.53 ±0.55^{afi}	1.14 ±0.95^{be}	1.83 ±1.46^{cgj}	2.56 ±1.89^{dhk}	0.20 ±0.21^{adefg}	0.85 ±1.31^{abcde}	0.83 ±1.37^{abcde}	1.99 ±2.98^{bchi}
Zygophyllum morgsana	16.15 ±4.11^{ab}	10.22 ±6.17^{cd}	8.48 ±2.48^{cd}	27.4 ±3.13^{ef}	0.16 ±0.07^{ace}	0.79 ±1.11^{ace}	2.07 ±3.59^{abce}	1.48 ±1.93^{bdf}	0.16 ±0.18^{abce}	0.10 ±0.15^{abce}	0.51 ±0.58^{abce}	0.20 ±0.33^{abce}
Lampranthus suavissimus	0.32 ±0.23^{ab}	1.73 ±0.44^{cd}	8.57 ±10.55^{ef}	3.71 ±2.46^{ef}	7.44 ±11.68^{abce}	3.53 ±1.38^{aceg}	5.84 ±1.10^{bdfh}	8.29 ±2.77^{bdfh}	0.57 ±0.96^{aceh}	1.60 ±2.5^{dbfg}	1.71 ±1.94^{bdfg}	3.99 ±5.63^{abce}

Table 6.6: χ^2 and P-values evaluating the distribution of individuals across size classes between three sites (n=3) at Namakwa Sands. The three sites are as described in Figure 6.3. Significant values (p<0.05) are in bold.

Species	*Othonna cylindrica*				*Ruschia versicolor*				*Zygophyllum morgsana*				*Lampranthus suavissimus*			
Sampling	Feb-04	Sep-04	Feb-05	Sep-05	Feb-04	Sep-04	Feb-05	Sep-05	Feb-04	Sep-04	Feb-05	Sep-05	Feb-04	Sep-04	Feb-05	Sep-05
χ^2	79.1	178.59	111.6	116.36	22.53	16.34	82.87	94.7	55.91	25.77	19.67	30.53	22.43	83.34	46.7	56.6
P	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.032**	0.09	**<0.001**	**<0.001**	**<0.001**	**0.028**	0.141	**0.033**	**0.013**	**<0.001**	**<0.001**	**<0.001**

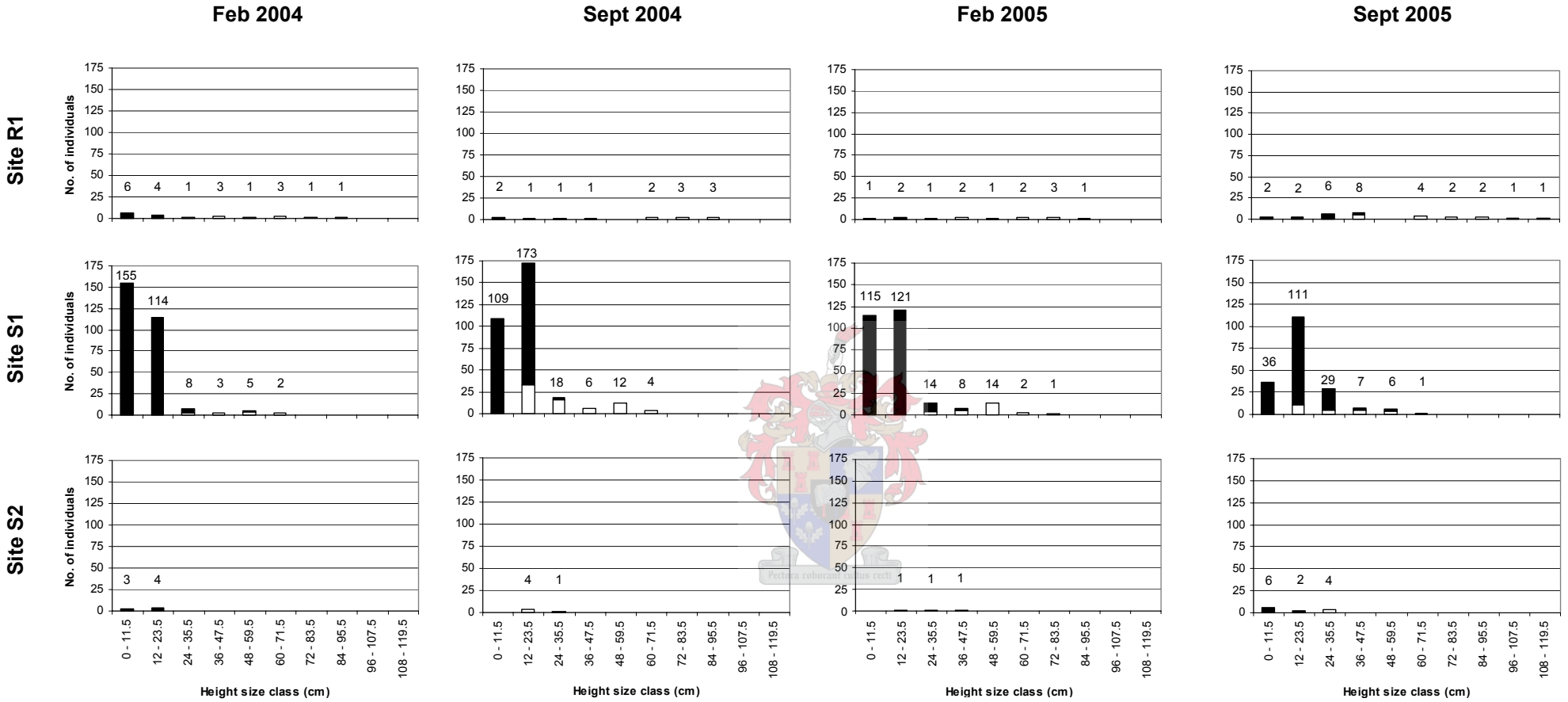

Figure 6.5: Size class distributions of *O. cylindrica* showing the number of sexually mature (white) and immature (black) individuals per 480 m^2 at three Namakwa Sands sites (n=3) over four seasons. The three sites are as described in Figure 6.3.

