Towards ecological restoration strategies for Peninsula Shale Renosterveld: testing the effects of disturbance-intervention treatments on seed germination on Devil's Peak, Cape Town

by
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Dissertation presented in fulfilment of the requirements of the degree of Master of Science at the University of Cape Town, Department of Environmental and Geographical Sciences
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September 2013

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Declaration

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Abstract

The ecological restoration of Peninsula Shale Renosterveld is essential to redress its conservation-target shortfall. The ecosystem is Critically Endangered and, along with all other renosterveld types in the Cape lowlands, declared 'totally irreplaceable'. Further to conserving all extant remnants, ecological restoration is required to play a critical part in securing biodiversity and to meeting conservation targets. Remnants of Peninsula Shale Renosterveld are situated either side of the Cape Town city bowl and, despite formal protection, areas of the ecosystem are degraded and require restoration intervention.

The body of research addressing restoration in renosterveld is small, yet growing, and seed-based restoration efforts have achieved limited success. This study primarily set out to further understand community recovery through determining the effects of interventions, implemented to mimic ecological drivers, on several seed germination criteria. Thirty-two interventions (comprised of five, crossed factors: burning, tillage, herbicide-application, rodent-exclusion and seeding) were incorporated into a field experiment situated in an area that was most likely ploughed over a century ago and is currently dominated by alien, annual grasses. Additional aspects of the study included determining the physiological status of the seed from 31 harvested species (through viability and germinability tests) towards identifying key restoration species, and, assessing intervention cost-effectiveness as a measure of intervention feasibility.

The majority of the harvested species exhibited moderate to high levels of viability and germinability and occurred in the middle or upper key-restoration-species index range, indicating their potential for use in future restoration efforts. Seeding contributed considerably to overall community attributes, significantly increasing indigenous seedling density, species richness and canopy cover. Due to the dominance of alien, annual grasses, seeding alone was relatively ineffectual and recruitment was considerably improved when seeding was implemented in conjunction with one or more of the other interventions. These findings indicate that a lack of available seed is not the only barrier to community recovery and that competition exerted by the alien grass component plays a large role in inhibiting seedling recruitment of desirable species. Intervention effectiveness increased with the number of factors per intervention yet, fortuitously, the most effective interventions were not necessarily the most costly. Some interventions resulted in good performances and have the potential to restore self-perpetuating communities with a semblance of ecosystem composition, structure and function.

There is a clear and promising way forward incorporating these findings into feasible, implementable, landscape-scale, ecological restoration strategies for Peninsula Shale Renosterveld and potentially other Critically Endangered and 'totally irreplaceable' renosterveld ecosystems.

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Acknowledgements

This project benefitted from abundant and generous support from numerous people and organisations. I thank:

Cape Peninsula University of Technology: Lucienne de Villiers, Dr Reinette Snyman and Dr Mike Young for vision, encouragement and support.

Doug Euston-Brown (for assistance with in-field plant identification), Kay Waldon (for support and guidance throughout), Deon van Eeden (for reviewing the seed collection protocol and providing commercial seed collection costs) and Leonard Henning for guidance in herbicide selection.

South African National Parks: Ruth-Mary Fisher (permitting), Carly Cowell (seed collection expertise), Jannie du Plessis and Jackie Pretorius (game camp access), Blessing Mutiti and Tobie Brunk (assistance with seed processing), Philip Prins, Clinton Dilgee, Pierre Gallagher and team for nothing but enthusiasm and willingness to plan and implement the prescribed burn and for providing assistance with burn costing.

Seed collecting volunteers: Lisa Conradie, Jacqui L'Ange, Penny Brown, Geert Sprangers, Belinda and Chris Steyn, Gianpaolo Gilardi, Katherine Nurse, Marion Ogden, Peter Grey, Mary, Annelise, John, Mike and Joy Waller and Zukile Gqabuza for offering hours towards happy and productive harvesting in, at times, harrowing conditions.

Cape Nature: Carlo Arendorf for seed export permits.

Royal Botantic Gardens Kew: Jane Davies (for the implementation of viability tests) and Dr Rose Newton for guidance throughout, reviewing chapter two of the manuscript, providing valuable comment and improving the accuracy of the text.

Millennium Seed Bank: Olivia Tyambetyu and Victoria Wilman for seed processing guidance and assistance with preparing seed for export.

South African Biodiversity Institute (SANBI) (including Kirstenbosch Gardens, Bolus Herbarium and Seed Room): Les Powrie (for providing the georeferenced species list), Anthony Hitchcock and Trevor Adams (for guidance on pre-sowing seed treatments, willingness to assist with and house the germinability tests, and, germinability interpretation), Ernst van Jaarsveld (for guidance on pre-sowing seed treatments), John Manning (for assistance with species identification), Kuphumla Zenze (for seed collection expertise throughout), Deon Smith, Grant van Gusling, Pat Kettledas and Andrew Jenkins (for conversations, seed processing expertise and guidance), Enoch Sodladla (for seed smoking) and Ntsindiso Zide for watering the species in the glasshouse.

Liesl van der Walt, Dr Peter Linder and Fiona Powrie for assistance with the interpretation of germinability.

University of Cape Town (UCT) Zoology department: Dr Gary Bronner (for input into the rodent-exclusion cage design) and George du Plessis for assistance with and management of the cage construction process.

Tygerberg Nature Reserve: Penny Glanville for providing sampling data.

South African Environmental Observation Network (SAEON): Dr Nicky Allsopp (for guidance and input), Abri de Buys and Siyasanga Mpehle for installing the weather and soil moisture equipment and the monthly data collections.

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Kumbukani Imani, Blessing's Asoka and George Bannet for enthusiasm, stamina and attention to detail for the cage construction, tillage and on-site implementation.

My parents, John and Annelise Waller, for, in addition to unconditional support throughout, rallying to stir 64 buckets of seed, sweep plots and install cages for two days in the rain.

Moorosi Leshoele (for assisting with the July 2012 data collection and for the inclination for grace and philosophy) and Takunda Mambo for assisting with the September 2012, November 2012 and January 2013 data collections and for maintaining superb humour and the mantra 'things could always be worse'.

UCT Botany Department: Dawood Hattas for the use of the light reading equipment.

South African Weather Services (SAWS): Gail Linnow for providing weather data.

UCT Statistical Advisor: Katya Muaff for valuable support and assistance with experimental design and analysis.

UCT Environmental and Geographical Sciences Department: Sharon Adams for considerable time and mental clarity required for managing project finances.

Table Mountain Fund (for financing the costs of the project as one of the Cape Action for People and the Environment projects occurring within the Greater Cape Town project portfolio) and the Council for Scientific and Industrial Research for supplementing the project budget.

Harry Crossley Scholarship (2012), National Research Foundation/SAEON student bursary (2012 and 2013), Fynbos Forum Scholarship (2012 and 2013), Wilderness Wildlife Safaris Trust Scholarship (2012 and 2013) and UCT Masters Research Scholarship (2012 and 2013) for considerable personal financial support.

Dr Pat Holmes for providing guidance throughout, reviewing chapters two to four of the manuscript and for providing insightful and detailed feedback.

Dr Pippin Anderson for supervising with extraordinary wisdom, practicality, humour and care throughout.

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1 General Introduction

1.1 Rationale

Ecological restoration of Peninsula Shale Renosterveld, a Critically Endangered ecosystem (SANBI and DEAT 2009) with remnants situated on either side of the Cape Town city bowl, has been identified as essential due to extensive transformation (Rebelo et al. 2006). Peninsula Shale Renosterveld is a vegetation type of the West Coast Renosterveld Bioregion (Rutherford et al. 2006), the latter occupying the western portion of the highly transformed and fragmented Cape Renosterveld Lowlands (von Hase et al. 2003a) and comprised of the previously referred to Swartland and Boland Broad Habitat Units (Cowling and Heijnis 2001). The ecosystem status of renosterveld in the Cape lowlands, comprised of the West Coast Renosterveld and East Coast Renosterveld Bioregions, is Critically Endangered and has been classified as 'totally irreplaceable' rendering all extant remnants essential for meeting conservation targets (von Hase et al. 2003a, Rouget et al. 2006).

Ecological restoration has an important part to play in contributing towards securing biodiversity and meeting conservation targets (Cowling et al. 2003). The conservation strategy and action plan for the Cape Floristic Region (CFR), implemented by Cape Action for People and the Environment (CAPE) and formally adopted by a major executing agency Western Cape Nature Conservation Board (Cape Nature) (Younge and Fowkes 2003), aims to achieve not only sustainable conservation by 2020, but in addition, the restoration of degraded habitat (Lochner et al. 2003, SANBI 2011, CAPE 2013). Ecological restoration has been identified as essential where habitat transformation is extensive and extant remnants are not adequate to meet conservation targets, despite the potentially high opportunity costs associated with agricultural landscapes (Cowling et al. 2003, Pressey et al. 2003). Thus, the reclamation of abandoned agricultural fields and plantations (Rebelo 1995, Krug and Krug 2007), which make up sizeable areas (Heelemann 2012), is required to meet conservation targets and are important areas for future restoration efforts (Krug and Krug 2007, Heelemann 2012). With an immediate aim to promote the persistence of biodiversity, ecological restoration should, in addition to reclamation, set out to increase the size of extant remnants, link remnants (Rebelo 1995, Pressey et al. 2003, Krug and Krug 2007), buffer remnants and provide habitat for threatened species (Krug and Krug 2007).

Despite remnants of Peninsula Shale Renosterveld being formally protected within Table Mountain National Park (TMNP) (Rebelo et al. 2011), portions of the ecosystem are degraded and require restoration intervention (Rebelo et al. 2006, City of Cape Town 2008, Terblanche 2011, Cowan 2013). However due to large-herbivore extinctions and severe habitat transformation, there is limited knowledge of the ecology of pristine renosterveld (Rebelo 1995). This lack of pre-transformation data is problematic for restoration due to an absence of representative reference systems (Rebelo 1995) contributing to a limited understanding of ecosystem drivers (Krug et al. 2004). Furthermore, the body of research pertaining to renosterveld restoration is small and restoration efforts, particularly seed-based restoration efforts, have achieved limited success to date (Holmes 2002, Midoko-Iponga 2004). Although renosterveld studies, predominantly over the past 10

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years, have incrementally contributed to an understanding of ecological processes and responses to restoration interventions, there are still considerable knowledge gaps. More research in renosterveld restoration is required to further our understanding of community recovery in response to restoration interventions with a view to optimizing the effectiveness and efficiency of restoration outcomes.

Through the different components of this research (seed physiology, intervention responses and intervention feasibility), this study has set out to make empirical and theoretical contributions to the small, but growing, body of renosterveld restoration research. The outcomes of this study are relevant to determining feasible, landscape-scale restoration strategies for Peninsula Shale Renosterveld and potentially other closely associated renosterveld types. The outcomes of this research thus align with the broader conservation strategy for the CFR and are well-placed to contribute towards attaining conservation targets of 'totally irreplaceable' ecosystems.

1.2 Project description and thesis structure

This project has set out to grow the knowledge base of ecological restoration in Peninsula Shale Renosterveld. At the heart of the study, through in-field experimentation, the research tests several aspects of community recruitment response to 32 different restoration interventions (including the control). The interventions are surrogates for ecological drivers of recruitment in renosterveld and include burning, tillage, herbicide-application, rodent-exclusion and seeding. Supporting the core of the study are two additional aspects of great relevance to ecological restoration. The first is that of determining the physiological status of seed which is important to ascertain prior to commencing landscape-scale restoration so as to optimize restoration outcomes and avoid costly failures (Lippitt et al. 1994). The second is to ascertain the cost-effectiveness of interventions so as to promote well-informed decision-making with respect to intervention feasibility at the landscape-scale.

The thesis has been arranged into five chapters. The first chapter contains the project aim and objectives and a review of the literature. The literature review presents a broad view of the theory of restoration ecology followed by aspects relevant to the restoration of Peninsula Shale Renosterveld including its regional context, historical influences, and, contemporary phytosociological understanding of renosterveld and Peninsula Shale Renosterveld. Literature supporting the rationale underpinning Chapter 2 (addressing the importance of ascertaining the physiological status of seed), Chapter 3 (the importance of testing ecological responses to restoration interventions) and Chapter 4 (the importance of ascertaining intervention feasibility towards the determination landscape-scale restoration strategies) are presented in each of the respective chapters. The second chapter addresses the physiological status of seed collected from 31 renosterveld species. Thereafter, each of the 31 species is assessed with respect to how useful it is likely to be for use in future seed-based ecological restoration efforts. Chapter three is centred around the in-field experimentation which tested the effects of 31 restoration interventions on several aspects of community recruitment. The fourth chapter carries out an assessment of intervention effectiveness in concert with intervention cost per hectare to determine intervention cost-effectiveness as a measure of intervention feasibility. Lastly,

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chapter five provides the research conclusions and recommendations guiding future research and implementation of renosterveld restoration.

1.3 Aim and objectives

The principal aim of this research is to determine the effectiveness of each of the 32 interventions (including the control) with respect to several seed germination criteria.

Research objectives:

1) To determine the germination responses for:
   a) The 31 seeded species:
      i) seedling density (total number of seedlings per m^2 and number of seedlings per species per m^2);
      ii) species richness (number of species per m^2);
      iii) canopy cover (percentage canopy cover per species per m^2);
      iv) seedling height (height per seedling (cm) per species per m^2).
   b) The existing indigenous species:
      i) seedling density (total number of seedlings per m^2 and number of seedlings per species per m^2);
      ii) species richness (number of species per m^2);
      iii) canopy cover (percentage canopy cover per species per m^2);
   c) The weed species:
      i) canopy cover (combined canopy cover of all alien, invasive species per m^2);
2) To determine the percentage *germinability* of each of the 31 species.
3) To determine the percentage *viability* of each of the 31 species.
4) To consider each of the 31 species (in terms of germinability, viability and several additional criteria) towards evaluating it as a key species for inclusion in future restoration strategies.
5) To determine the feasibility of each intervention (with respect to effectiveness (in terms of seed germination criteria) and cost) for inclusion in landscape-scale restoration strategies.

1.4 Theoretical framework

This portion of the literature review motivates the need for the restoration of Peninsula Shale Renosterveld. The review initially addresses the theory of restoration ecology and predominantly draws on international literature. Thereafter the phytosociology and conservation status of Peninsula Shale Renosterveld is considered within the context of its regional position, historical influences and contemporary phytosociological understanding of renosterveld.

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1.4.1 Restoration ecology theory

Restoration ecology

Ecological restoration is the process of assisting the recovery of degraded, damaged, or destroyed ecosystems (SER 2004). The science of restoration ecology and the practice of ecological restoration have mutually informed one another (Hobbs and Harris 2001) over the past 30 years contributing to the knowledge base of this relatively new discipline (Hobbs and Cramer 2008). Despite notable progress in the field, there are still considerable uncertainties and knowledge gaps (van Andel and Aronson 2012) emphasizing the need for further coordination of scientific research, innovative practice and policy (Hobbs and Harris 2001, Suding and Hobbs 2008). Ecological restoration is underpinned by several agendas and holistic efforts require consideration and realization of the ecological, socio-economic, cultural, and, political and legislative motivations for ecological restoration (MA 2005, Young et al. 2007, TEEB 2010, van Andel and Aronson 2012). This is challenging where in broad terms, the ecological agenda seeks to restore ecological processes, the conservation agenda seeks to recover biodiversity, the socio-economic agenda seeks to reinitiate ecosystem services and the cultural agenda endeavours to enhance communities and institutions (Clewell and Aronson 2013).

World-wide, ecosystem transformation and degradation, resulting predominantly from intensified economic development in the last 50 years, has resulted in a loss of ecosystem services and biodiversity (Holl et al. 2007) compounding socio-economic pressures (Aronson et al. 2007). Further to the effects of land-cover change, nitrogen enrichment, fragmentation and invasive species, pressure on ecosystems is escalated by a changing climate (Hobbs and Cramer 2008). Extensive ecosystem alteration has resulted in the greater likelihood of no-analogue ecosystem changes (Aronson et al. 2007) where the majority of ecosystems are so altered, with respect to functional and structural attributes, that they are more appropriately considered novel systems (Hobbs and Cramer 2008, Seastedt et al. 2008, Perring et al. 2013). From an ecological perspective, ecosystem degradation leads to ecosystem simplification (Clewell and Aronson 2013). This loss of ecosystem complexity manifests in a number of ways, including, the loss of specialized species and an increase in generalized and invasive species; reduction in community structure complexity; the alteration of microclimate and moisture regimes; loss of soil properties and nutrient retention; and, trophic cascades (Clewell and Aronson 2013). Ecosystem changes of this nature are experienced globally and are significant and unprecedented to the extent that the restoration of ecosystems in many instances is widely considered non-negotiable and an imperative (Aronson et al. 2007, McGhee et al. 2007).

Ecosystem recovery

Community assembly is the process by which a biotic community develops over time as species colonize, persist or become extinct within the ecosystem (van Andel et al. 2012). As per expectation, the sequence in which species colonize a site (termed Initial Floristic Composition by Grime (1979)) has bearing on the subsequent nature of the restoring community with respect to both structure and ecosystem processes. The species that establish first are thus more likely to contribute to these

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community traits (Clewell and Aronson 2013). In the context of natural regeneration, environmental filters (dispersal filters, tolerance/ecological filters and disturbance filters) (Clewell and Aronson 2013) function to restrict the species able to colonize an area (White and Jentsch 2004) and ultimately determine community species pools (Zobel 1997). In the context of ecological restoration these filters pertaining to species pools may be overridden to a large extent, for instance, by the selection of species for introduction by seed (Clewell and Aronson 2013). Applying the principle of Initial Floristic Composition (Grime 1979), as the species introduced in restoration have the potential to persist in a community, selection of the appropriate species is important and should be carried out with a view to reinstating a desired trajectory (Clewell and Aronson 2013). In addition to the history of species immigration being relevant to community assembly, the pre-existing site conditions affect the nature of the restoring community and include environmental factors such as climate and hydrology (Clewell and Aronson 2013) and factors relevant to land-use legacy such as soil properties, biotic community characteristics (including plant-soil-feedback) and seed-bank status (Memiaghe 2008, Cowan 2013).

Theories of ecosystems existing only in equilibrium and ecosystem changes being linear have increasingly been replaced by the understanding that most ecosystems are dynamic and may be nonlinear, complex and unpredictable (Hobbs and Cramer 2008, Clewell and Aronson 2013). It is acknowledged that an ecosystem may not necessarily occur in one state but may be simultaneously composed of a number of 'alternative states' (multiple equilibria) as a response to past disturbance and land-use practices (Milton et al. 1994, Hobbs and Cramer 2008). In contrast to the idea of multiple equilibria, the theory of nonequilibria proposes that ecosystems are, on a small spatial scale, continually in a state of flux and thus never reach a state of equilibrium (Pickett and White 1985). Whether attributing ecosystem regeneration to models of multiple equilibria or nonequilibria, restoration efforts must acknowledge the complexity and unpredictability of ecosystems (Hobbs and Cramer 2008) by incorporating several likely restoration outcomes into project goals (Hilderbrand et al. 2005).

Tying into the theory of multiple alternative states, Whisenant (1999, 2002) proposed the concept of 'restoration thresholds' where ecosystem degradation reaches a point beyond which the recovery of the ecosystem fails (Hobbs and Harris 2001, King and Hobbs 2006, Perring et al. 2013). Thresholds can therefore prevent an ecosystem from reverting to a healthier state and consequently may act as a barrier to ecological restoration (Hobbs and Harris 2001, Hobbs and Cramer 2008). Whisenant proposed a model where transition thresholds result from biotic interactions and abiotic limitations (Whisenant 1999, 2002). Similar models are increasingly being used to describe ecosystem dynamics that are nonlinear and nonequilibrium (Hobbs and Cramer 2008) and Gaertner et al. (2012) incorporate a third threshold for the biotic-abiotic positive feedbacks associated with invasive species. Gaertner et al. (2012) present this in a model noting the relative roles of biotic interactions and abiotic limitations and showing the progression of degradations to a culminating point of positive feedback where the degraded state becomes entrenched as biotic and abiotic elements establish a positive feedback loop.

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Where lightly degraded ecosystems appear to have the ability to restore naturally, a no-intervention approach may be adopted (Hobbs and Cramer 2008, Holl et al. 2007). However, where thresholds have been crossed, passive restoration may not suffice, and active implementation of abiotic and biotic interventions is required (Hobbs and Harris 2001). The implementation of interventions seeks to manipulate and redress the ecological drivers of degradation (Brudvig 2011). Through the manipulation of the site, interventions are intended to render conditions appropriate for a specific community (Brudvig 2011) thereby facilitating a more rapid ecological restoration process and restoration outcomes (Palmer et al. 1997, Brudvig 2011). Once an environment is beneficially altered, ecological restoration is initiated and the ecosystem may potentially be aligned with its developmental trajectory (Palmer et al. 1997, SER 2004).

Reversing degradation is challenging particularly where ecosystem degradation is severe and trapped in a positive feedback loop (Gaertner et al. 2012). The interventions required to initiate recovery are context-specific (Gaertner et al. 2012) and may involve a multi-faceted approach (Prober et al. 2009) for instance eradicating annual invasive grasses (biotic factor) in conjunction with managing fire frequency (abiotic factor) towards favouring indigenous species. Abiotic interventions seek to redress chemical or physical limitations (Hobbs and Cramer 2008) and in some instances, redressing abiotic limitations alone may be adequate to facilitate system recovery (Hobbs and Cramer 2008). Once the site has returned from beyond the abiotic threshold efforts may concentrate on implementing biotic interventions seeking to manipulate the biotic community (Hobbs and Cramer 2008). Examples of biotic interventions include species removal, managing grazing regimes and the reintroduction or introduction of species (Hobbs and Cramer 2008) by planting seedlings or sowing seed (Holmes 2002, Midoko-Iponga 2004, Waters et al. 2007). A relatively intact ecosystem requires minor interventions in the form of enhanced management (Hobbs and Suding 2009) such as keeping alien species cleared and maintaining appropriate fire regimes (Gaertner et al. 2012). To ensure resource efficiency and promote restoration outcomes, addressing the abiotic factors as a priority ahead of the biotic factors is required to ensure that ecosystem functions are reinitiated and capable of supporting efforts to correct ecosystem structure and composition (Hobbs and Harris 2001). Determining the state of degradation and identifying which thresholds are relevant, however, is complex and frequently requires experimentation prior to implementing interventions at the larger scale of adaptive management activities (Hobbs and Cramer 2008).

Restoration strategy

Too frequently restoration efforts are misdirected and spend significant resources without understanding the drivers of degradation (Hobbs and Harris 2001) emphasizing the importance of a conceptual and practical framework to guide efforts (Hobbs and Cramer 2008). In providing a conceptual framework upon which to build strategic restoration ecology practice, Hobbs and Harris (2001) describe the ecological restoration process in a number of steps (Hobbs and Harris, 2001) (Figure 1.1). Determining the ecological-response model for an ecosystem is the central element of their strategy and requires a thorough understanding of the ecosystem (Hobbs and Harris 2001)

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which ideally encompasses phytosociological knowledge of both the pre-transformation ecosystem and the contemporary ecosystem including historical and cultural influences (SER 2004).

An ecological-response model is assembled through collation of available scientific data and local expertise and functions to predict the likely outcome of restoration interventions (Hobbs and Harris 2001). In the event of a paucity of data pertaining to the ecosystem, experimentation testing intervention responses is recommended (Suding and Hobbs 2008). The data thus informs the ecological-response model which in turn informs the restoration options and, in accordance with stakeholder interests, the definition of the project goals (Hobbs and Harris 2001). Project goals may stipulate for instance a target measure of seedling density, species richness or canopy cover within a defined time period (Holmes and Richardson 1999, Hobbs and Harris 2001). Once the effectiveness of restoration interventions is measured through the experimentation (Suding and Hobbs 2008), Hobbs and Harris (2001) recommend assessing the costs of each so that restoration options can be tabled. Restoration stakeholders may then make informed decisions with regards to what options may be implemented, to what ends and at what cost (Hobbs and Harris 2001). The strategic restoration framework facilitates a cyclical process of continual improvement where the step of 'monitoring and evaluation' measures intervention success as well as feeds through the system to re-inform the key components of the framework (Hobbs and Harris 2001).

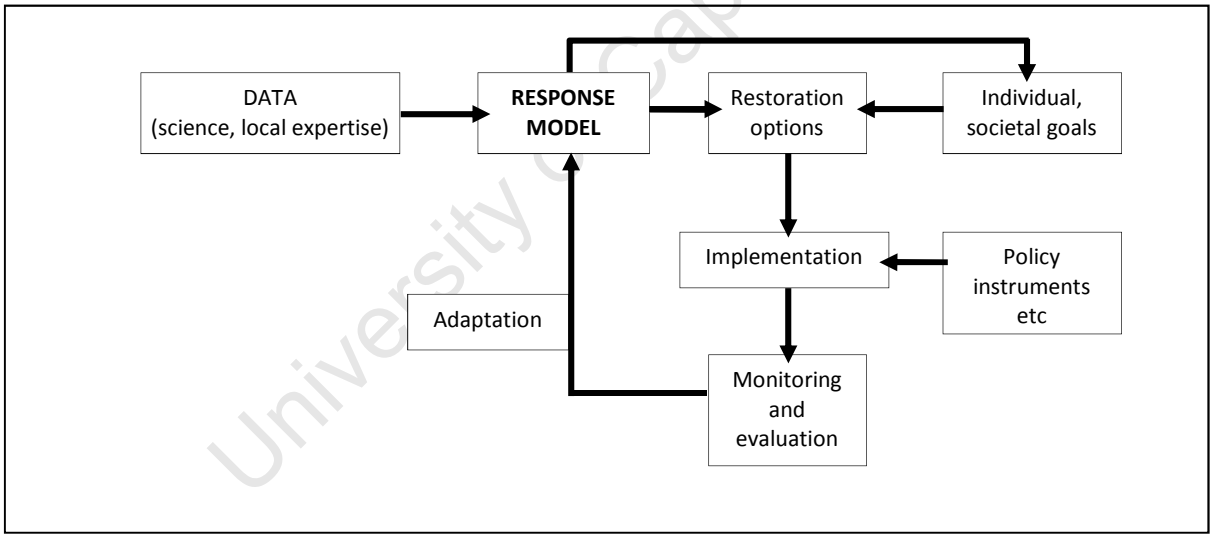

Figure 1.1: Conceptual framework for strategic ecological restoration efforts (Hobbs and Harris 2001).

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Restoration goals

Both the considerations of how far back in time one chooses to travel with respect to selecting a reference site and consequently which goals are appropriate for restoration efforts have been extensively debated in the literature (Hobbs and Harris 2001, Hobbs and Cramer 2008, Hobbs et al. 2011). It is repeatedly emphasized that goals need to be clearly defined in order to make monitoring and measurement relevant, yet to what end remains a widespread challenge (Farley and Gaddis 2007). In a broad view of ecological restoration, goals need to achieve multiple objectives including ecological, socio-economic, cultural (Young et al. 2007) and political and legislative objectives (van Andel and Aronson 2012). Attaining ecological attributes is often prioritized due to being necessary

for the realization of the other agendas (Clewell and Aronson 2013). Hobbs and Harris (2001) list typical ecosystem attributes identified in setting restoration goals which include ecosystem composition, structure, functioning, heterogeneity, ecological fidelity (compromised of structural or compositional replication, functional success and durability), system vigour, organisation and resilience. Clewell and Aronson (2013) propose 11 ecological attributes as appropriate restoration goals: landscape context, appropriate abiotic environment, species composition, community structure, ecological functionality, historic continuity, ecological complexity, self-organisation, resilience, self-sustainability and biosphere support. However, all attributes may not be appropriate to all projects or indeed feasible (Hobbs and Harris 2001) and manifestation of these attributes is likely to take many years (Clewell and Aronson 2013). Ecological restoration to what end remains unresolved, yet is central to efforts and determines restoration monitoring. Defining restoration goals in fire-driven system is even more complex where change in community attributes is frequent. Towards achieving species composition, community structure and ecosystem functioning in fynbos and fire-prone shrubland systems, Holmes and Richardson (1999) recommend monitoring goals such as certain measures of density, species richness and canopy cover for each of the functional groups for at least a two year period following the initial fire event.

1.4.2 Regional context of Peninsula Shale Renosterveld

Peninsula Shale Renosterveld occurs within the CFR (Rebelo 1995) (Figure 1.2). The CFR, located at the south western tip of Africa (Goldblatt and Manning 2002), is one of 34 internationally recognised biodiversity hotspots due to extensive habitat loss (Conservation International 2013) coupled with exceptionally high levels of floristic diversity and endemism (Goldblatt and Manning 2002, Helme and Trinder-Smith 2005). Peninsula Shale Renosterveld occurs within the West Coast Renosterveld Bioregion (Rutherford et al. 2006) which comprises the western portion of the Cape lowlands located 'to the north and east of the Cape Peninsula, (and) stretch out between the coastal forelands and inland Cape Fold Mountains' (von Hase et al. 2003a). The Cape lowlands are Critically Endangered, due to transformation resulting from predominantly agriculture, urban settlement and alien invasive species, and are declared 'totally irreplaceable' (von Hase et al. 2003a).

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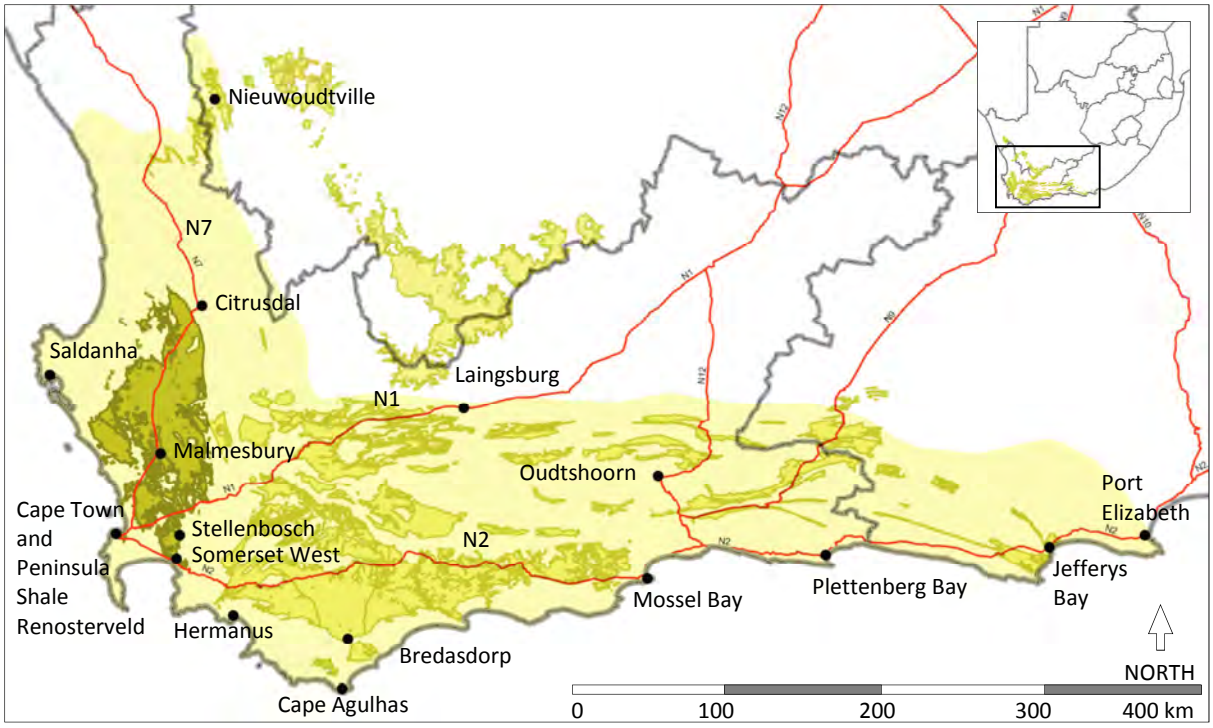

Figure 1.2: Historical distribution of the West Coast Renosterveld types and other renosterveld vegetation in relation to the CFR (adapted from Goldblatt and Manning 2002, Rebelo et al. 2006, SANBI 2013).

1.4.3 Historical influences on renosterveld

At the macro temporal-scale, West- and East Coast Renosterveld, over the past 20 000 years, are thought to have predominantly consisted of grasslands but shifted to and from open shrubland states (Krug et al. 2004, Newton and Knight 2004). Krug et al. (2004) and Newton and Knight (2004) postulate that 5 000 years ago, the advent of a distinctly warmer, drier climate marked the invasion of Asteraceous shrubs into the grasslands. Regular fire regimes (managed by the Khoekhoen herders within the last 2 000 years (Deacon 1983, Hall 1984b) and to a lesser degree by the San hunter-gatherers before them (Hall 1984b)) and the summer drought conditions in West Coast Renosterveld, promoted the proliferation of grasses and the suppression of shrubs (Newton and Knight 2004). In West Coast Renosterveld, the summer-drought conditions in combination with a high fire frequency, inhibited the success of the tropical C_4 grasses and promoted the abundance of C_3 grasses (Newton and Knight 2004). A number of other studies similarly suggest that renosterveld was historically grassier than is currently the case (Skead 1980, Cowling 1984, Rebelo 1995).

Early European accounts of the vegetation in the greater Cape Town area describe the vegetation as grassy and supporting large herds of game as well as domestic animals (Newton and Knight 2004)

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for instance 20 000 cattle and sheep were recorded in the summer of 1652 in 'Table Valley' (Skead 1980) in what would have been Peninsula Shale Renosterveld. Deacon (1983) states that large herds were in addition most likely maintained in these areas for at least 1 000 years prior to European settlement. With the establishment of the early Cape settlement, Peninsula Shale Renosterveld would have been the first ecosystem to experience transformation due to the agriculture required for the provisioning of crops and fruit (Anderson and O'Farrell 2012). By the 18th century, the majority of coastal renosterveld was being farmed (Rebelo 1995) and by the turn of the 20th century, although not supported by findings of isotope analysis by Stock et al. (1997), the literature generally reports a significant decline in grass cover and the advent of renosterbos invasion (*Dicerothamnus rhinocerotis*) (Levyns 1956, Cowling et al. 1986, Newton and Knight, 2004). An observation of Signal Hill in 1922, notes that after a fire, renosterbos (*Dicerothamnus rhinocerotis*) was distinctly dominant in the landscape (Michell 1922).

Given the extensive and rapid transformation in renosterveld, intensified by European farming coinciding with large-herbivore extinctions within 200 km from Cape Town by the early 1700s, estimating the role renosterveld played in the regional ecology of the CFR is not possible and an exercise of extrapolation (Rebelo 1995, Krug et al. 2004). Numerous authors acknowledge that the current understanding of renosterveld and its ecological processes and functions is limited (Wood and Low 1993, Rebelo 1995, von Hase et al. 2003a, Krug et al. 2004, Newton and Knight 2004, Rebelo et al. 2006, Milton 2007, Heelemann 2012). This limited understanding renders the ecological restoration of degraded areas challenging due to the absence of baseline data and reference sites to function as restoration benchmarks (Rebelo 1995, Milton 2007).

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1.4.4 Contemporary renosterveld phytosociology

Renosterveld is renowned for a high diversity in geophytes (Cowling 1983), and endemism, although marginally lower than in fynbos, is noteworthy in several families (particularly Iridaceae, Aizoaceae, Asteraceae and Fabaceae) (von Hase et al. 2003a). Renosterveld occurs on relatively fertile, clay-rich soils derived from predominantly shale, granite and silcrete (von Hase et al. 2003a, Rebelo et al. 2006). Renosterveld receives 370 mm mean annual precipitation (Rebelo et al. 2006) and the renosterveld-fynbos transition usually occurs in areas receiving over 500-800 mm whilst the renosterveld-succulent karoo shrublands transition occurs in areas usually receiving 250-300 mm (Cowling and Holmes 1992). Roughly speaking, renosterveld units west of the lower Breede River Valley receive predominantly winter rainfall whilst renosterveld units to the east receive even distribution in winter and summer (Rebelo et al. 2006).

Renosterveld is a very poorly-defined vegetation type (Milton 2007) and despite numerous descriptions, there is generally a lack of clarity (Walton 2006). Rebelo et al. (2006) have largely adopted the description by Moll et al. (1984) where renosterveld is described as an evergreen shrubland or grassland dominated by low, cuppressoid-leaved, Asteraceous shrubs with an understory composed of grasses and geophytes (Moll et al. 1984). Rebelo et al. (2006) in part define renosterveld as excluding typical fynbos elements (i.e. proteoid or ericoid components or a restioid component in excess of 5% to 10% cover) and including the tall clumps of thicket elements associated

with heuweltjies (soil mounds formed by termites). Considerable compositional variation exists in renosterveld (Milton 2007) and corresponds to the west-east rainfall gradient within the CFR with respect to quantity and temporal distribution (Rebelo et al. 2006) and this has seen the vegetation type variably sub-classified and these sub-classifications debated through time. Grasses in West Coast Renosterveld are predominantly C_3 species (such as *Ehrharta, Merxmuellera* and *Tenaxia* species) and are sparser in cover than those in East Coast Renosterveld which are predominantly C_4 species (for instance *Themeda triandra* and *Cymbopogon* species) (Moll et al. 1984, Rebelo et al. 2006). The diversity of deciduous geophytes and annuals is relatively high in West Coast Renosterveld (Rebelo et al. 2006). The over-storey canopy cover ranges from 50% to 90% and is greatest in West Coast Renosterveld (Moll et al. 1984). Sub-dominant shrub species include *Eriocephalus africanus* and *Leysera gnaphaloides* (Moll et al. 1984).

Renosterveld vegetation types are collectively the most transformed yet under-conserved areas within the CFR (Rouget et al. 2003a). Traditional conservation efforts overlooked the CFR lowlands (Rouget et al. 2003b) and their dire conservation status became recognized and documented from the 1980s (Jarman 1986, Hall 1984a, Burgers 1994). In 1982, 85% of all coastal renosterveld had been transformed (Hall 1984a), this estimation recently increasing to over 90% transformation (Rebelo et al. 2006). In 2003, less than 1% of these renosterveld lowlands was formally protected (Von Hase et al. 2003b). For West Coast Renosterveld specifically, it was estimated that 5% remained with only 0.28% formally protected (von Hase et al. 2003a). Broader conservation efforts for the CFR (Cowling et al. 1999) have identified renosterveld conservation as a priority and, further to reserve-based conservation, off-reserve efforts have become increasingly important (Gelderblom et al. 2003, CAPE 2013). The Biodiversity Stewardship Programme (with numerous non-statutory reserves and conservancies) and other initiatives (such as Conservation at Work and the Overberg Lowlands Conservation Trust) continue to play a critical role in the securing irreplaceable biodiversity.

Recovery in West Coast Renosterveld is slow where after approximately 35 years in old fields, recovered areas were comparable to natural vegetation with respect to structure and cover but species richness was found to be considerably lower due to a loss in predominantly geophytes (Walton 2006). Recovery of indigenous plant species, despite proximity to natural areas, was similarly found not to occur for decades due to limitations in seed dispersal and seedling recruitment (Shiponeni 2003). Restoration potential of renosterveld old fields has been found to be lower than that of pine plantations (Heelemann 2012) as confirmed by a seed-bank survey of the game camp in Groote Schuur Estate which found a dominance of alien invasive species seed (Cowan 2013). An estimated 1% of former renosterveld is thought to be composed of old fields and pine plantations and due to the challenges associated with recovery in these areas (Heelemann 2012), focusing restoration research and implementation in these remnants will be central to meeting conservation-target shortfalls (Cowling et al. 1999, Krug and Krug 2007).

1.4.5 Peninsula Shale Renosterveld phytosociology

The historical distribution of Peninsula Shale Renosterveld predominantly coincides with the Cape Town city bowl from the coastline to 350 m above mean sea level (SANBI and DEAT 2009).

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Remnants still occur in parts of the historical distribution area including Signal Hill and the lower, northern slopes of Table Mountain and Devil's Peak including within Groote Schuur Estate (Joubert and Moll 1992, Rebelo et al. 2006) (Figure 1.3).

Despite a lack of pre-transformation phytosociological data, there are several key sources mapping and describing West Coast Renosterveld and Peninsula Shale Renosterveld including Acocks (1988), Low and Rebelo (1998) and Rebelo et al. (2006). From the early 1980s onwards, Boucher conducted a number of studies of the West Coast foreland vegetation (for example Boucher 1983). Addressing the phytosociology of Peninsula Shale Renosterveld specifically, Joubert (1991) conducted a study of the vegetation on Signal Hill. Euston-Brown (2011) conducted fine-scale vegetation survey for Groote Schuur Estate and recently Cowan (2013) studied four renosterveld vegetation states occurring within the Estate.

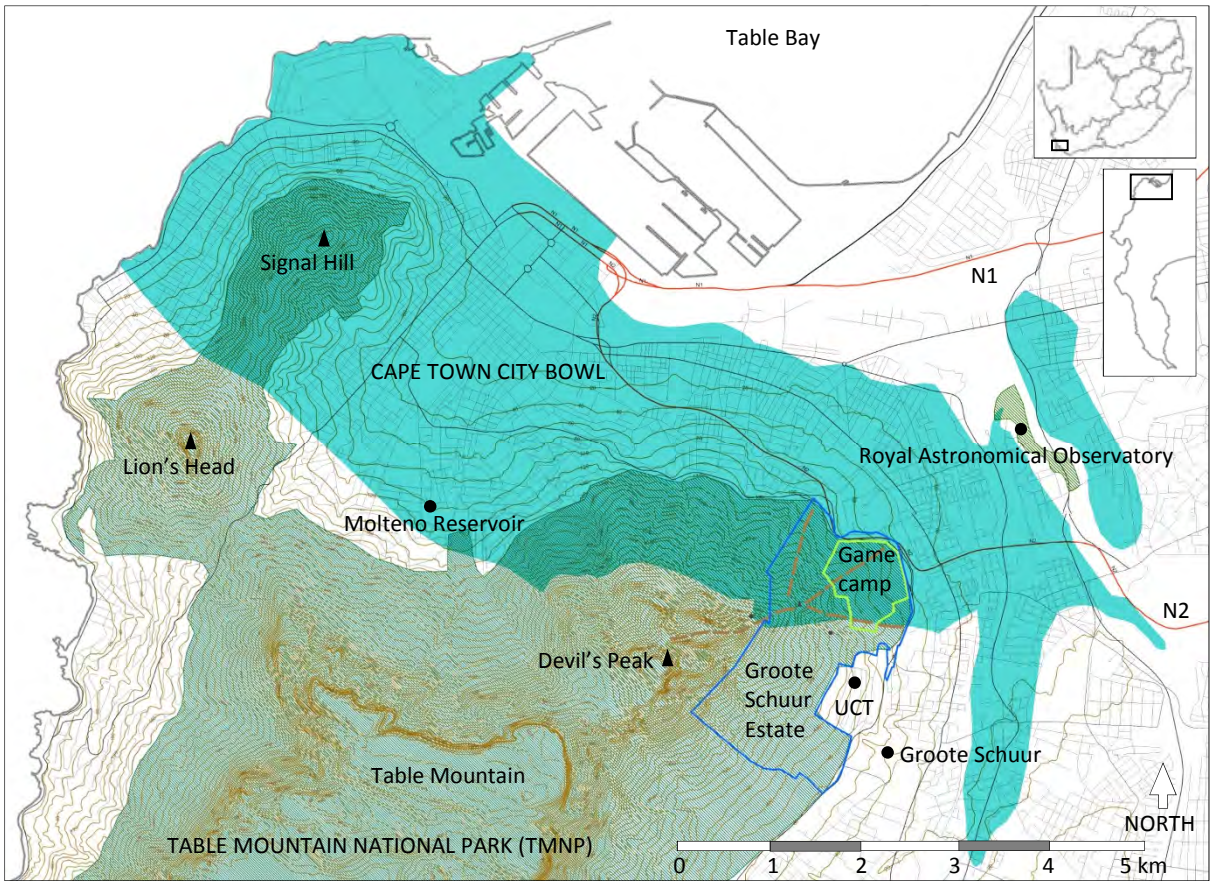

Figure 1.3: Historical distribution of Peninsula Shale Renosterveld predominantly coinciding with the Cape Town city bowl.

Rebelo et al. (2006) describe Peninsula Shale Renosterveld as a tall open shrubland and grassland occurring on gentle to steep gradients where renosterbos is not very common and frequent fires have promoted grass cover. Tall stands of *Searsia* species are dominant in the fire-protected areas of Signal Hill (Britton and Jackelman 1995, Rebelo et al. 2006). The prominent vegetation guilds include small trees or tall shrubs (thicket species), low shrubs, succulents, herbs, geophytic herbs and graminoids (Rebelo et al. 2006). Joubert (1991) identified one major floristic community over the whole Signal Hill study area (*Searsia-lucida–Chrysocoma coma-aurea*), and two sub-communities (*Hyparrhenia hirta–Searsia glauca* and *Cliffortia polygonifolia–Searsia tomentosa*) with a further three and two variants respectively (Joubert 1991, Joubert and Moll 1992).

Accounts of post-fire succession in Peninsula Shale Renosterveld, for Signal Hill and Groote Schuur Estate, have been recorded by Mitchell (1922) and by Britton and Jackelman (1995). On Signal Hill, after a long period of invasion by annual alien grasses (1950s-1960s), followed by a long fire-exclusion period of approximately 25 years, the vegetation recovery after an unplanned fire in the autumn of 1993 followed a very similar recovery sequence as that described by Mitchell (1922) almost 75 years earlier for the same area (Britton and Jackelman 1995). Recovery after a fire on Signal Hill in 1993 followed much the same recovery as that on Devil's Peak in February of 1991. Immediately after the Signal Hill fire in 1993, *Asparagus capensis* (low shrub), *Hyparrhenia hirta* (graminoid), *Haemanthus sanguineus* (geophyte), *Gladiolus priorii* (geophyte) and numerous *Oxalis* species (geophyte) resprouted and flowered within the first three months (Britton and Jackelman 1995). Many other shrubs resprouted within this period most notably *Searsia lucida* (tall shrub) (Britton and Jackelman 1995). In the winter and spring flowering phase (approximately six to nine months since the fire) many of the grasses and geophytes flowered and seedlings of the later seral stages were observable (Britton and Jackelman 1995).