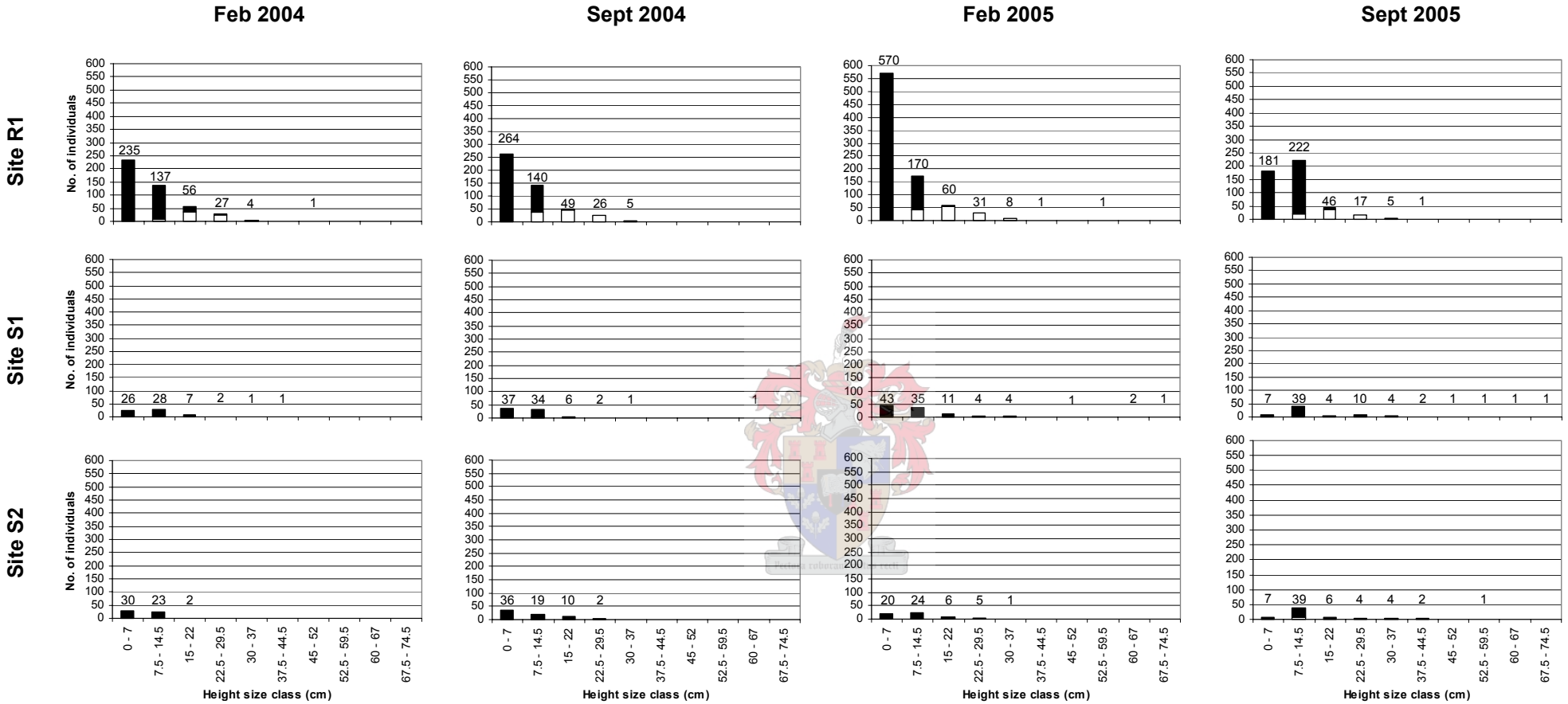

Figure 6.6: Size class distributions of *R. versicolor* showing the number of sexually mature (white) and immature (black) individuals per 480 m^2 at three Namakwa Sands sites (n=3) over four seasons. The three sites are as described in Figure 6.3.

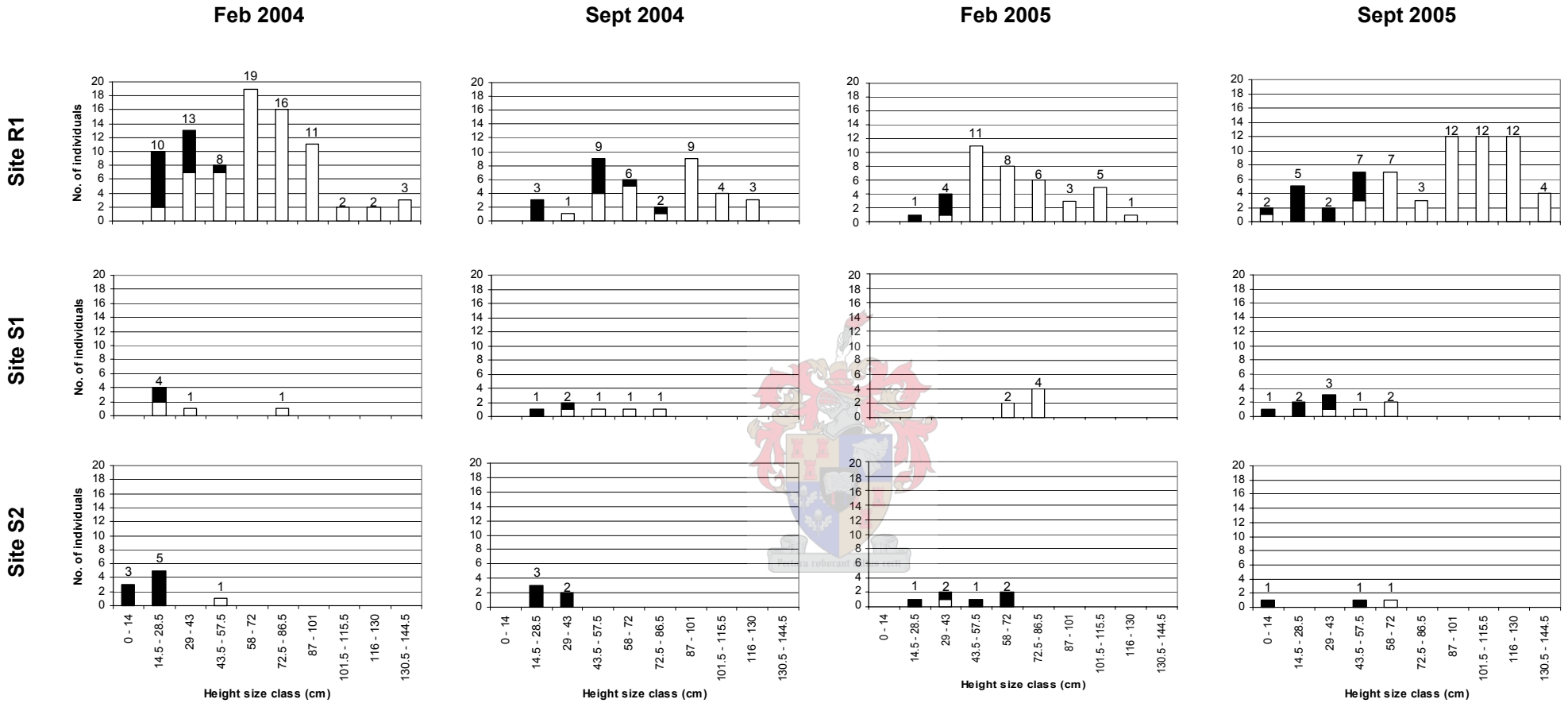

Figure 6.7: Size class distributions of *Z. morgsana* showing the number of sexually mature (white) and immature (black) individuals per 480 m^2 at three Namakwa Sands sites (n=3) over four seasons. The three sites are as described in Figure 6.3.