Peninsula Shale Renosterveld receives distinctly more rainfall than other renosterveld types with a range of 480 - 870 mm (720 mm mean annual precipitation) (Rebelo et al. 2006). The average daily minimum temperature for July is 7.8°C (with an annual incidence of two to three frost days) whilst the daily maximum temperature for February is 26.7°C (Rebelo et al. 2006).

Peninsula Shale Renosterveld predominantly occurs on Glenrosa, Mispah and Lamotte soil forms derived from the Malmesbury Group of the Tygerberg Formation (Rebelo et al. 2006). The City identifies two vegetation sub-types on the basis of substrate: recent non-aeolian colluvium (hill-wash occurring in depressions in between ridgelines derived of quaternary sands) and shale (exposed Malmesbury Group predominantly occurring on ridgelines) (Theron 1984, City of Cape Town 2008, Cowan 2013).

Approximately 87% of Peninsula Shale Renosterveld, coinciding with the City's footprint, is entirely transformed rendering the conservation target of 26% unachievable (City of Cape Town 2008, Rebelo et al. 2011). Of the remaining 13%, 11% is formally protected within TMNP (Rebelo et al. 2011), yet portions of the remnants are degraded predominantly due to alien vegetation and frequent fires (Rebelo et al. 2006) and land-use legacies (Cowan 2013). Approximately 27 hectares are in a degraded condition and require restoration (City of Cape Town 2008).

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1.4.6 Groote Schuur Estate

The portion of the Peninsula Shale Renosterveld remnant occurring within Groote Schuur Estate is situated in its northern extent. Subsequent to a fine-scale, in-field assessment of the vegetation in Groote Schuur Estate, two units of terrestrial renosterveld types were identified: Mesic Renosterveld and Dry Renosterveld (Euston-Brown 2011). The former, occurring at higher elevations, coincides with the renosterveld-shale fynbos transition whilst the latter is located on the lower, dry slopes (Euston-Brown 2011).

The land-use of the Estate is historically and culturally complex where the landscape has been anthropogenically transformed for over 2,000 years (CNdV 2000). Records as early as 1600s document Khoi pastoralists on the eastern slopes of Table Mountain practicing burning and grazing regimes (CNdV 2000). The intensity and diversity of land-uses escalated with European settlement and although the records of land-uses are not detailed, large parts of the Estate were farmed and afforested (CNdV 2000). In the 1700s Stone Pines (*Pinus pinea*) were extensively established on the Table Mountain slopes and from 1891-1899 Rhodes started consolidating farms in the area and thereafter set about transforming the landscape into open parkland (CNdV 2000).

The game camp precinct within Groote Schuur Estate is situated to the north east and is of particular relevance to this study as the field experimentation is sited here (Figure 1.3). The grassy paddocks, animals and Stone Pines (*Pinus pinea*) of the game camp occupy a visually and culturally significant position in the landscape as the gateway, via Mowbray Ridge, to and from the Cape Town city bowl (CNdV 2000). Topographically, the game camp is defined by Mowbray Ridge, which runs in a north east direction more or less from the King's Blockhouse through Plumpudding Hill and to the bottom of Hospital Bend. The ridgeline divides the game camp and the field experimentation is situated to its south. The gradients of these lower game camp slopes are more gradual than those of the upper slopes in the Estate. Although not definitive, through assessment of historical aerial photography, the game camp remained largely un-forested but was, in all likelihood, ploughed over 100 years ago (Cowan 2013) and today exhibits the characteristics of old lands (Euston-Brown 2011). The invasive annual alien grasses dominating the camp were most likely originally seeded as fodder (some may have been horticultural escapees) and have persisted in part due to management practices (Cowan 2013). It is possible that the recent translocation of the majority of grazing animals from the game camp, retaining just five animals of the Quagga breeding programme, has resulted in a reduction in grazing pressure and allowed the relatively unchecked proliferation of the dominant annual grass species (Cowan 2013). Vegetation recovery of Peninsula Shale Renosterveld following alien clearance of *Pinus* and *Eucalyptus* species was found to be slow and poor (with respect to species diversity, composition and cover) and insufficient to meet conservation targets (Terblanche 2011). The recent phytosociological study within Groote Schuur Estate found low levels of diversity in general, the game camp being particularly poor as invasive species, predominantly annual alien grasses, dominate above-ground cover as well as the seed bank and renosterveld elements are almost absent (Cowan 2013).

In closing, the ecosystem status of Peninsula Shale Renosterveld is Critically Endangered (DEAT and SANBI 2009) and there is a need to implement ecological restoration to redress its conservation-

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target shortfall (Rebelo et al. 2006). Despite formal protection, areas within the ecosystem are degraded and require restoration intervention (Terblanche 2011, Cowan 2013). The literature review of restoration ecology theory indicates that determining an ecological-response model for an ecosystem is a central element to drafting well-placed restoration strategies (Hobbs and Harris 2001). However, the understanding of the pre-disturbance ecology and ecological drivers of renosterveld is limited (Rebelo 1995, Krug et al. 2004), the body of renosterveld research is small (albeit growing) and seed-based renosterveld restoration efforts to date have achieved limited success. These factors reinforce to the need to conduct further research in Peninsula Shale Renosterveld, through experimentation designed to test the ecological responses to restoration interventions. Further to being relevant to the restoration and thus the conservation status of Peninsula Shale Renosterveld, the findings of this study are potentially relevant to other closely associated renosterveld types and thus support the broader, biodiversity conservation agenda of the CFR.

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1.5 References

Acocks, J.P.H. 1988. *Veld types of South Africa*. Pretoria: Botanical Research Institute.

Anderson, P.M.L and O'Farrell, P.J. 2012. An ecological view of the history of the city of Cape Town. *Ecology and Society*, 17(3), pp. 28-40.

Aronson, J., Milton, S. and Blignaut, J. 2007. Restoring Natural Capital: Definition and Rationale. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice*. Washington: Island Press, pp. 3-8.

Boucher, C. 1983. Floristic and structural features of the coastal foreland vegetation south of the Berg River, western Cape Province, South Africa. *Bothalia*, 14(3&4), pp. 669 – 674.

Britton, P. and Jackelman, J. 1995. Biological Management of Renosterveld: Some questions and lessons from Signal Hill and Devil's Peak. In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region.* Proceedings of a symposium. FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp.58-62.

Brudvig, L. A. 2011. The Restoration of Biodiversity: Where has research been and where does it need to go? *American Journal of Botany*, 98, pp. 549-558.

Burgers, C. J. 1994. *Report on progress with implementation of recommendations of the 'Jarman Report' on the conservation of the coastal lowlands of the Western and South-Western Cape*. Cape Town: Cape Nature Conservation.

Cape Action for People and the Environment (CAPE). 2013. *Draft CAPE Strategy: 2011-2020*. Available online: http://www.capeaction.org.za/index.php/resources/coordinated-action?view=category&id=36.

Chittenden Nicks de Villiers (CNdV). 2000. *Groote Schuur Estate: Conservation and Development Framework, Phase 1: Analysis.* Prepared for South African National Parks, Cape Town.

City of Cape Town. 2008. *City of Cape Town BioNet Terrestrial Systematic Conservation Plan Re-Analysis: Methods and Results.* Cape Town: City of Cape Town.

Clewell, A.F and Aronson, J. 2013. *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession, second edition.* Washington: Island Press.

Conservation International. 2013. *Hotspots revisited*. Available online: http://www.conservation.org/.

Cowan, O. S. 2013. *The Peninsula Shale Renosterveld of Devil's Peak: Phytosociology, System Drivers and Restoration Potential*. MSc thesis. University of Cape Town.

Cowling , R. M. 1984. A syntaxonomic and synecological study in the Humansdorp region in the Fynbos Biome. *Bothalia,* 15, pp. 175-227.

Cowling, R. M., Pierce, S. M. and Moll, E. J. 1986. Conservation and utilisation of South Coast Renosterveld, and endangered South African Vegetation Type. *Conservation Biology*, 37, pp. 363-377.

Cowling, R.M. and Holmes, P.M. 1992. Flora and Vegetation. In: R.M. Cowling, (ed). *The Ecology of Fynbos: nutrients, fire and diversity.* Cape Town: Oxford University Press, pp. 23-61.

Cowling, R.M., 1983. Diversity relations in Cape shrublands and other vegetation in the southeastern Cape, South Africa. *Vegetatio*, 54, pp. 103–127.

Cowling, R.M., Heijnis, C.E. 2001. The identification of broad habitat units as biodiversity entities for a systematic conservation planning in the Cape Floristic Region. *South African Journal of Botany*, 67, pp. 15–38.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Heijnis, C.E., Richardson, D.M. and Cole, N. 1999. *Framework for a conservation plan for the Cape Floristic region, South Africa.* IPC Report 0001 prepared for WWF, South Africa.

University of Cape Town

Cowling, R.M., Pressey, R.L., Rouget, M. and Lombard, A.T. 2003. *A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa.* Biological Conservation, 112, pp. 191-216.

Deacon, H.J. 1983. The peopling of the fynbos region. In: H.J. Deacon, O.B. Hendey and J.J.N. Lambrechts, (eds). 1983. *Fynbos palaeoecology: A preliminary synthesis*. South African National Science Programme, Report 75, Pretoria: CSIR, pp. 183-204.

Euston-Brown, D. 2011. *Description of natural vegetation on Groote Schuur Estate, Cape Town*. University of Cape Town, unpublished report.

Farley, J. and Gaddis, E. 2007. Restoring Natural Capital: An Ecological Economics Assessment. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 17-27.

Gaertner, M., Holmes, P.M. and Richardson, D. 2012. Biological Invasions, Resilience and Restoration. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.265-280.

Gelderblom, C.M., van Wilgen, B.W., Nel, J., Sandwith, T., Botha, M., Hauck, M., 2003. Turning strategy into action: implementing a conservation action plan in the Cape Floristic Region. *Biological Conservation*, 112, pp. 291–297.

Goldblatt, P. and Manning, J.C. 2002. Plant diversity of the cape region of Southern Africa. *Annals of the Missouri Botanical Garden,* 89(2), pp. 281-302.

Grime, J.P. 1979. *Plant Strategies and Vegetation Processes*. Chichester: Wiley.

Hall, A.V. 1984a. *Proposals for nature conservation areas in the coastal lowlands of the south-western Cape (The 'Fenn report').* Report prepared for the Minister of Environment Affairs, the Honourable Mr. J.W.E. Wiley. October 1984.

Hall, M. 1984b. The burden of tribalism: the social context of Southern African Iron Age studies. *American Antiquity*, 49(3), pp. 455-467.

Heelemann 2012. *Renosterveld remnants- Current ecological situation and restoration perspectives*. PhD thesis, University of Stellenbosch.

Helme, N. and Trinder-Smith, T.H. 2005. The endemic flora of the Cape Peninsula, South Africa. *South African Journal of Botany,* 72, pp. 205-210.

Hilderbrand, R.H., Watts, A.C. and Randle, A.M. 2005. The Myths of Restoration Ecology. *Ecology and Society*, 10(1): 19.

Hobbs, R. and Harris, J. 2001. Restoration ecology: Repairing the Earth's ecosystems in the new millennium. *Restoration Ecology*, 9(2), pp. 239-246.

Hobbs, R.J, Hallet, L.M., Ehrlich, P.R. and Mooney, H.A. 2011. Intervention Ecology: Applying Ecological Science in the Twenty-first Century. *BioScience,* 61(6), pp. 442, 450.

Hobbs, R.J. and Cramer, V.A. 2008. Restoration Ecology: Interventionist Approaches for Restoring and Maintaining Ecosystem Function in the Face of Rapid Environmental Change. *Annual Reviews Environmental Resources,* 33, pp. 39-61.

Hobbs, R.J. and Suding, K.N. 2009. Synthesis: Are New Models for Ecosystem Dynamics Scientifically robust and Helpful in Guiding Restoration Projects? In: R.J. Hobbs and K.N. Suding. (eds). *New Models for Ecosystem Dynamics and Restoration.* Washington: Island Press, pp. 325-334.

Holl. K., Pejchar, L. and Whisenant, S. 2007. Overcoming Physical and Biological Obstacles to Restore Natural Capital. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 249-255.

Holmes, P. M. 2002. *Renosterveld Restoration on Cultivated Lands at Eerste River, Western Cape: Preliminary Findings of a Pilot Study*, unpublished report to Lafarge Quarries, May 2002.

University of Cape Town

Holmes, P. M. and Richardson, D. M. 1999. Protocols for restoration based on Recruitment Dynamics, Community Structure, and Ecosystem Function: Perspectives from South African Fynbos. *Restoration Ecology*, 7(3), pp.215-230.

Jarman, M.L. 1986. *Conservation Priorities in Lowland Regions of the Fynbos Biome*. Pretoria: Foundation for Research Development CSIR.

Joubert, C. 1991. *History and description of contemporary vegetation, Signal Hill, Cape Town*. MSc thesis, University of Cape Town.

Joubert, C. and Moll, E. J. 1992. A phytosociological study of Signal Hill, Cape Town, utilizing both perennial and ephemeral species. *Bothalia*, 22, pp. 255-282.

King, E.G. and Hobbs, R.J. 2006. Identifying linkages among conceptual models of ecosystem degradation and restoration: towards and integrative framework. *Restoration Ecology*, 14, pp. 369-378.

Krug, C. B. and R. M. Krug. 2007. Restoration in Old Fields in Renosterveld: A Case Study in a Mediterranean-type Shrubland of South Africa. In: V. A. Cramer and R. J. Hobbs, (eds). *Old fields: Dynamics and restoration of abandoned farmland.* Washington: Island Press, pp. 265-285.

Krug, C. B., R. M. Krug, D. Midoko-Iponga, B. A. Walton, S. J. Milton, and N. N. Shiponeni. 2004. *Restoration of west coast renosterveld: facilitating the return of a highly threatened vegetation type*. In: Proceedings of the 10th international conference on Mediterranean climate ecosystems, Rhodes, Rhodos, Greece, pp. 107-119.

Levyns, M.R. 1956. Notes on the biology and distribution of the Rhenoster bush. *South African Journal of Science,* 52, pp. 141-143.

Lippitt, L., Fidelibus, M. W. and Bainbridge, D. A. 1994. Native Seed Collection, Processing, and Storage for Revegetation Projects in the Western United States. *Restoration Ecology*, 2(2), pp. 120-131.

Lochner, P., Weaver, A., Gelderblom, C., Paert, R., Sandwith, T., Fowkes, S., 2003. Aligning the diverse: the development of a biodiversity conservation strategy for the Cape Floristic Region. *Biological Conservation*, 112, pp. 29–43.

Low, A.B. and Rebelo, A.G, (eds). 1998. *Vegetation of South Africa, Lesotho and Swaziland*. Pretoria: Department of Environmental Affairs and Tourism.

McGhee, W., Craig, J., de Groot, R., Miller, J.S. and Bowers, K. 2007.Making Restoration Work: Nonmonetary Mechanisms. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 291-302.

Memiaghe, H. R. 2008. *Old field restoration: vegetation response to soil changes and restoration efforts in Western Cape Lowlands*. MSc thesis. University of Stellenbosch.

Michell, M.R. 1922. Some observations on the effects of a bush fire on the vegetation of Signal Hill. *Transactions of the Royal Society of South Africa,* 10, pp. 213-232.

Midoko-Iponga, D. 2004. *Renosterveld Restoration: The role of competition, herbivory and other disturbances*. MSc thesis, University of Stellenbosch.

Millennium Ecosystem Assessment (MA). 2005. *Ecosystems and Human Well-being: General Synthesis*. Washington: Island Press.

Milton, S. J. 2007. *Rapid Assessment Method for Renosterveld*. Renosterveld Assessment Workshop, unpublished report.

Milton, S. J., W. R. J. Dean, M. A. du Plessis and W. R. Siegfried. 1994. A Conceptual Model of Arid Rangeland Degradation: The Escalating Cost of Declining Productivity. *BioScience*, 44, pp. 70–76.

Moll, E. J., B. M. Campbell, R. M. Cowling, L. Bossi, M. L. Jarman, and C. Boucher. 1984. *Description of major vegetation categories in and adjacent to the Fynbos biome.* Report no. 84, Cape Town: CSIR.

University of Cape Town

Newton, I.P. and Knight, R.S. 2004. The Matrix. The evolutionary development of coastal renosterveld. *Veld and Flora,* 90(4), pp. 162-165.

Palmer, M. A., Ambrose, R. F. and LeRoy Poff, N. 1997. Ecological Theory and Community Restoration Ecology. *Restoration Ecology*, 5(4), pp. 291-300.

Perring, M.P., Standish, R.J. and Hobbs, R.J. 2013. Incorporating novelty and novel ecosystems into restoration planning and practice in the 21st century. *Ecological processes*, 2(18), pp. 1-8.

Pickett, S. T. A., and P. S. White. 1985. *The Ecology of Natural Disturbance and Patch Dynamics*. San Diego: Academic Press,

Pressey, R.L., Cowling, R.M., Rouget, M., 2003. Formulation of conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, 112, pp. 99–127.

Prober, S.M., Lunt, I.D. and Morgan, J.W. 2009. Rapid internal plant-soil Feedbacks Lead to Alternative stable states in Temperate Australian Grassy Woodlands. In: R.J. Hobbs and K.N. Suding. (eds). *New Models for Ecosystem Dynamics and Restoration*. Washington: Island Press, pp. 156-168.

Rebelo, A. G. 1995. Renosterveld: Conservation and Research. In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region*. Proceedings of a symposium. FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp: 32-42.

Rebelo, A. G. 1998. *Renosterveld*. In: Low A. B. and Rebelo A. G. (eds) Vegetation of South Africa, Lesotho and Swaziland. Department of Environmental Affairs and Tourism, Pretoria, pp.63-67.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. and Rutherford, M, C. 2006. Fynbos Biome. In: L. Mucina, L. and M.C. Rutherford, (eds). *The Vegetation of South Africa, Lesotho and Swaziland*. Pretoria: South African National Biodiversity Institute.

Rebelo, A.G., Holmes, P.M., Dorse, C. and Wood, J. 2011. Impacts of urbanization in a biodiversity hotspot: conservation challenges in metropolitan Cape Town. *South African Journal of Botany*, 77(1), pp. 20-35.

Rouget, M., D. M. Richardson, R. M. and Cowling. 2003b. The current configuration of protected areas in the Cape Floristic Region, South Africa—reservation bias and representation of biodiversity patterns and processes. *Biological Conservation*, 112, pp. 129-145.

Rouget, M., D. M. Richardson, R. M. Cowling, J. W. Lloyd, and A. T. Lombard. 2003a. Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. *Biological Conservation*, 112, pp. 63-85.

Rouget, M., Jonas, Z., Cowling, R.M., Desmet, P.G., Driver, A., Mohamed, B., Mucina, L., Rutherford, M.C. and Powrie, L.W. 2006. Ecosystem Status and Protection Levels of Vegetation Types. In: Mucina, L. and Rutherford, M.C. (eds). 2006. *The Vegetation of South Africa, Lesotho and Swaziland*. Pretoria: South African National Biodiversity Institute.

Rutherford, M, C, Mucina, L. and Powrie, L.W. 2006. Biomes and Bioregions of Southern Africa. In: Mucina, L. and Rutherford, M.C. (eds). 2006. *The Vegetation of South Africa, Lesotho and Swaziland*. Pretoria: South African National Biodiversity Institute.

SANBI and DEAT. 2009. *Threatened Ecosystems in South Africa: Descriptions and Maps*. DRAFT for Comment. Pretoria: South African National Biodiversity Institute.

Seastedt, T.R., Hobbs, R.J. and Suding, K.N. 2008. Management of Novel Ecosystems: Are Novel Approaches Required? *Frontiers in Ecology and the Environment*, 6(10), pp. 547-553.

Shiponeni, N.N. 2003. *Dispersal of seeds as a constraint of revegetation of old fields in renosterveld vegetation in the Western Cape, South Africa*. MSc thesis, University of Stellenbosch.

Skead, C.J. 1980. *Historical mammal incidence in Cape Province. The western and northern Cape*. Cape Town: The Department of Nature and Environmental Conservation, Cape Provincial Administration.

University of Cape Town

Society for Ecological Restoration International Science and Policy Working Group (SER). 2004. *The SER International Primer on Ecological Restoration*. Tucson: Society for Ecological Restoration International.

South African National Biodiversity Institute (SANBI). 2011. *CAPE.* [online] Available at: <http://www.capeaction.org.za/> Accessed 24 July 2013.

South African National Biodiversity Institute (SANBI). 2013. *BGIS.* [online] Available at: <http://www.plantzafrica.com> Accessed 13 May 2012.

Stock, W.D., Bond, W.J. and le Roux, D. 1993. Isotope evidence from soil carbon to reconstruct vegetation history in the south-western Cape Province. *South African Journal of Science*, 89, pp.153-154.

Suding, K.N., and Hobbs, R.J. 2008. Threshold models in restoration and conservation: a developing framework. *Trends in Ecology and Evolution*, 24(5), pp. 271-279.

Terblanche, K. 2011. *Meeting conservation goals: passive restoration of peninsula shale renosterveld following the clearance of alien invasive forestry species*. BSc Hons thesis, University of Cape Town.

The Economics of Ecosystems and Biodiversity (TEEB). 2010. In Pushpam Kumar (ed) 2010. *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations.* London: Earthscan.

Theron, J.N. 1984. *The Geology of Cape Town and Environs*. Explanations if 1:50000 sheets 3318CD and DC and 3418AB, AD and BA. Pretoria: Geological Survey of South Africa.

Van Andel, J. and Aronson, J. 2012. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell.

Van Andel, J., Grootjans, A.P. and Aronson, J. 2012. Unifying Concepts. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.9-22.

Von Hase, A., M. Rouget, K. Maze, and N. A. Helme. 2003a. *A fine-scale conservation plan for Cape Lowlands Renosterveld.* Full Report. Cape Town: Conservation Unit. Botanical Society of SA.

Von Hase, A., M. Rouget, K. Maze, and N. A. Helme. 2003b. *A fine-scale conservation plan for Cape Lowlands Renosterveld.* Summary Report. Cape Town: Conservation Unit. Botanical Society of SA.

Walton, B. J. 2006. *Vegetation patterns and dynamics of renosterveld at Agter-Groeneberg Conservancy, Western Cape, South Africa.* MSc thesis. University of Stellenbosch.

Waters, C., Young, A.G., and Crosthwaite, J. 2007. Genetic Integrity as a Target for Natural Capital Restoration: Weighing the Costs and Benefits. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 85-93.

Whisenant, S. G. 1999. *Repairing damaged wildlands.* Cambridge: Cambridge University Press.

Whisenant, S.G. 2002. Terrestrial systems. In: M.R. Perrow and A.J. Davy, (eds). 2002. *Handbook of Ecological Restoration*. Cambridge: Cambridge University Press, pp. 83-105.

White, P. S., and A. Jentsch. 2004. Disturbance, Succession, and Community Assembly in Terrestrial Plant Communities. In: V. M. Temperton, R. Hobbs, T. Nuttle, and S. Halle (eds). *Assembly Rules and Restoration Ecology: Bridging the Gap between Theory and Practice.* Washington: Island Press, pp. 342–66.

Wood, J. and A. B. Low. 1993. *Environmental survey and management guidelines for the Tygerberg and environs: Part two - Management guidelines.* Kirstenbosch: National Botanical Institute.

Young, M, Hajkowicz, S., Gaddis, E. and de Groot, R. 2007. A Decision-Analysis Framework for Proposal Evaluation Natural Capital. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 237-248.

Younge, A. and Fowkes, S., 2003. The Cape Action Plan for the Environment: overview of an ecoregional planning process. *Biological Conservation*, 112, pp. 15–28.

Zobel, M. 1997. The relative role of species pool in determining plant species richness: an alternative explanation of coexistence? *Trends in Ecology and Evolution*, 12, pp. 266-269.

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Appendix 1.1: Final list of species collected

Table 1.1: Final list of the 31 species collected.

SPECIES	FAMILY	GROWTH FORM	STATUS
Arctopus echinatus L.	APIACEAE	Forb	LC
Athanasia crithmifolia (L.) L.	ASTERACEAE	Tall shrub	LC
Babiana disticha= B. fragrans	IRIDACEAE	Geophyte	NT
Chironia baccifera L.	GENTIANACEAE	Low shrub	LC
Chrysocoma coma-aurea L.	ASTERACEAE	Low shrub	LC
Cymbopogon marginatus (Steud.) Stapf ex Burtt Davy	POACEAE	Graminoid	LC
Dimorphotheca pluvialis (L.) Moench	ASTERACEAE	Forb (annual)	LC
Ehrharta calycina Sm.	POACEAE	Graminoid	LC
Erepsia anceps (Haw.) Schwantes	MESEMBRYANTHEMACEAE	Succulent shrub	LC
Ericephalus africanus var africanus	ASTERACEAE	Low shrub	LC
Felicia filifolia (Vent.) Burtt Davy subsp. *filifolia*	ASTERACEAE	Low shrub	LC
Helichrysum cymosum (L.) D.Don subsp. *cymosum*	ASTERACEAE	Low shrub	LC
Helichrysum patulum (L.) D.Don	ASTERACEAE	Low shrub	LC
Hermannia hyssopifolia L	MALVACEAE	Low shrub	LC
Lachenalia fistulosa Baker	HYACINTHACEAE	Geophyte	LC
Lampranthus emarginatus (L.) N.E.Br.	MESEMBRYANTHEMACEAE	Succulent shrub	LC
Tenaxia stricta (Schrad.) = Merxmuellera stricta (Schrad.) Conert	POACEAE	Graminoid	LC
Moraea bellendenii (Sweet) N.E.Br.	IRIDACEAE	Geophyte	LC
Myrsine africana L.	MYRSINACEAE	Tall shrub	LC
Ornithogalum thyrsoides Jacq.	HYACINTHACEAE	Geophyte	LC
Othonna arborescens L.	ASTERACEAE	Succulent shrub	LC
Pelargonium cucullatum (L.) L'Hér. subsp. *tabulare Volschenk*	GERANIACEAE	Low shrub	LC
Pentaschistis airoides (Nees) Stapf subsp. *airoides*	POACEAE	Graminoid (annual)	LC
Podalyria sericea (Andrews) R.Br. Ex Aiton f.	FABACEAE	Low shrub	VU
Ruschia rubricaulis (Haw.) L.Bolus	MESEMBRYANTHEMACEAE	Succulent shrub	VU
Salvia africana-caerulea L.	LAMIACEAE	Low shrub	LC
Searsia laevigata (L.) F.A.Barkley var. villosa (L.f.) Moffett	ANACARDIACEAE	Tall shrub	LC
Searsia tomentosa (L.) F.A.Barkley	ANACARDIACEAE	Tall shrub	LC
Themeda triandra Forssk.	POACEAE	Graminoid	LC
Trachyandra muricata (L.f.) Kunth	ASPHODELACEAE	Geophyte	LC
*Ursinia anthemoides (L.) Poir.*subsp. *anthemoides*	ASTERACEAE	Forb (annual)	LC

Appendix 1.2: List of existing indigenous species (sampled)

Table 1.2: List of existing indigenous species sampled.

SPECIES	FAMILY	GROWTH FORM	STATUS
Cotula turbinata L.	ASTERACEAE	Forb (annual)	LC
Cynodon dactylon (L.) Pers.	POACEAE	Graminoid	LC
Ficinia sp. unknown	CYPERACEAE	Graminoid	na
Gladiolus watsonius Thunb.	IRIDACEAE	Geophyte	NT
Moraea collina Thunb.	IRIDACEAE	Geophyte	LC
Otholobium virgatum (Burm.f.) C.H.Stirt.	FABACEAE	Dwarf shrub	LC
Oxalis sp. unknown	OXALIDACEAE	Geophyte	na
Oxalis obtusa Jacq.	OXALIDACEAE	Geophyte	LC
Oxalis pes-caprae L. var. pes-caprae	OXALIDACEAE	Geophyte	LC
Oxalis purpurea L.	OXALIDACEAE	Geophyte	LC
Oxalis tenuifolia Jacq.	OXALIDACEAE	Geophyte	LC
Oxalis tomentosa L.f.	OXALIDACEAE	Geophyte	LC
Romulea rosea	IRIDACEAE	Geophyte	LC
Senecio pubigerus L.	ASTERACEAE	Low shrub	LC
Zantedeschia aethiopica (L.) Spreng.	ARACEAE	Geophyte	LC

Appendix 1.3: List of existing indigenous species (not sampled)

Table 1.3: List of existing indigenous species: observed in the burnt and unburnt blocks but not sampled.

SPECIES	FAMILY	GROWTH FORM	STATUS
Baeometra uniflora (Jacq.) G.J.Lewis	COLCHICACEAE	Geophyte	LC
Geissorhiza aspera Goldblatt	IRIDACEAE	Geophyte	LC
Gnidia sp. unknown	THYMELAEACEAE	Low shrub	na
Hyparrhenia hirta (L.) Stapf	POACEAE	Graminoid	LC
Lobelia erinus L.	LOBELIACEAE	Forb	LC
Moraea vegeta L.	IRIDACEAE	Geophyte	LC
Senecio pterophorus DC.	ASTERACEAE	Low shrub	LC

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Appendix 1.4: List of weed species

Table 1.4: List of weed species occurring within the field-experiment area.

SPECIES	FAMILY	GROWTH FORM	STATUS
Anagallis arvensis L. subsp. *arvensis*	PRIMULACEAE	Forb (annual)	Naturalised exotic
Avena fatua L.	POACEAE	Graminoid (annual)	Naturalised exotic
Brachypodium distachyon (L.) P.Beauv	POACEAE	Graminoid (annual)	Naturalised exotic
Briza maxima L.	POACEAE	Graminoid (annual)	Naturalised exotic
Briza minor L.	POACEAE	Graminoid (annual)	Naturalised exotic
Bromus diandrus Roth	POACEAE	Graminoid (annual)	Naturalised exotic
Panicum stapfianum Fourc.	POACEAE	Graminoid (annual)	LC, weedy, introduced from NE SA.
Pennisetum clandestinum Hochst. ex Chiov.	POACEAE	Graminoid (annual)	Naturalised exotic
Digitaria debilis (Desf.) Willd.	POACEAE	Graminoid (annual)	LC, weedy, introduced from NE SA.
Echium plantagineum L.	BORAGINACEAE	Low shrub (annual)	Naturalised exotic

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2 Seed Biology: determining the physiological status of harvested, wild seed through germinability and viability testing towards identifying key restoration species

2.1 Abstract

The ecological restoration of Peninsula Shale Renosterveld is critical for meeting conservation-target shortfalls yet, to date, seed-based restoration experimentation in renosterveld has achieved limited success. As seed of high quality is central to successful seed-based restoration outcomes, ascertaining the seed physiological status prior to commencing large-scale restoration is necessary to prevent costly failures. In ecological restoration, the selection of appropriate species with an ability to establish successfully, compete and contribute to a structurally representative and functioning community is of central importance. Scattered literature, addressing species selection in restoration, points to the need to address several considerations over a range of scales encompassing ecosystem- and population-scale criteria as well as the seed physiological aspects.

This portion of the study set out to ascertain the viability and germinability of each of the 31 selected species. Thereafter, each of the species was evaluated with respect to several criteria towards determining their usefulness in restoration. Three viability and germinability tests (X-ray viability; viability by germination and cut test method (hereafter referred to as 'viability'); and, germinability) were carried out for each of the 31 species. The outcomes of these tests were evaluated in conjunction with several additional criteria (proximity to restoration site, population size, extent of area, plant abundance, seeds per individual, conservation status) to determine an overall score, a key-restoration-species index, as a measure of how useful the species are expected to be in future restoration efforts.

Germinability, and to a lesser extent viability, outcomes varied among species. The majority of species (roughly 75%) exhibited reasonably high levels of viability. Approximately two thirds of the species attained levels of germinability within 25% of viability whilst overall, just over half of the species exhibited germinability greater than 50%. This latter finding is of some concern since germinability is a good indicator of likely in-field performance and thus restoration success. In evaluating key restoration species, the majority of species scored in the mid-range which was well-represented by all six of the growth forms. Over a quarter of the species were placed in the high-range which was fairly well-represented by four of the growth forms. The variable seed physiological test outcomes, particularly with respect to germinability, highlight the importance of ascertaining seed quality prior to initiating large-scale seed collection and restoration.

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2.2 Introduction

The need for ecological restoration is apparent yet it is a research area with considerable knowledge gaps. Ecological restoration bears both considerable cost and risk factors (Crookes et al. 2013) and as seed quality is a central element of successful seed-based restoration efforts (Lippitt et al. 1994), ascertaining the physiological status of the seed is central to reducing the risk of this fundamental aspect. Germinability and viability are empirical measures of seed physiology and are critical in considering species for use in restoration as they are indicators for likely and potential performance. With respect to experimental seed biological-trait data, the Royal Botanic Gardens Kew Seed Information Database (SID) (RBG Kew 2009) is the most comprehensive source yet germination data exists for only half the 31 species harvested for this project. There remains a paucity of seed-based information pertaining to renosterveld species in general, particularly for the species that are common and of least concern, these traits being characteristics of value in the consideration of target species for restoration. The role of species selection is of central importance to restoration yet the literature in general fails to bring the complex issues underlying species selection and seed physiology together. Seed-based restoration efforts hinge on the selection of appropriate species to improve the success of species establishment in the short-term and over time the assembly of a desirable community. In considering species selection, scattered literature points to the need to consider informants at both the ecosystem- and population-scale as well as at the seed biology- and seed ecology-scale. The former refers to consideration of the resilience of the source population, conservation status of the species and the genetic integrity and diversity of the source population/s whilst seed physiology criteria require consideration with respect to seed viability, seed dormancy, germination cues (in the form of pre-treatments, environmental parameters and growth medium) and germinability. Going about species selection in a way that draws all these considerations together has not previously been done and the model proposed here is a novel approach to addressing the complex issues surrounding species selection. This portion of the research makes an empirical contribution in supplementing specific seed physiology information on the little-studied renosterveld species in question, and in addition a theoretical contribution in proposing a model that considers the empirical seed physiology criteria in the context of wider ecological- and population-scale criteria and allows for the thorough and efficient consideration of key species for inclusion in restoration. This introduction addresses the role of seeding in restoration; considerations relevant to the selection of species; the purpose of ascertaining seed quality (though viability and germinability testing); and, the principle factors affecting seed physiology and germination for their relevance to viability and germinability test outcomes.

2.2.1 Seeding in restoration

In degraded ecosystems where thresholds have been crossed (Whisenant 1999), passive restoration alone may not suffice, and the implementation of interventions is required to initiate restoration (Hobbs and Harris 2001). In ecological restoration the introduction of species by seed supplementation is commonly implemented (Holmes 2002a, Hufford and Mazer 2003, Midoko-

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Iponga 2004, Holmes 2005, Krauss and He 2006, Broadhurst et al. 2008, Holmes 2008, Mortlock 2008, Turnbull et al. 2000). Where the ability of a community to regenerate is poor (Stevensen et al. 2008) seeding may be required to overcome limitations in seed dispersal (Shiponeni 2003, Walton 2006), seed-bank depletion (Poschold et al. 1996, Cione et al. 2002, Heelemann 2012) and where species or functional groups are missing from a community (Holmes and Richardson 1999, Holmes 2008).

The source of seed for restoration projects is contentious (Kaye 2001, Wilkinson 2001, Hufford and Mazer 2003, Bischoff et al. 2006) with respect to issues of genetic integrity and diversity. Using seed material of local provenance (origin) in restoration is more commonly advocated than using seed from introduced genotypes as, over and above numerous risks associated with introduced genotypes, seed sourced from locally adapted genotypes is expected to establish more successfully and achieve greater restoration success (Holmes and Richardson 1999, Hufford and Mazer 2003, Bischoff et al. 2006, Bautista et al. 2009, Mijnsbrugge et al. 2010, Clewell and Aronson 2013). This concept of the appropriateness of local provenance is often referred to as home-site advantage where local genotypes, being better adapted to local habitat conditions, result in greater population fitness (Montalvo and Ellstrand 2000, Hufford and Mazer 2003, Bischoff et al. 2006). Delineating 'regions of provenance' is important (Bautista et al. 2009, Mijnsbrugge et al. 2010) yet may be challenging as genetic differentiation between populations or even within a population can be spatially restricted (Mijnsbrugge et al. 2010) and in the field this may potentially translate to non-local genotypes occurring within one kilometre of each other (Holmes and Richardson 1999). Instead of focusing on restricting the distance between the seed source and receiving restoration site, emphasis is rather placed on matching the habitats, abiotic and biotic factors of both sites (Wilkinson 2001, Hufford and Mazer 2003, Way 2003, Bautista et al. 2009, Mijnsbrugge et al. 2010). Once an appropriate target population has been determined, collection within the population must not be too narrow (Kaye 2001, Hufford and Mazer 2003, Mijnsbrugge et al. 2010). This caution is based on the assertion that a collection should be a good genetic representation of the population (Way 2003) and be adequately genetically diverse so as to prevent genetic 'bottlenecks' (Kaye 2001, Hufford and Mazer 2003, Mijnsbrugge et al. 2010). As a practical guideline the Kew Millennium Seed Bank set an approximate minimum number of plants for a seed collection at 50 plants per population (Way 2003, Way and Gold 2008).

The question of the quantity of seed required for a project, as determined by the project scope, goals and objectives, relates to issues surrounding the resilience of the source population and its ability to tolerate the effects of harvesting. In planning seed collection one should be mindful that harvesting seed from an area will have a negative impact on the source community or population (Guerrant et al. 2004, Broadhurst et al. 2008, Mortlock 2008). The greater the resilience of the community or population (for instance, the less threatened the species or the larger the population size) the smaller the impact experienced (Broadhurst et al. 2008). At the individual plant-scale, the Millennium Seed Bank restricts the quantity of seed collected per plant to a maximum of 20% of the available mature seeds observed on the day (Way and Gold 2008). From project to project, seeding rates (expressed as the number of seeds per area or more commonly as weight per area) vary

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significantly due to the variability of seed viability between species (Holmes and Richardson 1999), the degree and nature of the degradation and the specific ecosystem and project requirements. For example, restoration projects in renosterveld and at the fynbos-succulent karoo interface, rates of 50 kg uncleaned seed per hectare (Holmes 2002a) and 300 kg uncleaned seed per hectare (Holmes 2005) were sown respectively.

The selection of appropriate species that will establish successfully in restoration attempts is central to restoration outcomes yet there is little guidance on species selection in the literature which in general fails to coordinate ecosystem- or population-scale considerations and seed physiology. A limited literature addresses the selection of species for forest restoration. In response to a need to advance restoration practice of Amazonian forest and a lack of information pertaining to the component species, Knowles and Parrotta (1995) established a systematic approach to species selection whereby species were screened and rated by a numerical scoring system according to, for instance, phenological data, seed viability and post-establishment performance. In the screening of trees species for selecting species for restoration trials in dry tropical forest in northern Thailand, Blakesley et al. (2000) build their model of species selection on the 'framework species method' for tropical forest restoration proposed by Goosem and Tucker (1995) and Lamb et al. (1997). Some of the criteria used for species selection address issues of seed physiology, post-establishment performance and species abundance which are measured by germinability, in-field seedling survival, vigour and rarity (Blakesley et al. 2000). In terms of initiating the species selection process, however, Way (2003) recommends a desktop, species-based approach as a good starting point whereby information pooled from various sources functions to collate a list of species that are characteristic of the reference ecosystem. Refinement of this composite species list, in accordance with criteria as determined by the purpose and goals of the project in combination with in-field prospecting, is required to prioritise species and thus identify which species to target for seed collection (Way 2003).

A positive relationship is generally found between species richness and ecosystem processes and there is growing consensus that plant functional diversity is an appropriate determinant of ecosystem functioning (Diaz and Cadibo 2001), which, in conjunction with community structure, are appropriate aims in restoration (Holmes and Richardson 1999). Consequently, selecting species for restoration on the basis of functional traits is commonly advocated in the literature. Species can be grouped according to their responses to environmental and biotic factors as well as by their effects on ecosystem processes (Diaz and Cadibo 2001, Lavorel and Garnier 2002; Funk et al. 2008). Del Moral et al. (2007) discuss the links between succession theory and restoration ecology and recommend selecting species that are unlikely to disperse into the area (to overcome dispersal limitation and promote colonisation of desirable species) and species from a variety of life-forms towards redressing niche limitation and the associated challenges of species colonisation. Harvesting from more species than required redresses the likelihood of low survival and the selection of 'seral' species (from seral stages of successional sequence) functions to direct the restoration trajectory (del Moral et al. 2007). Holmes and Richardson (1999), in proposing restoration protocols based on the ecology of fynbos (and other fire-adapted shrublands) with respect to recruitment dynamics, community structure and ecosystem functioning, recommend the selection of species from the major guilds (growth form,

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nutrient-acquisition and regeneration guilds) and as many species as possible as the full representation of species recruits post-fire. In application of the redundant species theory, towards increasing ecosystem resilience and resistance to environmental change, the higher the number of species per each general functional type in a community, the higher the level of functional redundancy and the greater the chances of species survival and maintenance of ecosystem functionality (Holmes and Richardson 1999, Diaz and Cadibo 2001, Clewell and Aronson 2013). To date, few restoration efforts have selected target species on the grounds of their having similar ecological roles to the likely problematic invasive species (Funk et al. 2008). This approach to species selection and community assembly consequently seeks to build the ability of the community to resist invasive species (Funk et al. 2008). With this objective, Funk et al. (2008) propose the selection of species that share similar traits to invasive species and the selection of species with a high diversity of functional traits (Funk et al. 2008). Where a community has been invaded by alien species and the invasion has been facilitated by disturbance as opposed to species traits, selecting species on the basis proposed by Funk et al. (2008) has considerable merit (Vallejo et al. 2012).

2.2.2 Ascertaining seed physiological status

Seed-based restoration relies on successful seed germination and thus on seed of a high quality. Determining the seed physiological status prior to use in restoration is critical if costly restoration failures are to be averted (Lippitt et al. 1994). The processes involved with testing seed quality may be costly and time-consuming. There are essentially two types of tests for determining the physiological status and likely performance namely germination and viability tests (Gosling 2003). A germination test is a direct measure of the proportion of seeds capable of germinating (comprised of the number of seedlings germinated) whilst a viability test is an indirect measure of germination that determines the proportion of seeds that are alive and theoretically capable of germinating (comprised of germinated normal and abnormal seedlings and seeds that are fresh when cut or stained with Tetrazolium (TZ)) (Gosling 2003). Germination tests are often considered more relevant than viability tests particularly when one is testing a seed sample intended for a purpose where germinability is a key trait (Gosling 2003). Viability tests are more intensive but are favoured for their relatively rapid production of results or where seeds are known or expected to be dormant or recalcitrant (Gosling 2003). The X-ray of seeds is an additional method of estimating potential viability and is particularly useful for wild seed collections which may contain large quantities of partially-filled or empty seeds (Gosling 2003). Determining viability with X-rays requires the categorisation of the seed contents on the basis of interpreting varying colour intensities from light to dark (Gosling 2003). It is generally easy to categorise seeds as filled, partially-filled, empty, infested, diseased or mechanically damaged (Kamra 1976) but disadvantages include X-rays distinguishing damaged or missing tissue but not dead or living tissue and X-rays portraying a two-dimensional image of three-dimensional structures (Gosling 2003).

There is considerable value in comparing the methods and outcomes of the different seed quality tests, as the underlying factors responsible for discrepancies in the results can be narrowed down or identified. Whether the comparison involves test results of samples from the same seed lot or

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different seed lots invokes a different set of variables relevant to each comparison. Through such comparisons one can build up a cross-referenced repository for each species with respect to the effects of pre-treatments and parameters on species performance. Ultimately the purpose being to identify, among species, the optimal pre-treatments and environmental parameters required for high yields. The comparison of the different test outcomes may in addition reveal problematic species that perform poorly across the tests, highlighting the need for further germination cue-based experimentation in order to understand the dormancy breakage requirements of the species in question. The accumulation of seed physiological information of this nature is of great relevance to restoration for a number of reasons. The quality of the seed for a species can be used to inform the seed collection protocol with respect to which species to target as it is an indicator of the likely or potential in-field performance and a species may be included or excluded as a target species on the basis of their viability/germinability. Furthermore, the seed physiological status may inform the quantity of seed required for collection, for instance a species with a low measure of viability/germinability may be included for collection on the grounds of it being a desirable component of the assembled community in which case a larger quantity of seed would need to be collected to compensate for the proportion expected to be non-viable or non-germinable. Family- and species-specific knowledge of seed physiology with respect to germination cues (pre-treatments and environmental parameters) in addition has bearing on the timing and methodology of restoration in so far as informing how and when to make the most efficient use of a species with a view to optimising in-field performance.

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2.2.3 Factors affecting seed production and quality

Several factors affect seed development, and thus seed quality, and have the ability to considerably affect viability and germinability test outcomes. The main factors addressed here include genetic integrity and diversity, resource limitation and seed collection protocol considerations.

In keeping with the explanations offered for seed provenance, seed collected from local genotypes that are adapted to local conditions (Hufford and Mazer 2003, Bischoff et al. 2006, Mijnsbrugge et al. 2010) are more likely to result in high levels of germinability. Conversely, narrow seed collection within a population or from small, isolated or sparse populations (Lamont et al. 1993) may result in inbreeding effects which impact seed set and germinability (Keller and Waller 2002, Fenner and Thompson 2005). Further to the genetic risks of small populations, fragmentation size and isolation in renosterveld have been found to negatively affect pollinator diversity and seed set in several plant species (Donaldson et al. 2002).