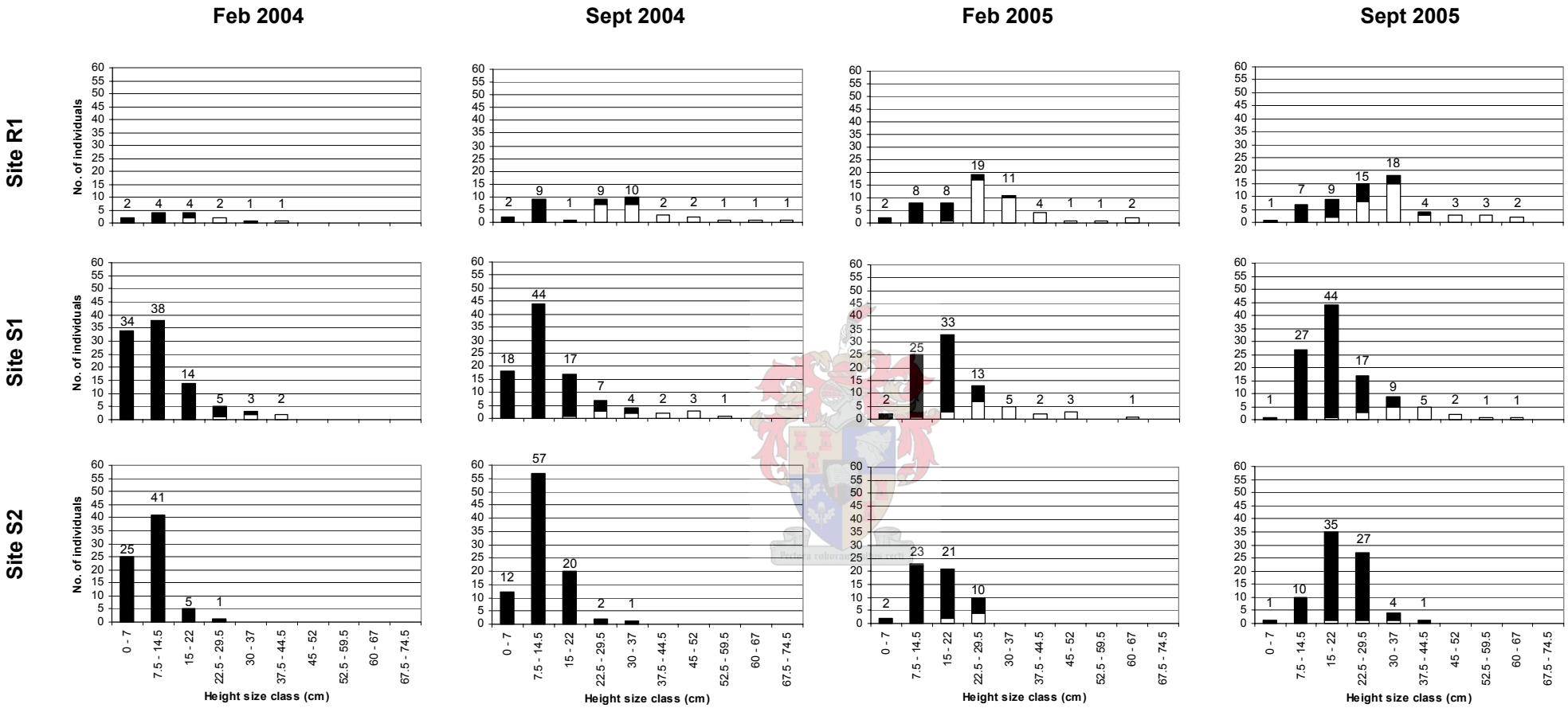

Figure 6.8: Size class distributions of *L. suavissimus* showing the number of sexually mature (white) and immature (black) individuals per 480 m^2 at three Namakwa Sands sites (n=3) over four seasons. The three sites are as described in Figure 6.3.

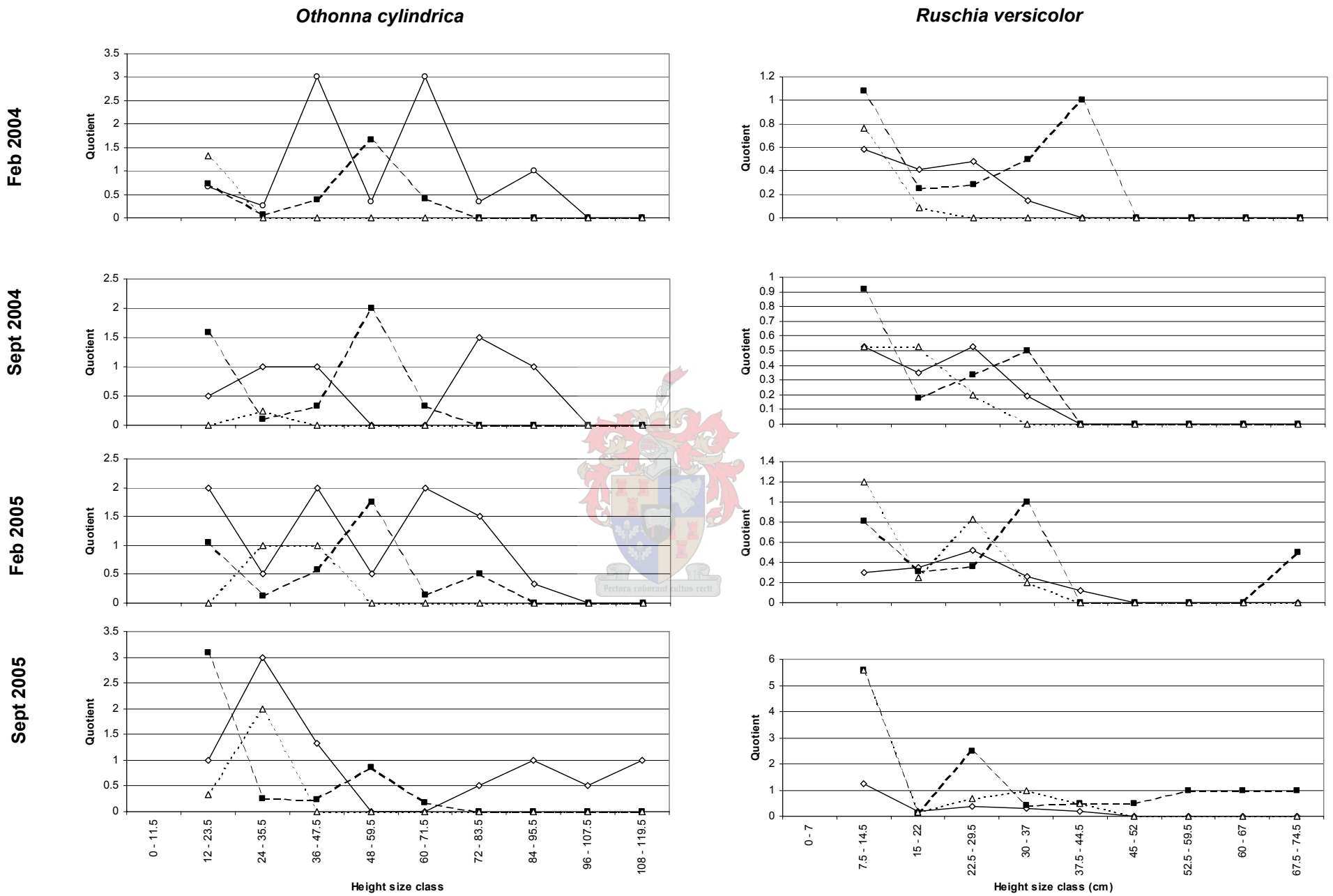

Figure 6.9: Quotients between the number of individuals in successive size classes of *O. cylindrica* and *R. versicolor* per 480 m^2 at three Namakwa Sands sites (n=3) over four seasons. Sites R1 (—◇—), S1 (– –■– –) and S2 (···△···) are as described in Figure 6.3.

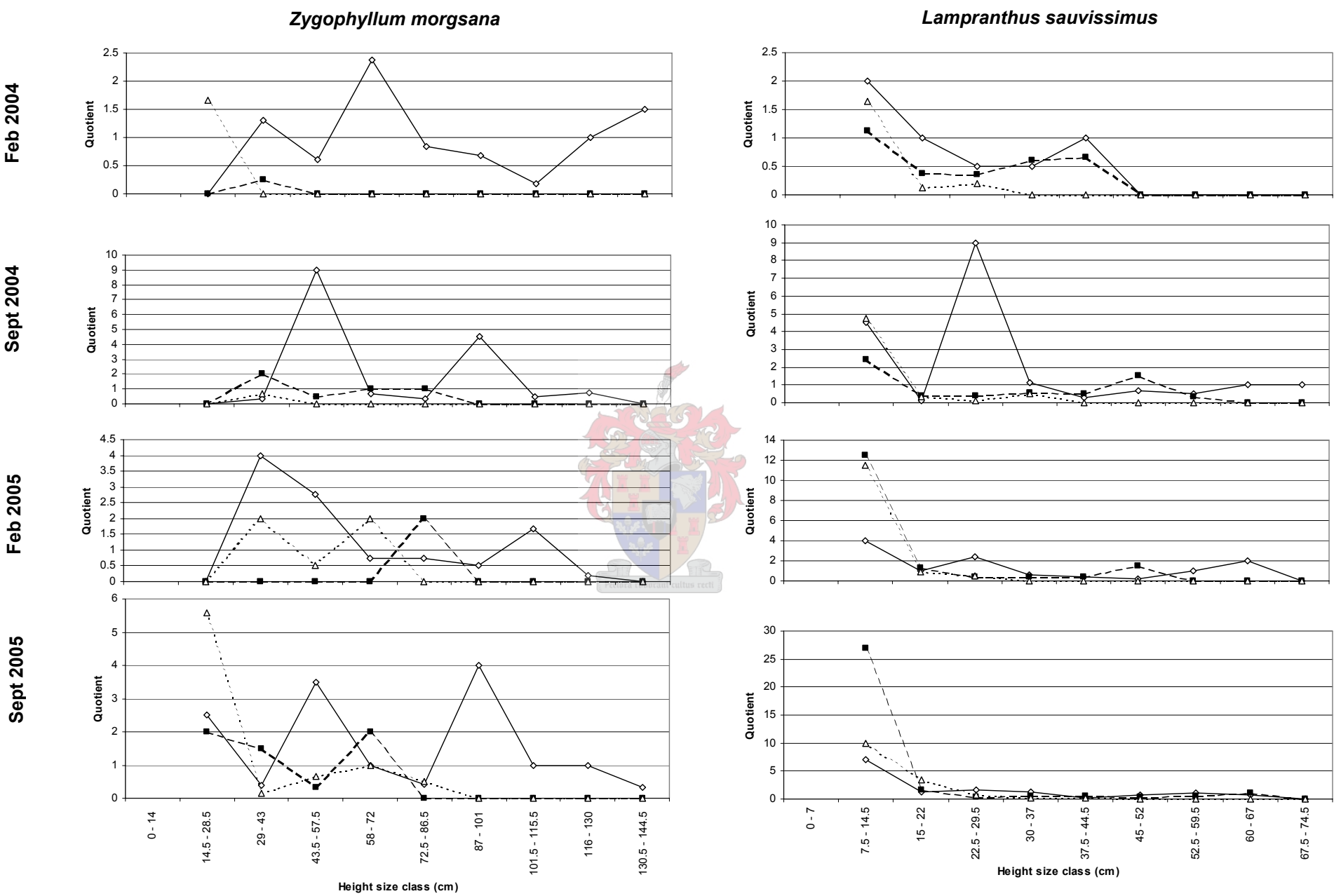

Figure 6.10: Quotients between the number of individuals in successive size classes of *Z. morgsana* and *L. suavissimus* per 480 m^2 at three Namakwa Sands sites (n=3) over four seasons. Sites R1 (—◇—), S1 (–■–) and S2 (···△···) are as described in Figure 6.3.

R. versicolor

The distribution of *R. versicolor* individuals over the size classes at the three sites was significantly different ($p<0.05$), except in the September 2004 sampling (Table 6.6). Site R1 was characterised by a relatively high density of *R. versicolor* seedlings in the first two size classes (i.e. greater recruitment), whereas sites S1 and S2 had relativity few individuals in these classes (Figure 6.6). This finding could be related to the fact that site R1 also had a greater number of sexually mature individuals than sites S1 and S2. There was a significant positive correlation between recruitment and the number of sexually mature individuals in sites R1 and S1 (Table 6.4). Sites S1 and S2 had a similar distribution of *R. versicolor* individuals in the smaller size classes. In general, site S1 was the only site that had *R. versicolor* individuals present in the larger few size classes. Site S1 had a greater number of sexually mature individuals than site S2, which is possibly a result of the mature individuals translocated by Mahood (2003a).

Z. morgsana

The distribution of *Z. morgsana* individuals over the size classes was significantly different ($p<0.05$) at the three sites, except in the February 2005 sampling (Table 6.6). Site R1 had a larger number of *Z. morgsana* individuals in the middle to larger size classes compared to sites S1 and S2, where they were distributed over the middle to lower size classes (Figure 6.7). It is interesting to note that site R1 had very few seedlings (i.e. poor recruitment), even though it had a relatively large number of sexually mature individuals compared to sites S1 and S2. Site S1 had a greater number of sexually mature individuals than site S2, which could be related to the mature individuals translocated in site S1 by Mahood (2003a). There was no significant correlation between recruitment and the number of sexually mature individuals at any of the sites (Table 6.4).

Lampranthus suavissimus

The distribution of *L. suavissimus* individuals over the size classes was significantly different ($p<0.05$) at the three sites (Table 6.6). In general, site R1 had a greater number of sexually mature individuals than sites S1 and S2. However, sites S1 and S2 had a relatively greater density and cover of *R. versicolor* seedlings in the first three size classes than site R1 (Figure 6.8). This finding was supported by there being no significant correlation between recruitment and the number of sexually mature individuals at any of the sites (Table 6.4). Site S1 had a greater number of sexually mature *L. suavissimus* individuals, as well as individuals present in larger size classes, than site S2. This could be related to the larger mature individuals that were translocated in site S1 by Mahood (2003a).

6.4 DISCUSSION

Strategies for rehabilitation usually involve augmenting, enhancing or accelerating changes in species composition and succession (Pyke & Archer 1991). Translocation of species in multi-species clumps is one proposed strategy to facilitate or accelerate the successional processes needed to create a self-sustaining vegetation at Namakwa Sands (de Villiers *et al.* 2001; Mahood 2003a), where vegetation growth is restricted by saline soils, aridity, wind and nutrient-poor soils (Milton 2001; Blignaut & Milton

2005). This study investigates the size and age structure of the five species translocated by Mahood (2003a) and compares seedling establishment of the five translocated species to a reference site (*Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld) and an area where only topsoil was replaced.

As stated earlier, several studies in arid ecosystems have demonstrated that seedling recruitment and survival are facilitated by the presence of shrub species compared to open inter-clump areas. However, other studies have shown that there is no evidence to support this hypothesis (de Villiers et al. 2001; Blignaut & Milton 2005). The success of translocating individuals in multi-species clumps in disturbed areas is dependent, amongst other things, on the species selected for translocation. This is possibly related to the size, morphology and life history attributes of the species used (Monalvo *et al*. 1997; Blignaut & Milton 2005). Life-history attributes tend to correlate with colonisation ability, population structure and population growth rates (Monalvo *et al*. 1997). Site conditions are initially harsh and colonising species must be able to cope with the lack of nutrients and often poor water availability (Jochimsen 2001). Therefore, successful plant translocation requires the selection of suitable plant species, as the structural similarity of plant species selected for translocation together with dry conditions may exacerbate the negative effects of clumping (Blignaut & Milton 2005).

Approximately four years after Mahood's (2003a) translocation trials were initiated, the success of the five species differed in terms of cover and average density of individuals and seedlings (i.e. recruitment). In translocation site S1, *O. cylindrica* had the greatest cover and average density of individuals and seedlings, followed by *L. suavissimus*, *R. versicolor* and *Z. morgsana* (Table 6.2 & 6.3). No translocated *Asparagus* spp. individuals had survived by the first sampling period in February 2004. Therefore, under the current climatic conditions *O. cylindrica* appeared to be the most successful species translocated, while *Asparagus* spp. was the least successful species. These findings support those of Mahood (2001a). The recommendation made by Mahood (2003a) that *Asparagus* spp. should not be considered for any future large-scale translocation purposes is supported. The long-term viability of rehabilitated *Z. morgsana* populations is in doubt due to the almost complete lack of recruitment. Populations such as this, with few seedlings relative to adults, are assumed to be in decline (Johnson *et al.* 2004). However, it should be noted that recruitment of *Z. morgsana* individuals was also poor in the reference site (Figure 6.7).

From an ecological perspective, the value of plant translocations lies in their ability to become self-perpetuating over the long-term (Morgan 1999). Recruitment patterns result from a broad suite of factors including seed production or fecundity, seed dispersal, seed bank, climate (temperature; duration and timing of rainfall events), safe sites and predation (Esler 1999; Coates 2002). Since these factors influence the germination and establishment of seedlings, they may explain why *O. cylindrica* proved to be the most successful species translocated.