Levels of seed viability have been shown to change in response to climate between years (Chambers 1989) and variable seed-crop size, incomplete seed development, seed mortality and variable seed size have been linked to the competition of resources such as rainfall, light hours and nutrients (Fenner and Thompson 2005) as well as the reproductive traits of species (Shibata et al. 2002). Unfavourable conditions with respect to unusually dry seasons (O'Connor and Pickett 1992) induce fruit and seed abortion (Lovett-Doust, 1988, Ruiters et al. 1993). Seed size may differ among populations, plants (Wulff 1986), inflorescences, fruits (Fenner and Thompson 2005), geographic

locations (Khan and Stofella 1985), seasonal timing (Kane and Cavers 1992) and years (O'Connor and Pickett 1992). Larger seeds are usually the result of higher nutrient availability (Fenner 1992) and smaller seeds are usually produced in response to higher temperatures (Kiniry and Musser 1988), drought (Benech Arnold et al. 1991), shade (Gray et al. 1986), and inter plant competition (Bhaskar and Vyas 1988). Although smaller seeds may not necessarily have lower seed viability (Kane and Cavers 1992, Holmes and Newton 2004) in general, larger seeds perform better in response to variable environmental conditions due to greater energy reserves (Westoby at el. 1996) and thus may exhibit greater levels of seed germination, establishment and competitive ability (Roach and Wulf 1987, Fenner 1992).

The timing of seed collection is important as seed collected when incompletely formed is typically unviable whilst seed collected when at its optimum stage of development (when the seed equilibrium relative humidity (RH) begins to decrease) maximises seed viability and longevity (Hay and Smith 2003; Way 2003). In the field, this point of maturity typically equates to when the seed is ready to disperse naturally (Hay and Smith 2003; Way 2003). A rapid in-field assessment of seed quality prior to large-scale collection is prudent whereby a cut-test of approximately 10-20 seeds determines the number of full, empty, infested or immature seeds (Way 2003; Way and Gold 2008).

Several aspects of post-harvest handling have implications for seed longevity and performance. Reducing the moisture content of seeds considerably improves seed longevity (Probert 2003, Linnington 2003, Lippitt et al. 1994). For long-term conservation purposes, seed is ideally dried to approximately 15% equilibrium RH at 15°C (Linnington 2003, Gold 2008, Linnington and Manger 2009) however these conditions may not be necessary or possible for most short-term restoration projects. Open storage may sometimes be the only viable option (Hartman and Kester 2011) in which case seed is initially dried in ambient conditions in a covered, well-ventilated, outside area, and once drier (Schmidt and Thomsen 2003), placed in a dry, cool, dark place (Krug 2004) with protection from insect seed predators (for example Karbadust insecticide by Efekto) (Holmes 2002a). The storage time of seed should preferably be limited to sowing in the next appropriate season. Cleaning seed has been shown to improve longevity and germinability by reducing susceptibility to disease (Lippitt et al. 1994, Terry and Bertenshaw 2008) but again this degree of processing may be over specified for a restoration application. Over-vigorous cleaning has the potential to damage the seed coat and compromise seed longevity and germinability (Dickie and Stuppy 2003).

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2.2.4 Environmental factors affecting germination

Several environmental factors are vital for seed germination and are consequently central to viability and germinability outcomes. Temperature, water availability, gaseous exchange and light are the essential determinants of seed germination and affect its timing, success and rate (Hartmann and Kester 2011). Mediterranean climate species typically require low temperatures to germinate and may be inhibited by temperatures outside of the optimum range (Hartmann and Kester 2011). Germination of many fynbos species, (a useful point of reference as the better-studied adjacent vegetation type to renosterveld), results from the diurnal temperature fluctuation of autumn (Holmes 2002b) thus seed should ideally be sown into a post-fire environment in autumn (Holmes and

Richardson 1999). Water availability is a determinant of seed imbibition and germination. As a seed imbibes, water moves down a water potential gradient from the soil into the seed and the soil surrounding the seed requires moisture replenishment from the adjacent soil (Hartmann and Kester 2011). Consequently, seed microsites where the seeds are in close contact with the soil are important for sustained germination (Holmes 2001) as low moisture conditions reduce both seed germination and seedling emergence rates (Hartmann and Kester 2011). In addition to moisture, germination is reliant on the exchange of gases between the seed embryo and the soil (Hartmann and Kester 2011). Germination can be inhibited by excessive moisture where oxygen availability is restricted (Gosling 2003, Hartmann and Kester 2011). Although fynbos species may not be as reliant on light as other floras (Holmes and Newton 2004), light plays a role in inducing and breaking seed dormancy in many species with small-seeded species being generally more reliant on light than large seeded species (Milberg et al. 2000). In natural sunlight, the light receptor phytochrome, in an imbibing seed, changes to an active state and germination is stimulated, thus, where light conditions are low, germination is inhibited (Hartmann and Kester 2011).

Most plant taxa display seed dormancy (Baskin and Baskin 2003) as do the majority of fynbos species (Le Maître and Midgley 1992) and consequently do not germinate under otherwise favourable conditions due to additional requirements resulting from dormancy (Hartmann and Kester 2011). In the context of ecological restoration or viability and/or germinability testing, dormant seeds require the application of pre-sowing treatments that function to break dormancy and stimulate germination (Hartmann and Kester 2011). The nature of the pre-treatment required to break dormancy is a function of the exogenous and endogenous seed traits and therefore the class of dormancy (physiological, morphological, morphophysiological, physical and combinational) (Baskin and Baskin 2003). Dormancy class, level and/or type although often a family trait has been found to vary within families for example Fabaceae and Anacardiaceae (Hartmann and Kester 2011).

This chapter aims to determine the physiological status of each of the 31 species through three tests: X-ray viability, viability and germinability. Thereafter, each of the 31 species is considered with respect to several criteria (proximity to restoration site, population size, extent of area, plant abundance, seeds per individual, conservation status, viability and germinability) towards evaluating it as a key restoration species. The viability and germinability test outcomes contribute empirically through adding and in some instances generating physiological data for the 31 renosterveld species, which are in general, little-studied. This portion of the study proposes a novel approach to identifying species as key to restoration efforts whereby a simple model draws together and systematically evaluates the multifarious issues pertaining to species selection: seed performance data (viability and germinability) in conjunction with ecosystem- and population-scale considerations.

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2.3 Methods

2.3.1 Study area

Seed was collected from Groote Schuur Estate on the lower north eastern slopes of Devil's Peak and Signal Hill, Cape Town. Both locations occur within the current distribution of the vegetation type (Figure 1.3). The areas in Groote Schuur Estate ranged approximately 500 m to 1.3 km from the restoration site whilst the areas on Signal Hill were across the Cape Town city bowl and approximately 6.6 km to 7 km from the restoration site.

2.3.2 Target species selection

A composite list of 668 species was compiled from the following sources: species recorded for Peninsula Shale Renosterveld by Rebelo et al. (2006); the species recorded for Peninsula Shale Renosterveld by SANBI; the species list from the Signal Hill (Joubert 1991); a list of species from the survey of Groote Schuur Estate (Euston-Brown 2011) and the species list used for renosterveld restoration in Eerste River (Holmes 2002a).

This composite list was then reduced to a selection of 150 species on the basis of their being recorded by two or more of the above-mentioned sources, listed by Rebelo et al. (2006), recorded as being of conservation concern by the Red List of South African Plants (SANBI 2011), utilised by Holmes (2002a), and, if recorded as common on Signal Hill by Joubert (1991). The list was subsequently arranged according to growth forms (graminoid, geophyte, forb, succulent shrub, low shrub and tall shrub) to ensure representation from the major growth form guilds.

With this version of the list areas of Groote Schuur Estate and Signal Hill were assessed with the purpose of reducing the list to a more manageable number for actual collection according to 'ground-truthing' of the following: identification and location of populations, size of populations, and the likelihood of the populations to produce adequate seed for harvest within the time-frame identified for seed collection (August 2012 to May 2013). Tree species which are uncommon in renosterveld (Rebelo et al. 2006) were dropped from the list. Although collection of as many species as possible is appropriate for restoration (Holmes and Richardson 1999) it was decided that approximately 25 to 35 species would be appropriate for the experiment but that additional species would be included on the target list to provide a degree of collection lee-way. Ultimately, seed was collected from 31 species (five graminoids, five geophytes, three forbs, four succulent shrubs, ten low shrubs and four tall shrubs).

2.3.3 Seed collection, processing and storage

Seed from the 31 species was collected from August 2011 till May 2012. The bulk of the collection was made from August 2011 to November 2011 with just a few species being collected in the late summer months- *Searsia* species (considered potentially recalcitrant) and *Chironia baccifera* (spontaneous collection) were collected two days before in-field sowing (which commenced 19th May

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2012). Seed was collected over weekends (a field assistant, volunteers and I spent 154 hrs, 75 hrs and 186 hrs respectively). As the seed was collected, it was temporarily stored in a dry, well-ventilated room and an initial phase of cleaning took place to remove debris and reduce bulk, then placed in brown paper bags and fumigated (with a Doom Fogger insecticide) prior to transportation to the dry room at Kirstenbosch Botanical Garden's Seed Room facility and stored at 15% equilibrium RH at 15°C. Four fumigations and trips were made to Kirstenbosch. The seed of each species was weighed upon delivery. The total weight of newly delivered seed (prior to the drying process and second phase of cleaning and excluding the fresh collections of *Chironia baccifera, Searsia tomentosa, S. laevigata var. villosa* was approximately 18.71 kg. After approximately six months of drying and the second phase of cleaning, the total weight of the collection was approximately 8.33 kg (resulting in a 44.5% reduction in weight). The total weight (including the fresh collections) was approximately 9.57 kg of semi-cleaned seed which over the area of the experimentation is a rate of approximately 57.42 kg semi-cleaned seed per hectare. The seed for the germinability test (glasshouse) and viability tests (X-ray viability and viability (by germination and cut test method)) was fully cleaned, for instance, all debris was removed and the seed from the following species had been removed from their respective covering structures: *Cymbopogon marginatus, Chironia baccifera, Ehrharta calycina, Eriocephalus africanus var. africanus, Myrsine africana, Pentameris airoides* subsp. *airoides, Searsia laevigata var. villosa, S. tomentosa, Tenaxia stricta* and *Themeda triandra.* The seed used in the field experiment however remained semi-cleaned and the seed from all species were kept inside their covering structures (with the exception of *Myrsine africana* which had been soaked and the flesh removed by rubbing in a sieve). Heat treatment is a method of scarification and increases seed coat permeability to initiate imbibition (Hartmann and Kester 2011). Hot water treatment was carried out for three species (*Arctopus echinatus, Myrsine africana* and *Podalyria sericea*) as a sowing pre-treatment in the viability and germinability tests and field experiment. The seed for each species was placed in approximately 250 ml of water at 80°C, allowed to cool and soak for eight hours. The seed used in the germinability test and field experiment was smoked to promote germination (Brown et al. 2000, Brown et al. 2003, Brown and Botha 2004) by placing it inside the Kirstenbosch tent filled by smoke derived from wet and dry fynbos material and the smoke allowed to settle for three hours. In an attempt to replicate the smoke pre-treatment for the seed used in the viability test, the seed was soaked overnight in Kirstenbosch smoke primer (containing growth stimulants including gibberellic acid) dissolved in water (one smoke primer disc placed in 250 ml water per species).

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2.3.4 X-ray viability

Seed export permits were granted by Cape Nature to courier seed to Royal Botanic Gardens Kew where both the X-ray viability and the viability (by germination and cut test method) were conducted. Random samples of 100 seeds (plus approximately 10 extra seeds per species as spares) per each of the 31 species were X-rayed. Each replicate of 25 seeds (of which there were four replicates per species) was placed inside a Petri dish and X-rayed using a Faxitron digital X-ray machine (Qados, Sandhurst, UK), set at the standard Millennium Seed Bank settings for seed X-radiography (22 kV and 0.3 mA for 20 s). The digital image was saved in jpeg format.

2.3.5 Viability (germination and cut test method)

Once the seeds were X-rayed, they were placed in dry storage (15% equilibrium RH at 15°C) until required for the germination and cut test. It is relevant to note that the pre-treatments and parameters implemented in the viability tests were generally done so as to mimic the conditions of the glasshouse germinability test (and conditions of the in-field experiment) and for this reason, the methodology of the viability tests conducted for these species may differ to the methodology of tests previously carried out by Kew for the same (or similar species). The tests were initiated in three batches: on 5th, 12th and 20th September 2012. Four replicates of 25 seeds per species were tested (except where indicated otherwise for 17 species where n ≠ 100, refer to Table 2.1). Dry Petri dishes containing all seed replicates were put over water (100% RH in a closed polyethylene container at 20°C for 24 hrs to prevent imbibition damage). The same three species (*Arctopus echinatus, Myrsine africana* and *Podalyria sericea*) were then hot water treated in approximately 250 ml of water at 80°C and allowed to cool. Thereafter, seeds of all species were soaked for five minutes in a 10% Domestos solution (to sterilise the seed surface and prevent fungal growth), rinsed for one minute in distilled water, soaked overnight (18 hrs at 15°C) in Kirstenbosch smoke primer solution, plated on 1% agar in 90 mm Petri dishes and placed in an incubator at an alternating temperature of 20/10°C with lighting (8/16 hrs) provided in the warm phase. The agar was changed as required. Replicates were checked each week and germinated seedlings removed from the plate and added to the running total. Germination was scored when the radicle was at least 2 mm long. After 12 weeks of no germination, or after four weeks of no further germination where some germination had occurred, a cut test of the non-germinated seeds was carried out to determine the number of seeds that was fresh (i.e. likely to be viable) or mouldy (non-viable) on dissection. If the majority of seeds were fresh on cut-testing, they were placed into a 1% solution of TZ for 72 hrs at 30°C and the staining evaluated. Seeds that stained dark red were interpreted as viable (i.e. dormancy may not have been broken). For the first batch, seeds were at a constant 20°C for the first week due to incubator failure; thereafter plates were alternating at 20/10°C. For three species, additional treatments were implemented compared with those in the germinability test. Where the germination response in the laboratory was very low yet a sub-sample of seeds was fresh and identified as viable with a cut test or TZ test, protocols known to improve germination were used in an attempt to break seed dormancy. This was carried out in the laboratory for *Hermannia hyssopifolia* (the seeds were placed into 25°C and the replicates chipped), *Pelargonium cucullatum* subsp. *tabulare* (on cut testing three seeds were firm and bright green on the inside so the remaining seeds were planted in sand) and *Searsia laevigata var. villosa* (the fruit pulp was removed with a scalpel).

2.3.6 Germinability (glasshouse test)

Each of the 31 species were sown into polystyrene seedling trays at a rate of one seed per cavity (each cavity approximately 40 × 40 × 90 mm deep) where there were five replicates of 25 seeds per species. The substrate used was Kirstenbosch general mix (eight parts bark, three parts sand, three parts compost) mixed with 5% peat moss (whereas fynbos mix is eight parts bark, three parts sand). Three species were hot-water treated and all species were sown into the trays, smoked, placed inside

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the glasshouse and watered (with fungicide diluted into the water, Apron XL with active ingredient mefenoxam from UAP Paarl). Hand-watering was carried out roughly three times per week. Apron XL was reapplied to all seedlings 17th May, 8th August and 2nd October 2012.

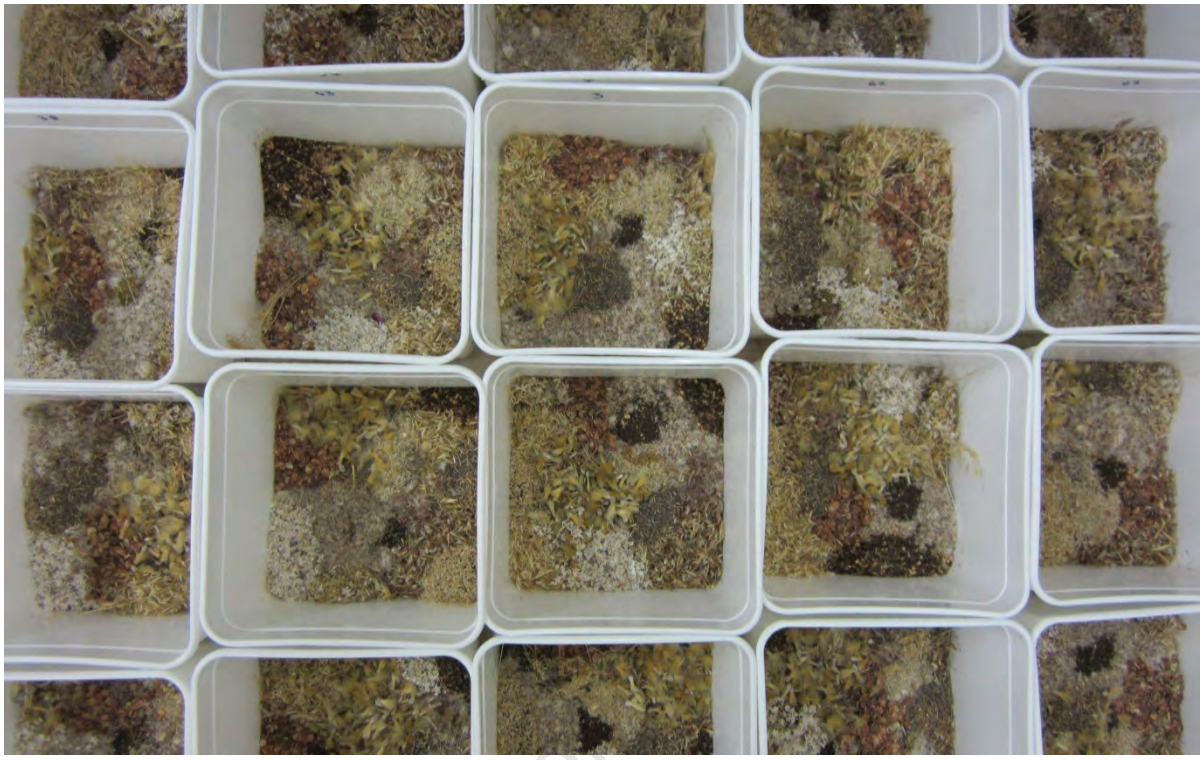

Photo plate 2.1: Seed mix weighed and placed into 64 containers for smoking prior to sowing in the field experiment.

Photo plate 2.2: Seed-smoking tent at Kirstenbosch. Plant material burnt in the drum (refer arrow) and pumped into tent containing the batches of seed (seed trays for the germinability test and containers for the field experiment).

Photo plate 2.3: Germinability test in the Kirstenbosch glasshouse: seed (here *Othona arborescens*) sown into trays at a rate of 1 seed per cavity and five replicates of 25.

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2.3.7 Criteria for a key restoration species

During and after the relevant processes (seed collection, germinability and viability testing), information pertaining to the following was collated for each species: proximity of seed source to restoration site (Groote Schuur Estate, Signal Hill or both), population size (ranging from <25 to >30 000 individuals); extent of area (ranging from <0.25 to >10 ha), plant abundance (rare, occasional, frequent, common, abundant), seeds per individual (ranging from <10 to >1 000 seeds), conservation status (as classified in the Red List of South African Plants), % germinability (as determined by the glasshouse test), % viability (as determined by the germinability and cut test). Information for the seed collection criteria were estimates derived solely from the areas from which seed was harvested for this project and may not be representative of other areas within the ecosystem or the ecosystem as a whole.

2.4 Data collection

2.4.1 X-ray viability

Once the X-ray images were taken and saved as jpegs, no further data collection was required.

2.4.2 Viability (germination and cut test method)

Data were collected more or less on a weekly basis for each of the three test batches (from 12th September 2012 to 13th January 2013). A running total of the number of germinated seedlings was kept. The species were dissected and some were stained with TZ and the number of fresh (i.e. viable), mouldy (i.e. non-viable), empty and infested seeds was recorded.

2.4.3 Germinability (glasshouse test)

Data were collected monthly from June 2013 till January 2013. The number of seedlings per replicate was counted and the height (cm) of five seedlings per species was measured. Photographs of each species were taken to record seedling emergence and development which served to assist in-field identification.

2.4.4 Criteria for a key restoration species

Information for the seed collection criteria were estimates derived solely from the areas from which seed was harvested for this project. Areas were mapped and measured on Google Earth (2011). Plant abundance, population size and seeds per individual were estimated based on in-field observations. No additional data collection (further to that conducted for the germinability and viability tests) occurred for the performance-related criteria.

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2.5 Data analysis

2.5.1 X-ray viability

The X-ray image of each of the four replicates of 25 seeds per species was studied and the number of seeds for the following categories was recorded: A (0-½ filled: empty, unlikely to germinate), B (½-¾ filled: partially filled, incompletely formed, may/may not germinate depending on the species), C (¾ filled-full: full, expect seed/s to germinate) and D (insect and physically damaged: may/may not germinate depending on the position and extent of damage). The number of seeds recorded in category C per replicate (those expected to germinate) were averaged to determine the mean percentage X-ray viability per species.

2.5.2 Viability (germination and cut test method)

The number of seedlings germinated was added to the number of fresh seeds for each of the four replicates and the totals averaged to determine the mean percentage viability per species. Instances where TZ staining affected viability are reported (Appendix 2.2) and in each case have been addressed in the discussion.

2.5.3 Germinability (glasshouse test)

The maximum number of seedlings per each of the five replicates were averaged to determine the mean percentage germinability per species.

2.5.4 Key restoration species

The criteria (proximity to restoration site, population size (individuals), extent of area (ha), plant abundance, seeds per individual (no.), conservation status, germinability (%) and viability (%)) were divided into categories relevant to the range of the effect size and assigned values (ranging from zero to eight). Three categories (conservation status, % germinability and % viability) were weighted to increase the magnitude of their score and avoid losing the value in these performance related criteria: germinability and viability being of great relevance to restoration success. Scores per species were tallied across all categories to produce a final key-restoration-species index. The lowest possible score=6 and highest possible score=49 and the following categories were determined: low (<20), moderate (20-30), high (>30).

2.6 Results

This section of the chapter firstly deals with seed physiology and reports the outcomes of the seed performance tests (X-ray viability, viability and germinability). In comparing the results from the different tests, problem thresholds were utilised to detect species with potential problems relating to, for instance, poor seed quality, dormancy and germination cue variables. Thereafter the results of the key restoration species assessment are presented.

2.6.1 Seed performance

The purpose of this component of the analysis was to gain insight into the physiological status of the seed of each species through setting up a comparison of the seed performance results generated by the three individual test methods (X-ray viability, viability and germinability) (refer Table 2.1 for summary results and Appendices: 2.1, 2.2 and 2.3 for individual test results). Through comparisons of the test conditions and associated results, the underlying factors responsible for the different test outcomes have been narrowed down and identified where possible.

The results were analysed according to a four-fold comparative process made from particular standpoints from which to identify potential problem areas. The first three comparisons were inter-procedural comparisons whilst the fourth looked discretely at species performance with respect to germinability. The problem areas were identified by thresholds and the species that occur in each problem area have been listed. Lastly, the species that did not occur within any of the four problem areas have been listed as non-problematic. The four sets of comparison were as follows:

a) Comparison of viability with a 'reference' germinability (the highest recorded % germinability for the species recorded on SID) (refer Appendix 2.4 for comparative test conditions): the purpose of this comparison was to measure the viability of each of the 31

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species against a benchmark germinability so as to indicate relative seed quality. The problem area was identified on the basis of the 'reference' germinability exceeding viability by more than 25%. There were many variables in the respective tests resulting from, most notably, the fact that the tests are conducted on samples from different seed lots. Where the species was not recorded on SID (for 12 species), a species of the same genus was used as a surrogate (in all cases except one, species from the Cape Peninsula or Western Cape were selected) but where no South African species of the genus were recorded (for three species) no comparison was made.

b) Comparison of viability and germinability: the purpose of this comparison was to identify species where viability exceeded germinability by more than 25%. The relationship where viability exceeds germinability is expected, but it was where this difference exceeded 25% that determined the problem threshold. Cases where germinability exceeded viability have been reported.

c) Comparison of X-ray viability with either viability or germinability (whichever was greater): the purpose of this comparison was to detect species where X-ray viability exceeded either viability or germinability by more than 25%.

d) Assessment of germinability: the purpose of this comparison was to indicate where the germinability of a species was poor and measured less than 50%.

Comparison of 'reference' germinability with viability

The eight species where the 'reference' germinability exceeded viability by more than 25% or more were *Athanasia crithmifolia, Erepsia anceps, Eriocephalus africanus var. africanus, Hermannia hyssopifolia, Pentameris airoides* subsp. *airoides, Searsia laevigata var. villosa, Searsia tomentosa* and *Themeda triandra.*

Comparison of viability with germinability

The 11 species where viability exceeded germinability by 25% or more were *Dimorphotheca pluvialis, Hermannia hyssopifolia, Lachenalia fistulosa, Lampranthus emarginatus, Myrsine africana, Pelargonium cucullatum* subsp. *tabulare, Pentameris airoides* subsp. *airoides, Ruschia rubricaulis, Tenaxia stricta, Trachyandra muricata* and *Ursinia anthemoides* subsp. *anthemoides*.

The four species where germinability exceeded viability were *Babiana fragrans, Erepsia anceps, Eriocephalus africanus var. africanus* and *Podalyria sericea.* The magnitude of the difference between values here was low and ranged from 1.2% to 16.1%.

Comparison of X-ray viability with viability or germinability

The following two species showed a large discrepancy where X-ray viability exceeded viability or germinability (whichever was greater) by more than 25%: *Erepsia anceps* and *Pentameris airoides* subsp. *airoides.*

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Germinability of less than 50%

The 15 species with germinability lower than 50% were *Arctopus echinatus, Dimorphotheca pluvialis, Erepsia anceps, Hermannia hyssopifolia, Lampranthus emarginatus, Myrsine africana, Pelargonium cucullatum* subsp. *tabulare, Pentameris airoides* subsp. *airoides, Ruschia rubricaulis, Searsia laevigata var. villosa, Searsia tomentosa, Tenaxia stricta, Themeda triandra, Trachyandra muricata* and *Ursinia anthemoides* subsp. *anthemoides.*

Non-problematic species

Thirteen species (approximately 42%) were classified non-problematic as they avoided detection from the problem-thresholds by exhibiting viability within 25% of the 'reference' germinability; germinability within 25% of viability; either viability/germinability within 25% of the X-ray viability; and, germinability in excess of 50%. These species were *Babiana fragrans, Chironia baccifera, Chrysocoma coma-aurea, Cymbopogon marginatus, Ehrharta calycina, Felicia filifolia, Helichrysum cymosum* subsp. *cymosum, Helichrysum patulum, Moraea bellendenii, Ornithogalum thyrsoides, Othonna arborescens, Podalyria sericea* and *Salvia africana-caerulea.* Fabaceae species are often insect-infested (Way 2003) and although seed-infestation was confirmed in *Podalyria sericea* seed (insects of the Coleoptera order visible on the X-rays) the seed quality was largely unaffected.

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Table 2.1: Seed performance, summary results for X-ray viability, viability and germinability for each of the 31 species.
X-ray viability (%) ± S.D: n=100 except for *Eriocephalus africanus* where n=216 (†). Germinability (%) ± S.D: n=125. Viability (%) ± S.D: n=100 except for 17 species where n ≠ 100 (#): *Arctopus echinatus* n=101, *Athanasia crithmifolia* n=98, *Chrysocoma coma-aurea* n=97, *Cymbopogon marginatus* n=98, *Eriocephalus africanus var. africanus* n=218, *Felicia filifolia* n=99, *Helichrysum cymosum* subsp. *cymosum* n=96, *Helichrysum patulum* n=98, *Hermannia hyssopifolia* n=99, *Lachenalia fistulosa* n=96, *Lampranthus emarginatus* n=99, *Moraea bellendenii* n=97, *Pentameris airoides* subsp. *airoides* n=95, *Salvia africana-caerulea* n=99, *Searsia laevigata var. villosa* n=98, *Searsia tomentosa* n=98 and *Tenaxia stricta* n=97. One of the data points (viability of *Lachenalia fistulosa*) was corrected to 100% from 102% (as germination was possibly counted twice on day 42 for rep 2).

SPECIES	FAMILY	GROWTH FORM	X-RAY VIABILITY n=100 except where indicated (†)	VIABILITY n=100 except where indicated (#)	GERMINABILITY (n=125)
			Mean % ± S.D	Mean % ± S.D	Mean % ± S.D
Arctopus echinatus	APIACEAE	Forb	38.0 ± 19.18	# 27.7 ± 27.22	6.4 ± 3.58
Athanasia crithmifolia	ASTERACEAE	Tall shrub	67.0 ± 13.61	# 64.3 ± 5.69	53.6 ± 13.45
Babiana fragrans	IRIDACEAE	Geophyte	94.0 ± 5.16	95.0 ± 6.00	99.2 ± 1.79
Chironia baccifera	GENTIANACEAE	Low shrub	76.0 ± 7.30	76.0 ± 8.76	67.2 ± 21.24
Chrysocoma coma-aurea	ASTERACEAE	Low shrub	92.0 ± 0.00	# 93.8 ± 2.31	84.8 ± 9.12
Cymbopogon marginatus	POACEAE	Graminoid	100.0 ± 0.00	# 98.0 ± 2.36	76.0 ± 8.94
Dimorphotheca pluvialis	ASTERACEAE	Forb (annual)	96.0 ± 3.27	88.0 ± 4.62	4.0 ± 4.00
Ehrharta calycina	POACEAE	Graminoid	89.0 ± 6.00	96.0 ± 3.27	85.6 ± 9.63
Erepsia anceps	MESEMBRYANTHEMACEAE	Succulent shrub	99.0 ± 2.00	13.0 ± 10.00	28.0 ± 10.58
Eriocephalus africanus var africanus	ASTERACEAE	Low shrub	† 45.8 ± 17.09	# 46.3 ± 3.83	62.4 ± 7.80
Felicia filifolia	ASTERACEAE	Low shrub	82.0 ± 7.66	# 79.0 ± 5.67	57.6 ± 4.56
Helichrysum cymosum subsp. *cymosum*	ASTERACEAE	Low shrub	95.0 ± 5.03	# 88.5 ± 7.90	70.4 ± 10.43
Helichrysum patulum	ASTERACEAE	Low shrub	91.0 ± 6.00	# 81.6 ± 14.27	75.2 ± 12.46
Hermannia hyssopifolia	MALVACEAE	Low shrub	57.0 ± 11.49	# 65.7 ± 8.50	0.0 ± 0.00
Lachenalia fistulosa	HYACINTHACEAE	Geophyte	99.0 ± 2.00	# 100.0 ± 3.91	51.2 ± 7.16
Lampranthus emarginatus	MESEMBRYANTHEMACEAE	Succulent shrub	97.0 ± 2.00	# 82.8 ± 4.05	22.4 ± 12.52
Moraea bellendenii	IRIDACEAE	Geophyte	100.0 ± 0.00	# 97.9 ± 5.16	84.8 ± 9.12
Myrsine africana	MYRSINACEAE	Tall shrub	99.0 ± 2.00	93.0 ± 6.00	25.6 ± 4.56
Ornithogalum thyrsoides	HYACINTHACEAE	Geophyte	98.0 ± 2.31	98.0 ± 2.31	77.6 ± 4.56
Othonna arborescens	ASTERACEAE	Succulent shrub	95.0 ± 5.03	80.0 ± 4.62	64.8 ± 7.69
Pelargonium cucullatum subsp. *tabulare*	GERANIACEAE	Low shrub	82.0 ± 5.16	84.0 ± 13.47	7.2 ± 3.35
Pentameris airoides subsp. *airoides*	POACEAE	Graminoid (annual)	98.0 ± 4.00	# 53.7 ± 10.01	0.0 ± 0.00
Podalyria sericea	FABACEAE	Low shrub	80.0 ± 10.83	79.0 ± 11.94	89.6 ± 5.37
Ruschia rubricaulis	MESEMBRYANTHEMACEAE	Succulent shrub	95.0 ± 2.00	83.0 ± 18.29	0.0 ± 0.00
Salvia africana-caerulea	LAMIACEAE	Low shrub	69.0 ± 8.87	# 59.6 ± 17.31	52.8 ± 13.08
Searsia laevigata var. villosa	ANACARDIACEAE	Tall shrub	0.0 ± 0.00	# 1.0 ± 2.00	0.0 ± 0.00
Searsia tomentosa	ANACARDIACEAE	Tall shrub	4.0 ± 3.27	# 5.1 ± 3.83	0.0 ± 0.00
Tenaxia stricta	POACEAE	Graminoid	96 ± 3.27	# 91.8 ± 3.21	20.0 ± 4.90
Themeda triandra	POACEAE	Graminoid	50.0 ± 7.66	25.0 ± 29.30	29.6 ± 4.56
Trachyandra muricata	ASPHODELACEAE	Geophyte	100.0 ± 0.00	10.0 ± 2.31	11.2 ± 5.93
Ursinia anthemoides subsp. *anthemoides*	ASTERACEAE	Forb (annual)	95.0 ± 3.83	79.0 ± 18.29	36.8 ± 9.55

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2.6.2 Key restoration species

The model developed to identify key restoration species assigned values to each of the categories so as to rank each species within the indices range (refer Table 2.2). The possible minimum and maximum scores were six and 49 respectively. The actual restoration potential scores ranged from 16 to 42 and three categories were determined as follows: low (indices <20), moderate (indices from 20 to 30) and high (indices >30).

Low (<20)

Five species (16%) exhibited low indices (<20): *Themeda triandra* (graminoid), *Arctopus echinatus* (forb), *Pelargonium cucullatum* subsp. *tabulare* (low shrub), *Myrsine africana* (tall shrub) and *Searsia laevigata var. villosa* (tall shrub).

Moderate (20 to 30)

The mid-range (indices 20-30) was comprised of 18 species (58%) and was well-represented by species from each of the six growth forms: *Ehrharta calycina, Pentameris airoides* subsp. *airoides* and *Tenaxia stricta* (graminoid), *Moraea bellendenii, Trachyandra muricata* (geophyte), *Dimorphotheca pluvialis, Ursinia anthemoides* subsp. *anthemoides* (forb), *Erepsia anceps, Lampranthus emarginatus, Othonna arborescens, Ruschia rubricaulis* (succulent shrub), *Chironia baccifera, Eriocephalus africanus var. africanus, Felicia filifolia, Helichrysum cymosum* subsp. *cymosum, Helichrysum patulum, Hermannia hyssopifolia* (low shrub) and *Searsia tomentosa* (large shrub).

High (>30)

Eight species (26%) occurred in the high-scoring category (>30). In evaluating the characteristics of these species, four out of the six growth forms were represented and six different families (out of a possible 14): graminoid (*Cymbopogon marginatus*), geophyte (*Babiana fragrans, Lachenalia fistulosa* and *Ornithogalum thyrsoides*) low shrub (*Chrysocoma coma-aurea, Podalyria sericea* and *Salvia africana-caerulea*) and tall shrub (*Athanasia crithmifolia*).

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Table 2.2: Key-restoration-species indices.

KEY	(0)	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Proximity to restoration site		Signal Hill	Signal Hill & Groote Schuur Estate	Groote Schuur Estate					
Population size (individuals)		<250	250-1000	1000-5000	5000-10000	10000-20000	20000-30000	>30000	
Extent of area (ha)		<0.25	0.25-0.5	0.5-2.5	2.5-5	5-7.5	7.5-10	>10	
Plant abundance		Rare	Occasional	Frequent	Common	Abundant			
Seeds per individual (no.)		<10	10-50	50-100	100-250	250-500	500-1000	>1000	
Conservation status		Least concern			Conservation concern				
Germinability %	0		0-25		25-50		50-75		75-100
Viability %	0		0-25		25-50		50-75		75-100

SPECIES	FAMILY	GROWTH FORM	PROXIMITY TO RESTORATION SITE	POPULATION SIZE	EXTENT OF AREA	PLANT ABUNDANCE	SEED ABUNDANCE PER INDIVIDUAL	CONSERVATION STATUS	GERMINABILITY (Glasshouse test) (n=125)	VIABILITY (Germination and cut test) (n=100 except where specified)	**TOTAL SCORE**	**LOW / MODERATE / HIGH**
Arctopus echinatus	APIACEAE	Forb	1	1	4	2	1	1	2	0	**12**	**L**
Athanasia crithmifolia	ASTERACEAE	Tall shrub	3	7	4	5	7	1	6	6	**39**	**H**
Babiana fragrans	IRIDACEAE	Geophyte	1	7	7	4	3	4	8	8	**42**	**H**
Chironia baccifera	GENTIANACEAE	Low shrub	1	1	2	3	7	1	6	8	**29**	**M**
Chrysocoma coma-aurea	ASTERACEAE	Low shrub	3	4	3	4	7	1	8	8	**38**	**H**
Cymbopogon marginatus	POACEAE	Graminoid	2	5	4	3	4	1	8	8	**35**	**H**
Dimorphotheca pluvialis	ASTERACEAE	Forb (annual)	1	3	2	2	2	1	2	8	**21**	**M**
Ehrharta calycina	POACEAE	Graminoid	1	2	3	2	2	1	8	8	**27**	**M**

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Erepsia anceps	MESEMBRYANTHEMACEAE	Succulent shrub	2	3	3	3	4	1	4	2	**22**	**M**
Eriocephalus africanus var africanus	ASTERACEAE	Low shrub	1	3	3	3	2	1	6	4	**23**	**M**
Felicia filifolia	ASTERACEAE	Low shrub	1	2	3	3	5	1	6	8	**29**	**M**
Helichrysum cymosum subsp. *cymosum*	ASTERACEAE	Low shrub	1	2	4	2	3	1	6	8	**27**	**M**
Helichrysum patulum	ASTERACEAE	Low shrub	1	2	3	2	4	1	8	8	**29**	**M**
Hermannia hyssopifolia	MALVACEAE	Low shrub	2	7	6	4	2	1	0	6	**28**	**M**
Lachenalia fistulosa	HYACINTHACEAE	Geophyte	1	7	7	4	4	1	6	8	**38**	**H**
Lampranthus emarginatus	MESEMBRYANTHEMACEAE	Succulent shrub	1	2	1	2	4	1	2	8	**21**	**M**
Tenaxia stricta	POACEAE	Graminoid	1	3	2	2	4	1	2	8	**23**	**M**
Moraea bellendenii	IRIDACEAE	Geophyte	1	1	3	1	2	1	8	8	**25**	**M**
Myrsine africana	MYRSINACEAE	Tall shrub	3	1	3	2	2	1	2	2	**16**	**L**
Ornithogalum thyrsoides	HYACINTHA-CEAE	Geophyte	1	7	3	5	4	1	8	8	**37**	**H**
Othonna arborescens	ASTERACEAE	Succulent shrub	1	4	3	3	2	1	6	8	**28**	**M**
Pelargonium cucullatum subsp. *tabulare*	GERANIACEAE	Low shrub	2	3	3	3	3	1	2	2	**19**	**L**
Pentameris airoides subsp. *airoides*	POACEAE	Graminoid (annual)	1	5	3	3	3	1	0	6	**22**	**M**
Podalyria sericea	FABACEAE	Low shrub	1	3	2	4	4	4	8	8	**34**	**H**
Ruschia rubricaulis	MESEMBRYANTHEMACEAE	Succulent shrub	1	3	2	3	6	4	0	8	**27**	**M**
Salvia africana-caerulea	LAMIACEAE	Low shrub	2	3	7	3	4	1	6	6	**32**	**H**
Searsia laevigata var. villosa	ANACARDIACEAE	Tall shrub	2	1	2	2	7	1	0	2	**17**	**L**
Searsia tomentosa	ANACARDIACEAE	Tall shrub	3	2	3	4	7	1	0	2	**22**	**M**
Themeda triandra	POACEAE	Graminoid	1	3	3	2	2	1	4	2	**18**	**L**
Trachyandra muricata	ASPHODELACEAE	Geophyte	1	3	7	4	6	1	2	2	**26**	**M**
Ursinia anthemoides subsp. *anthemoides*	ASTERACEAE	Forb (annual)	1	3	1	3	3	1	4	6	**22**	**M**

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2.7 Discussion

The majority of species exhibited moderate to high levels of viability and germinability. The vast majority of the 31 species were comparable with the 'reference' germinability yet just over half of the species achieved germinability in excess of 50%. The latter finding raises some concern in so far as germinability is the most accurate indicator for likely in-field performance. What is emphasised by these findings is the need to ascertain the physiological status of each species prior to large-scale collection and secondly the need to build to the knowledge base of these generally little-studied species towards informing the selection of likely high-yield species. Analysis of the key-restoration-species indices revealed that the majority of species scored in the moderate range which was represented by all six of the growth forms while just over a quarter of the species scored in the high range which was represented by four of the six growth forms. This discussion firstly addresses the four sets of comparisons, each defined by 'problem' thresholds, and the issues brought to the fore by each comparison. Thereafter, the criteria relevant to the evaluation of key restoration species are discussed.

2.7.1 'Reference' germinability : viability : germinability : X-ray viability

A comparison of the different seed quality tests is useful in identifying the underlying factors responsible for discrepancies in performance. Through a comparison of the results relative to the test-condition variables (pre-treatments, environmental parameters and growing-medium) one may develop a better understanding of the possible reasons for observed poor performance (for example poor seed quality, seed dormancy) among species. This information is of great value to restoration as it may be used to inform how and when to make the most efficient use of a species or when further experimentation into the germination cue requirements of a species is necessary.

Comparison of 'reference' germinability with viability

Roughly a quarter of the species were detected by this comparison, indicating that three quarters of the species achieved a comparable level of viability and of these, 16 species had viability greater than 75% (Appendix 2.4). In addition to the test-condition variables (such as different pre-treatments, environmental parameters and growing-medium), other variables relevant to inter-seed lot comparisons of this nature are linked to factors affecting seed quality such as genetic variability within a population and among populations; variability of resource limitation affecting seed production (the winter prior to seed collection for this study was considerably drier than the five year mean, receiving 42% less rainfall) (SAWS, 2013); seed age (the age of the 'reference' seed is unknown whilst the age of the project seed ranges from seven months to two days); timing of seed collection; seed infestation by insects; and, damage in processing and storage.

To reiterate, with the exception of three of the 31 species, the pre-treatments and parameters implemented in the viability test were done so as to mimic in as far as possible the glasshouse conditions of the germinability test (and conditions of the in-field experiment). For this reason, methodologies of the 'reference' tests and viability tests differ, to a greater or lesser degree with respect

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to the test conditions, for the majority of the 31 species. In the viability tests, all species were soaked in smoke extract (containing growth stimulants) whilst none of the species in the 'reference' tests was soaked.

In addition to the factors affecting seed quality, considering the family-specific traits and test-condition variables (pre-treatments, environmental parameters and growing-medium) helps to understand species performance. Low seed viability may be a family-specific trait, for instance, Poaceae and Asteraceae are known to produce a high number of empty seeds (Way and Gold 2003) and this trait may account for the relatively lower seed viability of *Athanasia crithmifolia, Eriocephalus africanus var. africanus, Pentameris airoides* subsp. *airoides* and *Themeda triandra.* As the glasshouse germinability of *Eriocephalus africanus var. africanus* was greater than the viability, this indicates that laboratory germination conditions were not optimal. In the 'reference' test, the species germinated better under cooler conditions. The low viability of *Pentameris airoides* subsp. *airoides* was unexpected (P Linder, personal communication, March 15, 2013) as it is an annual species and dependant on rapid seed-based recruitment. Considering its X-ray viability was very high, imperfect viability test conditions are more likely the cause of low viability. In the 'reference' germinability test for *Themeda triandra*, the pericarp was excised whilst no similar pre-treatment occurred in the viability test. *Themeda triandra* generally germinates in its second year after sowing, which, considering the seed coats were not nicked with a scalpel in the viability test, is a likely reason for the poor result. The proxy 'reference' test for *Erepsia anceps* is for *E. aspera* (a Western Cape species) and records nicking the seed coat which was absent from the viability test, and this disparity in pre-treatments is most likely responsible for the large difference of almost 90%. Furthermore, as with *Lampranthus emarginatus*, capsules from the previous year may have been collected resulting in old, deteriorated seed. Seed produced by the Malvaceae family have impermeable seed coats (Dickie and Stuppy 2003) yet since the seed coats for *Hermannia hyssopifolia* were nicked in the viability test and the seeds subsequently placed in 25°C (conditions similar to the 'reference' test which achieved 100% germination), the differential of almost 40% must be attributed to other factors affecting seed quality. This in corroborated by the X-ray viability, where over 40% of the seeds appeared partially-filled or empty. *Searsia* species are generally recorded as having low viability resulting from high levels of insect infestation or partially-filled seeds (Newton 2012). These factors are most likely responsible for only one seedling of *Searsia laevigata var. villosa* emerging in the viability test and zero seedlings, from either *Searsia laevigata var. villosa* or *S. tomentosa* emerging in the glasshouse. The seed of *Searsia laevigata var. villosa* was fresh when viability was tested while the seed from *S. tomentosa* was harvested when already mature and dry on the plant.

Comparison of viability with germinability

In the majority of germinability tests, some seeds fail to germinate either due to factors pertaining to poor seed quality or to factors relating to dormancy (Kamra 1976). As per definition, the value of viability is expected to exceed germinability and in this comparison, roughly two thirds of the species avoided detection by achieving germinability within 25% of the viability. In considering the potential factors responsible for the poor germinability performance of a third of the species detected by the

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'problem' threshold, because the seed samples for both tests were from the same seed lot, factors pertaining to variable seed quality can be precluded which instead implicates test-condition variables (pre-treatments, environmental parameters and growing-medium), family-specific traits and/or the interpretation of the viability tests (cut and TZ) as the potential factors responsible for discrepancies. Cross examination of the variables in the respective tests will help understand the drivers responsible for sustaining or alleviating seed dormancy in unequal measures. The potential test-condition variables, resulting from conditions in the laboratory versus those in the glasshouse include: water (variable moisture regime in glasshouse due to hand-watering versus continual moisture supply by the 1% agar growing medium in the laboratory); temperature (variable temperature in the glasshouse versus the controlled incubator temperature of 20/10°C 8/16 hrs); light (indirect light in the glasshouse due to seed being sown into soil versus the direct light in the laboratory); substrate (seed sown into Kirstenbosch general mix versus 1% agar in the laboratory) and smoke (the glasshouse seed trays were smoked whilst in the laboratory seeds were soaked overnight in smoke extract solution containing germination stimulants). The ideal sowing period for these species is March to April when diurnal fluctuations are greater than in May (A Hitchcock, personal communication, March 14, 2013 and F Powrie, personal communication, March 15, 2013), consequently sowing in May 2012 may have affected germinability in general. The monthly mean minimum and maximum temperatures for Kirstenbosch weather station for 2012, for the relevant five month period from May 2012 to September 2012, indicate that each monthly mean was marginally cooler than that of the preceding four years and that the mean minimum and maximum temperatures across the whole five month period of 8.6 ºC (± 0.8) and 17.62 ˚C (± 1.5) (SAWS, 2013) respectively, were marginally cooler than the temperature regime used in the viability test. The lower temperatures in general may have affected germinability in some of the more temperature sensitive species.