The study area is characterised by hot, dry summers and sporadic winter rainfall, falling mainly in the months from May to July (EEU 1990; le Roux & Schelpe 1997). Seedlings generally emerge in autumn or early winter in response to rain and seeds do not generally germinate in response to summer rains.

Selection has evidently favoured species whose germination is inhibited by high temperatures (Milton 1995). Therefore, the length of the favourable growing period depends on the timing of the first sufficient winter rain and temperature. A positive correlation between recruitment and winter rainfall was not, however, always evident in the current study (Table 6.4). This could be related to the below average rainfall the study area has experience over the past few years with the summer samplings (i.e. September to January) receiving uncharacteristically more rainfall than the winter samplings (i.e. March to July) (Table 6.1). Water availability following germination is highly unpredictable in Namaqualand, which can lead to stress during the growing season (Steyn *et al.* 1996). Milton (1995) found that seedling survival in arid Karoo shrublands was strongly influenced by spring rainfall with survivorship being poor (<5%) except when there were follow up rains in winter and spring. The germination response of seeds of desert plants to sporadic rainfall (Higgins *et al.* 2000) results in stochastic (i.e. non-deterministic or variable) and episodic recruitment events (Milton *et al.* 1999). Due to the variance in seed bank densities (Esler 1999) and the germination response to rain, seedling densities can vary by orders of magnitude (Higgins *et al.* 2000) spatially and temporally. Therefore, the natural vegetation will exhibit periods with greater and lesser densities of mature individuals as different cohorts move to the largest height classes (Walker *et al.* 1986). The unevenness or lack of stability evident in the populations of *O. cylindrica, R. versicolor, Z. morgsana* and *L. suavissimus* in all sites (Figures 6.9 & 6.10) could be related to these expectedly stochastic recruitment events. The variance in recruitment rates can allow the coexistence of strongly competing species, provided that some life history attribute allows for the storage of reproductive potential across generations (Esler 1999; Higgins *et al.* 2000). The poor recruitment of *Z. morgsana* individuals in all sites could be related to the below average rainfall the study area has experienced over the past few years. The long-term viability of rehabilitated *Z. morgsana* populations is in doubt due to the almost complete lack of recruitment and it is likely to remain difficult to re-establish during periods of below average rainfall. In order to determine whether this species should be considered for any future large-scale translocation purposes the long-term viability of *Z. morgsana* populations needs to be determined after the next favourable germination period when winter rainfall is not below average.

Reproductive variation in space via dispersal or in time via dormancy and seed storage is critical for many plant species in unpredictable environments (Esler 1999). Species that are not able to store seed may be at risk of local extinction during unfavourable years (Higgins *et al.* 2000). In Namaqualand, the germination of many seeds is retarded for a few years, either by controlling seed release or germination behaviour, to ensure that not all seeds germinate after the first favourable rains (le Roux & Schelpe 1997). Maintaining a high degree of seed dormancy, confining germination to a relatively narrow range of conditions and retaining seeds for long periods in capsules are some of the traits that spread or confine risk of recruitment failure in disturbed environments (Esler & Cowling 1995; Milton *et al*. 1999). A population that is capable of storing reproductive potential increases its variance in recruitment thereby promoting persistence in areas with variable environmental conditions (Higgins et al. 2000). *L. suavissimus* and *R. versicolor* have hygrochastic capsules, which open in response to rain to release seeds when moisture conditions are favourable for germination and establishment (Esler & Cowling 1995). This may explain why the translocation of mature individuals of these two species performed better than *Z. morgsana* in terms of recruitment. The poor recruitment of *Z. morgsana* in the translocation

site and reference site could due to a combination of the conditions not being ideal for germination (below average rainfall) and its inability of store seed until germination conditions are more favourable. The assumption that conditions are not ideal for *Z. morgsana* seed production and germination is highlighted by the fact that the majority of sexually mature *Z. morgsana* individuals within the reference site did not have flowers or capsules during the study period (personal observation). However, this theory of seed retention does not explain why *O. cylindrica*, which is also not able to store seed, appeared to be the most successful in terms of recruitment, total numbers and cover.

A characteristic of many Namaqualand plants is their prolific production of seed (le Roux & Schelpe 1997). Seed bank densities in Namaqualand have been found to range from 5 000 per m^2 in areas with scattered perennials and a few annuals to 41 000 per m^2 in areas dominated by annuals (Esler 1999). Progeny of plants producing abundant seed (e.g. *R. versicolor* and *L. suavissimus*) have a greater chance of exploiting unusual climatic events or reaching rare, competition-free establishment sites than progeny of plants, which have had their reproductive potential reduced by herbivory (Milton 1994) or plants that produce fewer seeds (e.g. *Z. morgsana*). This may also provide an explanation why *R. versicolor* and *L. suavissimus* performed better than *Z. morgsana* in terms of recruitment.

Seed production is one of the processes most sensitive to herbivory among long-lived perennials and continued low levels of reproductive output under a heavy grazing regime results in these populations being unable to replace themselves and eventually becoming locally extinct when adults die off (Riginos & Hoffman 2003). Mahood (2003a) selected non-palatable species for translocation to limit the extent of grazing and thus the impact on seed production. Therefore, grazing is unlikely to have had an effect on the recruitment of the five species translocated species.

Seed dispersal strategies and seed morphology may indirectly play a role in seedling distribution patterns (de Villiers *et al.* 2001). Succulents in the southern Karoo generally have small seeds, an attribute that facilitates colonisation of bare, fine textured soils (Esler 1999). The smaller seeds of *O. cylindrica, L. suavissimus* and *R. versicolor* may explain the greater success of these species in terms of recruitment in rehabilitation sites that have large areas of bare soil compared to *Z. morgsana*, which has relatively larger seeds. However, this does not explain why the recruitment of *Z. morgsana* was also low in the reference site. This again suggests that the conditions were not ideal for *Z. morgsana* germination.

As mentioned earlier, the Mesembryanthemaceae (e.g. *L. suavissimus* and *R. versicolor*) have hygrochastic capsules, which open in response to train to release seeds when moisture conditions are favourable for germination and establishment (Esler & Cowling 1995). Water-dispersed seeds are dispersed over short distances of between 20 cm and 90 cm (Milton *et al.* 1999). Wind is an important element in the area (Washington 1990) and many seeds have adapted to wind dispersal with appendages like wings and thin hairs, making seeds easily transportable over distances (e.g. *O. cylindrica* and *Z. morgsana*) (le Roux & Schelpe 1997). Most large-seeded, wind dispersed species with wing-like appendages are ultimately trapped and establish under “nurse” plants (Esler 1999). Studies investigating microsites (closed-canopy versus open) showed that closed-canopy microsites contained

seed densities between two- and four-fold greater than open microsites (Esler 1993; Jones & Esler 2004). Long-distance seed dispersal can increase seedling survival due to increasing the possibility of finding a suitable germination site and reduced competition from parent plant and other seedlings. In contrast, species showing short-distance dispersal must have other mechanisms for avoiding mortality associated with their limited dispersal distance (Takeuchi *et al.* 2005). Based on the differing dispersal mechanisms of the translocated species and the large areas of bare soil within the translocation site, one would expect that *L. suavissimus* and *R. versicolor* seeds would germinate close to the translocated parent plant and that seeds from *O. cylindrica* and *Z. morgsana* would be blown out of the translocation site into surrounding areas. However, this did not seem to hold true as *O. cylindrica* had the greatest cover and average density of individuals and seedlings compared to the other species.

A comparison of the flowering phenologies and associated pollinators of the five translocated species has not been undertaken as part of this study. An understanding of the life-history characteristics and reproductive biology of the five species selected for translocation may provide a greater understanding of why *O. cylindrica* appeared to be the most successful and *Asparagus* spp. the least successful species selected for translocation.