Variables additional to the test-condition variables, potentially contributing to low germinability relative to viability, predominantly involve family-specific traits and the difficulty in interpreting TZ test staining. *Dimorphotheca pluvialis,* reported as having a viability of 88% may, depending on the interpretation of TZ staining, have had a much lower viability of 11% if the dark staining was indicative of seed bruising, a viability value more in line with the germinability value. Possible reasons for poor germination in *D. pluvialis* include decreased drainage resulting from the use of general mix compared with the fynbos mix (which has a higher proportion of sand) and the time of sowing (mid-May 2012) may have been contributing factors (A Hitchcock, personal communication, March 14, 2013). Despite these potentially inhibitive factors, expected germinability would still be in excess of 80%, particularly for seed collected from the Cape Peninsula, and that poor drainage and late sowing would be expected to impact seedling vigour more than it would seedling emergence (L van der Walt, personal communication, March 15, 2013). Due to poor initial germination of *Hermannia hyssopifolia* in the laboratory, and in the knowledge of seed coat impermeability for Malvaceae seeds (Dickie and Stuppy 2003), the seed coats were subsequently nicked in the laboratory (but not in the glasshouse) and this difference in pre-treatments is most likely responsible for the significant difference of almost two thirds viability versus a complete failure to germinate in the glasshouse. Similarly, the *Lachenalia fistulosa* seeds were chipped in the viability test but not in the germination test and is most likely the

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reason for this near 50% disparity. In the viability test, *Lampranthus emarginatus* seeds were dissected and stained with TZ. The viability dropped in the region of 60% for either interpretation of the TZ test, whether the pink-stained embryos were indicative of viability or non-viability, rendering the TZ-adjusted viability and germinability values more comparable. A possible reason for the low viability and germinability may be related to the collection of capsules produced the previous year containing deteriorated seed. There were no pre-treatment variables for the tests of *Tenaxia stricta* and *Pentameris airoides* subsp*. airoides,* implicating the other test-condition variables as responsible for poor performance, in particular the poor drainage resulting from the general mix (A Hitchcock, personal communication, March 14, 2013). For *Myrsine africana,* the pre-treatment of removing the outer covering structure was replicated in both tests yet viability considerably exceeded germinability. Taking the TZ staining into account, however, the viability dropped dramatically by over 90% to the extent where the TZ-adjusted viability becomes exceeded the germinability by almost 25%, indicating the TZ interpretation too conservative. *Myrsine africana* is generally a poor germinator (A Hitchcock, personal communication, March 14, 2013) and usually germinates in its second year and both these traits are likely reasons why performance was poor. Many of the 'non-fynbos' shrubs and trees germinate well in the warmer months too, for instance from mid-October to mid-January, (A Hitchcock, personal communication, March 14, 2013), and this may be why the germination of the species in the glasshouse started notably after most other species and seedling emergence was highest in February 2012 (nine months since sowing) when the final germinability data were collected. The four and seven month duration of the viability and germinability tests respectively may consequently have been too short for accurate results of this species. The *Pelargonium cucullatum* subsp. *tabulare* seeds were not nicked in either test. TZ staining suggested high viability and thus it is likely that germination was very low in both tests due to the physical seed dormancy not being broken in either. *Pelargonium* seeds are hard-coated (Holmes and Newton 2004) particularly in dry years and may only germinate in the second year (F Powrie, personal communication, March 15, 2013). The viability of *Trachyandra muricata* dropped by almost a third when interpreting the TZ staining yet still far exceeded germinability. As the pre-treatment variables from both the viability and germinability tests were comparable, the difference of over 50% can be attributed to other test-condition variables. Despite wide ethnobotanical use of the species (Philander 2011), no *Trachyandra* species are currently cultivated, although cultivation is expected to be non-problematic (SANBI 2013). The TZ interpretation for *Ursinia anthemoides* subsp. *anthemoides* resulted in a 30% reduction in viability which was consistent with the germinability value. The *Ruschia rubricaulis* seeds were not nicked in either the viability or germinability tests. Only four seedlings emerged in the viability test and the TZ adjusted viability values of *Ruschia rubricaulis* remained low whether considering the pink-stained embryos viable or non-viable. However, zero seedlings emerged in the nursery, suggesting the difference here can again be attributed to the other test-condition variables (environmental parameters and growing-medium). In contrast to the *Lampranthus emarginatus* and *Erepsia anceps* capsules, those collected for *Ruschia rubricaulis* were from the immediately preceding season and full of large, red seed and thus ruling out deteriorated seed as a possible explanation. The poor staining

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response to TZ is therefore more likely attributed to poor genetic fitness of the single, small, isolated population on Signal Hill from which it was collected.

Where germinability exceeded viability for four species the magnitude of the difference was low. The pre-treatments for all species were comparable in both tests so this trend is unexpected and results from the glasshouse variables (environmental parameters and growing-medium) favouring germination over the laboratory conditions. Compost inhibits fungal growth more so than agar (Gosling 2003) and may be a factor contributing to the relatively higher germinability values.

Comparison of X-ray viability with viability or germinability

As X-rays do not differentiate between living or dead tissue (Gosling 2003), where X-ray viability exceeds viability or germinability or both, this discrepancy may be attributed to TZ interpretation or may simply reflect poor seed quality. For instance a fully-formed seed that has deteriorated and lost viability over time may still appear viable on X-ray. The germinability of *Erepsia anceps* was considerably exceeded by the X-ray viability. The 'reference' test for *Erepsia aspera,* the proxy species for *Erepsia anceps*, recorded nicking the seed coat, and this pre-treatment not being implemented in the germinability test is most likely the largest contributing factor for the large discrepancy. The collection of old capsules may in addition be a plausible explanation for poor performance. *Pentameris airoides* subsp. *airoides* had X-ray viability of almost 100% yet a viability of almost half this. As no seedlings emerged in the glasshouse, despite pre-treatments of the viability and germinability tests being comparable, this suggests the species is sensitive to the environmental variables and growing-medium.

2.7.2 Germinability of less than 50%

While just over half of the species achieved germinability in excess of 50%, nearly half of the species failed to achieve this and five of the species failed to germinate at all. Germinability outcomes were thus very variable among species. As germinability is the most accurate indicator of likely in-field performance and thus restoration success, this very variable performance highlights the importance of determining the physiological status of seed prior to large-scale restoration efforts and furthermore the need to accrue species-specific data of this nature so as to inform the species selection process towards targeting likely high-yield species for future restoration efforts.

In cases where germinability was low, the quality of the seed may have been poor or the conditions required to stimulate germination were not met. In some cases, where little is known about the germination-cue requirements of a taxon, further experimentation, testing different pre-treatments under a range of conditions, is required. The X-ray viability of *Arctopus echinatus* revealed poor seed density in the majority of seeds in part explaining the low germinability. Roughly a quarter of the seeds were fresh on dissection yet stained very poorly with TZ. Despite the ethnobotanical importance of *Arctopus echinatus* (Magee et al. 2008) (common name "Siekentroos' translating to 'comforts the sick', has a broad spectrum medicinal use by the Khoi and European settlers and it is still available in Cape Town markets) (van Wyk and Gerike 2007), the species is horticulturally insignificant and there is currently no information regarding its cultivation (SANBI 2013). The remaining 14 species,

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performing poorly with respect to germinability, occurred in one or more of the problem areas discussed in the sections above. Although difficult to identify the source of variation (attributable to the numerous variables affecting seed quality and germination), it is interesting to note that in germination tests conducted by Midoko-Iponga (2004) on commercially sourced seed for *Eriocephalus africanus* and *Ehrharta calycina*, germinability measured 12.5% and 73.68% respectively, both of these values being lower than those measured for the same species in this study (62.4% and 85.6% respectively).

2.7.3 Criteria for a key restoration species

The model proposed in this portion of the study draws together criteria that are scattered in the literature yet frequently emphasized as important considerations for the selection of species for restoration. The criteria are relevant over a range of scales from the ecosystem- and population-scale through to seed-biology- and seed-ecology-scales. The criteria selected for inclusion in the model speak broadly to the appropriateness of a species for restoration and encompass themes of genetic integrity and diversity, source population resilience, conservation imperative, seed collection 'ease', and performance criteria based on seed physiology. These elements have been pulled together in what aims to be a simple and efficient rubric designed to address these complex decisions relevant to species selection for use in restoration.

Making use of local provenance seed is recommended for ensuring genetic integrity and improving the likelihood of local genotypes establishing and surviving in a restored community (Holmes and Richardson 1999, Hufford and Mazer 2003, Bischoff et al. 2006, Bautista et al. 2009, Mijnsbrugge et al. 2010, Clewell and Aronson 2013). Despite a preference for matching the habitats of the source population with that of the receiving population (Wilkinson 2001, Hufford and Mazer 2003, Way 2003, Bautista et al. 2009, Mijnsbrugge et al. 2010), using *proximity of the seed source population to the restoration site*, despite its limitations, is a practical surrogate for promoting the collection of locally adapted genotypes. Caution against collecting too narrowly within a population is made in order to prevent genetic 'bottlenecks' (Kaye 2001, Hufford and Mazer 2003, Mijnsbrugge et al. 2010) and ensure the collection is a good genetic representation of the whole population (Way 2003). The *population size* and *extent of area* of the source population are relevant indicators in this regard. *Population size,* in addition, is an indicator of the resilience of a population to the effects of seed harvesting (Broadhurst et al. 2008). This criterion is based on the premise that the greater the number of the individuals, the greater the ability of the population to absorb the negative impacts of seed harvesting (Broadhurst et al. 2008). The *conservation status* of the species is a criterion worth building into the model due to the conservation imperative and benefits associated with establishing a novel population of a species of conservation concern (Falk et al. 1996). Here, the consideration of the size of the population and the resilience of the population to harvesting is crucial as the benefits of attempting to establish a novel population must distinctly outweigh the negative impacts associated with compromising the source population through seed harvesting.

Germinability and viability of species are good indicators of likely and potential performance in restoration. As germinability test conditions and on-site conditions are more comparable, one may

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infer that germinability is likely to be a better indicator of likely on-site success than viability. In addition to incorporating germinability into the model, the viability of a species is worth including particularly if germinability is very low. Low germinability and high viability is likely to be indicative of a lack of understanding of the dormancy breakage and germination cue requirements of a species, a species which does, however, have a great potential to perform. Poor performance in both viability and germinability simply reflects poor seed quality and is consequently an accurate contributor to a low key-restoration-species index. The criteria *plant abundance* and *number of seeds per individual* relate to the time spent in the field with respect to locating plants and collecting the seed and consequently the degree of collection ease, efficiency and cost-effectiveness, practical aspects of paramount relevance.

The bulk of the species occurred in the mid key-restoration-species index range which was spread across all six growth forms, raising no concerns. However, five species were conspicuous for their low indices. Common traits amongst these species were their low to moderate plant abundance and their poor performance with respect to viability and/or germinability which contributed to their low scores. At the species level, of the five species, these findings are not unexpected in that germination may typically occur in the second year (for example *Themeda triandra, Pelargonium cucullatum* and *Myrsine africana*), poor viability is a trait (for example *Searsia laevigata*) or the species is horticulturally unknown and morphological dormancy may play a role in poor performance (for example *Arctopus*). In contrast, the eight species falling into the high index category exhibited very good performance with respect to viability and/or germinability which contributed to their high overall scores. Four out of the six growth forms were represented in this category, the succulent shrub and forb growth forms being absent from this range, yet all five succulent shrubs species and two of the three forb species fell into the mid-category. All of the high-scoring species (except *Lachenalia fistulosa* and *Athanasia crithmifolia*) were 'non-problematic' species and avoided detection by the performance-related test comparisons. *Lachenalia fistulosa* and *Athanasia crithmifolia* redeemed points by being, amongst other attributes, either common or abundant, having large populations and exhibiting moderate to high levels of germinability and viability.

The criteria incorporated into the key-restoration-species model, in temporal-scale, cover the phases of pre-seed collection planning, seed collection and pre-restoration performance, and in this instance the bulk of these phases were completed more or less within a year. The scope and temporal-scale of this key-restoration-species model focuses on the building blocks of a restoration project but do not extend to encompass certain ecological themes which are of the utmost relevance to ecological restoration efforts. There are additional post-sowing, performance-related criteria, relevant to community assembly dynamics and ecosystem processes, which come into play over a longer time-frame than the two years of this particular project and which are very relevant to a more accurate evaluation of just how useful these species are in restoration. Different species make contributions of variable importance to ecosystem functioning (Diaz and Cadibo 2001) making the longer-term monitoring of species with respect to their role within the community relevant to their selection for restoration. Pywell et al. (2003) proposed trait-based performance evaluation of species in terms of their ability to colonize, competitive interactions, regeneration strategy and habitat associations.

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Although this is beyond the scope of this project, how each of the 31 introduced species performs and interacts over the longer-term, will have an impact on the kind of community that assembles which in turn will have bearing on how desirable the species is for inclusion in future restoration projects.

Expanding the proposed model through the addition of longer-term monitoring data incorporating performance-related seedling ecology information, such as the criteria proposed by Pywell et al. (2003), would dramatically increase its value to large-scale restoration. Increasing the span of such a model would raise the overall accuracy of the evaluation of species performance and thus provide a more authentic measure of its appropriateness for use in restoration efforts. Further to expanding the model over time, the rubric may in addition be tailored to address project-specific needs. In this instance, the effect sizes and corresponding scores of the model are relevant to these 31 species and to this ecosystem and would require revision if employed for other species in other ecosystems. The eight assessment criteria, however, remain relevant to all restoration projects but are by no means exhaustive. Based on project needs, additional criteria may easily be incorporated into the model, for instance, a project applying the alien niche-preoccupation theory proposed by Funk et al. (2008) would be in the position to introduce assessment criteria that would bring to light species with the traits appropriate for satisfying the project-specific restoration agenda.

2.8 Conclusion

Seed-based restoration experimentation in renosterveld has achieved limited success. As restoration is in part reliant on quality seed to ensure successful germination and establishment of seedlings, ascertaining the physiological status of seed through viability and germinability testing at an early stage, to raise concerns or affirm likely in-field performance, is critical to restoration outcomes. In selecting species for restoration the literature points to the need to consider informants at a range of scales from ecosystem- and population-scale considerations through to seed physiology considerations.

The outcomes from the seed physiology tests, with respect to both germinability and viability, were variable and emphasize the sense in ascertaining seed quality prior to embarking on large-scale seed collection and restoration. Detailed consideration of the factors underpinning large discrepancies between viability, which was generally high, and germinability, highlights the role of dormancy-alleviation in achieving germination success and demonstrates the value of this level of testing and cross-testing in understanding seed physiology and how important this is in informing the 'when' and the 'how' of restoration efforts. In a few instances, findings point to a need for more work towards understanding unexplained outcomes and discrepancies. Here the value of cross-referencing is clearly demonstrated. Future work could use these findings as a point of departure from which to proceed down particular family lines, or within certain functional types to start exploring for patterns of responses.

The key-restoration-species model makes a first attempt at drawing together a number of suggested criteria into one single approach in the challenging space of species selection. In this study it proved useful where the vast majority of species exhibited moderate to high potential. The model in addition pulled the poor-performing species into focus, prompting consideration of the trade-off

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between collection-time in the field versus likely reward in the context of how desirable an element the species is for the future community. The model, while relevant to the specific parameters of this study, warrants expansion, further examination and testing on both these species in this ecosystem and other species in other ecosystems. One starting point might be to use it on older restoration studies, incorporating criteria reflecting longer-term in-field performance, to provide a more faithful measure of the usefulness of species and furthermore to see if it holds out over time.

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2.9 References

Baskin, J., M. and Baskin, C. C. 2003. Classification, biogeography, and phylogenic relationships of seed dormancy. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice.* Kew: Royal Botanic Gardens, pp. 517-544.

Bautista, S., Aronson, J. and Vallejo, R.V. 2009. *Land Restoration to Combat Desertification: Innovative Approaches, Quality Control and Project Evaluation.* Spain: Fundación Centro de Estudios Ambientales del Mediterráneo.

Benech Arnold, R.L., Fenner, M. and Edwards, P.J. 1991. Changes in germinability, ABA content and ABA embryonic sensitivity in developing seeds of *Sorghum bicolor* (L.) Moench. induced by water stress during grain filling. *New Phytol*, 118, pp. 339-347.

Bhaskar, A. and Vyas, K.G. 1988. Studies on competition between wheat and *Chenopodium album* L. *Weed Research*, 28(1), pp. 53-58.

Bischoff, A., Vonlanthen, B. Steinger, T. and Muller-Scharer, H. 2006. Seed provenance matters: Effects on germination of four plant species used for ecological restoration. *Basic and Applied Ecology*, 7, pp. 347 - 359.

Blakesley, D., Anusarnsunthorn, V., Kerby, J., Nnavakitbumrang, P., Kuarak, C., Zangkum, S., Hardwick, K. and Elliot, S. 2000. Nursery technology and tree species selection for restoring forest biodiversity in northern Thailand. In: S., J. Elliott, D. Kerby, K. Blakesley, K. Hardwich, V. Woods and V. Anusarnsunthorn, (eds). *Forest Restoration for Wildlife Conservation,* pp.207-222.

Broadhurst, L. M. Lowe, A., Coates, D. J., Cunningham, S. A. McDonald, M., Vesk, P. A., and Yates, C. 2008. Seed supply for broadscale restoration: maximizing evolutionary potential. *Evolutionary Applications,* pp. 587–597.

Brown, N. A. C. and Botha, P. A. 2004. Smoke seed germination studies and a guide to seed propagation of plants from the major families of the Cape Floristic Region, South Africa. *South African Journal of Botany*, 70, pp. 559-581.

Brown, N. A. C., Botha, P. A., Johnson, T. and Prosch, D. S. 2000. *Propagation of Cape Wildflowers From Seed*. Paper presented at the Fifth International Botanic Gardens Conservation Congress.

Brown, N. A. C., van Staden, J., Johnson, T. and Daws, M., I. 2003. A Summary of Patterns in the Seed Germination Response to Smoke in Plants from the Cape Floral Kingdom. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 563-574.

Chambers, J.C. 1989. Seed Viability of Alpine Species: Variability within and among Years. *Journal of Range Management,* 42(4), pp. 304-308.

Cione, N.K., Padgett, P.E. and Allen, E.B. 2002. Restoration of a Native Shrubland Impacted by Exotic Grasses, Frequent Fire, and Nitrogen Deposition in Southern California. *Restoration Ecology*, 10(2), pp. 376-384.

Clewell, A.F and Aronson, J. 2013. *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession*, *second edition.* Washington: Island Press.

Crookes, D.J., Blignaut, J.N., de Wit, M.P., Esler, K.J., le Maître, D.C., Milton, S.J., Mitchell, S.A., Cloete, J., de Abreu, P., Fourie (nee Vlok), H., Gull, K., Marx, D., Mugido, W., Ndhlovu, T., Nowell, M., Pauw, M. and Rebelo, A. 2013. System dynamic modelling to assess economic viability and risk trade-offs for ecological restoration in South Africa. *Journal of Environmental Management*, 120, pp. 138-147.

Del Moral, R., Walker, L. R. and Bakker, J. P. 2007. Insights Gained from Succession for the Restoration of Landscape Structure and Function. In: L.R. Walker, J. Walker and R.J. Hobbs, (eds). 2007. *Linking Restoration and Ecological Restoration.* New York: Springer Science.

Diaz, S. and Cadibo, M. 2001. Vive la différence: plant functional diversity matters to ecosystem processes. *Elsevier Science,* 16(11), pp. 646-655.

University of Cape Town

Diaz, S., Cabido, M. and Casanoves, F. 1998. Plant Functional Traits and Environmental Filters at a Regional Scale. *Journal of Vegetation Science*, 9(1), pp. 113-122.

Dickie, J. B. and Stuppy, W. H. 2003. Seed and fruit Structure. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 255-279.

Donaldson, J., Nänni, I., Zachariades, C. and Kemper, J. 2002. Effects of habitat fragmentation on pollinator diversity and plant reproductive success in renosterveld shrublands of South Africa, *Conservation Biology*, 16(5), pp. 1267-1276.

Euston-Brown, D. 2011. *Description of natural vegetation on Groote Schuur Estate, Cape Town*. University of Cape Town, unpublished report.

Falk, D.A., Millar, C.I. and Olwell, M. 1996. *Restoring Diversity: Strategies for Reintroduction of Endangered Plants*. USA: Island Press.

Fenner, M. 1992. Environmental influences on seed size and composition. *Horticultural Reviews*, 13, pp. 183-213.

Fenner, M. and Thompson, K. 2005. *The Ecology of Seeds*. Cambridge: Cambridge University Press.

Funk, J. L., Cleland, E. E., Suging, K. N. and Zavaleta, E. S. 2008. Restoration through reassembly: plant traits and invasion resistance. *Trends in Ecology and Evolution*, 23(12), pp. 695-703.

Gold, K. 2008. *Post-harvest handling of seeds*, Technical Information Sheet 04, Seed Conservation Department, Kew: Royal Botanic Gardens.

Google Earth, 2011. Image date December 16, 2011, accessed September 2012 to February 2013. California, USA.

Goosem, S. P. and N. I. J. Tucker, 1995. *Repairing the Rainforest – Theory and Practice of Rainforest Re-establishment in North Queensland's Wet Topics*. Cairns: Wet Tropics Management Authority.

Gosling, P. 2003. Viability Testing. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Pritchard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 445-481.

Gray, D., Steckel, J.R.A. and Ward, J.A. 1986. The Effect of Cultivar and Cultural Factors on Embryo-sac Volume and Seed Weight in Carrot (*Daucus carota* L.). *Annals of Botany*, 58(5), pp. 737-744.

Guerrant E. O Jr., Havens K. and Maunder M (eds). 2004. *Ex situ plant conservation*. Washington: Island Press.

Hartmann, H. T. and Kester, D. E, (eds). 2011. *Plant propagation: Principles and practices*. Upper Saddle River: Prentice Hall.

Hay, F. R. and Smith, R. D. 2003. Seed Maturity: when to collect seeds from wild plants. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 97-133.

Heelemann 2012. *Renosterveld remnants - Current ecological situation and restoration perspectives*. PhD thesis, University of Stellenbosch.

Hobbs, R. and Cramer, V. 2008. Restoration Ecology: Interventionist Approaches for Restoring and Maintaining Ecosystem Function in the Face of Rapid Environmental Change. *Annual Review of Environment and Resources,* pp. 39-61.

Hobbs, R. and Harris, J. 2001. Restoration ecology: Repairing the Earth's ecosystems in the new millennium. *Restoration Ecology*, 9(2), pp. 239-246.

Holl. K., Pejchar, L. and Whisenant, S. 2007. Overcoming Physical and Biological Obstacles to Restore Natural Capital. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice*. Washington: Island Press, pp. 249-255.

Holmes, P. M. 2001. Shrubland Restoration Following Woody Alien Invasion and Mining: Effects of Topsoil Depth, Seed Source, and Fertilizer Addition. *Restoration Ecology*, 9(1), pp. 71-84.

University of Cape Town

Holmes, P. M. 2002a. *Renosterveld Restoration on Cultivated Lands at Eerste River, Western Cape: Preliminary Findings of a Pilot Study*, unpublished report to Lafarge Quarries, May 2002.

Holmes, P. M. 2002b. Depth distribution and composition of seed-banks in alien-invaded and uninvaded fynbos vegetation. *Austral Ecology*, 27, pp. 110–120.

Holmes, P. M. 2005. Results of a lucerne old-field restoration experiment at the Fynbos-Karoo interface. *South African Journal of Botany,* 71, pp. 326-338.

Holmes, P. M. and Richardson, D. M. 1999. Protocols for restoration based on Recruitment Dynamics, Community Structure, and Ecosystem Function: Perspectives from South African Fynbos. *Restoration Ecology*, 7(3), pp. 215-230.

Holmes, P.M and Newton, R.J. 2004. Patterns of Seed Persistence in South African Fynbos. *Plant Ecology*, 172(1), pp. 143-158.

Holmes, P.M. 2008. Optimal ground preparation treatments for restoring lowland Sand Fynbos vegetation on old fields. *South African Journal of Botany*, 74, pp. 33-40.

Hufford, K. M. and Mazer, S, J. 2003. Plant ecotypes: genetic differentiation in the age of ecological restoration. *Trends in Ecology and Evolution*, 18(3), pp.147-155.

Joubert, C. 1991. *History and description of contemporary vegetation, Signal Hill, Cape Town*. MSc. University of Cape Town.

Kamra, S. K. 1976. *Use of X-ray radiography for studying seed quality in tropical forestry*. Sweden: Department of Reforestation, Royal College of Forestry.

Kane, M. and Cavers, P.B. 1992. Patterns of seed weight distribution and germination with time in a weedy biotype of proso millet (*Panicum miliaceum*). *Canadian Journal of Botany*, 70(3), pp. 562-567.

Kaye, T.N.2001.Common ground and controversy in native plant restoration: The SOMS debate, source distance, plant selections, and a restoration oriented definition of native. In D. L. Haase and R. Rose, (eds), *Proceedings: native plant propagation and restoration strategies.* Nursery Technology Cooperative and Western Forestry and Conservation Association, pp. 5–11.

Keller , L.F. and Waller, D.M. 1992. Inbreeding effects in wild populations. *Trends in Ecology and Evolution*, 17(5), pp. 230-241.

Khan, B.A. and Stofella, P.J. 1985. Yield Components of Cowpeas Grown in Two Environments. *Crop Science*, 25, pp. 179-182.

Kiniry, J.R. and Musser, R.L. 1988. Response of Kernel Weight of Sorghum to Environment Early and Late in Grain Filling. *Agronomy Journal*, 80(4), pp. 606-610.

Knowles, O.H. and Parrotta, J.A, 1995. Amazonian forest restoration: an innovative system for native species selection based on phenological data and field performance indices. *Commonwealth Forestry Review*, 74(3). pp. 230-243.

Krauss, S., L. and He, H., T. 2006. Rapid genetic identification of local provenance seed collection zones for ecological restoration and biodiversity conservation. *Journal for Nature Conservation* 14, pp. 190-199.

Krug, C. 2004. *Practical Guidelines for the Restoration of Renosterveld*, University of Stellenbosch.

Lamb, D., J. Parrotta, R. Keenan, and N. I. J. Tucker, 1997. Rejoining habitat remnants: restoring degraded rainforest lands. In: W. F. Laurence and R. O. Bierregaard Jr. (eds) *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities*. Chicago: University of Chicago Press, pp. 366-385.

Lamont, B.B., Klinkhamer, P.G.L. and Witowski, E.T.F. 1993. Population Fragmentation May Reduce Fertility to Zero in *Banksia goodii:* A Demonstration of the Allee Effect. *Oecologia*, 94(3), pp. 446-450.

Lavorel, S. and Garnier, E. Predicting changes in community composition and ecosystem functioning from plant traits: revisiting the Holy Grail. *Functional Ecology*, 16, pp. 545–556.

University of Cape Town

Le Maître, D. and Midgely, J. J. 1992. Plant reproductive ecology. In: R.M. Cowling, (ed). *The ecology of Fynbos: nutrients, fire and diversity*. Cape Town: Oxford University Press, pp. 135-174.

Linington, S.H. 2003. The design of seed banks. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Pritchard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 593-636.

Linington, S.H. and Manger, K. 2009a. *Seed bank design: seed drying rooms*, Technical Information Sheet 11, Seed Conservation Department, Kew: Royal Botanic Gardens.

Lippitt, L., Fidelibus, M. W. and Bainbridge, D. A. 1994. Native Seed Collection, Processing, and Storage for Revegetation Projects in the Western United States. *Restoration Ecology*, 2(2), pp. 120-131.

Lovett-Doust, J. and Lovett-Doust, L. 1988. *Plant Reproductive Ecology: Patterns and Strategies*. Oxford University Press: UK.

Magee, A. R, van Wyk, B. E., Tilney, P. M. and van der Bank, M. 2008. A taxonomic revision of the South African endemic genus *Arctopus* (Apiaceae, Saniculoideae). *Annals of the Missouri Botanical Garden*, 95, pp.471-486.

Midoko-Iponga, D. 2004. *Renosterveld Restoration: The role of competition, herbivory and other disturbances*. MSc thesis, University of Stellenbosch.

Mijnsbrugge, K. V., Biscoff, A. and Smith, B. 2010. A question of origin: Where and how to collect seed for ecological restoration. *Basic and Applied Ecology*, 11, pp. 300-311.

Milberg P., Andersson L. and Thompson K. 2000. Large-seeded species are less dependent on light for germination that small seeded ones. *Seed Science Research,* 10, pp. 99-104.

Montalvo, A.M. and N.C. Ellstrand. 2000. Transplantation of the subshrub *Lotus scoparius*: testing the home-site advantage hypothesis. *Conservation Biology*, 14, pp. 1034-1035.

Mortlock, W. 2008. Local seed for revegetation: Where will all that seed come from? *Ecological Management and Restoration,* 1(2), pp. 93-101.

Newton, R. 2012. *Report on seed germination of four Searsia species from SANParks*. Wakehurst: Royal Botanic Gardens Kew.

O'Connor, T.G. and Pickett, G.A. 1992. The Influence of Grazing on Seed Production and Seed Banks of Some African Savanna Grasslands. *Journal of Applied Ecology*, 29(1), pp. 247-260.

Philander, L. A. 2011. An ethnobotany of Western Cape Rasta bush medicine. *Journal of Ethnopharmacology*, 138, pp. 578-594.

Poschlod, P., J. Bakker, S. Bonn, and S. Fischer. 1996. Dispersal of plants in fragmented landscapes. In: J. Settele, C. Margules, P. Poschlod, and K. Henle, editors. *Species Survival in Fragmented Landscapes*. Kluwer Academic Publishers, London, pp. 123-127.

Probert R. J. 2003. Seed viability under ambient conditions and the importance of drying. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Pritchard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 337-365.

Pywell, R. F, Bullock, J., M., Roy, D., B. Warman, L., Walker, K., J. and Rothery, P. 2003. Plant traits as predictors of performance in ecological restoration. *Journal of Applied Ecology*, 40, pp. 65–77.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. and Rutherford, M, C. 2006 Fynbos Biome. In: L. Mucina, L. and M.C. Rutherford, (eds). *The Vegetation of South Africa, Lesotho and Swaziland.* Pretoria: South African National Biodiversity Institute.

RBG (Royal Botanic Gardens Kew). 2009. *Seed Information Database* (SID). Downloaded from http://data.kew.org/sid/sidsearch.html on 2012/03/24 and 2013/03/13.

Roach, D.A. and Wulff, R.D. 1987. Maternal Effects in Plants. *Annual Review of Ecology and Systematics*, 18, pp. 209-235.

University of Cape Town

Ruiters, C., McKenzie, B. and Raitt, L. M. 1993. Life-History Studies of the Perennial Geophyte *Haemanthus pubescens* L. Subspecies *Pubescens* Amaryllidaceae in Lowland Coastal Fynbos, South Africa. *International Journal of Plant Sciences*, 154(3), pp. 441-449.

Schmidt, L.H. and Thomsen, K.A. 2003. Tree Seed Processing. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Pritchard and R.J. Probert, (eds). *Seed Conservation: turning science into practice.* Kew: Royal Botanic Gardens, pp. 283-306.

Shibata, M., Tanaka, H., Iida, S., Abe, S., Masaki, T., Niiyama, K. and Nakashizuka, T. 2002. Synchronized Annual Seed Production by 16 Principal Tree Species in a Temperate Deciduous Forest, Japan. *Ecology*, 83(6), pp. 1727-1742.

Shiponeni, N.N. 2003. *Dispersal of seeds as a constraint of revegetation of old fields in renosterveld vegetation in the Western Cape, South Africa.* MSc thesis, University of Stellenbosch.

South African National Biodiversity Institute SANBI. 2011. *Red List of South African Plants.* [online] Available at: <http://redlist.sanbi.org/> Accessed 15 May 2011.

South African National Biodiversity Institute SANBI. 2013. *Plantzafrica.* [online] Available at: http://www.plantzafrica.com/index.html.

South African Weather Service (SAWS) 2013. *Temperature and precipitation data for Groote Schuur, Kirstenbosch, The Royal Astrological Observatory and Molteno Reservoir for 2008 to 2013.* Extracted 2013/02/05.

Stevenson, M. J., Ward, L. K. and Pywell, R. F. 2008. Re-creating Semi-natural Communities: Vacuum Harvesting and Hand Collection of Seed on Calcareous Grassland. *Restoration Ecology,* 5(1), pp. 66–76.

Terry, J. and Bertenshaw, K. 2008. *Cleaning seed collections for long-term conservation*, Technical Information Sheet 14, Seed Conservation Department, Kew: Royal Botanic Gardens.

Turnbull, L.A., Crawley, M.J. and Rees, M. 2000. Are Plant Populations Seed-Limited? A Review of Seed Sowing Experiments. *Oikos,* 88, pp. 225-238.

Vallejo, R.V., Allen, E.B., Aronson, J., Pausas, J.G., Cortina, J. and Gutierrez, J.R. 2012. Restoration of Mediterranean-type Woodlands and Shrublands. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition*. UK: Wiley-Blackwell, pp.130-144.

Van Wyk, B. and Gericke, N. 2007. *Peoples Plants: A Guide to Useful Plants of Southern Africa.* Pretoria: Briza.

Walton, B. J. 2006. *Vegetation patterns and dynamics of renosterveld at Agter-Groeneberg Conservancy, Western Cape, South Africa.* MSc thesis. University of Stellenbosch.

Way, M and Gold, K. 2008. *Assessing a potential seed collection,* Technical Information Sheet 02, Seed Conservation Department, Kew: Royal Botanic Gardens.

Way, M. J. 2003. Collecting seed from non-domesticated plants for long-term conservation. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice.* Kew: Royal Botanic Gardens, pp. 163-201.

Westoby, M., Leishman, M., Lord, J., Poorter, H. and Schoen, D.J. 1996. Comparative Ecology of Seed Size and Dispersal. Philosophical Transactions of the Royal Society of London, *Biological Sciences*, 351, pp. 1309-1318.

Whisenant, S. G 1999. *Repairing damaged wildlands*. Cambridge: Cambridge University Press.

Wilkinson, D. M. 2001. Is Local Provenance Important in Habitat Creation? *Journal of Applied Ecology*, 38(6), pp. 1371-1373.

Wulff, R.D. 1986. Seed Size Variation in *Desmodium Paniculatum*: I. Factors Affecting Seed Size. *Journal of Ecology,* 74(1), pp. 87-97.

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Appendix 2.1: X-ray viability summary results

Table 2.3: X-ray viability results.
Untransformed data, mean percentage ± standard deviation; n=100 (four replicates of 25 per species) except for Eriocephalus africanus where n=216.

CATEGORY

E: 0 – ½ filled (empty, unlikely to germinate)
P: ½ - ¾ filled (partially full, incompletely formed, may/may not germinate - depending on the species)
F: ¾ filled – full *(full, expect seed/s to germinate)
I: infested or physically damaged (may/may not germinate depending on the position and extent of damage)

SPECIES	FAMILY	GROWTH FORM		REP 1	REP 2	REP 3	REP 4	MEAN %	±SD
Arctopus echinatus	APIACEAE	Forb	E	0	1	1	2	4.0	3.27
			P	21	10	11	16	58.0	20.26
			F	4	14	13	7	*38.0	19.18
			I	0	0	0	0	0.0	0.00
Athanasia crithmifolia	ASTERACEAE	Tall shrub	E	4	4	8	6	22.0	7.66
			P	1	3	5	1	10.0	7.66
			F	20	17	12	18	*67.0	13.61
			I	0	1	0	0	1.0	2.00
Babiana fragrans	IRIDACEAE	Geophyte	E	2	0	1	0	3.0	3.83
			P	1	0	1	1	3.0	2.00
			F	22	25	23	24	*94.0	5.16
			I	0	0	0	0	0.0	0.00
Chironia baccifera	GENTIANACEAE	Low shrub	E	4	3	2	5	14.0	5.16
			P	2	5	2	0	9.0	8.25
			F	18	17	21	20	*76.0	7.30
			I	1	0	0	0	1.0	2.00
Chrysocoma coma-aurea	ASTERACEAE	Low shrub	E	0	2	2	2	6.0	4.00
			P	2	0	0	0	2.0	4.00
			F	23	23	23	23	*92.0	0.00
			I	0	0	0	0	0.0	0.00
Cymbopogon marginatus	POACEAE	Graminoid	E	0	0	0	0	0.0	0.00
			P	0	0	0	0	0.0	0.00

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			F	25	25	25	25	*100.0	0.00
			I	0	0	0	0	0.0	0.00
Dimorphotheca pluvialis	ASTERACEAE	Forb (annual)	E	0	0	1	1	2.0	2.31
			P	1	1	0	0	2.0	2.31
			F	25	24	23	24	*96.0	3.27
			I	0	0	0	0	0.0	0.00
Ehrharta calycina	POACEAE	Graminoid	E	0	0	0	0	0.0	0.00
			P	2	5	2	2	11.0	6.00
			F	23	20	23	23	*89.0	6.00
			I	0	0	0	0	0.0	0.00
Erepsia anceps	MESEMBRYANTHEMACEAE	Succulent shrub	E	0	0	0	0	0.0	0.00
			P	0	0	1	0	1.0	2.00
			F	25	25	24	25	*99.0	2.00
			I	0	0	0	0	0.0	0.00
Eriocephalus africanus var africanus	ASTERACEAE	Low shrub	E	31	29	25	21	49.1	17.74
			P	0	3	5	3	5.1	8.25
			F	30	20	26	23	*45.8	17.09
			I	0	0	0	0	0.0	0.00
Felicia filifolia	ASTERACEAE	Low shrub	E	3	2	6	2	13.0	7.57
			P	2	1	1	1	5.0	2.00
			F	20	22	18	22	*82.00	7.66
			I	0	0	0	0	0.0	0.00
Helichrysum cymosum subsp. *cymosum*	ASTERACEAE	Low shrub	E	0	0	0	0	0.0	0.00
			P	1	0	0	1	2.0	2.31
			F	24	24	25	22	*95.0	5.03
			I	0	1	0	2	3.0	3.83
Helichrysum patulum	ASTERACEAE	Low shrub	E	2	0	1	2	5.0	3.83
			P	1	1	0	0	2.0	2.31
			F	22	24	24	21	91.0	6.00
			I	0	0	0	2	2.0	4.00
Hermannia hyssopifolia	MALVACEAE	Low shrub	E	8	6	12	12	38.0	12.00
			P	2	1	1	1	5.0	2.00

			F	15	18	12	12	*57.0	11.49
			I	0	0	0	0	0.0	0.00
Lachenalia fistulosa	HYACINTHACEAE	Geophyte	E	0	0	0	0	0.0	0.00
			P	1	0	0	0	1.0	2.00
			F	24	25	25	25	*99.0	2.00
			I	0	0	0	0	0.0	0.00
Lampranthus emarginatus	MESEMBRYANTHEMACEAE	Succulent shrub	E	0	0	1	1	2.0	2.31
			P	0	1	0	0	1.0	2.00
			F	25	24	24	24	*97.0	2.00
			I	0	0	0	0	0.0	0.00
Moraea bellendenii	IRIDACEAE	Geophyte	E	0	0	0	0	0.0	0.00
			P	0	0	0	0	0.0	0.00
			F	25	25	25	25	*100.0	0.00
			I	0	0	0	0	0.0	0.00
Myrsine africana	MYRSINACEAE	Tall shrub	E	0	1	0	0	1.0	2.00
			P	0	0	0	0	0.0	0.00
			F	25	24	25	25	*99.0	2.00
			I	0	0	0	0	0.0	0.00
Ornithogalum thyrsoides	HYACINTHACEAE	Geophyte	E	0	1	1	0	2.0	2.31
			P	0	0	0	0	0.0	0.00
			F	25	24	24	25	*98.0	2.31
			I	0	0	0	0	0.0	0.00
Othonna arborescens	ASTERACEAE	Succulent shrub	E	0	1	0	3	4.0	5.66
			P	1	0	0	0	1.0	2.00
			F	24	24	25	22	*95.0	5.03
			I	0	0	0	0	0.0	0.00
Pelargonium cucullatum subsp. *tabulare*	GERANIACEAE	Low shrub	E	2	1	5	2	10.0	6.93
			P	2	3	1	1	7.0	3.83
			F	20	21	19	22	*82.0	5.16
			I	1	0	0	0	1.0	2.00
Pentameris airoides subsp. *airoides*	POACEAE	Graminoid (annual)	E	0	0	1	0	1.0	2.00
			P	0	0	0	0	0.0	0.00

			F	25	25	23	25	*98.0	4.00
			I	0	0	1	0	1.0	2.00
Podalyria sericea	FABACEAE	Low shrub	E	1	0	0	1	2.0	2.31
			P	1	0	0	0	1.0	2.00
			F	19	24	18	19	*80.0	10.83
			I	4	1	7	5	17.0	10.00
Ruschia rubricaulis	MESEMBRYANTHEMACEAE		E	0	1	1	1	3.0	2.00
		Succulent shrub	P	1	0	1	0	2.0	2.31
			F	24	24	23	24	*95.0	2.00
			I	0	0	0	0	0.0	0.00
Salvia africana-caerulea	LAMIACEAE	Low shrub	E	4	8	10	3	25.0	13.22
			P	3	1	0	2	6.0	5.16
			F	18	16	15	20	*69.0	8.87
			I	0	0	0	0	0.0	0.00
Searsia laevigata var. villosa	ANACARDIACEAE	Tall shrub	E	18	20	6	21	65.0	27.78
			P	7	5	19	4	35.0	27.78
			F	0	0	0	0	*0.0	0.00
			I	0	0	0	0	0.0	0.00
Searsia tomentosa	ANACARDIACEAE	Tall shrub	E	0	2	2	3	7.0	5.03
			P	25	22	21	21	89.0	7.57
			F	0	1	2	1	*4.00	3.27
			I	0	0	0	0	0.0	0.00
Tenaxia stricta	POACEAE	Graminoid	E	0	0	0	0	0.0	0.00
			P	0	1	1	2	4.0	3.27
			F	25	24	24	23	*96.0	3.27
			I	0	0	0	0	0.0	0.00
Themeda triandra	POACEAE	Graminoid	E	13	11	15	11	50.0	7.66
			P	0	0	0	0	0.0	0.00
			F	12	14	10	14	*50.0	7.66
			I	0	0	0	0	0.0	0.00
Trachyandra muricata	ASPHODELACEAE	Geophyte	E	0	0	0	0	0.0	0.00
			P	0	0	0	0	0.0	0.00

			F	25	25	25	25	*100.0	0.00
			I	0	0	0	0	0.0	0.00
Ursinia anthemoides subsp. *anthemoides*	ASTERACEAE	Forb (annual)	E	1	2	1	0	4.0	3.27
			P	0	0	1	0	1.0	2.00
			F	24	23	23	25	*95.0	3.83
			I	0	0	0	0	0.0	0.00
Total mean category F	**Total mean % (±SD)**							**80.0**	**27.10**

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Appendix 2.2: Viability (germination and cut test method) summary results

Table 2.4: Viability (germination and cut test method) results.
Untransformed data, mean percentage ±standard deviation, n=100 (four replicates of 25 per species) except where specified (see notes).

G	Germinated seedlings (normal and abnormal)	% viability = (G + F) and/or (G + TZ)
F	Fresh (once cut)	
TZ	Stained with TZ (interpretations (i) and (ii))	
M	Mouldy	% non- viable
E	Empty	
I	Infested	

SPECIES	FAMILY	GROWTH FORM	CAT	REP 1	REP 2	REP 3	REP 4	MEAN %	± S.D.
Arctopus echinatus	APIACEAE	Forb	G	0	0	0	1	1.0	2.00
			F		10	3	14	26.7	25.68
Notes: n = 101. Cut test - 17/1/13. 3 seeds from Rep 2 were lost. Fresh seeds were put into TZ. 0% viability with all repetitions.			TZ	0	0	0	0	0.0	0.00
			M	25	11	22	8	65.4	32.35
			E		1	1	2	4.0	3.27
			I					0.0	0.00
% V = (G+F)				0	10	3	15	**27.7**	27.22
% V = (G+TZ)				0	0	0	1	**1.0**	2.00
Athanasia crithmifolia	ASTERACEAE	Tall shrub	G	18	15	15	15	64.3	5.69
			F					0.0	0.00
Notes: n = 98. Cut test 7/11/12. Rep 3: One of the 'empty' seeds from the cut test was probably the seed coat of one of the seeds that germinated.			M	7	6	6	5	24.5	2.60
			E		4	5	3	12.2	8.66
			I					0.0	0.00
% V = (G+F)				18	15	15	15	**64.3**	5.69
Babiana fragrans	IRIDACEAE	Geophyte	G	22	25	23	25	95.0	6.00
			F					0.0	0.00
Notes: cut test 14/11/12.			M	3		2		5.0	6.00

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			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				22	25	23	25	**95.0**	6.00
Chironia baccifera	GENTIANACEAE	Low shrub	G	18	17	21	19	74.3	6.85
			F			1		1.0	2.00
Notes: n = 101. 26 seeds were sown for Rep 4 rather than 25. Cut test on the 6/12/12.			M	7	8	4	7	25.7	6.85
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				18	17	22	19	**75.3**	8.76
Chrysocoma coma-aurea	ASTERACEAE	Low shrub	G	22	22	23	24	93.8	2.31
			F					0.0	0.00
Notes: n = 97. Cut test 7/11/12.			M	2	0	1	1	4.1	3.4
			E		1	1		2.1	2.41
			I					0.0	0.00
% V = (G+F)				22	22	23	24	**93.8**	2.31
Cymbopogon marginatus	POACEAE	Graminoid	G	24	23	24	25	98.0	2.36
			F					0.0	0.00
Notes: n = 98. Cut test 22/11/12.			M	1	1			2.0	2.36
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				24	23	24	25	**98.0**	2.36
Dimorphotheca pluvialis	ASTERACEAE		G	3	4	2	2	11.0	3.83
Notes: Rep 1: 3 seeds were lost. 4 seeds from rep 2 were cut test 26/11/12. They were fresh so the remainder were returned. TZ on 27/11/12: 2 with very faint staining at tips, 2 completely unstained, 1 mouldy. Moved to 10 degrees on 27/11/12. Cut test 17/1/13: Rep 1 = 66%Viable, rep 2 = 76%, rep 3 = 90%, rep 4 =85%. Large proportion were stained very dark red, these were counted as viable but AOSA/SCST TZ handbook says that extensive bruising may be 'seen as dark red or grey'. If this was the cause then the seeds were non-viable.		Forb (annual)	F	18	17	21	21	77.0	8.25
			TZ (i)	12	13	19	18	61.4	14.05
			TZ (ii)	0	0	0	0	0.0	0.00
			M			2	1	3.0	3.83
			E	1			1	2.0	2.31
			I					0.0	0.00

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% V = (G+F)				21	21	23	23	**88.0**	4.62
% V = (G+TZ (i))				15	17	21	20	**72.4**	11.02
% V = (G+TZ (ii))				3	4	2	2	**11.0**	3.83
Ehrharta calycina	POACEAE	Graminoid	G	25	23	24	24	96.0	3.27
Notes: Stopped after 2 weeks as remaining two seeds were evidently soft and mouldy. Day 42, rep 4 = on transferring to new agar remaining seed disintegrated. Rep 3 : 1 seed was lost.			F					0.0	0.00
			M		2		1	3.0	3.83
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				25	23	24	24	**96.0**	3.27
Erepsia anceps	MESEMBRYANTHEMACEAE	Succulent shrub	G	2	7	2	2	13.0	10.00
			F					0.0	0.00
Notes: cut test 26/11/12.			M	23	17	23	23	86.0	12.00
			E		1			1.0	2.00
			I					0.0	0.00
% V = (G+F)				2	7	2	2	**13.0**	10.00
Eriocephalus africanus var africanus (% cleaned)	ASTERACEAE	Low shrub	G	27	20	28	22	44.5	3.83
Notes: n = 218. There were multiple seeds per fruit. The seeds of 100 fruits were sown. Cut test on 17/1/13, fresh seeds were placed in TZ. All were non-viable.			F	1	1		2	1.8	1.71
			M	4	3	7	7	9.6	4.24
			E	29	27	23	17	44.0	7.85
			I					0.0	0.00
% V = (G+F)				28	21	28	24	**46.3**	3.83
Felicia filifolia	ASTERACEAE	Low shrub	G	19	21	18	21	79.0	5.67
			F					0.0	0.00
Notes: n = 99. Some going mouldy- 12/9/12. Cut test: 12/12/12. Rep 4: 2 seeds were lost.			M	3	4	2		9.1	6.91
			E	3		5	2	10.1	8.37
			I					0.0	0.00
% V = (G+F)				19	21	18	21	**79.8**	5.67
Helichrysum cymosum subsp. *cymosum*	ASTERACEAE	Low shrub	G	22	21	23	19	88.5	7.90

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Notes: n = 96. An 'empty' seed from the cut test of Rep 2 is likely to be a seed coat of one of the seeds that had germinated.			F					0.0	0.00
			M	2	1		2	5.2	4.02
			E	1	4		2	7.3	6.87
			I					0.0	0.00
% V = (G+F)				22	21	23	19	**88.5**	7.90
Helichrysum patulum	ASTERACEAE	Low shrub	G	20	23	22	15	81.6	14.27
Notes: n = 98. Day 35, rep 3 = Abnormal growth (dark coloured root) cut test: 7/11/12. Rep 1: empty seed coats may be included in the germination results. Rep 3: one of the 'empty' seeds is probably a seed coat of a germinated seed. Rep 4: two of the 'empty' seeds were probably seed coats of germinated seeds.			F					0.0	0.00
			M	4	1	1	8	14.3	13.83
			E	4		3	4	11.2	7.72
			I					0.0	0.00
% V = (G+F)				20	23	22	15	**81.6**	14.27
Hermannia hyssopifolia	MALVACEAE	Low shrub	G	10	18	13	16	57.6	13.33
Notes: n = 99. 26/11/12: 26/11/12: Rep1 cut test. 7 mouldy, 4 fresh, 4 into TZ. All stained red. 27/11/12, Seeds placed into 25 degrees and Rep 1 was chipped. Three seeds were either lost or damaged when chipping. Other reps were chipped on 5/12/12. Rep 4: Quantity fresh and mouldy unknown. Seeds that are unaccounted for were either mouldy/empty/ infested on chipping.			F	4				4.0	8.33
			M	7				7.1	14.58
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				14	18	13	16	**61.6**	8.50
Lachenalia fistulosa	HYACINTHACEAE	Geophyte	G	23	25	20	24	95.8	3.29
Notes: n = 96. Cut test 3/1/12. 1 fresh seed from rep 3 placed into TZ overnight and it stained dark red all over. Germination was possibly counted twice on day 42 for rep 2. This would account for the 5 extra seeds.			F				1	1.0	2.0
			M	2				2.1	4.0
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				23	25	20	25	**96.9**	3.91
Lampranthus emarginatus	MESEMBRYANTHEMACEAE	Succulent shrub	G	4	7	8	5	24.2	7.16
Notes: n = 99. Rep 1: One seed was cut test on the 7/11/12 (making the total of fresh seeds 21). Rep 2: One of the 'empty' seeds was likely to have been the			F	20	16	16	16	68.7	7.66
			TZ (i)	1	1	4	1	67.0	5.97
			TZ (ii)	0	1	1	1	3.0	2.03

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seed coat of a germinated seed.			M				1	1.0	2.08
Rep 3: 1 seed was lost.			E		3		2	5.1	6.06
Rep 4: There were 16 fresh seeds at the end of the test but not all were not put into TZ as the seeds had been cut in half.									
TZ data: If a pale pink is viable: Rep 1: 1/9 = 11%, Rep 2: 1/16=31%, Rep 3: 4/15=26%, Rep 4: 1/6=1.6%			I					0.0	0.00
If a pale pink in not viable: Rep 1: 0/9 = 0%, Rep 2: 1/16 = 6.25%, Rep 3: 1/15 = 6.7%, Rep 4: 1/6 = 1.6%.									
% V = (G+F)				24	23	24	21	**92.9**	4.05
% V = (G+TZ (i))				5	8	12	6	**31.2**	12.20
% V = (G+TZ (ii))				4	8	9	6	**27.2**	8.77
Moraea bellendenii	IRIDACEAE	Geophyte	G	22	26	24	21	94.9	8.64
Notes: n = 98. Cut test on the 2/1/13. Fresh seeds were placed into TZ overnight. They both stained patchy pink suggesting they were non-viable. Rep 2: 26 seeds were sown. Rep 4: 3 seeds were lost.			F				2	2.0	4.00
			M			1		1.0	2.00
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				22	26	24	23	**97.0**	5.16
Myrsine africana	MYRSINACEAE	Tall shrub	G	0	0	2	0	2.0	4.00
Notes: Day 14 - Individual seeds starting to grow mould. 17/1/13 - only rep 1 was cut test and put into TZ. Rest were kept as seeds appear fresh for all reps. All the fresh seeds remained unstained after TZ testing = 0% viable.			F	21	24	22	24	91.0	6.00
			TZ	0	0	0	0	0.0	0.00
			M	2	1	1	1	5.0	2.00
			E					0.0	0.00
			I	1				1.0	2.00
% V = (G+F)				21	24	24	24	**93.0**	6.00
% V = (G+TZ)				0	0	2	0	**2.0**	4.00
Ornithogalum thyrsoides	HYACINTHACEAE	Geophyte	G	24	25	25	24	98.0	2.31
			F					0.0	0.00
			M		1		1	2.0	2.31
Notes: Cut test on 28/11/12.			E					0.0	0.00
			I					0.0	0.00
G+F				24	25	25	24	**98.0**	2.31

Othonna arborescens	ASTERACEAE	Succulent shrub	G	21	19	21	19	80.0	4.62
			F					0.0	0.00
Notes: Mould already visible by day 7.			M	3	5	4	5	17.0	3.83
			E	1	1		1	3.0	2.00
			I					0.0	0.00
% V = (G+F)				21	19	21	19	**80.0**	4.62
Pelargonium cucullatum subsp. *tabulare*	GERANIACEAE	Low shrub	G	1	5	1	2	9.0	7.57
Notes: Seed coat not removed (day 24 rep 2). Plates mouldy (24/10). 2/1/13: Seeds mouldy (some more than others). 17/1/13 - cut test of three seeds which were firm and bright green on the inside. Planted in sand (22/1/13) following plantzafrica guidance. TZ data- Rep 1 18/19 were viable. Rep 2: all were viable. Rep 3: 15/16 were viable. Rep 4:19/20 were viable.			F	19	20	16	20	75.0	7.57
			TZ	18	20	15	19	72.0	8.64
			M	2	1	3		6.0	5.16
			E	3		5		8.0	9.80
			I					0.0	0.00
% V = (G+F)				20	25	17	22	**84.0**	13.47
% V = (G+TZ)				19	25	16	21	**81.0**	15.10
Pentameris airoides subsp. *airoides*	POACEAE	Graminoid (annual)	G	16	14	10	11	53.7	10.01
			F					0.0	0.00
Notes: n = 95. Rep 3: 2 seeds were lost. Rep 4: 1 seed was lost.			M	7	4	10	1	23.2	15.11
			E	2	7	3	7	20.0	12.84
			I					0.0	0.00
% V = (G+F)				16	14	10	11	**53.7**	10.01
Podalyria sericea	FABACEAE	Low shrub	G	19	24	17	19	79.0	11.94
			F					0.0	0.00
Notes: Cut test: 6/12/12. Rep 1: 1 seed was lost.			M		1			1.0	2.00
			E					0.0	0.00
			I	4	1	8	6	19.0	11.94
% V = (G+F)				19	24	17	19	**79.0**	11.94
Ruschia rubricaulis	MESEMBRYANTHEMACEAE	Succulent shrub	G	2	2	0	0	4.0	4.62

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Notes: Rep 1: There were 12 fresh seeds at the end of the test. There were also 4 fresh seeds that were cut tested on the 27/11/12. None of these 4 seeds appeared viable from the TZ test. A further 9 seeds appeared fresh when cut tested on the 27/11/12 but were not put into TZ solution as the seeds had been cut in half. Rep 2: One seed was lost. Seeds moved to 15°C on the 27/11/12. TZ data: If a pale pink embryo is viable: Rep 1: 2/12 = 16%, Rep 2: 1/22 = 4.5%, Rep : 2/22 = 9%, Rep 4: 2/22 = 10% If a pale pink embryo is not viable: Rep 1: 2/12 = 16% viable, Reps 2,3, and 4 = all 0% viable.			F	12	22	22	23	79.0	20.75
			TZ (i)	2	1	2	2	7.0	2.00
			TZ (ii)	2	0	0	0	2.0	4.00
			M			2	1	3.0	3.83
			E			1	1	2.0	2.31
			I					0.0	0.00
% V = (G+F)				14	24	22	23	**83**.0	18.29
% V = (G+TZ (i))				4	3	2	2	**10.9**	3.83
% V = (G+TZ (ii))				4	2	0	0	**6.0**	7.66
Salvia africana-caerulea	LAMIACEAE	Low shrub	G	19	10	12	18	59.6	17.31
			F					0.0	0
Notes: n = 99. Cut tested 17/1/13. One seed was lost from each of Rep 1, Rep 2 and Rep 4.			M	5	14	12	6	37.4	18.14
			E					0.0	0
			I					0.0	0
% V = (G+F)				19	10	12	18	**59.6**	17.31
Searsia laevigata var. villosa	ANACARDIACEAE	Tall shrub	G	0	0	0	1	1.0	2.00
Notes: fresh germination.18/5 RH = 94.6%. 21/5 RH=87.1%. 21/5 -fruit pulp removed with scalpel. 22/5 = smoke solution. 1% sol domestos 5 min overnight soak (16h). Smoke solution (15°C).			F					0.0	0.00
			M	3	1	7		11.0	12.38
			E	13	20	12	16	61.0	14.38
			I	12	5	2	8	27.0	17.09
% V = (G+F)				0	0	0	1	**1.0**	2
Searsia tomentosa	ANACARDIACEAE	Tall shrub	G	0	0	0	0	0.0	0.00
			F		1	2	2	5.1	3.83
Notes: n = 98. Rep 4: There was also one seed that appeared part full on cut testing.			M	4	1	3	4	12.2	6.02
			E	15	17	16	15	64.3	3.32
			I	4	6	4	3	17.4	4.99

% V = (G+F)				0	1	2	2	**5.1**	3.83
Tenaxia stricta	POACEAE	Graminoid	G	23	21	22	23	91.8	3.21
			F					0.0	0.00
Notes: n = 97. Day 42- 1 abnormal (growing but very mouldy) Cut test: 7/11/12. Rep 2: 1 seed was lost.			M	1	1	3	2	7.2	3.70
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				23	21	22	23	**91.8**	3.21
Themeda triandra	POACEAE	Graminoid	G	0	2	1	1	4.0	3.27
Notes: TZ of Rep. 1 on 28/11/12. 0-30% may have been viable. (TZ sheet in folder). 3/1/13 - seeds going mouldy. Cut test 17/1/13. Rep 2: One of the seeds recorded as 'empty' is likely to have been a seed coat of a germinated seed. Rep 3: 3 seeds were lost. Rep 4: 2 seeds were lost. Fresh seeds were put into TZ, most were completely unstained. Rep 1: 21 seeds went into TZ. 3 had a red embryo. Rep 2: 3 had a red embryo.			F	21	6	3	12	42.0	31.75
			TZ	1	3	0	0	4.0	5.66
			M		8	9	1	18.0	18.62
			E	4	11	9	9	33.0	11.94
			I					0.0	0.00
% V = (G+F)				21	8	4	13	**46.0**	29.30
% V = (G+TZ)				1	5	1	1	**8.0**	8.00
Trachyandra muricata	ASPHODELACEAE	Geophyte	G	3	0	5	2	10.0	8.33
Notes: The TZ information for the family could not be found. The TZ percentages are based on the assumption that both the embryo and endosperm need to be red. % viable = TZ data: Rep 1: 10/20 = 50%, Rep 2: 18/21 = 85.7%, Rep 3: 12/20 = 60%, Rep 4: 17/22 = 72.7%. 2 fresh seeds from rep 1 and 3 fresh seeds from rep 2 were not put into TZ.			F	22	24	20	22	88.0	6.53
			TZ	10	18	12	17	57.0	15.45
			M		1		1	2.0	2.31
			E					0.0	0.00
			I					0.0	0.00
% V = (G+F)				25	24	25	24	**98.0**	2.31
% V = (G+TZ)				13	18	17	19	**67.0**	10.52
Ursinia anthemoides	ASTERACEAE	Forb (annual)	G	1	0	2	2	5.0	3.83

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subsp. *anthemoides*							
Notes: TZ of 5 seeds 27/11/12. Seeds firm but 4 were completely unstained (and one empty). Moved to 10 degrees. 17/11/13: the rest were cut tested and fresh seeds placed into TZ. Rep 1 = 37.5% viable, Rep 2 = 40%, Rep 3 = 40%, Rep 4 = 55%.	F	21	15	15	23	74.0	16.49
	TZ	8	6	6	13	33.0	13.22
	M	2	1	6		9.0	10.52
	E		1	2		3.0	3.83
	I					0.0	0.00
% V = (G+F)		22	15	17	25	**79.0**	18.29
% V = (G+TZ)		9	6	8	15	**38.0**	15.49

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Appendix 2.3: Germinability of species (glasshouse test) summary results

Table 2.5: Germinability of species (glasshouse test) results. Untransformed data, mean percentage ± standard deviation; n=125 (five replicates of 25).

SPECIES	FAMILY	GROWTH FORM	REP 1	REP 2	REP 3	REP 4	REP 5	MEAN %	±SD
Arctopus echinatus	APIACEAE	Forb	3	1	1	2	1	6.4	3.58
Athanasia crithmifolia	ASTERACEAE	Tall shrub	11	12	19	14	11	53.6	13.50
Babiana disticha	IRIDACEAE	Geophyte	25	25	24	25	25	99.2	1.790
Chironia baccifera	GENTIANACEAE	Low shrub	23	21	17	11	12	67.2	21.20
Chrysocoma coma-aurea	ASTERACEAE	Low shrub	20	18	22	24	22	84.8	9.12
Cymbopogon marginatus	POACEAE	Graminoid	20	20	20	20	15	76.0	8.94
Dimorphotheca pluvialis	ASTERACEAE	Forb (annual)	1	0	2	2	0	4.0	4.00
Ehrharta calycina	POACEAE	Graminoid	23	22	24	20	18	85.6	9.63
Erepsia anceps	MESEMBRYANTHEMACEAE	Succulent	11	8	6	6	4	28.0	10.60
Eriocephalus africanus var africanus	ASTERACEAE	Low shrub	19	15	14	15	15	62.4	7.80
Felicia filifolia	ASTERACEAE	Low shrub	14	15	14	13	16	57.6	4.56
Helichrysum cymosum subsp. *cymosum*	ASTERACEAE	Low shrub	15	21	18	19	15	70.4	10.40
Helichrysum patulum	ASTERACEAE	Low shrub	16	21	20	15	22	75.2	12.50
Hermannia hyssopifolia	MALVACEAE	Low shrub	0	0	0	0	0	0.0	0.00
Lachenalia fistulosa	HYACINTHACEAE	Geophyte	10	15	13	13	13	51.2	7.16
Lampranthus emarginatus	MESEMBRYANTHEMACEAE	Succulent shrub	4	11	3	5	5	22.4	12.50
Moraea bellendenii	IRIDACEAE	Geophyte	24	23	19	19	21	84.8	9.12
Myrsine africana	MYRSINACEAE	Tall shrub	7	6	5	6	8	25.6	4.56
Ornithogalum thyrsoides	HYACINTHACEAE	Geophyte	18	20	21	19	19	77.6	4.56
Othonna arborescens	ASTERACEAE	Succulent shrub	19	16	17	15	14	64.8	7.69
Pelargonium cucullatum subsp. *tabulare*	GERANIACEAE	Low shrub	1	2	1	3	2	7.2	3.35
Pentameris airoides subsp. *airoides*	POACEAE	Graminoid (annual)	0	0	0	0	0	0.0	0.00
Podalyria sericea	FABACEAE	Low shrub	23	21	23	24	21	89.6	5.37
Ruschia rubricaulis	MESEMBRYANTHEMACEAE	Succulent shrub	0	0	0	0	0	0.0	0.00
Salvia africana-caerulea	LAMIACEAE	Low shrub	10	17	16	10	13	52.8	13.10
Searsia laevigata var. villosa	ANACARDIACEAE	Tall shrub	0	0	0	0	0	0.0	0.00
Searsia tomentosa	ANACARDIACEAE	Tall shrub	0	0	0	0	0	0.0	0.00
Tenaxia stricta	POACEAE	Graminoid	5	3	6	5	6	20.0	4.90
Themeda triandra	POACEAE	Graminoid	7	6	8	7	9	29.6	4.56
Trachyandra muricata	ASPHODELACEAE	Geophyte	5	3	2	1	3	11.2	5.93
Ursinia anthemoides subsp. *anthemoides*	ASTERACEAE	Forb (annual)	8	11	9	12	6	36.8	9.55
	Total mean % (±SD)							**43.15**	**33.2**

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Appendix 2.4: Comparison of 'reference' germinability with viability

Table 2.6: Comparison of 'reference' germinability with viability.
Mean percentage and parameter details for the respective seed physiology tests.

Ref	The 'reference' germinability used in this comparison is in each case the highest recorded % germinability for the species on SID. Where the species has not been recorded on SID (for 12 species), a species of the same genus was used as a proxy (preferably a species from the Cape Peninsula or Western Cape). Where no South African species of the genus were recorded on SID (for three species) no comparison has been made. Symbol (#) indicates where a proxy has been used or where no comparison is made.
V	% viability (by germination and cut test method) for each of the 31 species. Parameters: Dry petri dishes containing all seed replicates were put over water (100% RH in a closed polyethylene container at 20°C for 24 hrs). Three species (*Arctopus echinatus, Myrsine africana* and *Podalyria sericea*) were hot water treated in approximately 250 ml of water at 80°C and allowed to cool. Thereafter, seeds of all species were soaked for five minutes in a 10% Domestos solution, rinsed for one minute in distilled water, soaked in smoke solution (with growth stimulants including gibberellic acid) overnight (18 hrs at 15°C), plated on 1% agar and placed in an incubator at an alternating temperature of 20/10°C with lighting (8/16 hrs) provided in the warm phase.
Ref less V	Difference between 'reference' germinability and viability. Note: **bold underlined** indicates 'reference' germinability exceeds viability by more than 25%.

SPECIES	Ref	V	Ref less V	'REFERENCE' GERMINABILITY: TEST PARAMETERS FOR PREVIOUS TESTS RESULTS RECORDED ON KEW SID	VIABILITY: TEST PARAMETERS IN ADDITION TO THOSE CARRIED OUT UNDER ROUTINE TESTING
Arctopus echinatus	#	27.7		No record of % germination for the species on Kew SID.	Notes: n = 101. Cut test - 17/1/13. Fresh seeds were put into TZ. 0% viability with all repetitions.
Athanasia crithmifolia	100.0	64.3	**35.7**	100% (15°C, 8/16).	
Babiana fragrans	95.0	95.0	0.0	95% (15°C, 8/16).	
Chironia baccifera	100.0	76.0	24.0	100% (20°C, 8/16 and 25/10°C)	
Chrysocoma coma-aurea	95.0	93.8	1.2	95% (20°C, 8/16).	
Cymbopogon marginatus	#	98.0		No record of % germination for the species on Kew SID.	
Dimorphotheca pluvialis	100.0	88.0	12.0	100% (11°C, 12/12).	Notes: 4 seeds from rep 2 were cut test 26/11/12. They were fresh so the remainder were returned. TZ on 27/11/12: 2 with very faint staining at tips, 2 completely unstained, 1 mouldy. Moved to 10 degrees on 27/11/12. Cut test 17/1/13: Rep 1 = 66%Viable, rep 2 = 75%,

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					rep 3 = 90%, rep 4 =85%. Large proportion were stained very dark red, these were counted as viable but AOSA/SCST TZ handbook says that extensive bruising may be 'seen as dark red or grey'. If this was the cause then the seeds were non-viable.
Ehrharta calycina	# 100.0	96.0	4.0	No record of % germination for the species on Kew SID: most comparable species = *E. thunbergii* (Western Cape): 100% (15°C, 8/16, seed scarified (covering structure removed)).	
Erepsia anceps	# 100.0	13.0	**87.0**	No record of % germination for the species on Kew SID: most comparable species = *E. aspera* (Western Cape): 100% (15°C, 8/16 and 25/10°C, 8/16, seed scarified (chipped with scalpel), germination medium = 1% agar + 250 mg/l gibberellic acid (GA3)).	
Eriocephalus africanus var africanus	96.0	46.3	**49.7**	96% (10°C, 8/16 and 15°C, 8/16).	Notes: n = 218. Cut test on 17/1/13, fresh seeds were placed in TZ. All were non-viable.
Felicia filifolia	100.0	79.0	21.0	100% (15°C, 8/16).	
Helichrysum cymosum subsp. *cymosum*	# 100.0	88.5	11.5	No record of % germination for the species on Kew SID: most comparable species = *H. asperum* (Western Cape): 100% (20°C, 8/16).	
Helichrysum patulum	# 100.0	81.6	18.4	No record of % germination for the species on Kew SID: most comparable species = *H. asperum* (Western Cape): 100% (20°C, 8/16).	
Hermannia hyssopifolia	100.0	65.7	**34.3**	100% (25°C, 12/12 and 30°C, 12/12 and 35°C, 8/16 seed sterilised (immersed in 10% Domestos solution for 5 mins), then seed scarified (chipped with scalpel)).	Notes: n = 99. 26/11/12: Rep1 cut test. 6 mouldy, 4 fresh, 4 into TZ. All stained red. 27/11/12, Seeds placed into 15 degrees and Rep 1 was chipped. Other reps were chipped on 5/12/12. Seeds that are unaccounted for were either mouldy/empty/ infested on chipping.
Lachenalia fistulosa	# 100.0	102.1	-2.1	No record of % germination for the species on Kew SID: most comparable species = *L. aloides* (Western Cape): 100% (15°C, 8/16).	Notes: n = 99. 26/11/12: Rep1 cut test. 6 mouldy, 4 fresh, 4 into TZ. All stained red. 27/11/12, Seeds placed into 15 degrees and Rep 1 was chipped. Other reps were chipped on 5/12/12. Seeds that are unaccounted for were

					either mouldy/empty/ infested on chipping.
Lampranthus emarginatus	# 100.0	82.8	17.2	No record of % germination for the species on Kew SID: most comparable species = *L. bicolor* (Cape peninsula): 100% (15°C, 8/16 seed scarified (chipped with scalpel) and 20°C, 8/16 and 25/10°C, 8/16 seed scarified (chipped with scalpel); germination medium = 1% agar + 250 mg/l gibberellic acid (GA3)).	Notes: n = 99. 7/11/12 - 1 seed from rep 1 cut test. It was fresh so the remainder were returned. Cut test 17/1/13 and fresh seeds placed into TZ. % viability depends on the viability interpretation of having a pale pink embryo. On the basis that pale pink embryos are viable: Rep 1 = 11% viable, Rep 2 = 31%, Rep 3 = 26%, Rep 4= 6.25%. On the basis that pale pink embryos are non-viable: Rep 1 = 0% viable, rep 2 = 6%, rep 3 = 6%, rep 4 = 6.25%.
Moraea bellendenii	100.0	97.9	2.1	100% (10°C, 8/16 and 16°C, 12/12).	Notes: n = 97. Cut test on the 2/1/13. Fresh seeds were placed into TZ overnight. They both stained patchy pink suggesting they were non-viable.
Myrsine africana	100.0	23.0	**77.0**	100% (20°C, 8/16, seed scarified (covering structure removed)).	Notes: Day 14 - Individual seeds starting to grow mould. 17/1/13 - only rep 1 was cut test and put into TZ. Rest were kept as seeds appear fresh. Rep 1 = 0% viable. The other reps will now be cut test/tz tested.
Ornithogalum thyrsoides	100.0	98.0	2.0	100% (10°C, 8/16 and 15°C, 8/16).	
Othonna arborescens	#	80.0		No record of % germination for the species on Kew SID.	
Pelargonium cucullatum subsp. *tabulare*	100.0	9.0	**91.0**	100% (15°C, 8/16 and 20°C, 8/16, seed scarified (covering structure removed and seed coat chipped)).	Notes: Seed coat not removed (day 24 rep 2). Plates mouldy (24/10). 2/1/13: Seeds mouldy (some more than others). 17/1/13 - cut test of three seeds which were firm and bright green on the inside. Planted in sand (22/1/13) following plantzafrica guidance.
Pentaschistis airoides subsp. *airoides*	# 80.0	53.7	**26.3**	No record of % germination for the species on Kew SID: most comparable species = *Pentameris setifolia* (Eastern Cape): 80% (20°C, 8/16).	

Podalyria sericea	100.0	79.0	21.0	100% (15°C, 8/16 and 20°C, 8/16, seed scarified (arils removed and seed coat chipped)).	
Ruschia rubricaulis	# 93.0	83.0	10.0	No record of % germination for the species on Kew SID: most comparable species = *R. macowanii* (Western Cape): 93% (35/20°C, 8/16, imbibed on 1% agar for 4 weeks at 35/20°C, then seed scarified (chipped with scalpel)).	Notes: 27/11/12 - 4 seeds (Rep 1) into TZ. Remainder were just cut tested and appeared fresh. All were unstained. Seeds placed in 15 degrees. Will be cut tested if no germination. 17/1/13 - remainder were cut test. It was hard to distinguish between fresh and mouldy so seeds recorded as fresh and placed into TZ. Rep 1 = 16% viable, rep 2 = 13%, rep 3= 9%, rep 4=10%.
Salvia africana-caerulea	83.0	59.6	23.4	83% (15°C, 8/16, germination medium = 1% agar + 250 mg/l gibberellic acid (GA3)).	
Searsia laevigata var. villosa	# 75.0	1.0	**74.0**	No record of % germination for the species on Kew SID: most comparable species = *S. glauca* (Western Cape): 75% germination (25/10°C, 8/16, pre-sowing treatments = seed scarified (chipped with scalpel))	
Searsia tomentosa	# 75.0	5.1	**69.9**	No record of % germination for the species on Kew SID: most comparable species = *S. glauca* (Western Cape): 75% germination (25/10°C, 8/16, pre-sowing treatments = seed scarified (chipped with scalpel))	
Tenaxia stricta	# 100.0	91.8	8.3	No record of % germination for the species on Kew SID: most comparable species = *M. lupulina* (Western Cape): 100% (20°C, 8/16, seed scarified (covering structure removed)).	
Themeda triandra	85.0	25.0	**60.0**	85% (35/20°C, 8/16, imbibed on 1% agar for 1 day at 35/20°C, then seed scarified (pericarp excised from along proximal to distal ridge above embryo)).	Notes: TZ of Rep. 1 on 28/11/12. 0-30% may have been viable. (TZ sheet in folder). 3/1/13 - seeds going mouldy. Cut test 17/1/1. Fresh seeds were put into TZ, most were completely unstained but 3 from rep.2 had a red embryo but endosperm was unstained.
Trachyandra muricata	100.0	10.0	**90.0**	100% (10°C, 8/16).	

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Ursinia anthemoides subsp. *anthemoides*	# 97.0	79.0	18.0	No record of % germination for the species on Kew SID: most comparable species = *U. tenuifolia* subsp tenuifolia (Western Cape): 97% (15°C, 8/16).	Notes: TZ of 5 seeds 27/11/12. Seeds firm but 4 were completely unstained (and one empty). Moved to 10 degrees. 17/11/13: the rest were cut tested and fresh seeds placed into TZ. Rep 1 = 37.5% viable, Rep 2 = 40%, Rep 3 = 43.8%V, Rep 4 = 55%

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Appendix 2.5: Selection of X-ray images

Predominantly full seeds

Photo plate 2.4: *Cymbopogon marginatus* (replicate 3): all 25 seed are full and potentially viable.

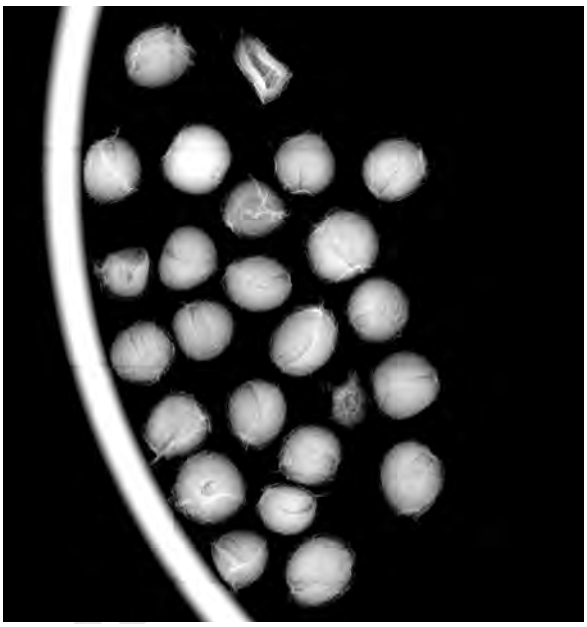

Photo plate 2.5: *Babiana fragrans* (replicate 3): 23 seed are full (potentially viable) and 2 seeds are non-viable (empty and incompletely formed).

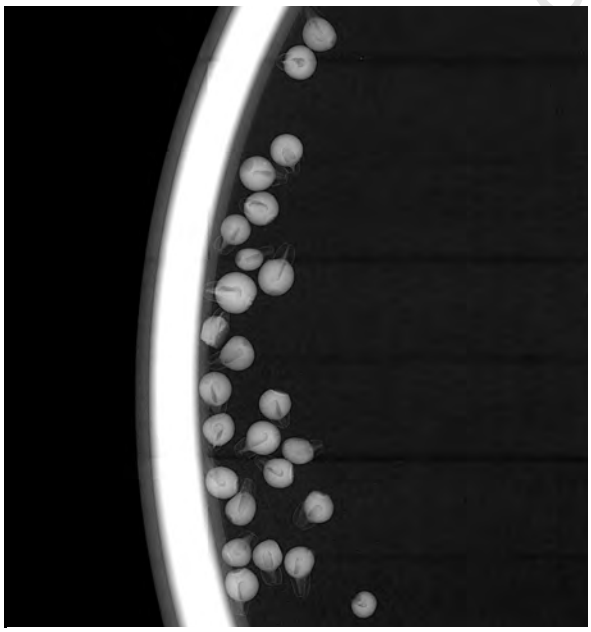

Photo plate 2.6: *Lachenalia fistulosa* (replicate 1): 24 seeds are full (potentially viable) and 1 seed is incompletely formed (non-viable).

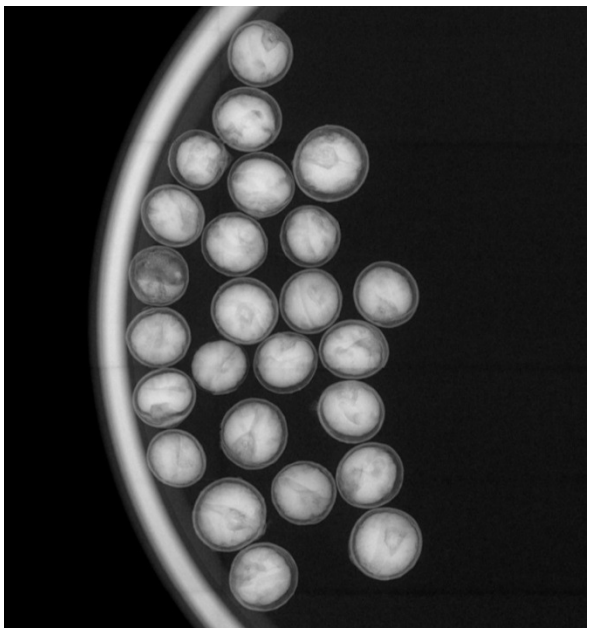

Photo plate 2.7: *Myrsine africana* (replicate 2): 24 full seeds (potentially viable) and 1 seed incompletely formed (non-viable).

Predominantly empty and partially-filled seeds

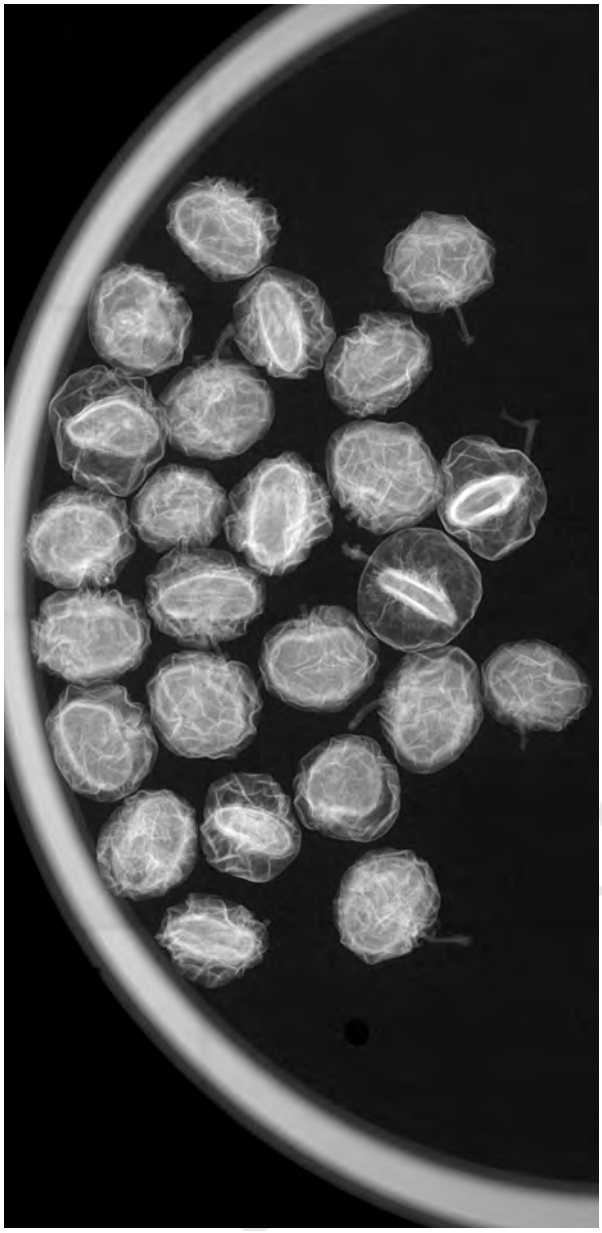

Photo plate 2.8: *Searsia laevigata var villosa* (replicate 3): zero seeds are full (potentially viable), six seeds are empty (non-viable) and 19 seeds are partially-filled and may or may not germinate depending on the species.

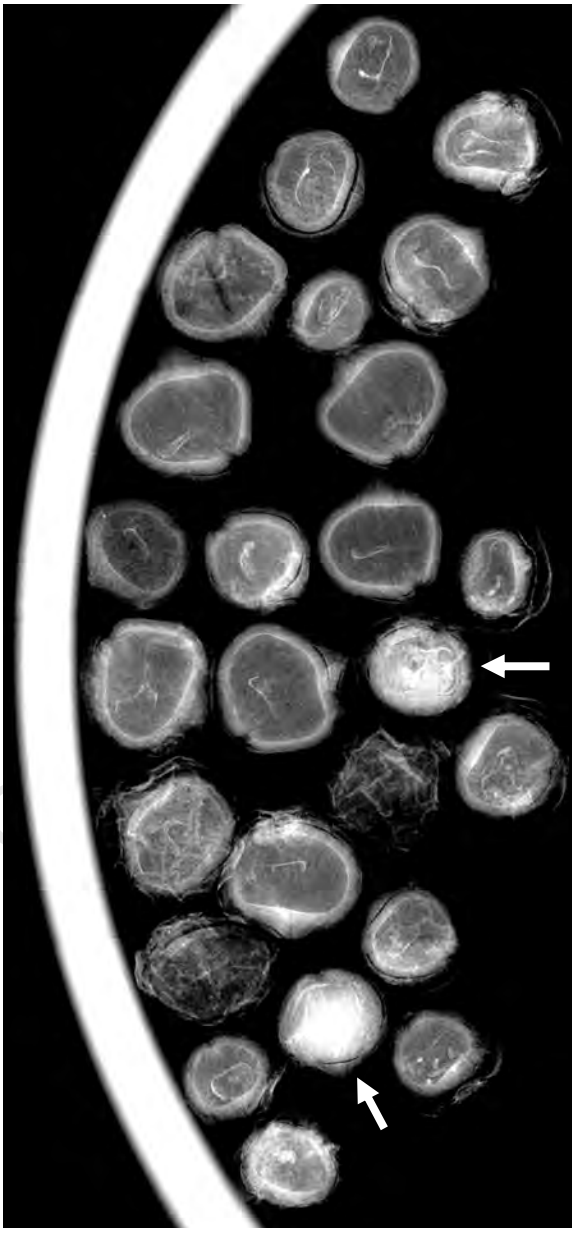

Photo plate 2.9: *Searsia tomentosa* (replicate 3): 2 full seeds (potentially viable, refer arrows), two seeds are empty (non-viable) and 21 seeds are partially-filled and may or may not germinate depending on the species.

Insect infested seed

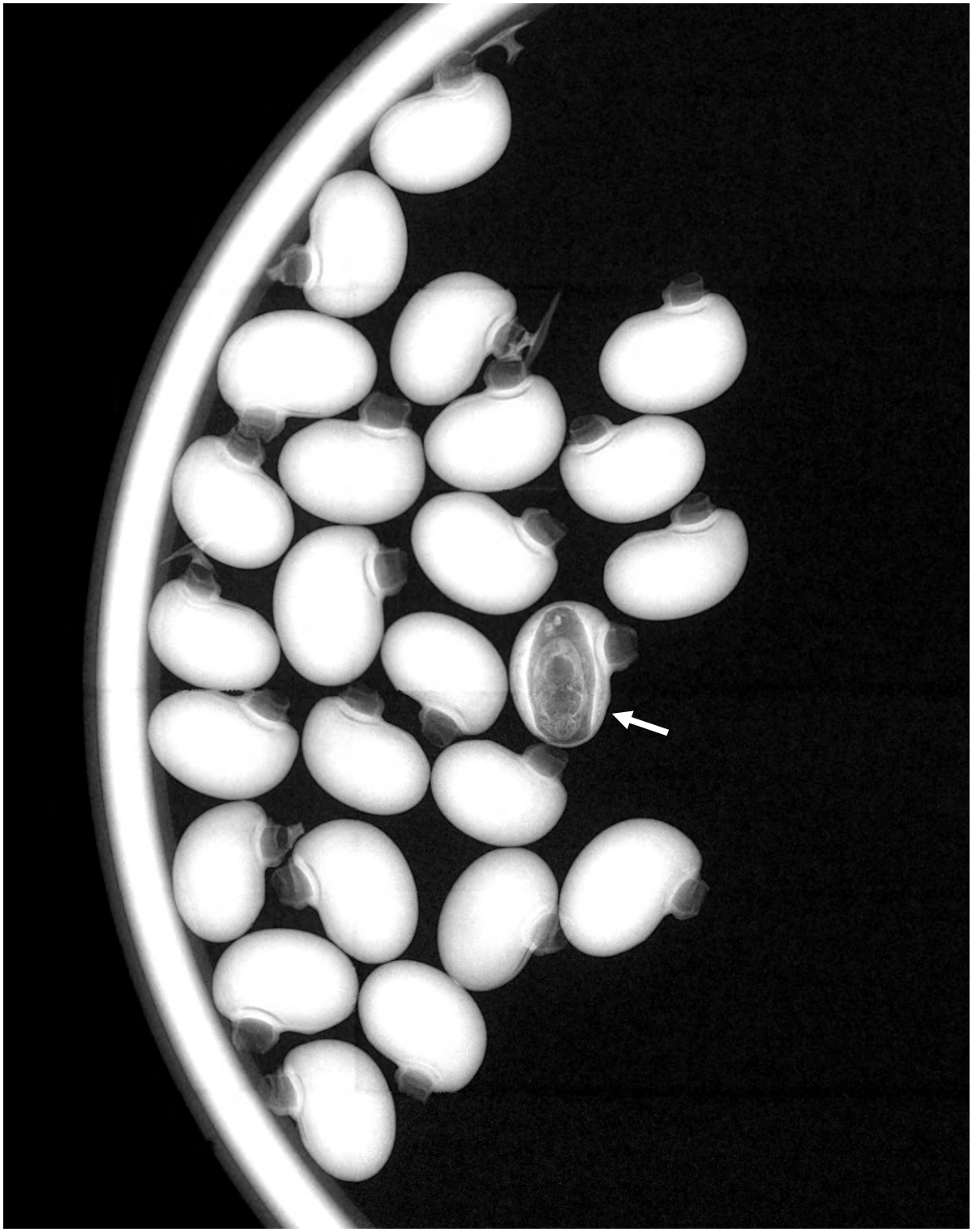

Photo plate 2.10: P*odalyria sericea* (replicate 2): 24 seeds are full (potentially viable) and one seed is infested (non-viable, refer arrow) due to the presence of an insect from the Coleoptera order.

Seed with high X-ray viability yet poor performance

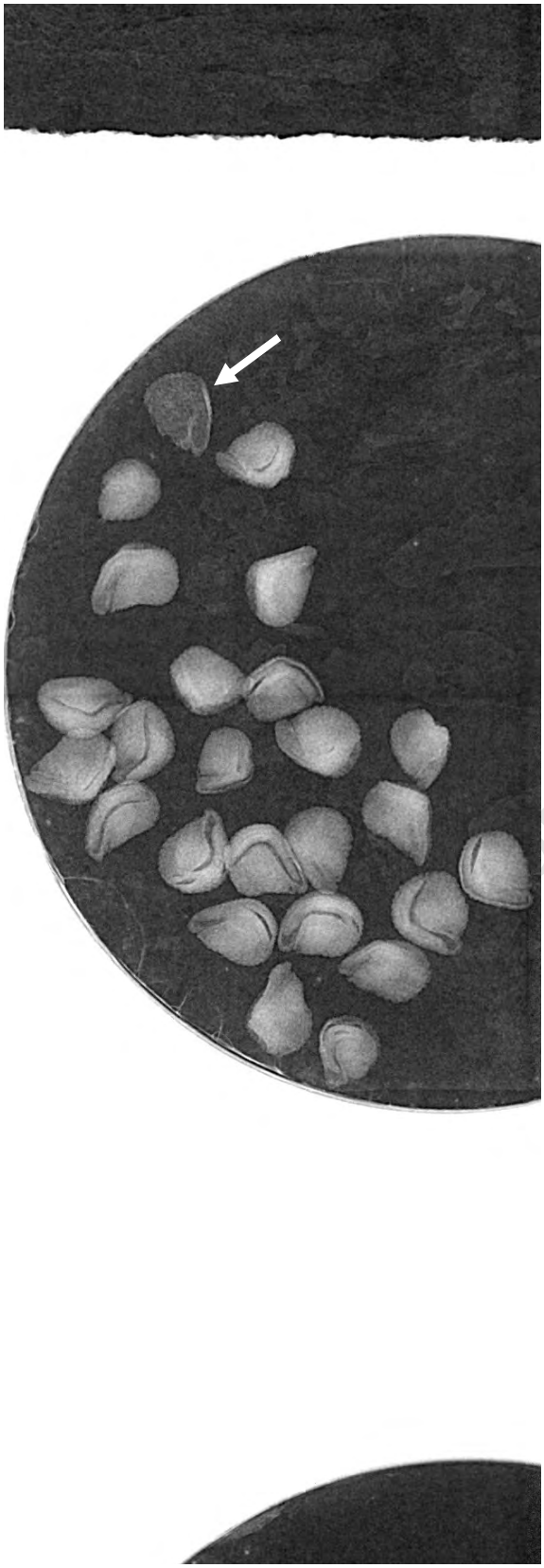

Photo plate 2.11: *Lampranthus emarginatus* (replicate 4): 24 seeds are full (potentially viable) and one seed is empty (non-viable, refer arrow). Despite a high X-ray viability, viability and germinability were relatively poor.