The cover and density of the four surviving translocated species differed between the two rehabilitation sites. In translocation site S1, *O. cylindrica* had the greatest average density and cover, followed by *L. suavissimus*, *R. versicolor* and *Z. morgsana* (Table 6.2 & 6.3). Whereas in site S2 where only topsoil was replaced, *L. suavissimus* had the greatest average density and cover, followed by *R. versicolor*. Both *Z. morgsana* and *O. cylindrica* performed poorly in site S2. If one compares the two rehabilitation methods (i.e. topsoil replacement versus topsoil replacement and plant translocation), the translocation of *O. cylindrica, R. versicolor* and *L. suavissimus* had enhanced, to varying degrees, the recruitment of seedlings within the translocation site compared to where just topsoil was replaced. This enhanced recruitment was particularly evident in the population structural comparisons of *O. cylindrica* (Table 6.3; Figure 6.5). This result provides support for de Villiers (2000), who suggested that adult perennial plants should be translocated during rehabilitation efforts because perennial species dominating the pre-mining vegetation were not well represented in the seed bank. De Villiers (2000) speculated that the poor representation of perennials in the seed bank might be as a result of long-lived perennial species generally having larger seeds, which are more likely to be predated, and a lower seed production, as well as factors such as the retention of seed in the canopy, seed dormancy and short seed longevity. In other areas of South Africa (e.g. southern Karoo), soil seed densities have also been found to be negatively correlated with densities of adult plants (Esler 1999). Esler & Cowling (1995) found a strong negative correlation between the degree of seed retention and maximum germination. Later successional species tend to produce seeds with more limited longevity than early successional species (Esler 1999). Long-lived perennials germinate rapidly and have high overall germination compared to short-lived species and it is their longevity which allows adult plants to risk the high probability of seedling mortality (Esler & Cowling 1995). Therefore, persistent seed banks seem to be of minor importance in the re-establishment of long-lived perennial plants (Milton 1995) and the regeneration of these species following disturbance seems to depend rather on recent seed production and suitable germination conditions (Milton 1995;

Milton *et al.* 1999). Thus, the translocation of *O. cylindrica, L. suavissimus* and *R. versicolor* and the subsequent production of seed by these individuals appears to have contributed to the recruitment of the translocated species.

Although the pre-mining vegetation is not well represented in the topsoil seed bank (de Villiers 2000; de Villiers *et al.* 2003), certain species (e.g. *R. versicolor* and *L. suavissimus*) do seem to be better represented in the seed bank than other species. The results of this study show that plant translocation and topsoil replacement generally resulted in only a slightly greater number of *R. versicolor* and *L. suavissimus* seedlings than where just topsoil was replaced (Table 6.3). These findings could be related to the large number of seeds produced by these two species (and associated seed bank) and the capability of storing reproductive potential in their hygrochastic capsules once established.

De Villiers (2000) found *Z. morgsana* to be one of the more prominent perennial species present in the seed bank of the study area. Therefore, it was speculated that topsoil replacement should be sufficient for the revegetation of that species. However, this did not seem to hold true as both rehabilitation methods resulted in poor recruitment of *Z. morgsana* seedlings (Table 6.3). It should again be noted that the recruitment of *Z. morgsana* individuals was also poor in the reference site (Figure 6.7). Therefore, it is speculated that the poor recruitment of *Z. morgsana* could be due to a combination of the conditions not being ideal for germination (below average rainfall) and its inability of store seed until germination conditions are more favourable.

Assuming that the number and distribution of individuals within the reference site is what could be expected during a period of below average rainfall in the natural vegetation, it is interesting to note that recruitment of *O. cylindrica* and *L. suavissimus* individuals in translocation site was greater than the reference site (Table 6.3). Since the reference site was not grazed by sheep, the reduced recruitment cannot be related to the loss of *O. cylindrica* flowers, and hence seed production and germination, through grazing. This result could be explained by the differences in the number of sexually mature individuals and parent plant abundance. The translocation site had a greater number of sexually mature *O. cylindrica* individuals in the middle to larger size class than the reference site. This would suggest that individuals were translocated at a greater density than what is found in the natural vegetation. Further support for this is provided if one compares the Importance Values[1] calculated for *O. cylindrica* and *L. suavissimus* in the translocation and the reference sites (Chapter 4). The Importance Values of *O. cylindrica* and *L. suavissimus* in the translocation site were between 2.5 and 5 times greater than in the reference site, which indicates that these species are more dominant in the translocation site.

The greater number of *O. cylindrica* and *L. suavissimus* seedlings present in the translocation site than the reference site could also be related to greater inter- and intra-specific competition at the reference site where the vegetation had a greater cover, species richness and diversity (results presented in Chapter 4). There are varying theories regarding competition among desert plants. Some authors are of the opinion that competition is infrequent or absent and may become relatively less important and facilitation

[1] Importance values are calculated using relative cover, relative frequency and relative density as parameters of importance summed to give a maximum importance value of 300 (Mueller-Dombois & Ellenberg 1974).

relatively more important as abiotic conditions become more limiting (Riginos *et al.* 2005). Other authors suggest that competition among plants is a common factor influencing plant populations (Miller & Huenneke 2000). In the southern Karoo, seedling survival is strongly influenced by competition from neighbouring plants, and competition from established plants is the major cause of mortality among emergent seedlings (Milton 1995). Seedling survival has also been found to be greater in disturbed microsites (Dean & Milton 1991). While translocation sites allow seedlings to establish with some consistency because they minimise inter-specific competition, their bare, exposed surfaces may be hostile for seedling establishment (Morgan 1999). However, this explanation regarding greater inter- and intra-specific competition at the reference site does not hold true for *R. versicolor*, as recruitment in both rehabilitation sites was poor relative to the reference site. The greater recruitment in the reference site could be related to the greater number of sexually mature individuals present (Table 6.4). Further support for this is provided if one compares the Importance Values calculated for *R. versicolor*, indicating that this species is more dominant in the reference site.

Some of the population structural comparisons of *O. cylindrica* and *R. versicolor* over the four sampling periods show a distribution where the average density decreases with increasing height or age. The initial relatively steeper slope of the smaller size classes compared to the flatter slope of the large size classes (i.e. the “reverse-J” or “L-shaped” population distribution) indicates a larger increment of change in the smaller size classes. This type of size-dependent (or age-dependent) distribution can reflect a population with a high mortality among seedlings (Rouset & Lepart 1999); a population where recruitment and mortality are continuous and density dependent rather than episodic (Widyatmoko *et al.* 2005); or a colonising or rapidly expanding population where germination and establishment of individuals is high (Chandrashekara & Sankar 1998). The “reverse-J” population distribution can be attributed to self-thinning within each site (Goodburn & Lorimer 1999) or to age-specific mortality (Milton *pers. comm.*). Therefore, if the “reverse-J” population distributions of *O. cylindrica* and *R. versicolor* are a result of a high mortality among seedlings and if mortality exceeds recruitment, these populations will decline in the long-term. It is suggested that further research be undertaken to determine the driving force behind these “reverse-J” population distributions, as there is no guarantee that translocation has resulted in the establishment of a self-perpetuating population of the translocated species in the long-term.

6.5 CONCLUSIONS

The population structures of all four species in all three sites appear to be unstable and therefore these populations will vary over time. This instability could be attributed to the sporadic Namaqualand winter rainfall and variable recruitment. Although populations showing variable recruitment make it difficult to project population trends over the short-term (Shackleton 1993) and the fact that short-term recruitment studies might provide misleading data (Coates 2002), it would appear that the translocation of *O. cylindrica, R. versicolor* and *L. suavissimus* has resulted in the recruitment of seedlings and the development of self-perpetuating populations within the translocation site. It would also seem that the germination of seeds in the topsoil seed bank (although to a lesser extent) resulted in the recruitment of seedlings of *R. versicolor* and *L. suavissimus*.