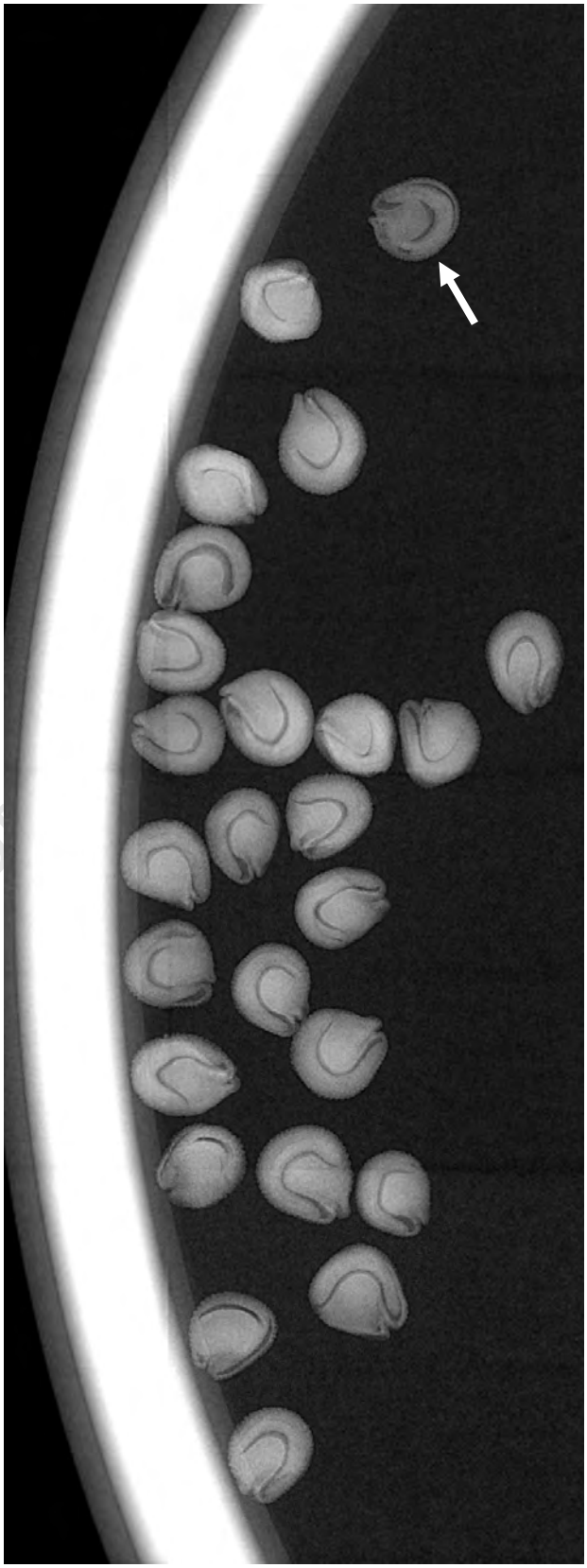

Photo plate 2.12: *Erepsia anceps* (replicate 3): 24 full seeds (potentially viable), one seed is partially-filled and may or may not germinate depending on the species (refer arrow). As with *Lampranthus emarginatus*, high X-ray viability and poor viability and germinability may be due to deteriorated seed.

3 Intervention responses: determining the effects of interventions (burning, tillage, herbicide-application, rodent-exclusion and seeding) with respect to seed germination

3.1 Abstract

Application of the ecological principles of community regeneration is central to optimising restoration outcomes and required to alleviate the critical conservation status of Peninsula Shale Renosterveld. Thus experimentation to test restoration ecological-response models of the ecosystem is imperative to inform restoration.

As surrogates for ecological drivers of recruitment in renosterveld, five restoration interventions (seeding, fire, tillage, herbicide-application and rodent-exclusion) were crossed to produce 32 different interventions (including a control) which were incorporated into a field experiment. This portion of the study set out to determine the effectiveness of the 31 interventions with respect to several seedling recruitment performance criteria: seedling density, species richness, canopy cover and seedling height of the 31 seeded species; seedling density, species richness and canopy cover of the existing indigenous species; and, canopy cover of the weed species. The actual experimental intervention responses were compared with expected responses as presented in an *a priori* ecological-response model.

The responses to the 31 interventions were variable with some interventions performing poorly and others performing well. Seeding made significant contributions to overall seedling density, species richness and canopy cover and is imperative for successful restoration efforts. In the context of extensive alien, annual grass invasion, sowing alone was almost ineffectual and was considerably enhanced when implemented in combination with another recruitment stimulating factor. These two-factor interventions have merit yet intervention effectiveness was considerably enhanced by the three- to five-factor interventions, several of which resulted in a full set of desired responses.

This chapter set out to make an empirical contribution through determining the measurement of intervention responses and a theoretical contribution towards informing the ecological-response model for Peninsula Shale Renosterveld more broadly, a contribution which also has relevance to other renosterveld types. This portion of the study has the potential to reduce some of the uncertainty and risk associated with restoration by informing practitioners how best to proceed with respect to selecting and incorporating appropriate interventions that are likely to result in a certain degree of effectiveness into landscape-scale restoration strategies.

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3.2 Introduction

In ecological restoration, the implementation of interventions may be necessary in the process of overcoming ecosystem degradation and initiating ecosystem recovery (Hobbs and Cramer 2008, Brudvig 2011). There is, however, a high level of uncertainty and risk associated with restoration projects (Crookes et al. 2013) and outcomes and ecosystem trajectories are often difficult to predict (Clewell and Aronson 2013). Thus, in determining which interventions are most appropriate for restoration, experimentation has a critical role to play in reducing risk through determining the likely ecological response of the system (Hobbs and Harris 2001). In order to maximise restoration results, the choice of intervention needs to acknowledge and take its cue from the ecological drivers in the ecosystem. In developing restoration protocols for fynbos and other fire-driven temperate shrublands, Holmes and Richardson (1999) emphasize the importance of founding restoration practices on ecological principles of regeneration so as to optimise restoration outcomes. Knowledge of the regional ecology and ecological processes in pristine renosterveld is poor due to extensive transformation, large-herbivore extinction and altered fire regimes (Rebelo 1995, Krug et al. 2004). Studies on renosterveld restoration are limited and with some exceptions, renosterveld restoration efforts have largely focused on West Coast Renosterveld and in the recovery dynamics in old fields. The renosterveld restoration knowledge base has been incrementally grown, predominantly in the last ten years, and has addressed the roles of, *inter alia*, soil (Memiaghe 2008, Cowan 2013), seed ecology (Kemper et al. 1999, Donaldson et al. 2002, Shiponeni 2003a, Shiponeni and Milton 2006, Muhl 2008, Heelemann 2012, Heelemann et al. 2012, Cowan 2013, Heelemann et al. 2013), fire (Midoko-Iponga 2004, Musil et al. 2005, Radloff 2008, Heelemann et al. 2013), grazing (Midoko-Iponga 2004, Midoko-Iponga et al. 2005, Radloff 2008), competition (Midoko-Iponga 2004, Musil et al. 2005, Muhl 2008, Sharma et al. 2010, Terblanche 2011) and plant reintroduction through seeding (Holmes 2002, Midoko-Iponga 2004), rooted material (Holmes 2002) and seedlings (Midoko-Iponga 2004). The role of seed and seedling predation by small mammals with respect to restoration in renosterveld has been addressed (Holmes 2002, Dreyer 2012) yet to date the effect on restoration efforts has not been empirically tested. Interactions between some of the above-mentioned factors have been investigated to some degree, for instance seeding/propagated seedlings, grazing/grass-clearing/herbicide and fire (Musil et al. 2005, Midoko-Iponga 2004, Midoko-Iponga et al. 2005, Memiaghe 2008) yet more than three factors have not been crossed in a factorial manner to determine restoration responses of the multiple resultant interactions. In addition to the limited success achieved by seed-based renosterveld restoration experimentation (Holmes 2002, Midoko-Iponga 2004) restoration is a costly endeavour (Crookes et al. 2013). Furthermore, the recovery in renosterveld old fields, despite proximity to natural areas, is slow and does not occur for decades (Midoko-Iponga 2004). Experimentation therefore has the important role of building knowledge of how these ecological drivers and their interactions affect seedling recruitment in a restoring system towards establishing empirical measurement of intervention effectiveness and to developing the conceptual restoration framework. Incorporation of effective interventions into ecological restoration strategies may reduce some of the risk associated with restoration by increasing the likelihood of achieving empirically-defined restoration objectives. This introduction addresses the role of

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ecological factors in driving vegetation regeneration in renosterveld particularly with respect to seedling recruitment and serves as rationale and context for the experimental work presented in this chapter.

3.2.1 State and transition in renosterveld

It is thought that historically, fire and grazing were the principle ecosystem drivers in renosterveld (Krug et al. 2004, Radloff 2008) and the periodic, natural disturbance resulting from these factors is an important driver maintaining biodiversity richness (Boucher 1995, Rebelo 1995, de Villiers et al. 2005, Walton 2006). Grazing and fire have been found to affect the interplay between grassy and shrubby states in renosterveld yet the mechanisms responsible for these shifts are poorly understood (de Villiers et al. 2005, Rebelo et al. 2006, Radloff 2008). Tying into the theory of multiple equilibria (Clewell and Aronson 2013) conceptual models proposed by Cowling et al. (1986) and expanded, amongst others, by Rebelo (1995), Krug (2004) and Milton (2007) illustrate how the type and timing of such interventions affects community composition and the resultant transition between different vegetation states. For instance a grassy shrubland or sweet grassland, through inappropriate burning and grazing regimes, may be converted to degraded states such as a species depauperate shrubland (McDowell 1988, McDowell 1995, Krug 2004, Milton 2007, Radloff 2008) or a state dominated by weedy grasses and annuals (Krug 2004, Milton 2007). The positive feedback of a degraded vegetation state dominated by weedy grasses and annuals (Milton 2004, Memiaghe 2008) may be maintained by grazing (Milton 2007) whilst through the implementation of ploughing, herbicide-application and seeding, this state may be shifted and the ecosystem trajectory altered by initiating the transition from a degraded state to a more desirable grassy shrubland state (Milton 2007).

3.2.2 Seed ecology

The fundamental germination cues for most species are the availability of light and favourable temperature and moisture regimes, these factors being considerably different in gaps than in dense vegetation (Fenner 2005). Gaps provide seedlings with the opportunity to germinate with reduced competition and an absence of gaps in a community consequently inhibits recruitment (Fenner 2005). In natural systems gaps are created by plant mortality and disturbance such as fire and faunal activity (Fenner 2005) and in the context of restoration, to create gaps artificially, burning, clearing vegetation (mechanically and/or chemically) and ploughing or tilling the soil are commonly implemented (Milton 2007). Alleviating crusted (Mills and Fey 2004) or compacted soils in degraded areas, through for instance tillage, may be necessary in restoration to enable root and shoot development (Whisenant 2002, Bassett et al. 2005) and is expected to promote seedling recruitment. The number of safe sites for seed germination occurring within a gap is a function of the microtopography of the soil surface (Fenner 2005, Hartmann and Kester 2011). Thus to promote seedling recruitment, a secondary soil-preparation process which fine-grades the soil optimises soil-seed contact (Fenner 2005), for instance harrowing after ploughing (Holmes 2005).

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The ability of species to re-colonize by seed is in part a function of the seed dispersal distance and thus of the dispersal syndrome (Shiponeni 2003a, 2003b). In Swartland Shale Renosterveld, dispersal distances were found to be variable (Shiponeni 2003a). Animal and wind dispersed seeds have a greater ability to reach new sites whilst the tumbling seeds of geophytes are relatively constrained to areas in close proximity to the source plants (Shiponeni 2003b). However, due to the presence of desirable species in the seed bank of old fields but a lack of above-ground representation, it was concluded that seed dispersal limitation plays a role in the recovery of old fields but may not be the primary constraint (Shiponeni 2003a). In renosterveld, myrmecochory (the dispersal and burial of seed by ants) is almost absent (Rebelo et al. 2006) yet ornithochory (the dispersal of seed by birds) is common in renosterveld thicket vegetation (Le Maître and Midgely 1992). Seed dispersal by scatter-hoarding (by the Cape Spiny Mouse, *Acomys subspinosus*) (Rusch 2012) is likely to occur in a small yet widespread guild of non-serotinous plants (Midgely et al. 2002) and indeed the Cape Spiny Mouse (*Acomys subspinosus*) has been recorded in Peninsula Shale Renosterveld remnants on Devil's Peak (Dreyer 2012).

In fire-driven ecosystems, soil-stored seed banks are largely the determinants of a post-fire community with respect to species composition and diversity (Holmes and Cowling 1997). The composition, status and genetic integrity of the seed bank is consequently of great value in restoration and projects should aim to optimise stimulation of this potentially considerable resource to enhance restoration outcomes (Holmes and Richardson 1999). In renosterveld, seed-bank persistence is thought to be short-lived and in the case of renosterbos (*Dicerothamnus rhinocerotis*) shrubland, seed longevity is in the order of seven (Levyns 1935, 1956) to one year (Shiponeni 2003b). A recent study has found renosterveld seed-bank persistence to be variable and subject to the nature of degradation, where, after many decades of invasion in two sites, the seed bank in old fields was found to be depleted yet viable in an area recently cleared of invasive *Pinus* species (Heelemann et al. 2013). In Peninsula Shale Renosterveld, succession following alien clearance was characterised by a poor colonisation of indigenous species and an abundance of predominantly invasive annual grasses, pointing to a depleted seed bank (Terblanche 2011). This finding was confirmed by Cowan (2013) who found that, with the exception of an old stand of renosterveld, the seed bank was depleted and dominated by alien species.

In addressing potential seed dispersal limitation and short-term seed-bank persistence, the reintroduction or supplementation of guilds by seeding may be necessary in an ecosystem where guilds are absent or poorly represented (Holmes and Richardson 1999, Holmes 2008). In fynbos where the majority of species have short dispersal modes, seed supplementation is recommended in degraded areas larger than 50 m in diameter as even where degraded areas border on natural areas, dispersal limitation results in slow recolonization that may require several fire cycles (Holmes and Richardson 1999). In renosterveld, as seed dispersal distances are variable (Shiponeni 2003a), it is likely that areas larger than 50 m would mostly require seed supplementation for the short dispersal-mode species. As resprouting species, characteristic of renosterveld, (Rebelo et al. 2006) tend to have smaller seed banks and slower growing seedlings than reseeding species, their re-establishment is more challenging (Holmes and Richardson 1999). Consequently, restoring resprouting species may

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require special attention such as collecting additional seed or propagation and transplanting seedlings into the post-fire environment (Holmes and Richardson 1999).

3.2.3 Fire

Like fynbos, recruitment in Renosterveld occurs predominantly in response to fire (Boucher 1995, Rebelo et al. 2006). Renosterveld communities tend to be maintained at an early seral stage due to a high fire frequency as well as grazing pressure and consequently seral succession is renosterveld communities is not well-studied (Rebelo et al. 2006). Within the first autumn after fire, a full representation of the early to late successional species recruits (Holmes and Richardson 1999, Rebelo et al. 2006). Recruitment following fire occurs by resprouting (van Wilgen and Forsythe 1992) or the germination of surface or soil-stored seed (Holmes 2002, Holmes and Newton 2004). Growth forms have different recruitment modes where geophytes and graminoids predominantly regenerate by resprouting, forbs or annuals by short-lived reseeding and shrubs predominantly regenerate by short- and long-lived reseeding and to a slightly lesser extent by resprouting (Holmes and Cowling 1997). The dominance of grasses in Renosterveld results in a larger portion of the system resprouting in response to fire relative to fynbos which is relatively poorly represented in this regard (Rebelo et al. 2006). Fire cues experienced by seeds may be physical [for example a heat pulse (Kilian and Cowling 1992, Bond and van Wilgen 1996)]; chemical [for example smoke (Bond and van Wilgen, 1996, Brown et al. 2000; Brown et al. 2003, Brown et al. 2004, van Staden et al. 2004)]; or, indirect [resulting in more light and available soil moisture (Bond and van Wilgen 1996), higher soil oxygen levels and a greater diurnal temperature range due to increased soil exposure (Brits 1986)].

Several interacting factors pertaining to the nature of the fire affect the characteristics of an emergent, post-fire community with respect to species composition, community structure and ecosystem function (Holmes and Richardson 1999) and subsequently have considerable bearing on how to incorporate fire into ecological restoration efforts. These interacting factors include: fire regime and the fire-return interval (Holmes and Richardson 1999, Rebelo et al. 2006), the seasonal timing of fire (Cowling et al. 1986, Rebelo et al. 2006) and fire intensity (Scott and van Wyk 1992, Bond and van Wilgen 1996, Holmes and Richardson 1999, Kemper et al. 1999, Rebelo et al. 2006). The ideal fire regime in renosterveld is unknown (Von Hase et al. 2003, Rebelo et al. 2006) but fire-return intervals tend to be more frequent in renosterveld than in fynbos (Boucher 1983) due to a relatively large proportion of graminoids which rapidly regenerates as flammable biomass (Holmes 2002, Rebelo et al. 2006). Estimates of the ideal fire return interval in renosterveld range from two to 10 years and may be as high as 40 years (Rebelo 1992). De Villiers et al. (2005) propose an interval of 10-15 years in wetter renosterveld types and longer intervals in drier types (de Villiers et al. 2005). Frequent fires favour the resprouting and short-lived reseeding species to the detriment of slow growing reseeding species whilst fires occurring over longer return intervals favour long-lived, reseeding species to the detriment of bulbs, herbs and grasses (de Villiers 2005, Rebelo et al. 2006). Fire occurring in late summer or early autumn allows for the winter germination of seedlings and thus a long, moist period to facilitate the development of a more extensive root system and thus a greater ability to survive the summer drought (Holmes and Richardson 1999). In renosterveld, an

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aseasonal fire, for instance an early spring fire, would intercept seed development in annuals, geophytes and most graminoids and thus prevent seed set (Cowling et al. 1986, Rebelo et al. 2006) and consequently favour winter-seeding less-palatable Asteraceous shrubs (Milton, 2007). Very hot fires, predominantly resulting from high biomass of woody invasive species, can have detrimental effects such as increasing the mortality of resprouting species, reducing the propagule density of vesicular-arbuscular mycorrhiza and thus the level of seedling infectivity (Klopatek et al. 1988, Vilarino and Arines 1991, Allsopp and Stock 1994) and soil water repellency (Scott and van Wyk 1992, Bond and van Wilgen 1996) causing localised drought which is likely to affect seedlings with shallow root systems (Bond and van Wilgen 1996). In addition, hot fires can damage the surface and shallow-stored seed (Holmes and Richardson 1999, Rebelo et al. 2006) thereby favouring the species with deeper buried bulbs and seeds, of which the larger seeds with greater resources emerge more successfully (Bond and van Wilgen 1996).

Fire is critical for initiating recruitment of the soil-stored seed bank and is expected to provide an ideal environment for the recruitment of sown seed, yet, also has the potential to result in undesired outcomes. Given the numerous effects fire has on community characteristics, the burn implemented to trigger restoration should in as far as possible be manipulated to mimic the natural fire conditions with respect to frequency, season and intensity (Holmes and Richardson 1999). Where burning is not possible, surrogate cues to mimic fire can be substituted, for instance clearing the vegetation, heat treating hard-coated seeds and smoking seeds prior to sowing (Holmes and Richardson 1999).

3.2.4 Seed and seedling predation

Through both granivory and herbivory, seed and seedling predators have an impact on seed germination and seedling development. In restoration the presence or exclusion of these predators is likely to inhibit or promote seedling recruitment respectively. Likely seed and seedling predators occurring in renosterveld are small mammals, insects and molluscs (Midoko-Iponga et al. 2005). The majority of small-mammal species in fynbos are granivorous (Rusch 2012), while many eat shoots and stems of grasses, sedges, reeds and other plants and some species eat insects (Stuart and Stuart 2007). Fossorial rodents (for example mole rats) subsist predominantly on the below-ground parts of geophytes (Rebelo et al. 2006). The dominant small-mammal species occurring in the game camp, *Otomys irroratus* (Vlei Rat) (Dreyer 2012) favours grassy hillsides and specialises in feeding on shoots and stems of grasses, sedges, reeds and other plant material (Stuart and Stuart 2007).

Small-mammal population dynamics are affected by fire events and consequently there are implications for restoration where burning is implemented. Studies, indicating both positive and negative responses of small-mammal populations to fire, predominantly indicate a reduction in small-mammal density and/or levels of granivory within the first several months following a fire (Midgley and Clayton 1990, van Hensbergen et al. 1992, Bond and van Wilgen 1996, Dreyer 2012). Bond (1984) found that herbivory of post-emergence *Proteaceae* seedlings in the unburnt areas was very high whilst negligible in the burnt areas. However, small-mammal abundance, after initially declining in response to fire (and remaining low for six to nine months), has been found to increase

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to a greater number than before the fire (Kruger and Bigalke 1984, van Hensbergen et al. 1992, Dreyer 2012).

3.2.5 Competition

The invasion of alien, annual grasses is problematic in renosterveld (van Rooyen 2003, Musil et al. 2005, Stanway 2007, Muhl 2008) which may be particularly susceptible to invasion due to the low occurrence of indigenous, annual grass species (Le Maître and Midgely 1992). Succession following disturbance is often characterised by an increase of alien, annual grasses which suppress indigenous species (Stylinski and Allen 2001, Milton 2004, Allen et al. 2005, Musil et al. 2005, Keeley 2006). Alien, annual grasses respond positively to disturbance (Dowling 1996, Milton 2004, Muhl 2008, Terblanche 2011) and areas become increasingly susceptible due to disturbance resulting from fires, vegetation clearance, ploughing, soil nitrogen enrichment (predominantly from agricultural run-off and nitrogen-fixing alien species) (Milton 2004) grazing and fragmentation (Kemper et al. 2000, van Rooyen 2003). In the Cape Metropolitan Area, increased air pollution, as nitrogen from car exhausts, has been identified amongst other factors as a threat as it promotes grass cover which in turn leads to a higher fire frequency (Wilson et al. 2009). Alien, annual grasses have been found to alter their environment to their own benefit providing a positive feedback which perpetuates the invasion (Milton 2004). They have been found to produce larger seed banks than indigenous annual grasses (Milton 2004) and in degraded areas alien grass seed may even dominate seed banks (Milton 2004, Shiponeni and Milton 2006, Muhl 2008, Heelemann 2012, Cowan 2013). Annual grass invasion is widespread on both Signal Hill and Devil's Peak (Britton and Jackelman 1995). In the game camp specifically, vegetation cover is dominated by the alien grasses *Panicum stapfianum, Avena fatua, Briza maxima, B. minor* and *Brachypodium distachyon* (this study). In other areas of Peninsula Shale Renosterveld in Groote Schuur Estate, these invasive annual grasses are less prominent with just less than 50% cover recorded in the *Pinus pinea* stands, approximately 5% cover in the lightly degraded renosterveld and are absent from the older extant renosterveld patches (Cowan 2013).

Competition between neighbouring plants is the greatest factor affecting seedling mortality and competitive ability varies within and among species (Fenner 2005). Alien, annual grasses germinate and grow faster than native species (Young and Allen 1997) and have been found to exert direct competitive pressure on native species (Sharma 2010). Studies, predominantly from Mediterranean-type systems, indicate how alien grasses have wide ranging impacts affecting biodiversity, fire regimes, forage stability, soil water content and soil decomposition dynamics which consequently have an effect on ecosystem structure, function and resources (Hobbs 2001, Vila et al. 2000, Lenz et al. 2003, Musil et al. 2005). Local studies have shown the suppression of renosterveld species in response to alien, annual grass dominance (Vlok 1988, Le Maître and Midgely 1992, Musil et al. 2005, Sharma 2010). In addressing the competitive effect of alien grasses, through broad-spectrum herbicide-application for example, weed species (and indigenous species in leaf) are expected to decrease whilst the subsequent recruitment response of indigenous sown seedlings is expected to increase.

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In addition to the effects of direct competition, annual grass leaf litter has been found to have numerous impacts in many diverse ecosystems outside of South Africa (Carson and Peterson 1990, Facelli and Pickett 1991, Facelli 1994, Lenz et al. 2003). The physical presence of litter has been found to inhibit seedling emergence and establishment (Facelli and Pickett 1991, Facelli 1994) and chemical leachates from the litter may also play a role in seedling inhibition (Lenz at el. 2003). Litter has been found to alter soil temperature regimes (Lenz et al. 2003) and soil moisture availability (Facelli 1994) and reduce levels of light (Carson and Peterson 1990, Facelli and Pickett 1991) which may inhibit or delay seed germination and establishment (Lenz et al. 2003, Hartman and Kester 2011), cause seedling etiolation and poor root system development (Groves et al. 2003) and promote fungal attack that increases mortality (Facelli et al. 1999, Lenz et al. 2003). Grass litter has been found to increase the presence of seed and seedling predators such as plant-chewing arthropods (Facelli 1994). In consideration of the numerous factors addressed above, which contribute to the competitive advantage of alien, annual grasses over indigenous species, the reduction and management of invasive, alien grasses emerges as critical to ecological restoration particularly in the implementation of seed-based interventions and a post-fire environment.

The above consideration of the role of ecological drivers in renosterveld recruitment indicates how employment and manipulation of these ecological drivers is central to restoration due to their ability to considerably shape restoration outcomes and trajectories. In order to optimise restoration outcomes these drivers need to be appropriately incorporated into restoration protocols. However, our understanding of the mechanisms underpinning these drivers (Rebelo et al. 2006) and their effects on a restoring community is deficient as reflected by the limited success in renosterveld restoration to date. Experimentation to test these driving factors and their interactions on seedling recruitment is critical to advancing the science of renosterveld restoration ecology and is the pragmatic next step.

Based on the literature review and local expert opinion, this portion of the study sets out to test an *a priori* ecological-response model (Table 3.1) incorporating five restoration interventions (burning, tillage, herbicide-application, rodent-exclusion and seeding) expected to enhance restoration outcomes with respect to several seed germination criteria. Adapted from increaser-decreaser models, the proposed ecological-response model has been simplified to predict the majority response of the community within the first season following intervention implementation.

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Table 3.1: Proposed ecological-response model.
The model proposed and tested in this study incorporates five factors expected to enhance restoration outcomes. Predicted responses represent the anticipated majority response within the first season of intervention implementation where '+'=increase, '0'=no change and '–'=decrease.

INTERVENTION	SEEDED SPECIES		EXISTING INDIGENOUS SPECIES		WEED SPECIES	
		predicted		predicted		predicted
BURNING	Density	+	Density	+		
	Richness	+	Richness	+		
	Cover	+	Cover	+	Cover	+
	Height	+				
TILLAGE	Density	+	Density	–		
	Richness	+	Richness	–		
	Cover	+	Cover	–	Cover	+
	Height	+				
HERBICIDE	Density	+	Density	–		
	Richness	+	Richness	–		
	Cover	+	Cover	–	Cover	–
	Height	+				
RODENT-EXCLUSION	Density	+	Density	0		
	Richness	+	Richness	0		
	Cover	+	Cover	0	Cover	0
	Height	+				
SEEDING	Density	+	Density	0		
	Richness	+	Richness	0		
	Cover	+	Cover	0	Cover	0
	Height	+				

This chapter sets out to empirically determine the effectiveness of 32 interventions (including the control) (comprised of all combinations of seeding, burning, tillage, herbicide-application and rodent-exclusion) with respect to eight seed germination criteria: seedling density, species richness, canopy cover and seedling height of the 31 seeded species; seedling density, species richness and canopy cover of the existing indigenous species and canopy cover of the weed species. Thereafter, the comparison of the empirical experimental outcomes of the main interventions in relation to the predicted outcomes makes a theoretical contribution through refining the conceptual ecological-response model for restoration in Peninsula Shale Renosterveld.

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3.3 Methods

3.3.1 Study area

Location

The field experiment was situated in the game camp in Groote Schuur Estate (33°56'53.57"S, 18°27'56.34"E, 43 m amsl) which occurs within TMNP on the north-eastern slopes of Devil's Peak (Figure 3.1). The field experimentation was located in a shallow valley between two ridgelines and immediately to the south east of a minor drainage line. Refer Chapter 1 (theoretical framework) for a description of the vegetation, land-use, geology, soil and climate.

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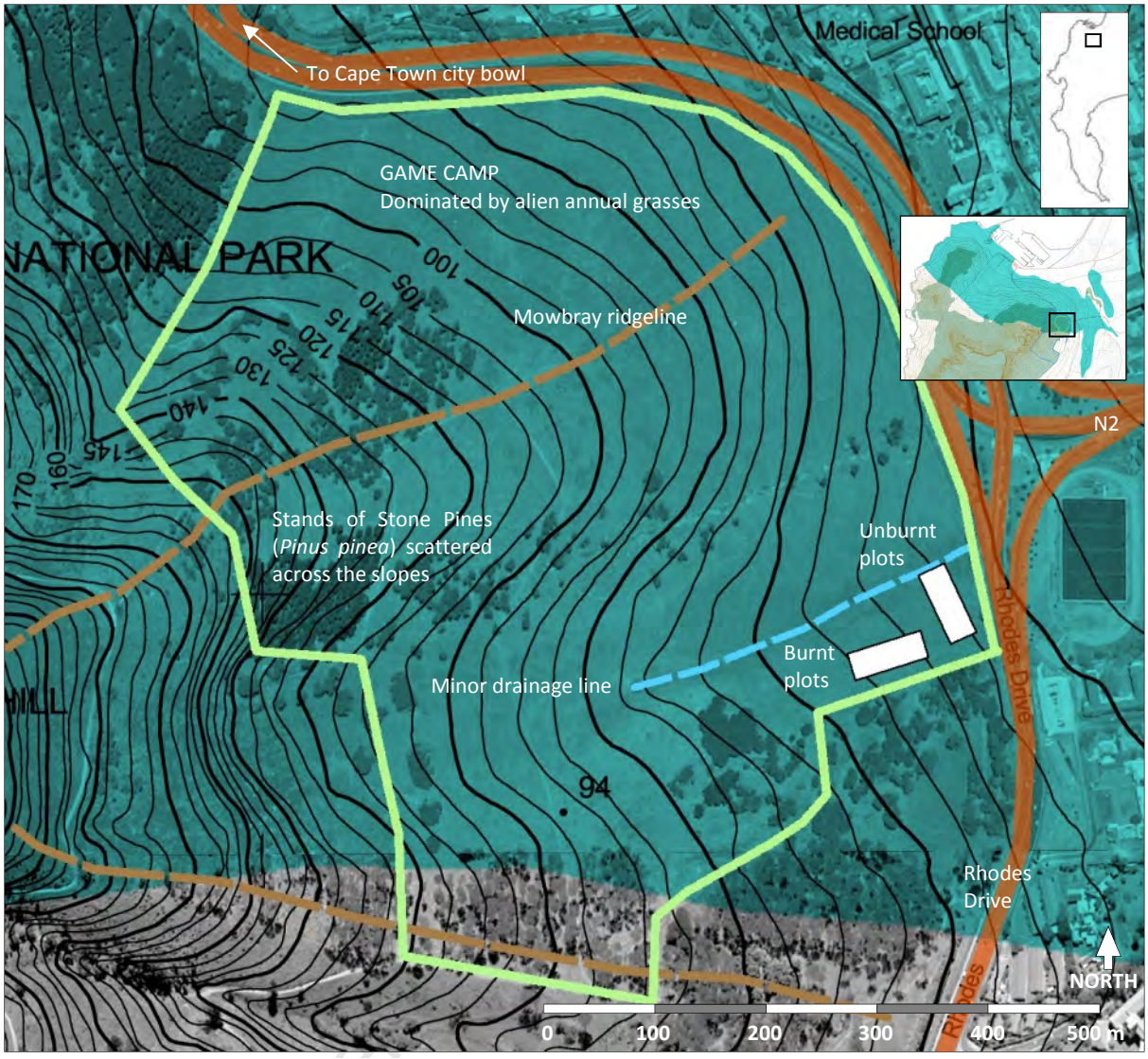

Figure 3.1 Plan of field-experiment study area.
Position of the experimental field site (burnt and unburnt plots) within the game camp of Groote Schuur Estate as incorporated into TMNP. The plots are situated in a shallow valley between two ridgelines, immediately to the south east of a minor drainage line and on a slope gradient more gradual than 1:10.

3.3.2 Field trial

The experimental design is complete factorial as the field experiment implemented 32 interventions (including a control) comprised of five crossed factors (burning, tillage, herbicide-application, rodent-exclusion and seeding) at two levels (implemented and not implemented), and replicated four times (replicate A, B, C and D) (refer Appendix 3.1 Experimental design). Due to the impracticality of implementing grazing over so many treatment levels in such small areas, grazing has

not been incorporated. In the remaining sections the 32 interventions are referred to with acronyms (refer Table 3.2).

Table 3.2: Intervention acronyms.
Where C=control, S=seeding, B=burn, T=tillage, H=herbicide-application and R=rodent-exclusion.

UNSEEDED INTERVENTIONS		SEEDED INTERVENTIONS	
1.	C	17.	S
2.	B	18.	BS
3.	T	19.	TS
4.	H	20.	HS
5.	R	21.	RS
6.	BT	22.	BTS
7.	BH	23.	BHS
8.	TH	24.	THS
9.	BR	25.	BRS
10.	TR	26.	TRS
11.	HR	27.	HRS
12.	BTH	28.	BTHS
13.	BTR	29.	BTRS
14.	BHR	30.	BHRS
15.	THR	31.	THRS
16.	BTHR	32.	BTHRS

A slope analysis was conducted and the plots were sited in an area with a gradual gradient of greater than 1:10 to minimise soil erosion. Due to permitting, logistical and cost constraints, the burnt and unburnt plots were consolidated into two areas as burning was restricted to a single event and area (refer Limitations). Seeded plots were placed in alternate rows yet sown as individual units. The interventions T, H and R were arranged randomly with the exception of 12 interventions, from which soil moisture was measured, which were moved closer if occurring more than 7 m from the data logger stations.

The Newlands Working on Fire team implemented the prescribed burn on 26th March 2012. A 20 m perimeter firebreak was brushcut and foam retardant applied to the outskirts. The fire was ignited and promoted by a diesel-petrol mix, applied sparingly by two individuals who laced their way across the area. The low biomass, consisting predominantly of alien, annual grass litter, burnt rapidly. The area (81 × 21 m) burned evenly and burnt out within 11 minutes. Once burnt, the plots were measured, pegged, demarcated with fishing line and labelled. Each intervention was implemented in a 25 m² plot (5 × 5 m) with a 9 m² core plot (3 × 3 m). Each block (burnt and unburnt) was fenced to prevent herbivory from the five resident Quagga Project animals. Trenching along the fence lines enabled the extension of the fencing mesh to 300 mm below ground level to exclude porcupine (*Hystrix africaeaustralis*) which typically target roots, bulbs and tubers (Stuart and Stuart 2007) and have been found to act as ecosystem engineers (Bragg et al. 2005). The fencing mesh apertures were large enough to permit small mammals passage (approximately 50 × 50 mm diamond mesh). Tillage, carried out by two individuals, commenced with the first rains in the second week of April 2012 and was completed by the end of the month. Tillage was carried out by hand and entailed turning the soil to the depth of a garden fork (approximately 150 mm). Herbicide (non-specific, post-emergent

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glyphosate 'Round-up 360' by Dow Agro-Sciences) was applied on 7th and 8th May 2012 in low wind conditions and a rectangular spray hood was attached to the lance to increase accuracy and minimise drift into neighbouring plots. Allowing for a herbicide breakdown period of 11 days (7 days recommended by manufacturers) the seed mix was broadcast sown into each of the 64 plots on 19th and 20th May 2012 (refer methodology from chapter 2 for a full account of seed collection and processing). During the two weeks prior to sowing, the seed was weighed, divided into 64 batches and placed into 5 ℓ ice-cream containers whilst the five species requiring hot water treatment were similarly weighed, divided and placed in small cotton bags in preparation for soaking. The 64 containers and 64 cotton bags were placed in batches in the smoking tent at Kirstenbosch and smoked on 16th and 17th May 2012. The night prior to sowing, the cotton bags were placed in 80˚C water and allowed to cool and soak overnight. On sowing, on a plot-by-plot basis, each container of seed mix received seed from one cotton bag, was topped-up with approximately 2.5 ℓ of horticultural sand, moistened with a water-fungicide dilution (Apron XL with active ingredient mefenoxam) and thoroughly stirred. The moist sand-seed mix from one container was broadcast into one 25 m² plot and the area lightly swept with a stiff broom to promote soil-seed contact. Immediately thereafter, if the plot required a rodent-exclusion cage, a cage was placed over the 1 m² subplot and secured in place with approximately 60 wire pegs (4 mm diameter and 300 mm length). Each cage protected an area just larger than the 1m² subplot (approximately 1.2 × 1.2 m), measured approximately 0.5 m high and had a horizontal perimeter skirt at the base of the cage of approximately 0.25 m to discourage rodents from burrowing underneath. Each cage was cut with an angle grinder from two sheets of mild steel mesh (ungalvanised), portions bent and the seams pop-riveted. Mesh with a very small aperture size (3 × 6 mm) was selected in order to exclude even the smallest granivorous rodents expected to occur on the site, the Pygmy Mouse (*Mus minutoides*). Once positioned and pegged, soil was raked to the edge of the cage skirt to close all gaps. The field work was fully implemented by 20 May 2012. With the aim to reduce the seed set from the dominant invasive annual grasses (*Avena fatua, Briza maxima* and *B. minor*) in October 2012 the grasses in all plots were brushcut to a height of approximately 400 mm as the presence of indigenous seedlings prevented cutting closer to the ground. The effect of this was not measured but it appeared to be ineffectual particularly with respect to the *Avena fatua* plants which subsequently produced inflorescences.

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Photo plate 3.1: The prescribed burn carried out by the Newlands Working on Fire team (March 2012).

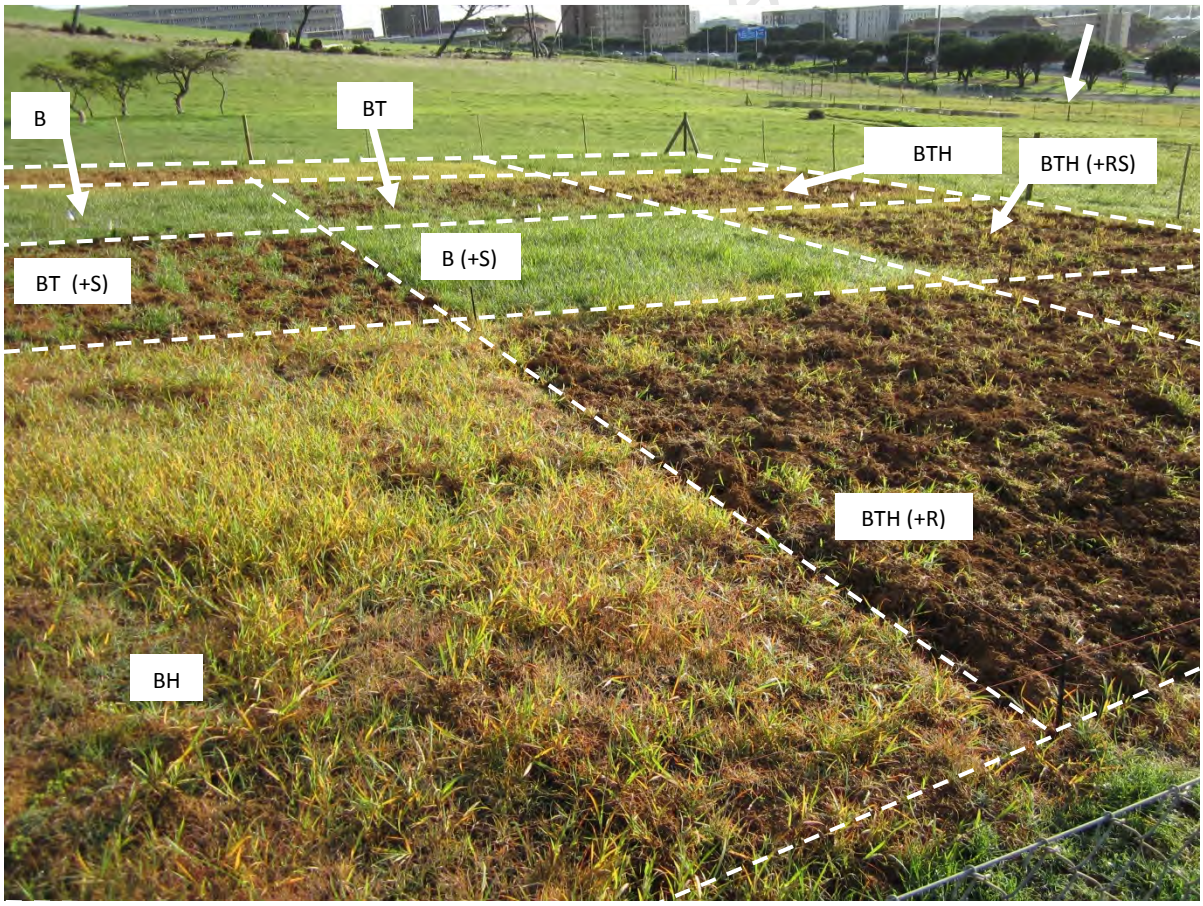

Photo plate 3.2: Within the burnt block, the matrix of plots where tillage and herbicide-application had already been implemented. Tillage exposed the soil and initially reduced grass cover, herbicide plots were starting to turn yellow (left foreground). Interventions still to be implemented indicated in brackets. Perimeter fencing of the unburnt block is visible top right of frame (top right arrow) (taken early May 2012).

Photo plate 3.3: Seed-sand-fungicide mix: each bucket of seed mix contained seed from a 5 ℓ container (64th of each species), seed from one cotton bag (of the hot-water treated species), was topped-up with approximately 2.5 ℓ of horticultural sand, moistened with a water-fungicide dilution and thoroughly stirred prior to broadcast sowing in each seeded plot.

Photo plate 3.4: Rodent-exclusion cages placed over 1 m² subplot and pegged in place. The soil is still to be raked to the edges of the horizontal cage skirt.

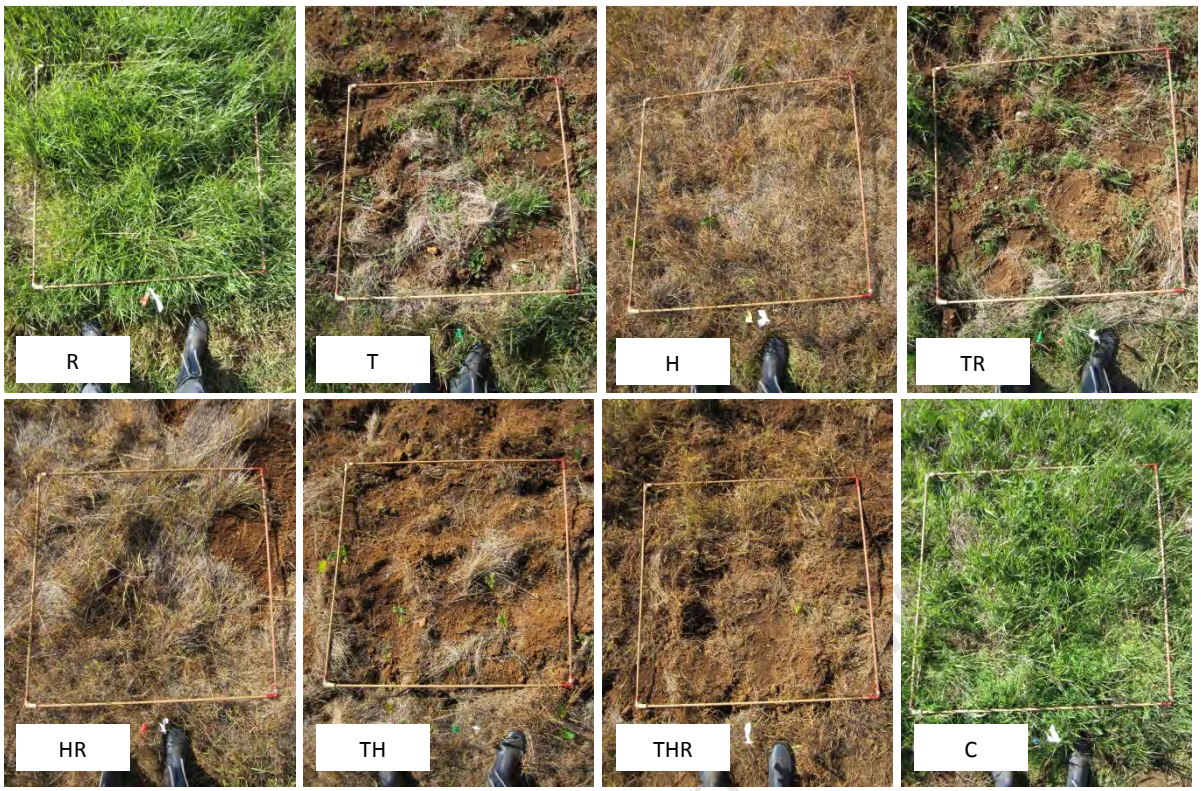

Photo plate 3.5: Unburnt plots and control (with 1 m² quadrat indicating scale). Photographs taken immediately prior to sowing and cage installation (19th and 20th May 2012) to illustrate the different environments into which seed was sown. There is great variability within the unburnt plots as well as among the unburnt and burnt plots, most notably high to low grass cover, also more thatch than in the burnt plots (refer below).

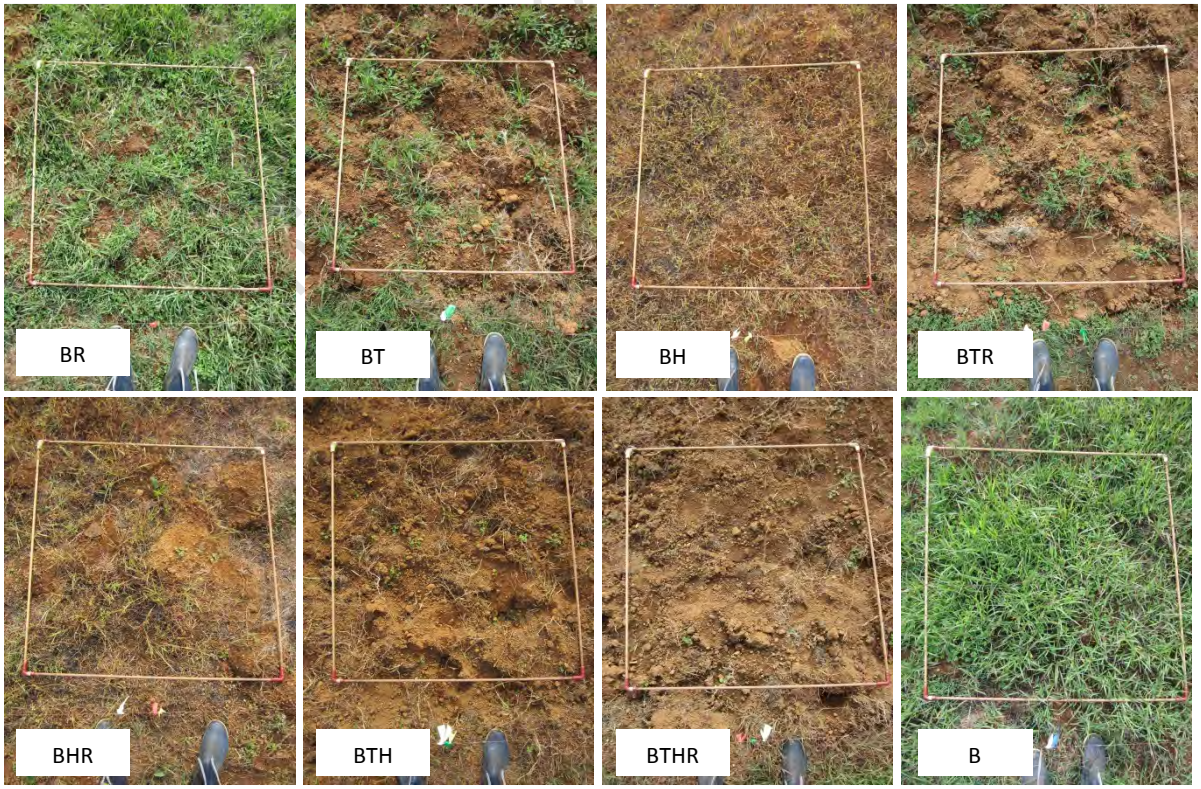

Photo plate 3.6: Burnt plots (with 1 m² quadrat indicating scale). Photographs taken immediately prior to sowing and cage installation (19th and 20th May 2012) to illustrate the different environments into which seed was sown. There is great

variability within burnt plots (low to high grass cover) as well as among the burnt and unburnt plots, with marginally less grass cover and greater soil exposure than in the unburnt plots (refer above).

Photo plate 3.7: Burnt plots fully implemented: burnt, tilled, herbicide applied, seed sown and rodent cages installed (May 2012).

Photo plate 3.8: Unburnt plots fully implemented: burnt, tilled, herbicide applied, seed sown and rodent cages installed (May 2012).

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3.3.3 Soil moisture

Soil moisture sensors were installed by SEAON within the experimental site to ascertain differences across the site. All sensors were installed 10 cm below the surface of the soil. Four data loggers were installed (refer Appendix 3.1: Field experiment design) each logger was installed with six Campbell Scientific CL616 volumetric water content sensors. Moisture sensors were installed in 12 different interventions (across different levels of burn, seeding, herbicide-application and tillage) and replicated twice. Prior to sensor installation, soil samples were taken for soil sensor calibration and bulk density tests.

3.3.4 Light readings

Light readings were measured with a light meter inside and outside the rodent-exclusion cages to compare the levels of light reaching the soil and/or vegetation layer. In addition, light readings were

taken the same day inside and outside of the glasshouse at Kirstenbosch Botanical Gardens where the germinability tests took place. In each of the four locations, three replicates of ten readings were measured. Light was measured as ×10 μmol. $m^{-2}.s^{-1}$ and converted to a percentage.

3.4 Data collection

3.4.1 Precipitation and temperature

Precipitation and temperature data were obtained from SAWS for three weather stations located nearest to the field site: the Molteno Reservoir and the Astronomical Observatory (both located in the historic distribution of Peninsula Shale Renosterveld) and Groote Schuur residence. In addition, SAEON set up a weather station on 2nd May 2012 situated approximately 350 m upslope from the burnt and unburnt blocks. Data were downloaded from the weather station by SAEON, more or less on a monthly basis.

3.4.2 Soil moisture and temperature

Each soil moisture station recorded data at two different frequencies, hourly and daily, and stored these data in two separate tables. The loggers were configured to scan the sensors every 30 seconds. Table 1 data was averaged and stored hourly. Table 2 data recorded maximum and minimum data recorded daily. Data were downloaded from the logger stations by SAEON, more or less on a monthly basis.

3.4.3 Light readings

One data collection took place on the 19th November 2012 from the four locations (inside and outside the rodent-exclusion cages, and, inside and outside the glasshouse).

3.4.4 Field trial

Data were collected from 128 plots (32 interventions replicated four times). One 1 m^2 subplot was sampled from each of the plots where rodent-exclusion was implemented (due to only one cage per plot) whilst two 1 m^2 subplots were sampled from the plots without rodent-exclusion resulting in a total of 192 (1 m^2) subplot samples per observation. The subplots were initially randomly selected and thereafter repeatedly sampled. Four observations took place and occurred alternate months starting July 2012 and ending January 2013. Data were collected for the following per 1 m^2 quadrat: the number of seedlings per species (for the existing indigenous species and each of the 31 seeded species), height of five seedlings per species (just for the seeded species); and, percentage canopy cover per species (for the existing indigenous species and seeded species). In addition, a combined canopy cover measurement was taken for all the weed species. Height and canopy cover were only recorded in the second, third and fourth observations. All seeded species and existing indigenous

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species were counted individually except the *Chrysocoma coma-area* and *Felicia filifolia* seedlings which were counted together due to their seedlings being indistinguishable.

3.5 Data analysis

Transformed (ANOVA) and untransformed data (Kruskal-Wallis tests and correlations) were analysed with Genstat statistical package 15th edition (VSN International Ltd. 2012) whilst standard deviations, medians and inter-quartile ranges were calculated using R Statistical software (R Core Team 2013). Graphs were plotted in Microsoft excel (Microsoft Office 2010).

3.5.1 Precipitation and temperature

Of the three weather stations closest to the site, data from Molteno Reservoir were selected for analyses due to being most comparable with the on-site weather station data. Mean monthly in-field precipitation and temperature were analysed for the period January 2012 to January 2013 and compared with the 50 and 12 year means for precipitation and temperature respectively.

3.5.2 Soil moisture

Kruskal-Wallis tests were run for soil moisture on the untransformed data for each of the following three group factors: B (burnt), S (seeded) and location.

3.5.3 Light

Untransformed data were used for the analysis of both the glasshouse and rodent-exclusion cage observations. Kruskal-Wallis tests were run separately for the following group factors: glasshouse (inside) glasshouse (outside) and cage (inside) and cage (outside).

3.5.4 Field trial

Data from the field trial were analysed with respect to eight response variables: seeded species (seedling density, species richness, canopy cover and seedling height), existing indigenous species (seedling density, species richness and canopy cover) and weed species (canopy cover).

Seeded species

The data from the two 1 m² subplots per intervention were averaged (except in rodent-exclosure plots where there was only one 1 m² subplot). For the analyses of the criteria pertaining to the seeded species (density, richness, canopy cover and height), the data sets for each were severely positively skewed due to the experimental design where half of the plots were unseeded resulting in a high proportion of zero values. As such, no data sets could be adequately transformed. Consequently, to establish the effect of seeding versus not seeding on each of the response variables, a Kruskal-Wallis test was run with the group factor S (seeding). Three additional Kruskal-Wallis tests were run to

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ascertain the effect of seeding on the overall indigenous community with respect to density, richness and canopy cover (for two variates: existing indigenous species, and, existing indigenous species plus seeded species). Density and richness data from the July 2012, September 2012, November 2012 and January 2013 observation were analysed and canopy cover data from the September 2012, November 2012 and January 2013 observations were analysed. Thereafter, data from the unseeded plots were deleted from the respective data sets, reducing the analysis from 32 interventions to the 16 seeded interventions. The seeded-only data were acceptably transformed for each response variable. Once transformed, a general ANOVA was run to determine the main effects and all interaction effects for burn, tillage, herbicide-application and rodent-exclusion and Tukey post-hoc tests were run. For a description of transformation, power and residuals for each ANOVA refer Appendix 3.2. In the analysis of seedling height of the seeded species, *in lieu* of averaging the mean height (cm) across all of the species recorded in a subplot, it was deemed more accurate to use the height of the Chryso-Fel category (*Chrysocoma coma-area* and *Felicia filifolia* seedlings) as a surrogate for seedling height as these seedlings were virtually ubiquitous and recorded in most plots over the four observations. A separate comparison of height per species was made for the seedlings on site versus those in the glasshouse. A Kruskal-Wallis test was run to ascertain the effect location (glasshouse versus in-field experiment) had on seedling height.

In addition to seedling density, species richness, canopy cover and seedling height; seedling survival of the 31 seeded species per observation was analysed with respect to survival per growth, per species and per intervention.

Existing indigenous species

Existing indigenous species were analysed with respect to seedling density, species richness (from September 2012) and canopy cover (from September 2012, November 2012 and January 2013) all analyses using seeded and unseeded data. General ANOVA analyses were run for the main factors (burning, tillage, herbicide-application, rodent-exclusion and seeding) and all interactions. For a description of transformation, power and residual for each ANOVA refer Appendix 3.3.

Weed species

An ANOVA was run on data (seeded and unseeded) from November only due to an inability to successfully transform the data from the three observations (September 2012, November 2012 and January 2013). The November data were selected as a surrogate as weed cover was greatest during this observation. Thereafter a general ANOVA was run on arcsine transformed data for all interactions of the main factors (burn, tillage, herbicide-application, rodent-exclusion and seeding) (Appendix 3.4).

Limitations

Being a complete factorial design (five crossed factors over two levels) ideally the 64 burnt and 64 unburnt plots would have been randomly located within the study site. However, several constraints

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informed the in-field layout of the experimentation whereby the burnt and unburnt plots were consolidated out of necessity.

Locational constraints: the permitting authority, SANParks, stipulated the plots be sited on slopes more gradual than 1:10. A slope analysis was conducted, identifying only two possible locations meeting this condition. Furthermore, access for fire-fighting vehicles and proximity to a fire hydrant was required restricting the two possible options to just the single location. Spatial constraints: each burnt area required a 20 m perimeter firebreak. In addition, within the single feasible location identified, the experimentation needed to be sited in between three sets of vehicle tracks (to avoid compacted soils) and a stipulated distance from an *Acacia* tree where film shoots frequently take place. Cost constraints: the area requiring firebreak preparation and thus the cost of the workforce to create and defend it was restricted. An initial cost estimate was conducted and the project budget, with respect to large-herbivore exclusion fencing, was in addition limited to approximately 200 m of fencing.

Given the consolidation of the burnt and unburnt plots into two areas, the statistical analyses as conducted in this portion of the study, did not account for unwanted influence of extraneous environmental variables resulting from the field (field effects). As such, the results pertaining to burnt plots versus unburnt plots need to be interpreted in this context, where the mechanisms responsible for the measured effects may not be restricted to the burn alone but may in part be due to other environmental variables. With the exception of soil moisture (which confirmed a soil moisture difference in one of the four locations) the remaining potential environmental variables were not tested.

Had the locational, spatial and cost constraints been less inhibitive, an alternative in-field layout would have been to consolidate just the 32 plots per replicate (16 burnt and 16 unburnt plots separated by a 20 m firebreak) into four blocks and to site these in four different locations. This alternative layout of partial consolidation, although in this instance precluded due to unfeasibility, would have enabled an analysis that tested for the effect of the burn factor whilst taking variability resulting from field effects into account.

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Canopy cover correlation

A nonparametric correlation test (Spearman's rank correlation) was run for each observation (September 2012, November 2012 and January 2013) on untransformed data (seeded plots only) for percentage canopy cover for the variates: seeded species, existing indigenous species and weed species.

Vegetation structure

The canopy cover data were plotted for each of the 16 seeded interventions for September 2012, November 2012 and January 2013: firstly with respect to seeded species, existing indigenous species and weed species; and secondly with respect to the seeded species as categorised into growth forms (graminoids, geophytes, forbs, succulent shrubs, low shrubs and tall shrubs).

3.6 Results

3.6.1 Precipitation and temperature

Groote Schuur weather station, although closest to the site, received a far greater mean rainfall than that of the on-site weather station for the period May 2012 to January 2013 (Figure 3.2). For the same period, the on-site weather station received marginally less rainfall than Molteno Reservoir in general, and the trends in rainfall from the two sites followed each other closely.

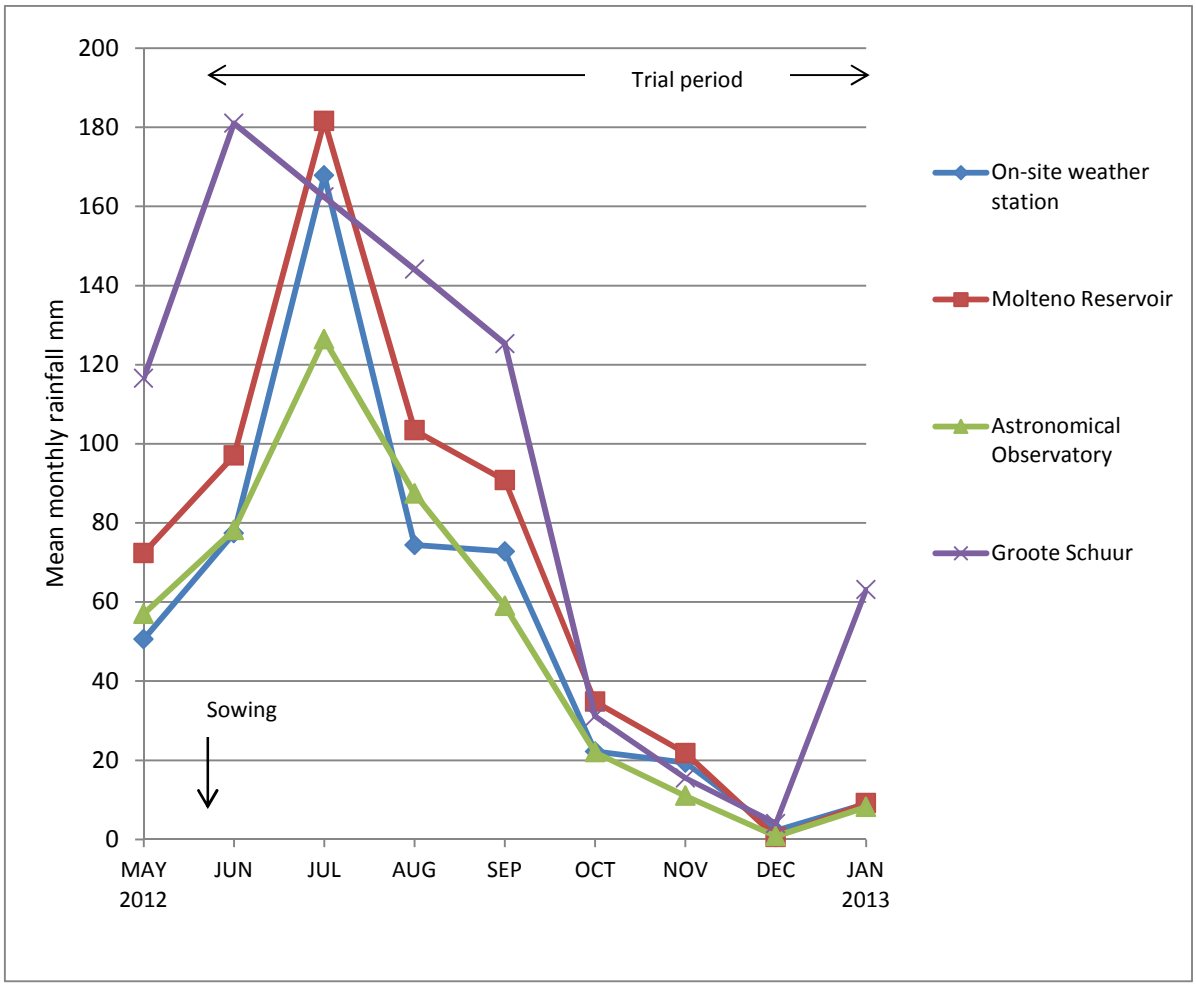

Figure 3.2: Mean monthly rainfall (May 2012 to January 2013) for Molteno reservoir, Astronomical Observatory, Groote Schuur and on-site weather station.
The rainfall pattern of the on-site weather station, although marginally lower in general, closely followed the pattern of the Molteno Reservoir.

For the period May 2012 to January 2013 (Figure 3.3), Molteno reservoir recorded approximately 100 mm (14%) less precipitation than the 50 year mean. The temporal distribution of precipitation for the period also differed from that of the 50 year mean where, with the exception of the mid-winter peak, fewer mm were recorded for the majority of months. As the rainfall patterns of Molteno Reservoir and the on-site weather station tracked each other closely, it likely that similar relative trends were experienced on site.

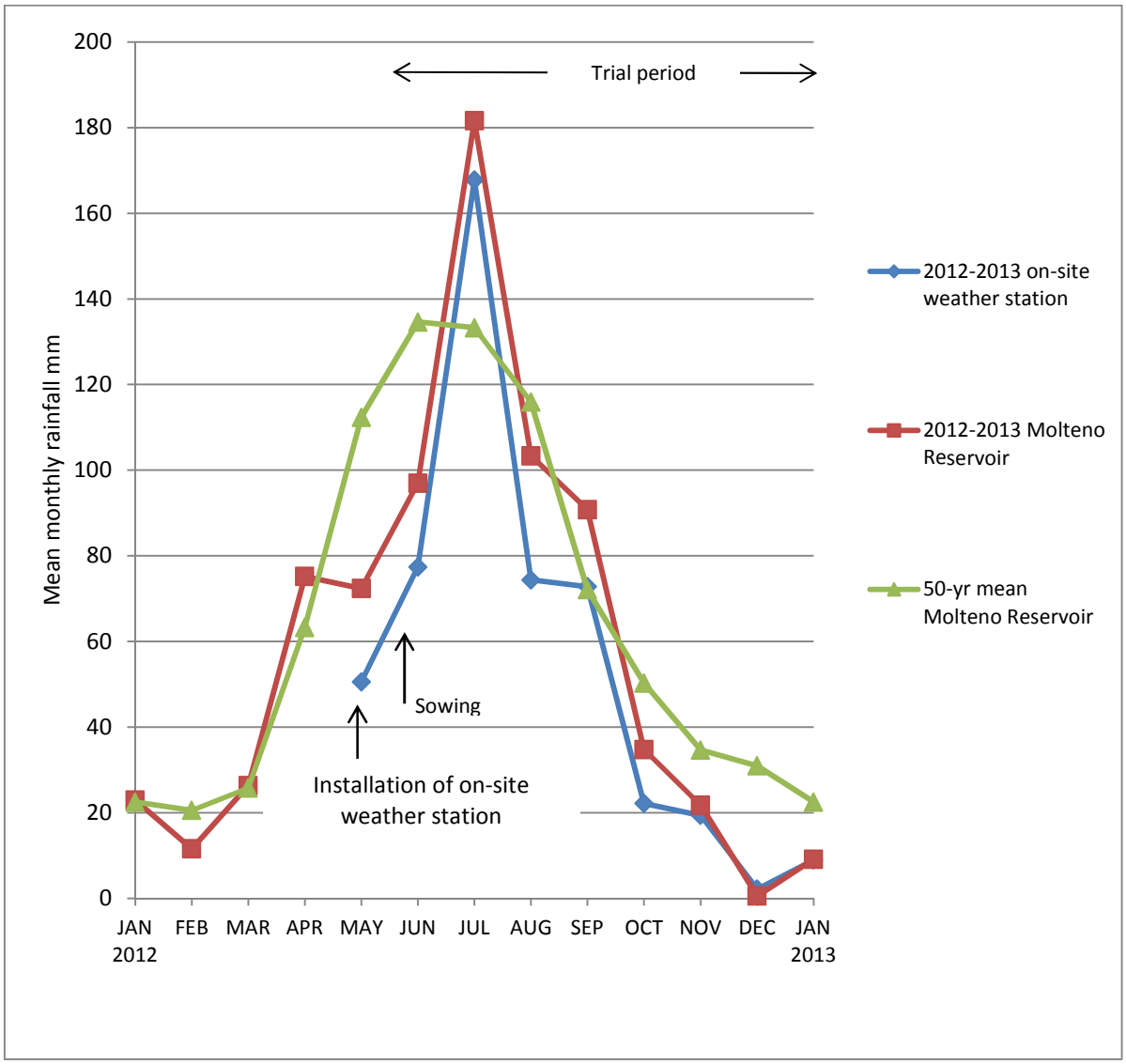

Figure 3.3: Mean monthly rainfall (January 2012 to January 2013) for Molteno reservoir and the on-site weather station with 50-yer mean for Molteno Reservoir.
The mean monthly rainfall for the site was most likely lower than the 50 year mean with most months being drier with the exception of the mid-winter July peak.

The mean monthly temperatures of Molteno Reservoir (for the period 2012 to 2013) were in general cooler than those of the 12 year mean with the exception of December 2012 which was the only month exceeding the 12 year mean (Figure 3.4). The temperature trends of Molteno Reservoir and the site were very similar (with the site being marginally cooler at times) so one may expect that the 2012-2013 trends experienced at Molteno Reservoir were similarly experienced on the study site: cooler temperatures for the trial period in general with the exception of a hotter December 2012.

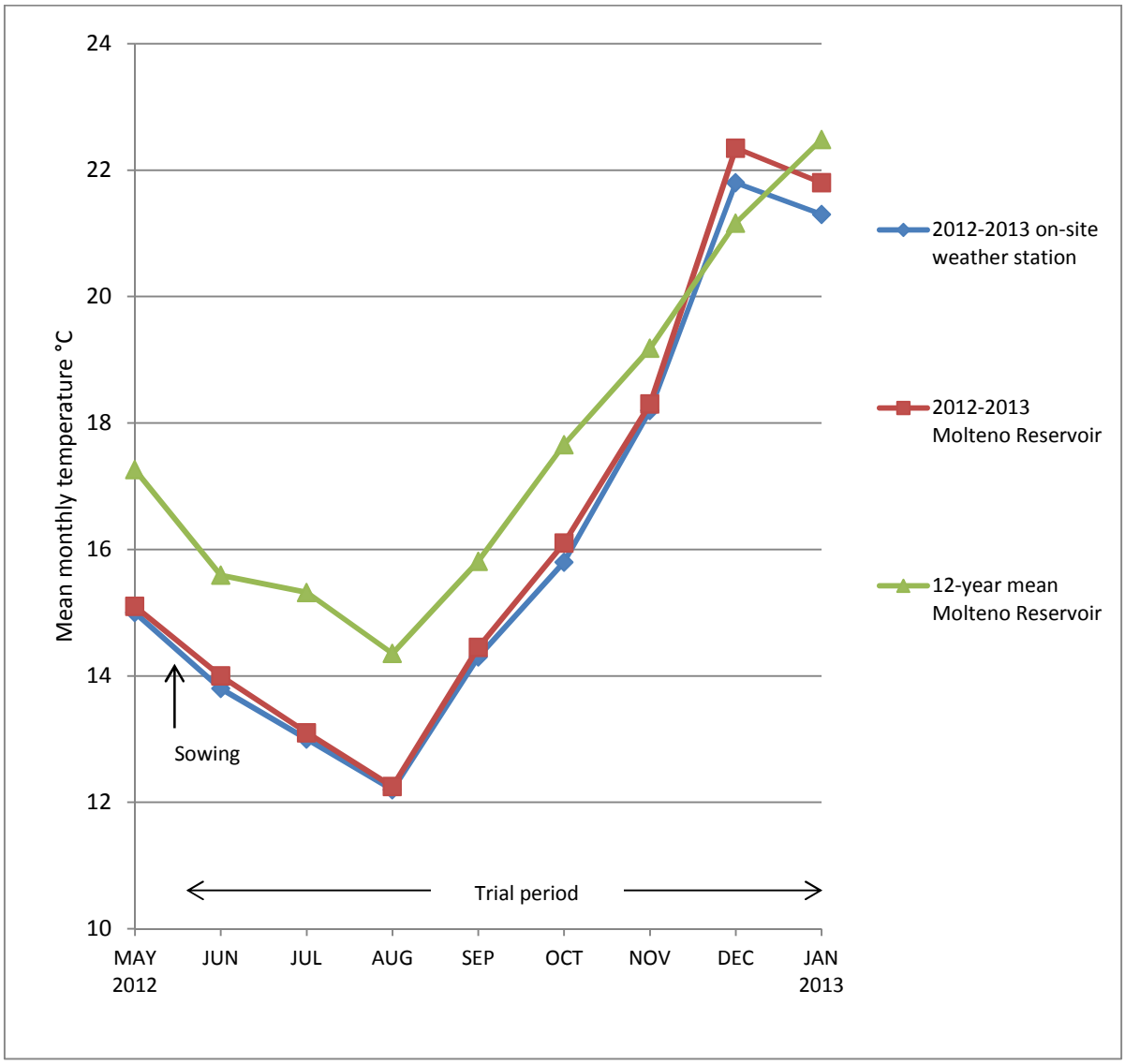

Figure 3.4: Mean monthly temperature (May 2012 to January 2013) for Molteno reservoir and the on-site weather station with 12-yer mean for Molteno Reservoir.
The mean monthly temperatures experienced on site, with the exception of a hotter December 2012, were most likely notably cooler in general.

3.6.2 Soil moisture

Over the seven month period from July 2012 to January 2013, the effect of seeding on soil moisture was non-significant (x^2=1.578, p=0.209) yet both the effects of burning and location were found to be significant (x^2=4.278, p=0.039 and x^2=22.500, p=<0.001 respectively) (refer Appendix 3.5).

The significant effect of location (Figure 3.5) indicated that soil moisture was fairly uniform across the field experimental site with the exception of the one area (Unburnt block D) which had a median % volumetric water content of approximately 8% higher than the other three logger locations over the same period (Figure 3.5). This significantly higher % soil moisture in Unburnt block D was due to its relative proximity to a minor drainage line situated immediately to the north of the experimental plots (refer Figure 3.1). This variability in soil moisture has the potential to influence seedling recruitment.

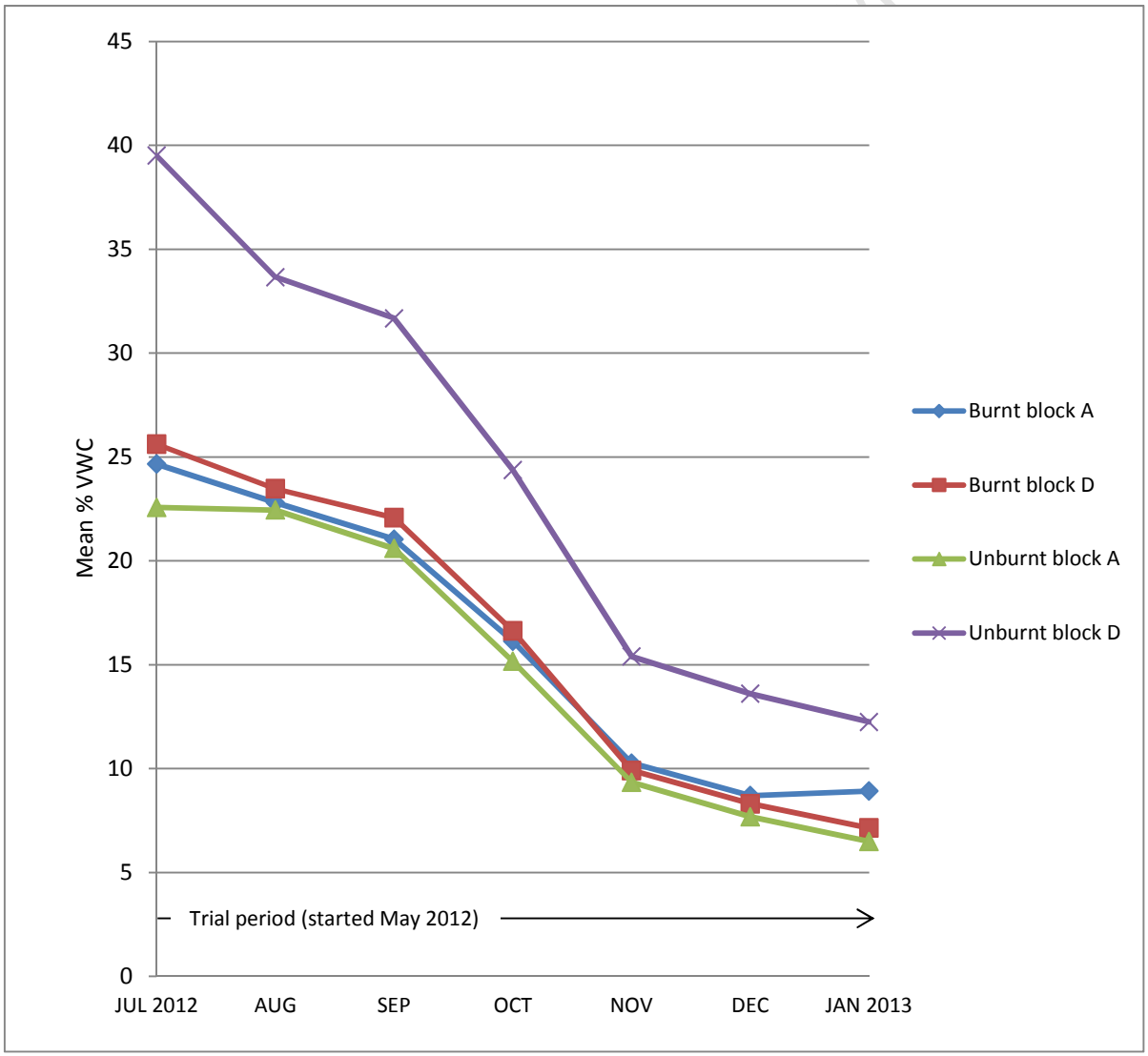

Figure 3.5: Mean soil volumetric water content (%) from July 2012 to January 2013.
Soil moisture was relatively uniform across the site with the exception of one location (Unburnt block D) which was consistently higher due to proximity to a minor drainage line.

3.6.3 Light

Both the rodent-exclusion cages and the glasshouse resulted in significant reductions in light (× 10 µmol.m^{-2}.s^{-1}) relative to light conditions outside of the cages and glasshouse (x^2=44.260, p=<0.001 for both) (refer Appendix 3.6 and Figure 3.6). The glasshouse cut out 76.0% of the light whilst the rodent-exclosure cages cut out 55.7% of the light. The significant reduction in light conditions inside rodent-exclusion cage is likely to have an impact on plant performance and in addition may affect the within-cage microclimate such as reduce soil temperature, alter soil moisture, reduce circulation and consequently affect seed germination, establishment and height.

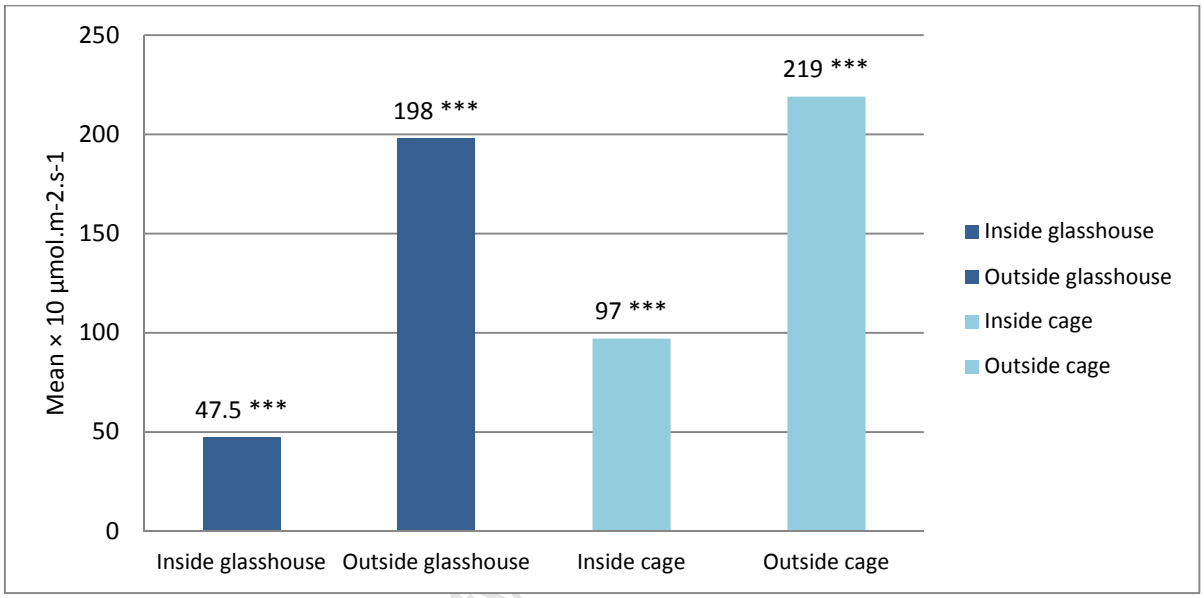

Figure 3.6: Light readings taken from inside and outside the rodent-exclusion cages and the glasshouse.
The rodent-exclusion cages and glasshouse intercepted a large proportion of light resulting in significant reductions. Median and significant effects: * for p<0.05, ** for p<0.01 and *** for p<0.001.

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3.6.4 Field trial

Responses to seeding

Seeding had a significant effect on the density (x^2=298.300, p=<0.001), richness (x^2=298.300, p=<0.001), cover (x^2=204.100, p=<0.001) and height (x^2=250.900, p=<0.001) of seeded species criteria as none of the seeded species were counted in the unseeded plots (refer Appendix 3.7a). The effect of seeding on overall density, richness and canopy cover (i.e. comparing density, richness and canopy cover of existing indigenous species versus the combined density, richness and cover of existing indigenous species plus seeded species), indicated that seeding contributed significantly to all three criteria (density (x^2=8.116, p=0.004), richness (x^2=19.23, p=<0.001) and canopy cover (x^2=4.258, p=0.039)) (refer Appendix 3.7b). Seeding increased overall indigenous seedling density by approximately 18%, species richness by approximately 30% and canopy cover by approximately 18%. Seeding had a significant positive effect on existing indigenous species richness (F=4.77, p=0.031).

Seedling height

Seedlings on site were significantly taller than those in the glasshouse (x^2=6.465, p=0.011) (refer Appendix 3.8a) This comparison of species height (cm) between the two locations is limited to 20 species as fewer species germinated on site (22 out of 31 in the field versus 26 out of 31 in the glasshouse). One species (*Searsia tomentosa*) which did not germinate in the nursery did germinate on site. Among species, only four species were more vigorous in the glasshouse than in the field: *Dimorphotheca pluvialis, Lachenalia fistulosa, Podalyria sericea* and *Tenaxia stricta* (refer Appendix 3.8b).

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Intervention responses

Table 3.3: Summary ANOVA table, chi squared, variance ratio and statistical probability for the 32 interventions (including control).
Derived from Kruskal-Wallis tests and general ANOVA. Significant effects:* for p<0.05, ** for p<0.01 and *** for p<0.001.

SEEDED INTERVENTIONS								
	SEEDED SPECIES				EXITING INDIGENOUS SPECIES			WEED SPECIES
	DENSITY	RICHNESS	COVER	HEIGHT	DENSITY	RICHNESS	COVER	COVER
	4th root transformed	square root transformed	fourth root transformed	square root transformed	square root transformed	untransformed	third root transformed	arcsine transformed
S	x^2 = 298.3 ***	x^2 = 298.3 ***	x^2 = 204.1 ***	x^2 = 250.9 ***	F = 0.53	F = 4.77 *	F = 1.16	F = 0.78
BS	F = 5.52 *	F = 14.17 ***	F = 0	F = 3.23	F = 2.30	F = 0.04	F = 0.01	F = 0.00
TS	F = 201.64 ***	F = 166.6 ***	F = 87.28 ***	F = 0.39	F = 0.04	F = 0.04	F = 0.00	F = 0.39
HS	F = 103.26 ***	F = 109.22 ***	F = 162.19 ***	F = 65.43 ***	F = 2.33	F = 0.59	F = 0.01	F = 1.97
RS	F = 7.27 **	F = 0.03	F = 5.63 *	F = 1.68	F = 0.05	F = 2.00	F = 0.07	F = 0.54
BTS	F = 2.74	F = 4.76 *	F = 0.27	F = 0.09	F = 0.66	F = 1.06	F = 0.40	F = 0.94
BHS	F = 0.65	F = 0.09	F = 2.04	F = 1.04	F = 0.19	F = 0.70	F = 0.64	F = 0.29
THS	F = 21.13 ***	F = 43.91 ***	F = 32.5 ***	F = 9.56 **	F = 0.31	F = 0.70	F = 0.13	F = 0.53
BRS	F = 0.00 ***	F = 1.08	F = 0.31	F = 6.81 *	F = 2.46	F = 1.19	F = 1.89	F = 0.01
TRS	F = 1.27	F = 2.67	F = 4.09 *	F = 0.00 5	F = 0.20	F = 2.58	F = 0.25	F = 0.20
HRS	F = 3.91 *	F = 8.17 **	F = 0.94	F = 2.17	F = 0.00	F = 1.06	F = 0.08	F = 1.02
BTHS	F = 0	F = 0.45	F = 0	F = 0.12	F = 0.14	F = 1.65	F = 0.01	F = 1.51
BTRS	F = 5.11 *	F = 0.27	F = 3.6	F = 4.65 *	F = 0.10	F = 2.38	F = 0.71	F = 0.04
BHRS	F = 0.9	F = 0	F = 0.07	F = 0.6	F = 0.04	F = 0.50	F = 0.15	F = 1.37
THRS	F = 0.03	F = 0.08	F = 0.38	F = 3.52	F = 0.21	F = 1.49	F = 0.88	F = 0.06
BTHRS	F = 8.18 **	F = 7.38 **	F = 1.42	unbalanced	F = 2.20	F = 0.26	F = 0.99	F = 0.11

UNSEEDED INTERVENTIONS								
	SEEDED SPECIES				EXITING INDIGENOUS SPECIES			WEED SPECIES
	DENSITY	RICHNESS	COVER	HEIGHT	DENSITY	RICHNESS	COVER	COVER
	4th root transformed	square root transformed	fourth root transformed	square root transformed	square root transformed	untransformed	third root transformed	arcsine transformed
C								
B					F = 161.16 ***	F = 66.56 ***	F = 4.25 *	F = 18.44 ***
T					F = 65.13 ***	F = 47.24 ***	F = 5.86 *	F = 0.24
H					F = 0.01	F = 5.35 *	F = 16.79 ***	F = 200.30 ***
R					F = 8.79 **	F = 0.00	F = 2.44	F = 48.28 ***
BT					F = 0.23	F = 3.24	F = 1.49	F = 8.76 **
BH					F = 6.13 *	F = 0.33	F = 0.01	F = 7.24 **
TH					F = 1.49	F = 0.10	F = 0.26	F = 33.30 ***
BR					F = 0.01	F = 3.47	F = 0.67	F = 0.64
TR					F = 3.36	F = 0.33	F = 1.07	F = 6.53 *
HR					F = 20.27 ***	F = 2.38	F = 2.35	F = 2.04
BTH					F = 0.92	F = 7.28 **	F = 2.34	F = 0.39
BTR					F = 0.07	F = 0.07	F = 0.07	F = 1.99
BHR					F = 0.51	F = 0.93	F = 0.00	F = 2.57
THR					F = 0.05	F = 0.70	F = 0.05	F = 3.70
BTHR					F = 3.23	F = 0.15	F = 0.05	F = 0.02

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Table 3.4: Summary ANOVA table for the 32 interventions (including the control), response direction (+ve, neutral, negative), 0 (intervention not implemented), 1 (intervention implemented), transformed mean ± standard deviation, [untransformed median (lower quartile, upper quartile)].