O. cylindrica, R. versicolor and *L. suavissimus* should be considered for future large-scale translocation purposes in the winter growing season when the winter rainfall is not below average. However, the long-term viability of rehabilitated *Z. morgsana* populations is in doubt due to the almost complete lack of recruitment and is likely to remain difficult to re-establish during periods of below average rainfall. The long-term viability of rehabilitated *Z. morgsana* populations needs to be determined after the next favourable germination period in order to determine whether this species should be considered for any future large-scale translocation purposes. No translocated *Asparagus* spp. individuals survived and should therefore not be considered for any further large-scale translocation purposes.

There is no guarantee of translocation success and the establishment of a self-perpetuating population of the translocated species in the long-term. If the population structures of *O. cylindrica* and *R. versicolor* are a result of a high mortality among seedlings and if mortality exceeds recruitment these populations will decline in the long-term. Therefore, the germination and establishment of seedlings are of paramount importance for the maintenance of populations in rehabilitation areas (Contreras & Valverde 2002). If plant translocation is to become a valuable rehabilitation tool at Namakwa Sands, all translocation efforts should be monitored over the long-term to determine if these populations are in fact self-perpetuating. This long-term monitoring will allow the techniques and species selected for translocation to be refined and improved. In order to meet the long-term rehabilitation goal at Namakwa Sands (i.e. small-stock farming), it may be necessary to consider the translocation of palatable and non-palatable species in multi-species clumps. In this regard, it is recommended that further investigations be undertaken to compare seed viability, germination frequency and seedling survival / mortality of the five species in order to obtain a greater understanding of why certain species are more successfully translocated than others.

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CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS: MONITORING AND FUTURE REHABILITATION

7.1 INTRODUCTION

Large areas of mined land at Namakwa Sands have been rehabilitated using four rehabilitation techniques. Information on the changes to the local ecosystem due to rehabilitation is required for a better understanding of the techniques implemented, the interactions between the soil and vegetation and appropriate management of the rehabilitation areas (Su & Zhao 2003). In addition, monitoring rehabilitation success will ultimately determine whether rehabilitation goals and specific end points have been reached (Holl & Cairns 2002).

Namakwa Sands currently undertakes an annual vegetation survey to measure species richness and cover of selected reference and rehabilitation sites (Grobler 2003). Monitoring programmes should not be static as they may need to be refined or changed for various reasons, for example changes may be required to rehabilitation goals as society's view of management changes, additional ecosystem information arises, etc. (Cairns *et al*. 1993). This chapter makes various recommendations for the refinement of the current monitoring programme (vegetation survey) implemented at Namakwa Sands based on the findings of this thesis. In addition, the success of the four rehabilitation techniques is summarised and recommendations regarding adaptive management and future research are presented. Lastly, certain suggestions are made to improve the management of the rehabilitation programme and associated research.

7.2 REHABILITATION PLANNING AND MONITORING AT NAMAKWA SANDS

This section analyses various aspects of rehabilitation planning at Namakwa Sands, as well as the annual vegetation survey, and makes various recommendations to refine the rehabilitation programme.

7.2.1 REHABILITATION OBJECTIVES

If rehabilitation is to be successful in restoring biodiversity, the process must go beyond the reconstruction of composition and cover to restore structure, biological interactions, processes and integrity (Reay & Norton 1999). However, the short-term objectives of the Namakwa Sands rehabilitation programme only deal with vegetation species richness and cover. Therefore, it is recommended that Namakwa Sands develop additional rehabilitation objectives relating to the structure and function of the

natural vegetation. These objectives should be linked to the vegetation parameters recommended for long-term monitoring (Section 7.2.2). This will give a better indication of whether the rehabilitation sites are progressing towards the desired end point or whether adaptive management is required.

7.2.2 MONITORING PARAMETERS

The selection of effective indicators is the key to the overall success of any monitoring programme (Dale & Beyeler 2001). Namakwa Sands currently monitors only species richness and cover as part of their annual vegetation survey (Grobler 2003), which is presumably related to the short-term objectives. Although the structure of vegetation plays an important role in arid system functioning (Desmet & Cowling 1999) and should ideally be considered in the long-term monitoring programme, it would appear that monitoring clump size, number of clumps, inter-clump distance and clump height within rehabilitation sites will not provide valuable information regarding rehabilitation success. Based on the results of this study, it is recommended that species composition and similarity, species richness, species diversity, vegetation cover, species dominance (Importance Values), vertical structure and functional diversity of the vegetation (including both clumps and inter-clump areas) be monitored as part of the long-term monitoring programme and compared to reference sites.

The monitoring of soil parameters, in addition to the vegetation, is important because soil influences the establishment and survival of plant and animal species and if efforts are made to ameliorate the soil conditions, these parameters should be monitored (Holl & Cairns 2002). Therefore, it is also recommended that carbon, pH and sodium in soil be monitored as these parameters showed little evidence of returning to naturally occurring levels. Soil sampling should take into account the heterogeneous distribution of soil resources between clump and inter-clump areas.

It should be noted that this study only assessed the success of various vegetation parameters and soil chemistry of four rehabilitation sites in relation to two reference sites. Cognisance should be taken of other components of the system apart from vegetation rehabilitation and succession (Westman 1991). The parameters recommended for long-term monitoring at Namakwa Sands therefore should not be seen as exhaustive. The results of other studies on the fauna, mycorrhiza, insects, etc. undertaken at Namakwa Sands should also be taken into consideration and the monitoring parameters expanded accordingly.

7.2.3 REFERENCE AND REHABILITATION SITES

Due to the spatial variation of the vegetation within the mining area, reference sites must be carefully selected, especially once mining moves out of the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld (site R1) and into the *Ruschia tumidula – Tetragonia virgata* Tall Shrub (site R2). It is suggested that comparisons preferably be made against the vegetation community that existed prior to mining or alternatively the community that Namakwa Sands has identified as the end point. Namakwa Sands currently monitors only one reference site as part of their annual vegetation survey, which is

located within the *Ruschia versicolor – Odyssea paucinervis* Dwarf Shrub Strandveld. This site is similar to site R1 sampled in this study. Since most of the mining to date has taken place within this vegetation community and only small portions of this vegetation community remain intact, the location of the current reference site is considered satisfactory. However, once mining moves into the *Ruschia tumidula – Tetragonia virgata* Tall Shrub community, a reference site should be established in this community as well.

Namakwa Sands currently monitors six rehabilitation transects as part of their annual vegetation survey (Grobler 2003). The six rehabilitation sites are located in both the West (two sites) and East (four sites) mines. All monitored rehabilitation sites located within the West Mine are located in areas where topsoil was replaced, whereas all sites within the East Mine are located in areas without topsoil. Since most of the mined out areas have been rehabilitated with just the replacement of topsoil it is recommended that additional topsoiled rehabilitation sites be monitored within the East Mine. It is important when selecting additional monitoring sites to know the age of each site as well as the rehabilitation technique implemented there. It is also recommended that all four rehabilitation techniques (as assessed in this study) continue to be monitored, as this will provide further valuable information on the success of these rehabilitation techniques over the longer term.

7.2.4 MONITORING FREQUENCY

Due to the temporal variation of the vegetation within the study area between summer and winter, monitoring should ideally be undertaken on a bi-annual basis during the summer (November to March) and winter (May to August) months. If this is not feasible, sampling should be undertaken during the winter months in order to get a better representation of species richness and diversity of perennials and annuals.

7.2.5 MONITORING SUCCESS

Currently, rehabilitation success at Namakwa Sands is determined by cover and species richness, both being expressed as a percentage of the reference site values (Grobler 2003). These data should be analysed and synthesised in a way that is useful to decision- or policy-makers (Noss 1990; Holl & Cairns 2002). Various approaches to data analysis are presented in Chapter 2. In summary, quantitative approaches to determine if objectives have been met include performance curves, resemblance functions, modelling and resilience indices (Westman 1991; Pastorok *et al.* 1997).

A common approach is to monitor vegetation and soil changes occurring along a chronosequence. This approach is recommended for Namakwa Sands and this study could provide the first point in the chronosequence. Monitoring systems have more recently attempted to identify indices of landscape function that are quantitative, rapid, repeatable and sensitive to change, e.g. “Ecosystem Function Analysis” (Tongway *et al.* 1997; Holm *et al.* 2002). It may also be useful to use this approach at Namakwa Sands, as Tongway *et al.* (1997) showed that this procedure could potentially be used for a wide range of

mine types (including bauxite, mineral sands, coal, gold, uranium, nickel and iron ore) across a wide range of climatic conditions. This approach should ideally be tested as part of the ongoing research at Namakwa Sands.

7.3 REHABILITATION SUCCESS, ADAPTIVE MANAGEMENT AND FURTHER RESEARCH OPPORTUNITIES

The variation in rehabilitation design increases the probability that at least one technique is successful as well as providing information that might be key to future adaptive management considerations (Pastorok *et al.* 1997). This study provided an indication that topsoil replacement (sites S1 & S3) and translocation of plants in multi-species clumps (sites S3 & S4) facilitated the return of similarity, species richness, species diversity and cover to post-mined areas. Site S3, which had the greatest amount of biological input (topsoil replacement and plant translocation), appeared to be the most successful technique in facilitating vegetation recovery similar to reference site R1, followed by site S4 (seeding and plant translocation) and site S1 (topsoil replacement only). Site S2, where only seeding was implemented, appeared to perform the worst. Therefore, it is recommended that Namakwa Sands continue to replace topsoil in all future rehabilitation efforts and, when possible (e.g. after sufficient winter rain), continue to translocate plants, collected from the mining front, in multi-species clumps. It is not known whether the translocation of separate plants would facilitate vegetation recovery to a similar extent as found for the translocation of plants in multi-species clumps. Therefore, it is recommended that until this has been tested as part of a future research project that Namakwa Sands continue to translocate plants in clumps.

Site S2 did not meet the three-year species richness objective and therefore requires adaptive management (e.g. reseeding and / or translocation with selected species before the onset of the winter rains) in order to increase the establishment of more indigenous species and to facilitate natural successional processes. There is little optimism that any of the rehabilitation sites studied will meet the final species richness objective of at least 60% of the number of species occurring within reference sites in the near future. Species and functional diversity appears to be the most limiting factors within the rehabilitation sites. For example, bird pollinated plants and geophytes are not returning to the mine site. Namakwa Sands will not be able to meet their long-term objective of small-stock farming if diversity and the number of palatable species do not increase significantly. Species diversity is expected to increase over time (Reay & Norton 1999), but adaptive management (e.g. reseeding and / or translocation with selected species before the onset of the winter rains) should seriously be considered in order to speed up this process.

In terms of species selected for translocation by Mahood (2003), *Othonna cylindrica, Ruschia versicolor* and *Lampranthus suavissimus* should be considered for future large-scale translocation purposes in the winter growing season when the winter rainfall is not below average. However, *Zygophyllum morgsana* appears to be more difficult to re-establish under the current climatic conditions (below average rainfall). The long-term viability of rehabilitated *Z. morgsana* populations needs to be determined after the next favourable germination period in order to determine whether this species should be considered for any

future large-scale translocation purposes. No translocated *Asparagus* spp. individuals survived and should therefore not be considered for any further translocation purposes. It is recommended that further investigations be undertaken to compare seed viability, germination frequency and seedling survival / mortality of the five species selected for translocation in order to obtain a greater understanding of why certain species are more successfully translocated than others. In addition, it is recommended that all translocation efforts be monitored over the long-term to determine if these populations are in fact self-perpetuating. This long-term monitoring will allow the techniques and species selected for translocation to be refined and improved.

The grasses *Eragrostis curvula* and *Ehrharta calycina*, which were dominant in the sites seeded, could potentially be considered for future seeding. However, since *Eragrostis curvula* is not indigenous to the area, it is recommended that only *Ehrharta calycina* continue to be considered for future seeding. Other species, which were dominant within reference sites, such as *Tripteris oppositifolia, Eriocephalus racemosus* var. *affinis, Trichogyne repens, Hermannia cuneifolia, Hermannia disermifolia, Stoebe nervigera, Willdenowia incurvata* and the grasses*, Chaetobromus involucratus, Stipagrostis obtusa* and *Stipagrostis namaquensis*, could also be considered for future translocation and / or seeding trials. In addition, the species listed by de Villiers *et al.* (1999), considered ideal for restoring the former appearance and structure of the vegetation, could also be considered for future translocation and / or seeding trials in order to improve species diversity and palatability within rehabilitation sites.

More time is needed to ameliorate the soil chemistry within rehabilitation sites (especially carbon) to the same level as in reference sites. However, should long-term monitoring show that carbon, pH and sodium are affecting the success of the rehabilitation programme, various cost-effective amelioration measures may have to be considered for implementation by Namakwa Sands. In order to increase organic matter within rehabilitation areas and create additional germination sites, Namakwa Sands should consider creating clumps with cleared vegetation from the mining front. This proposal should be tested as part of ongoing research and rehabilitation development at Namakwa Sands. In addition, since this study only considered the amelioration of soil chemical properties it is recommended that further research be undertaken to assess the success of the four rehabilitation techniques in terms of restoring the physical properties of the soil.

7.4 REHABILITATION MANAGEMENT

One of the primary causes of rehabilitation failure is insufficient management and maintenance in terms of intensity (e.g. inadequate provision of funds) or type (e.g. misunderstanding of site function) (Harris *et al.* 1996). Namakwa Sands has experienced a high turnover of Environmental Managers over the past few years. Unfortunately this high staff turnover has resulted in inconsistent supervision and input from Namakwa Sands in terms of research being undertaken at the mine. It has also seemingly resulted in the loss of knowledge as to what research has been undertaken, where it was undertaken, what the associated recommendations have been and what still requires research or refinement.

The solution is better data storage and management. It is suggested that more people become involved with the rehabilitation programme at all levels of management and from various organisations. In particular, it is recommended that the rehabilitation contractor and / or the rehabilitation consultant become more actively involved with each research project and the findings of those projects. The rehabilitation consultant commissioned to monitor rehabilitation success should be aware of what the ideal monitoring parameters are, where monitoring should be undertaken, how best to analyse, interpret and present the information gathered, and what adaptive management solutions exist should rehabilitation not be proceeding as predicted. In this regard, Namakwa Sands should prepare a contingency plan based on research undertaken at the mine. This contingency plan should list the various recommendations and adaptive management options available to them.

Another important management consideration is when rehabilitation sites should be opened up to small-stock farming. It is recommended that only when all rehabilitation objectives have been met, species diversity is not significantly different to reference sites, when the number of palatable species increases and once a closure certificate has been issued by the Department of Minerals and Energy should rehabilitation areas be considered for grazing. Based on the findings of this study, Namakwa Sands should seriously consider adaptive management (e.g. reseeding and / or translocation with selected species before the onset of the winter rains) to increase species diversity and the number of palatable species in order to speed up this process. Alternatively, an appropriate grazing strategy, which is related to the *Tetragonia fruticosa* dominated vegetation within rehabilitation sites, would need to be determined and adopted.

This study found that the long-term rehabilitation goal had not been achieved. Since the rehabilitation sites are very young in rehabilitation terms (approximately four years old), it is difficult to say whether or not any of the rehabilitation sites or techniques are on a successional trajectory towards the desired endpoint. Namakwa Sands must continue to monitor plant and soil changes until the rehabilitation objectives and goal have been achieved.

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