SEEDED INTERVENTIONS									
		SEEDED SPECIES				EXISTING INDIGENOUS SPECIES			WEED SPECIES
		DENSITY 4th root transformed [untransformed]	RICHNESS square root transformed [untransformed]	COVER fourth root transformed [untransformed]	HEIGHT fourth root transformed [untransformed]	DENSITY square root transformed [untransformed]	RICHNESS [untransformed]	COVER third root transformed [untransformed]	COVER arcsine transformed [untransformed]
S		+ ve	+ ve	+ ve	+ ve	- ve	+ ve	- ve	+ ve
	0	0 ± 0 [0 (0, 0)]	0 ± 0 [0 (0, 0)]	0 ± 0 [0 (0, 0)]	0 ± 0 [0 (0, 0)]	14.282 ± 4.559 [206.5 (118.8, 292.8)]	[5.0 ± 2.0]	2.530 ± 1.265 [19.0 (4.4, 46.6)]	1.138 ± 0.435 [92.0 (73.9, 100.0)]
	1	1.452 ± 0.937 [6.5 (0.9, 18.6)]	1.100 ± 0.713 [1.25 (0.5, 6.6)]	0.654 ± 0.507 [0.5 (0, 1.3)]	1.342 ± 1.778 [3.8 (2.6, 5.3)]	13.962 ± 4.005 [205.5 (127.9, 274.6)]	[5.5 ± 1.9]	2.398 ± 1.180 [15.1 (4.2, 37.0)]	1.174 ± 0.421 [95.0 (79.8, 100.0)]
BS		- ve	- ve	+ ve	+ ve	+ ve (despite inhibitive effect of S)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (despite positive effect of S)
	0	2.542 ± 1.431 [40.8 (8.4, 193.6)]	1.993 ± 1.017 [4.5 (2.0, 7.0)]	1.246 ± 0.674 [3.0 (0.8, 6.5)]	1.584 ± 0.256 [5.9 (3.9, 9.7)]	11.173 ± 3.480 [118.5 (85.8, 203.9)]	[4.0 ± 1.6]	2.398 ± 1.166 [16.8 (5.9, 41.0)]	1.224 ± 0.385 [96.5 (81.8, 100.0)]
	1	2.288 ± 1.215 [34.5 (6.0, 90.0)]	1.706 ± 0.828 [3.5 (1.5, 5.0)]	1.247 ± 0.715 [3.3 (0.7, 6.6)]	1.662 ± 0.338 [7.5 (3.6, 12.8)]	16.408 ± 2.704 [264.8 (211.8, 333.0)]	[6.5 ± 1.5]	2.519 ± 1.326 [15.8 (5.0, 56.0)]	1.086 ± 0.471 [92.5 (72.0, 100.0)]
TS		+ ve	+ ve	+ ve	- ve	+ ve (despite inhibitive effect of S)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S	+ ve (no inhibitive effects)
	0	1.646 ± 1.146 [9.0 (1.0, 37.9)]	1.357 ± 0.898 [2.0 (0.5, 4.0)]	0.951 ± 0.763 [1.0 (0.0, 4.0)]	1.675 ± 0.349 [6.1 (4.3, 15.2)]	12.559 ± 3.935 [156.5 (105.0, 218.0)]	[4.2 ± 1.9]	2.378 ± 1.260 [18.8 (2.0, 42.3)]	1.141 ± 0.479 [95.8 (71.0, 100.0)]
	1	3.184 ± 1.025 [91.8 (39.6, 239.5)]	2.343 ± 0.682 [5.0 (3.5, 8.0)]	1.542 ± 0.455 [5.0 (2.9, 8.3)]	1.593 ± 0.267 [6.5 (3.4, 9.5)]	15.771 ± 3.646 [262.5 (179.4, 332.25)]	[6.4 ± 1.6]	2.543 ± 1.110 [17.0 (5.0, 40.0)]	1.197 ± 0.349 [94.5 (85.3, 100.0)]
HS		+ ve	+ ve	+ ve	+ ve	- ve (no positive effects)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (no positive effects)
	0	1.865 ± 1.424 [11.3 (0.5, 68.6)]	1.450 ± 1.036 [2.0 (0.5, 5.0)]	0.843 ± 0.625 [1.0 (0.0, 3.1)]	1.467 ± 0.198 [4.6 (2.8, 7.5)]	14.634 ± 5.186 [225.5 (133.6, 329.8)]	[4.6 ± 2.1]	2.285 ± 1.338 [13.3 (2.4, 41.3)]	1.453 ± 0.227 [100.0 (99.5, 100.0)]
	1	2.965 ± 0.958 [64.0 (30.0, 188.3)]	2.249 ± 0.606 [5.0 (3.0, 7.0)]	1.650 ± 0.496 [5.8 (3.0, 11.8)]	1.729 ± 0.314 [8.6 (4.9, 15.5)]	14.278 ± 3.504 [211.3 (148.0, 263.4)]	[5.7 ± 2.0]	2.655 ± 1.118 [20.0 (7.6, 42.8)]	0.916 ± 0.426 [81.0 (59.8, 94.3)]
RS		- ve	- ve	- ve	+ ve	- ve (no positive effects)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	+ ve (no inhibitive effects)
	0	2.561 ± 1.250 [47.0 (11.3, 164.6)]	1.856 ± 0.854 [3.5 (1.9, 6.0)]	1.322 ± 0.660 [3.9 (1.0, 7.0)]	1.591 ± 0.278 [6.2 (3.2, 10.2)]	14.883 ± 4.131 [223.75 (150.8, 318.5)]	[5.2 ± 1.9]	2.451 ± 1.177 [16.3 (3.9, 43.6)]	1.012 ± 0.418 [79.5 (60.6, 99.5)]
	1	2.269 ± 1.397 [26.5 (5.5, 96.3)]	1.843 ± 1.016 [4.0 (2.0, 6.0)]	1.171 ± 0.720 [3.0 (0.5, 5.6)]	1.662 ± 0.325 [7.3 (4.2, 10.3)]	13.266 ± 4.070 [191.5 (123.0, 261.3)]	[5.7 ± 2.1]	2.510 ±1.208 [19.5 (5.0, 40.3)]	1.330 ± 0.360 [100.0 (93.8, 100.0)]
BTS		+ ve (despite inhibitive effect of B)	+ ve (despite inhibitive effect of B)	+ ve (despite inhibitive effect of B)	neutral (no inhibitive effect)	+ ve (despite inhibitive effect of S)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (despite positive effects of T and S)
	0	1.684 ± 1.161 [10.5 (1.0, 44.3)]	1.417 ± 0.938 [2.5 (0.5, 4.6)]	0.934 ± 0.723 [1.0 (0.0, 3.5)]	1.625 ± 0.302 [5.6 (4.5, 11.3)]	9.733 ± 2.924 [105.0 (70.5, 119.4)]	[3.1 ± 1.1]	2.360 ± 1.186 [19.0 (5.5, 36.0)]	1.147 ± 0.473 [95.3 (59.5, 100.0)]
	1	2.967 ± 0.861 [77.8 (38.9, 178.4)]	2.116 ± 0.550 [5.0 (3.0, 6.0)]	2.116 ± 0.471 [4.8 (2.5, 8.8)]	1.625 ± 0.303 [7.5 (3.4, 10.0)]	18.145 ± 1.934 [313.0 (269.4, 376.0)]	[7.0 ± 1.5]	2.700 ± 1.248 [16.4 (5.5, 64.0)]	1.068 ± 0.383 [89.8 (74.6, 95.8)]
BHS		+ ve (despite inhibitive effect of B)	+ ve (despite inhibitive effect of B)	+ ve (no inhibitive effects)	+ ve (no inhibitive effect)	+ ve (despite inhibitive effects of H and S)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (no positive effects)
	0	2.036 ± 1.567 [12.0 (0.9, 95.8)]	1.606 ± 1.158 [2.8 (0.5, 7.0)]	0.888 ± 0.645 [1.1 (0.0, 3.8)]	1.454 ± 0.189 [4.8 (2.8, 6.1)]	11.077 ± 4.036 [129.3 (80.3, 180.8)]	[3.6 ± 1.6]	2.110 ±1.260 [8.5 (2.8, 31.5)]	1.495 ± 0.154 [100.0 (99.9, 100.0)]
	1	2.882 ± 0.828 [60.0 (31.0, 178.4)]	2.116 ± 0.517 [5.0 (3.0, 6.0)]	1.696 ± 0.503 [6.5 (4.0, 12.8)]	1.774 ± 0.356 [10.1 (4.6, 17.9)]	16.088 ± 2.088 [252.0 (211.8, 291.1)]	[6.8 ± 1.4]	2.832 ±1.217 [25.5 (10.5, 63.5)]	0.762 ± 0.425 [71.0 (35.4, 89.6)]
THS		+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of	- ve (no positive effects)

					T)			S)	
	0	0.847 ± 0.883 [1.0 (0.0, 4.9)]	0.704 ± 0.710 [0.5 (0.0, 1.6)]	0.367 ± 0.468 [0.0 (0.0, 0.5)]	1.372 ± 0.158 [3.0 (2.6, 5.0)]	13.300 ± 4.733 [156.5 (120.1, 295.6)]	[3.7 ± 1.5]	2.079 ± 1.338 [8.0 (1.2, 31.5)]	1.558 ± 0.034 [100.0 (100.0, 100.0)]
	1	3.486 ± 0.855 [169.3 (56.1, 277.6)]	2.489 ± 0.605 [6.3 (4.5, 8.0)]	1.765 ± 0.457 [7.0 (4.7, 16.9)]	1.686 ± 0.291 [8.5 (5.0, 11.6)]	16.232 ± 2.795 [264.8 (203.4, 306.0)]	[6.6 ± 1.6]	2.791 ± 0.977 [20.0 (11.4, 51.0)]	1.070 ± 0.407 [91.3 (70.1, 97.8)]
BRS		- ve (no positive effects)	- ve (despite + ve effect of R)	- ve (no positive effects)	+ ve (no inhibitive effect)	+ ve (despite inhibitive effects of S and R)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	+ ve (despite inhibitive effect of S and B)
	0	2.685 ± 1.304 [47.0 (10.8, 239.1)]	2.040 ± 0.898 [4.3 (2.4, 7.6)]	1.338 ± 0.626 [3.9 (0.9, 7.0)]	1.581 ± 0.265 [5.7 (3.9, 9.6)]	12.096 ± 3.426 [146.5 (104.1, 216.4)]	[4.5 ± 1.8]	2.453 ± 1.060 [16.3 (7.3, 36.7)]	1.084 ± 0.417 [90.8 (73.3, 99.6)]
	1	2.138 ± 1.230 [25.5 (3.8, 74.5)]	1.739 ± 0.886 [4.0 (2.0, 5.3)]	1.190 ± 0.734 [3.0 (0.5, 6.3)]	1.731 ± 0.375 [7.8 (4.2, 17.7)]	15.347 ± 2.456 [248.5 (197.0, 277.3)]	[6.8 ± 1.8]	2.597 ± 1.390 [19.5 (4.5, 60.0)]	1.223 ± 0.443 [96.5 (89.5, 100.0)]
TRS		+ ve (despite inhibitive effect of R)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of R)	- ve (no inhibitive effect)	+ ve (despite inhibitive effects of S and R)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	+ ve (no inhibitive effects)
	0	1.853 ± 1.140 [12.3 (1.5, 47.0)]	1.426 ± 0.816 [2.5 (1.0, 4.0)]	1.090 ± 0.778 [1.4 (0.3, 6.3)]	1.626 ± 0.329 [5.5 (3.3, 13.2)]	13.660 ± 3.548 [175.5 (119.5, 257.9)]	[4.2 ± 1.7]	2.332 ± 1.252 [15.8 (1.9, 39.0)]	1.076 ± 0.508 [90.3 (51.1, 100.0)]
	1	3.099 ± 1.129 [82.5 (25.0, 204.3)]	2.399 ± 0.710 [5.5 (4.0, 8.0)]	1.531 ± 0.502 [4.8 (2.0, 10.5)]	1.621 ± 0.297 [6.2 (4.0, 9.5)]	15.377 ± 3.794 [261.5 (204.0, 306.0)]	[6.8 ± 1.6]	2.749 ± 1.117 [22.8 (11.0, 51.0)]	1.395 ± 0.316 [100.0 (95.0,100.0)]
HRS		+ ve (despite inhibitive effect of R)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of R)	+ ve (no inhibitive effect)	- ve (no positive effects)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (despite positive effect of S and R)
	0	2.118 ± 1.342 [28.5 (1.4, 84.5)]	1.566 ± 0.972 [2.8 (1.0, 5.0)]	0.949 ± 0.586 [1.4 (0.3, 4.1)]	1.475 ± 0.210 [5.5 (2.7, 7.7)]	16.205 ± 4.296 [284.3 (205.3, 354.9)]	[5.1 ± 2.0]	2.316 ± 1.346 [8.9 (3.1, 52.5)]	1.336 ± 0.277 [99.5 (90.1, 100.0)]
	1	2.927 ± 0.984 [61.5 (29.3, 175.8)]	2.352 ± 0.606 [5.5 (4.0, 7.0)]	1.605 ± 0.489 [5.3 (3.0, 10.8)]	1.788 ± 0.330 [9.2 (5.7, 17.9)]	14.582 ± 3.470 [214.5 (188.5, 270.5)]	[5.9 ± 2.5]	2.844 ± 1.114 [26.5 (12.0, 51.0)]	1.121 ± 0.408 [93.5 (82.0, 97.0)]
BTHS		+ ve (despite inhibitive effect of B)	+ ve (despite inhibitive effect of B)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of T)	+ ve (despite inhibitive effect of S and H)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (despite positive effect of B and S)
	0	0.925 ± 0.953 [1.0 (0.0, 9.3)]	0.751 ± 0.761 [0.5 (0.0, 2.0)]	0.397 ± 0.493 [0.0 (0.0, 0.5)]	1.379 ± 0.158 [3.8 (2.6, 5.3)]	10.155 ± 3.872 [119.8 (80.3, 142.3)]	[3.0 ± 1.4]	2.105 ± 1.280 [11.3 (1.9, 31.5)]	1.558 ± 0.035 [100.0 (100.0, 100.0)]
	1	3.315 ± 0.701 [124.3 (56.1, 195.0)]	2.299 ± 0.506 [5.3 (4.5, 7.0)]	1.792 ± 0.444 [9.0 (5.2, 14.9)]	1.727 ± 0.342 [8.9 (4.0, 16.6)]	17.679 ± 1.713 [293.8 (266.1, 340.8)]	[7.3 ± 1.8]	2.903 ± 1.130 [17.5 (11.4, 64.3)]	0.888 ± 0.427 [72.8 (54.6, 94.3)]
BTRS		+ ve (despite inhibitive effect of B & R)	+ ve (despite inhibitive effect of B & R)	+ ve (despite inhibitive effect of B & R)	+ ve (despite inhibitive effect of T)	+ ve (despite inhibitive effect of S and R)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	+ ve (despite inhibitive effect of S, B and T)
	0	2.009 ± 1.104 [14.5 (2.6, 51.9)]	1.545 ± 0.795 [2.5 (1.0, 4.1)]	1.150 ± 0.741 [1.3 (0.5, 6.4)]	1.626 ± 0.315 [5.5 (4.5, 13.7)]	11.145 ± 2.427 [118.5 (104.1, 142.3)]	[3.3 ± 1.3]	2.412 ± 1.171 [17.5 (6.5, 35.4)]	1.035 ± 0.553 [80.8 (51.1, 100.0)]
	1	2.756 ± 1.020 [60.5 (13.5, 141.3)]	2.191 ± 0.605 [5.0 (3.8, 6.3)]	1.473 ± 0.536 [4.3 (2.0, 8.8)]	1.668 ± 0.348 [7.6 (4.0, 14.6)]	17.260 ± 1.296 [284.5 (261.8, 332.3)]	[7.4 ± 1.8]	2.805 ± 1.343 [25.0 (11.0, 69.3)]	1.273 ± 0.396 [96.5 (94.8, 100.0)]
BHRS		+ ve (despite inhibitive effect of B & R)	+ ve (despite inhibitive effect of B & R)	+ ve (despite inhibitive effect of R)	+ ve (no inhibitive effect)	+ ve (despite inhibitive effects of R, H and S)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of S)	- ve (despite positive effects of B, R and S)
	0	2.234 ± 1.326 [19.0 (1.5, 68.6)]	1.758 ± 1.009 [3.0 (1.0, 7.0)]	1.003 ± 0.541 [1.4 (0.4, 4.1)]	1.461 ± 0.199 [5.4 (2.7, 6.4)]	13.081 ± 3.773 [188.5 (133.6, 254.5)]	[4.4 ± 1.9]	2.304 ± 1.236 [11.6 (4.4, 43.9)]	1.420 ± 0.194 [99.8 (95.8, 100.0)]
	1	2.890 ± 0.882 [65.5 (39.3, 141.3)]	2.261 ± 0.549 [5.0 (4.0, 7.0)]	1.677 ± 0.519 [6.5 (4.0, 11.8)]	1.894 ± 0.370 [16.4 (6.0, 21.6)]	15.825 ± 1.385 [252.0 (212.5, 271.3)]	[7.4 ± 1.6]	3.007 ± 1.225 [40.0 (11.0, 73.5)]	0.940 ±0.472 [86.0 (63.8, 94.3)]
THRS		+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (despite inhibitive effect of T)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	- ve (despite positive effects of T , R and S)
	0	1.152 ± 0.949 [1.5 (0.0, 11.6)]	0.893 ± 0.718 [1.0 (0.0, 2.1)]	0.517 ± 0.503 [0.3 (0.0, 0.8)]	1.346 ± 0.145 [2.8 (2.4, 4.8)]	15.519 ± 3.240 [268.3 (150.8, 318.5)]	[4.1 ± 1.8]	2.187 ± 1.540 [10.0 (1.1, 52.6)]	1.546 ± 0.046 [100.0 (99.9, 100.0)]
	1	3.517 ± 0.834 [125.5 (72.5, 269.8)]	2.643 ± 0.553 [7.0 (5.0, 9.0)]	1.804 ± 0.455 [7.8 (5.4, 19.4)]	1.774 ± 0.299 [8.9 (5.9, 16.6)]	16.893 ± 2.070 [279.0 (249.5, 306.0)]	[7.4 ± 1.8]	3.145 ± 0.740 [31.5 (17.8, 54.8)]	1.284 ± 0.413 [97.5 (93.3, 100.0)]
BTHRS		+ ve	+ ve	+ ve	+ ve	+ ve	+ ve	+ ve	- ve

	0	(despite inhibitive effect of B & R) 1.452 ± 0.937 [6.5 (0.9, 18.6)]	(despite inhibitive effect of B) 1.100 ± 0.713 [1.3 (0.5, 2.6)]	(despite inhibitive effect of B) 0.655 ± 0.507 [0.5 (0.0, 1.3)]	(despite inhibitive effect of T) 1.379 ± 0.158 [3.8 (2.6, 5.3)]	(despite inhibitive effect of T, R, H and S) 12.727 ± 2.092 [146.5 (133.6, 178.1)]	(no inhibitive effects) [3.5 ± 1.9]	(no inhibitive effects) 2.414 ± 1.413 [19.5 (5.3, 45.9)]	(despite positive effects of B, T, R and S) 1.546 ± 0.050 [100.0 (99.9, 100.0)]
	1	3.427 ± 0.683 [112.5 (72.5, 195.0)]	2.579 ± 0.324 [6.5 (5.0, 7.3)]	1.835 ± 0.411 [9.0 (5.9, 14.1)]	1.835 ± 0.363 [12.8 (5.5, 18.8)]	16.929 ± 0.934 [280.5 (261.8, 306.0)]	[8.3 ± 1.5]	3.129 ± 0.900 [35.0 (13.5, 65.6)]	1.104 ± 0.515 [94.5 (79.5, 96.3)]
UNSEEDED INTERVENTIONS									
		SEEDED SPECIES				**EXISTING INDIGENOUS SPECIES**			**WEED SPECIES**
		DENSITY 4th root transformed [untransformed]	RICHNESS square root transformed [untransformed]	COVER fourth root transformed [untransformed]	HEIGHT fourth root transformed [untransformed]	DENSITY square root transformed [untransformed]	RICHNESS [untransformed]	COVER third root transformed [untransformed]	COVER arcsine transformed [untransformed]
C	1					12.727 ± 2.092 [146.5 (133.6, 178.1)]	[3.5 ± 1.9]	2.414 ± 1.413 [19.5 (5.3, 45.9)]	1.546 ± 0.050 [100 (99.9, 100)]
B						+ ve	+ ve	+ ve	- ve
	0					11.345 ± 3.515 [127.3 (82.5, 199.4)]	[4.3 ± 1.6]	2.338 ± 1.088 [15.4 (4.5, 32.8)]	1.243 ± 0.365 [96.5 (85.3, 100.0)]
	1					16.899 ± 2.974 [286.0 (212.8, 358.8)]	[6.2 ± 1.7]	2.591 ± 1.336 [18.8 (3.9, 57.6)]	1.069 ± 0.467 [89.8 (69.4, 100.0)]
T						+ ve	+ ve	+ ve	+ ve
	0					12.356 ± 3.720 [150.0 (105.0, 213.3)]	[4.4 ± 1.8]	2.316 ± 1.246 [15.1 (2.0, 37.9)]	1.146 ± 0.479 [95.8 (73.8, 100.0)]
	1					15.888 ± 4.087 [265.3 (192.4, 354.4)]	[6.1 ± 1.7]	2.613 ± 1.185 [17.8 (5.4, 50.3)]	1.166 ± 0.370 [93.0 (77.9, 100.0)]
H						- ve	+ ve	+ ve	- ve
	0					14.140 ± 4.835 [196.8 (120.1, 324.5)]	[5.0 ± 1.9]	2.213 ± 1.265 [9.4 (2.0, 34.1)]	1.443 ± 0.214 [100.0 (97.5, 100.0)]
	1					14.104 ± 3.674 [208.0 (140.5, 263.4)]	[5.5 ± 1.9]	2.716 ± 1.129 [23.3 (9.4, 46.6)]	0.869 ± 0.393 [77.5 (51.9, 91.6)]
R						- ve	neutral	+ ve	+ ve
	0					14.771 ± 3.976 [218.8 (143.9, 317.5)]	[5.3 ± 1.7]	2.369 ± 1.162 [14.0 (3.9, 34.6)]	1.015 ± 0.418 [86.8 (67.3, 98.5)]
	1					13.473 ± 4.496 [197.5 (116.3, 261.3)]	[5.3 ± 2.2]	2.560 ± 1.278 [20.0 (5.0, 48.0)]	1.297 ± 0.390 [100.0 (91.0, 100.0)]
BT						+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	- ve (despite positive effect of T)
	0					9.685 ± 2.786 [105.0 (70.5, 126.6)]	[3.2 ± 1.0]	2.264 ± 1.127 [15.0 (4.0, 32.1)]	1.173 ± 0.439 [95.8 (79.5, 100.0)]
	1					18.771 ± 2.262 [350.8 (287.3, 410.5)]	[6.9 ± 1.5]	2.814 ± 1.282 [21.8 (5.9, 69.8)]	1.019 ± 0.407 [86.8 (67.8, 95.8)]
BH						+ ve (despite inhibitive effect of H)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	- ve (no positive effects)
	0					10.821 ± 3.627 [123.3 (72.8, 170.3)]	[4.0 ± 1.6]	2.094 ± 1.128 [9.0 (2.9, 24.1)]	1.475 ± 0.161 [100.0 (98.5, 100.0)]
	1					16.340 ± 2.393 [252.0 (211.0, 309.8)]	[6.6 ± 1.6]	2.849 ± 1.240 [28.0 (10.4, 66.0)]	0.728 ± 0.373 [69.3 (40.9, 82.0)]
TH						+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	+ ve (no inhibitive effects)	- ve (no positive effects)
	0					12.642 ± 4.387 [142.0 (107.8, 257.9)]	[4.1 ± 1.6]	2.033 ± 1.258 [8.5 (1.0, 28.1)]	1.550 ± 0.064 [100.0 (100.0, 100.0)]
	1					16.137 ± 3.192 [264.8 (204.6, 334.5)]	[6.3 ± 1.5]	2.832 ± 1.076 [22.0 (10.8, 53.3)]	0.996 ± 0.393 [81.5 (67.8, 95.5)]
BR						+ ve	+ ve	+ ve	+ ve

		(despite inhibitive effect of R)	(no inhibitive effects)	(no inhibitive effects)	(despite inhibitive effect of B)
	0	11.972 ± 3.056 [143.3 (104.1, 205.3)]	[4.5 ± 1.6]	2.292 ± 1.038 [13.5 (4.5, 29.7)]	1.086 ± 0.390 [91.3 (75.4, 98.5)]
	1	16.229 ± 3.231 [248.5 (201.8, 332.3)]	[6.5 ± 1.9]	2.737 ± 1.385 [24.0 (5.8, 66.5)]	1.194 ± 0.468 [96.5 (78.5, 100.0)]
TR		+ ve	+ ve	+ ve	+ ve
		(despite inhibitive effect of R)	(no inhibitive effects)	(no inhibitive effects)	(no inhibitive effect)
	0	13.406 ± 3.560 [182.8 (199.8, 244.5)]	[4.5 ± 1.6]	2.284 ± 1.233 [15.0 (1.7, 34.6)]	1.057 ± 0.521 [92.0 (51.1, 100.0)]
	1	15.640 ± 4.268 [257.5 (176.8, 332.3)]	[6.2 ± 1.9]	2.772 ± 1.261 [23.8 (8.0, 60.8)]	1.359 ± 0.351 [100.0 (94.8, 100.0)]
HR		- ve	+ ve	+ ve	- ve
		(no positive effects)	(no inhibitive effects)	(no inhibitive effects)	(despite positive effect of R)
	0	15.774 ± 4.160 [277.5 (151.8, 336.8)]	[5.2 ± 1.8]	2.211 ± 1.261 [7.5 (2.7, 38.1)]	1.331 ± 0.250 [98.5 (90.5, 100.0)]
	1	14.440 ± 3.803 [212.0 (140.3, 270.5)]	[5.7 ± 2.3]	2.905 ± 1.189 [31.8 (10.9, 57.0)]	1.039 ± 0.410 [91.0 (76.3, 95.3)]
BTH		+ ve	+ ve	+ ve	- ve
		(despite inhibitive effect of H)	(despite inhibitive effect of H)	(no inhibitive effects)	(despite positive effect of B)
	0	9.638 ± 3.221 [104.0 (72.0, 135.0)]	[3.3 ± 1.2]	2.083 ± 1.144 [12.3 (1.9, 26.3)]	1.535 ± 0.086 [100.0 (100.0, 100.0)]
	1	18.269 ± 1.799 [315.5 (287.3, 372.6)]	[6.8 ± 1.7]	2.947 ± 1.243 [25.0 (11.0, 73.1)]	0.782 ± 0.375 [66.5 (45.4, 86.8)]
BTR		+ ve	+ ve	+ ve	+ ve
		(despite inhibitive effect of R)	(no inhibitive effects)	(no inhibitive effects)	(despite inhibitive effect of B and T)
	0	10.771 ± 2.513 [119.0 (97.6, 139.0)]	[3.5 ± 1.1]	2.298 ± 1.123 [15.1 (5.4, 32.6)]	1.039 ± 0.510 [89.3 (57.9, 100.0)]
	1	18.444 ± 2.598 [315.5 (268.0, 283.5)]	[7.1 ± 1.6]	3.008 ± 1.352 [30.0 (12.4, 73.5)]	1.224 ± 0.439 [96.5 (90.3, 100.0)]
BHR		+ ve	+ ve	+ ve	- ve
		(despite inhibitive effect of R & H)	(no inhibitive effects)	(no inhibitive effects)	(despite positive effects of B and R)
	0	12.589 ± 3.262 [148.5 (114.8, 240.4)]	[4.3 ± 1.7]	2.146 ± 1.137 [9.4 (3.2, 23.5)]	1.380 ± 0.185 [98.5 (95.4, 100.0)]
	1	16.498 ± 2.395 [252.0 (211.0, 309.8)]	[7.1 ± 1.7]	3.085± 1.296 [41.5 (11.6, 74.6)]	0.849 ± 0.438 [78.0 (57.0, 92.3)]
THR		+ ve	+ ve	+ ve	- ve
		(no inhibitive effects)	(no inhibitive effects)	(no inhibitive effects)	(despite positive effects of T and R)
	0	14.723 ± 3.727 [245.5 (135.0, 324.5)]	[4.3 ± 1.6]	2.082 ± 1.395 [6.8 (1.0, 38.8)]	1.529 ± 0.087 [100.0 (99.9, 100.0)]
	1	16.828 ± 2.888 [279.0 (214.3, 336.3)]	[6.7 ± 1.7]	3.099 ± 1.032 [33.5 (16.3, 64.5)]	1.179 ± 0.421 [94.5 (89.5, 100.0)]
BTHR		+ ve	+ ve	+ ve	- ve
		(despite inhibitive effect of R & H)	(no inhibitive effects)	(no inhibitive effects)	(despite positive effects of B, T and R)
	0	11.519 ± 2.120 [132.0 (111.9, 145.3)]	[3.6 ± 1.5]	2.214 ± 1.205 [13.5 (4.5, 32.1)]	1.499 ± 0.113 [100.0 (99.3, 100.0)]
	1	18.330 ± 2.071 [315.5 (289.8, 376.0)]	[7.6 ± 1.3]	3.258 ± 1.123 [48.0 (17.0, 74.6)]	0.942 ± 0.464 [88.5 (57.0, 94.3)]

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Due to the numerous interventions and response variables, the results below do not report statistical values. For chi squared, variance ratio, probability values refer Table 3.3 and for response direction, mean ± standard deviation, median and inter-quartile range, refer Table 3.4. To reiterate, acronyms reported here include: C=control, S=seeding, B=burning, T=tillage, H=herbicide-application and R=rodent-exclusion.

Seeded species

Seeded species: seedling density

Significant positive effects on density of seeded species resulted from S, TS, HS, THS, HRS, BTRS, and BTHRS whilst significant negative effects resulted from BS, RS and BRS.

Considering untransformed median values, S implemented individually (without another intervention) resulted in the lowest seedling density (6.5 seedlings per m^2). Of the two-factor interventions (S plus another intervention) the density was highest in response to TS (91.8 seedlings per m^2) followed by HS (64.0 seedlings per m^2). Both BS and RS had negative effects on seedling density when implemented. The following three- to five-factor interventions resulted in greater seedling density than TS (in ascending order): BTHRS, BTHS, THRS and THS which resulted in 169.3 seedlings per m^2.

Seeded species: species richness

Significant positive effects resulted from S, TS, HS, BTS, THS, HRS and BTHRS whilst, of great concern, a significant negative effect resulted from BS.

With respect to the untransformed medians, S resulted in the lowest species richness (1.25 species per m^2) with TS and HS both resulting in 5.0 species per m^2 (the highest of the two-factor interventions). All of the remaining three- to five-factor interventions (with the exception of BRS) resulted in species richness higher than that achieved by TS and HS. THRS resulted in the highest richness (6.5 species per m^2).

Seeded species: canopy cover

The interventions S, TS, HS, THS and TRS resulted in significant positive effects on seeded species canopy cover whilst RS resulted in a significant negative effect.

Seedling canopy cover was lowest when only S was implemented (0.5% canopy cover). Of the two-factor interventions HS resulted in the highest cover (5.8% canopy cover). Of the three- to five-factor interventions, canopy cover for HS was exceeded by six interventions (in ascending order): BHS, BHRS, THS, THRS and BTHS and BTHRS both resulting in the highest canopy cover of 9%.

Seeded species: seedling height

Significant positive effects on seedling height of the seeded species resulted from S, HS, THS, BRS and BTRS.

With the exception of a non-significant negative effect resulting from TS and a neutral effect resulting from BTS, all of the remaining intervention resulted in positive effects on seedling height.

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Seedling height was lowest in the S-only plots with a median seedling height which measured 3.8 cm. Looking at responses of the two-factor interventions, seedling height was greatest in response to HS (8.6 cm per seedling). Of the three- to five-factor interventions, seedling height for HS was exceeded by seven interventions (in ascending order): BRS, BTHS, THRS, BHS, HRS, BTHRS with the highest seedling height resulting from BHRS (16.4 cm per seedling).

Existing indigenous species

Existing indigenous species: seedling density

Significant positive effects on seedling density of existing indigenous species resulted from B, T and BH. Significant negative effects resulted from R and HR.

The control resulted in the lowest seedling density with 146.5 seedlings per m^2. Of the single-factor interventions, B resulted in the highest density (286.0 seedlings per m^2) and T thereafter with 265.3 seedlings per m^2. Of the two- to four-factor unseeded interventions, four interventions result in a density greater than B and each contains the factors B and T: BTH, BTR and BTHR (each with 315.5 seedlings per m^2) and BT achieved the highest density with 350.8 seedlings per m^2. Of the seeded interventions, the factors B and T similarly contribute to good performance where BTHS (293.8 seedlings per m^2) and BTS (313.0 seedlings per m^2) resulted in higher seedling density than B alone.

Existing indigenous species: species richness

Significant positive effects on existing indigenous seedling richness resulted from S, B, T, H and BTH. With the exception of R (which had a neutral effect), the remaining 30 interventions resulted in non-significant positive effects.

With respect to the untransformed medians, the lowest species richness was recorded in the control (3.5 species per m^2). Of the single-factor interventions, B resulted in the highest richness (6.2 species per m^2). Of the unseeded interventions, all of the two- to five-factor interventions (except HR) resulted in higher species richness than B, the highest being BTHR with 7.6 species per m^2. Considering the seeded interventions, the following two- to five-factor interventions resulted in higher richness than B, in ascending order: TS, BS, THS, BHS, BRS, TRS, BTS, BTHS, BTRS, BHRS, THRS and the highest, BTHRS, with 8.3 species per m^2.

Existing indigenous species: canopy cover

The interventions B, T and H resulted in significant positive effects with respect to canopy cover for existing indigenous species.

All interventions, with the exception of S, which had the lowest cover of 15.1%, resulted in positive responses. Considering the untransformed means, of the single-factor interventions, H resulted in the highest canopy cover (23.3% cover). Of the two- to four-factor unseeded interventions, eight interventions resulted in higher canopy cover than H, in ascending order: BR, BTH, BH, BTR, HR, THR, BHR and BTHR with 48% cover. Of the two- to five-factor seeded interventions, the following six interventions achieved a higher canopy cover than H alone: BTRS, BHS, HRS, THRS, BTHRS and BHRS with 40% canopy cover.

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Weed species: canopy cover

Significant positive effects, i.e. interventions promoting weed canopy cover, resulted from R and TR. Significant negative effects, i.e. interventions inhibiting weed cover, resulted from B, H, BT, BH and TH whilst a further 15 interventions apparently suppressed weed cover yet were statistically non-significant.

Median canopy cover of the control plots was 100% cover. Of the unseeded interventions, the most effective single-factor intervention in suppressing weed cover was H (77.5% cover). Of the remaining two- to four-factor unseeded interventions, BH was next most effective (69.3% cover) and BTH the most effective due to inhibiting weed cover to 66.5%. Of the two- to five-factor seeded interventions, only BTHS (72.8%) and BHS (71.0%) were more effective than H.

Ecological-response model: predicted and actual responses

A summary of the actual experimental intervention responses in relation to the predicted responses is provided in Table 3.5.

Table 3.5: Comparison of proposed ecological-response model and experimental responses.
Expected responses in relation to the actual experimental responses where '+'=increase, '0'=no change and '–'=decrease. Actual responses differing from expected responses indicated in red. Significant effects:* for p<0.05, ** for p<0.01 and *** for p<0.001.

INTERVENTION	SEEDED SPECIES			EXISTING INDIGENOUS SPECIES			WEED SPECIES		
		predicted	actual		predicted	actual		predicted	actual
BURNING	Density	+	– *	Density	+	+ ***			
	Richness	+	– ***	Richness	+	+ ***			
	Cover	+	+	Cover	+	+ *	Cover	+	– ***
	Height	+	+						
TILLAGE	Density	+	+ ***	Density	–	+ ***			
	Richness	+	+ ***	Richness	–	+ ***			
	Cover	+	+ ***	Cover	–	+ *	Cover	+	+
	Height	+	–						
HERBICIDE	Density	+	+ ***	Density	–	–			
	Richness	+	+ ***	Richness	–	+ *			
	Cover	+	+ ***	Cover	–	+ ***	Cover	–	– ***
	Height	+	+ ***						
RODENT-EXCLUSION	Density	+	– **	Density	0	– **			
	Richness	+	–	Richness	0	0			
	Cover	+	– *	Cover	0	+	Cover	0	+ ***
	Height	+	+						
SEEDING	Density	+	+ ***	Density	0	–			
	Richness	+	+ ***	Richness	0	+ *			
	Cover	+	+ ***	Cover	0	–	Cover	0	+
	Height	+	+ ***						

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Seedling survival

Just over a quarter (25.8%) of the total number of seedlings recorded in the September 2012 survived to the January 2013 census (Figure 3.7). What appears to be nearly a 75% seedling mortality rate was predominantly due to the high proportion of deciduous geophytes which in September 2012 comprised 72.04% of the total number of seedlings.

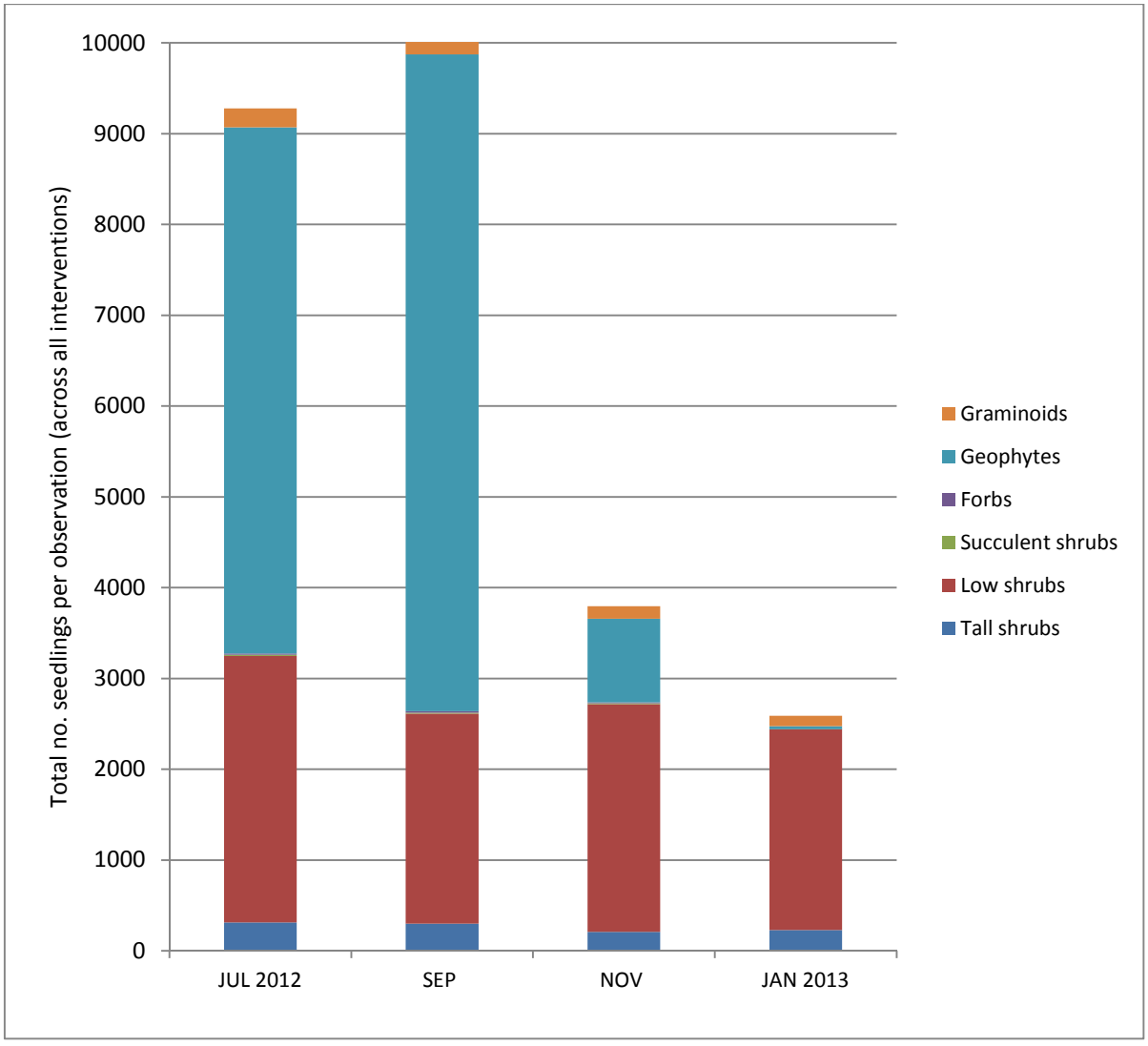

Figure 3.7: Total number of seedlings per growth form across interventions per observation (occurring within 64 m²).

The highest number of geophytes is recorded in September 2012 and due to dormancy, survival drops to 0.5% by January 2012. The succulent shrub survival rate is relatively low at 12.5%. Graminoid density tapers off slightly in January 2013 but maintains a 54.3% survival rate. Tall shrubs and low shrubs have the highest survival rates at 73.6% and 75.3% respectively.

The number of seedlings and survival at the individual species level is variable (ranging for instance from 0.5 seedlings for *Searsia tomentosa* and up to 8981 seedlings for *Ornithogalum thyrsoides*) per 64 m². Species have therefore been plotted on different axes according to magnitude (Figures 3.8a-d).

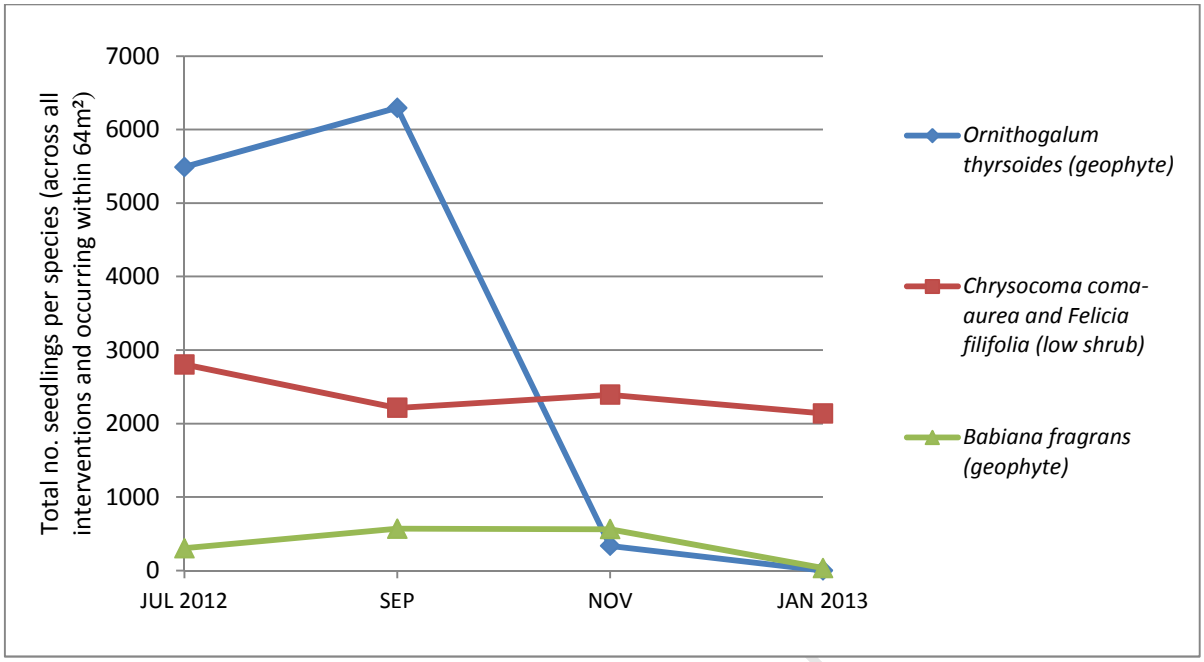

Figure 3.8a: Total number of seedlings per species, across all interventions per observation and occurring within 64 m²: *Ornithogalum thyrsoides, Chrysocoma coma-aurea* and *Babiana fragrans*.

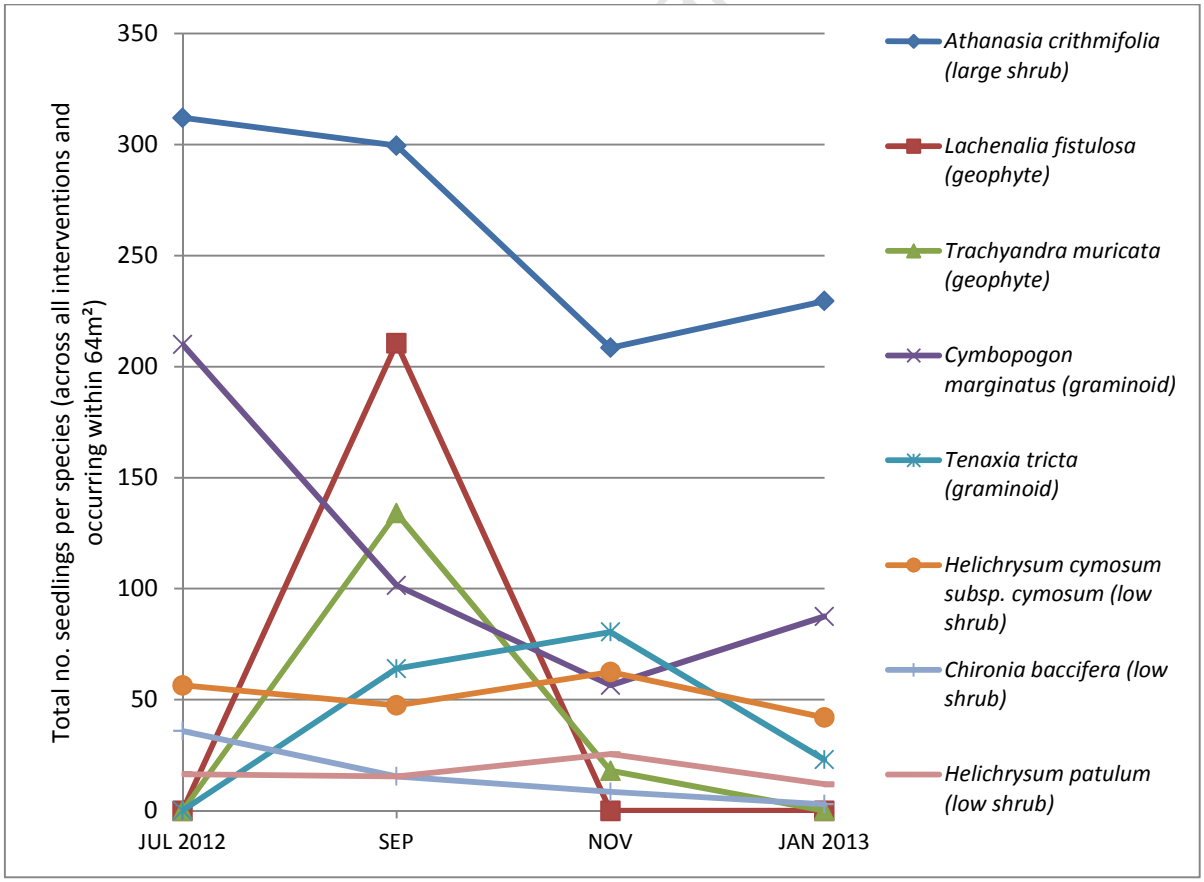

Figure 3.8b: Total number of seedlings per species, across all interventions per observation and occurring within 64 m²: *Athanasia crithmifolia, Lachenalia fistulosa, Trachyandra muricata, Cymbopogon marginatus, Tenaxia stricta, Helichrysum cymosum* subsp. *cymosum, Chironia baccifera* and *Helichrysum patulum*.

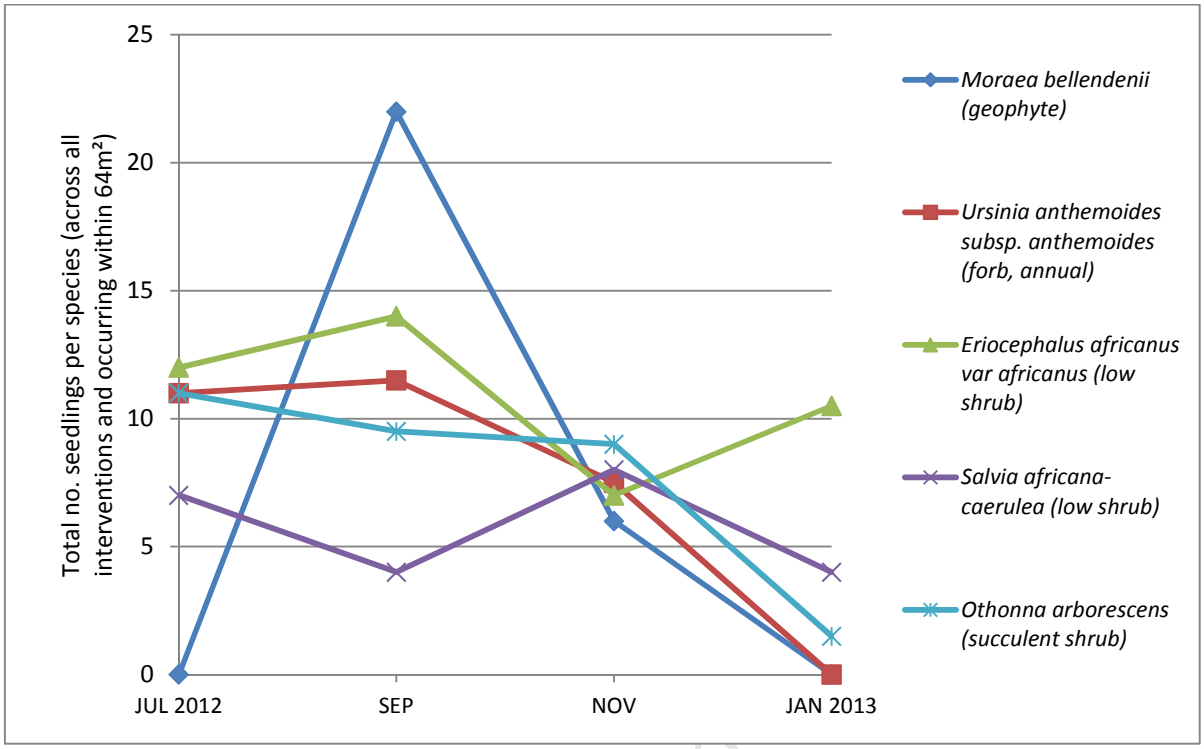

Figure 3.8c: Total number of seedlings per species, across all interventions per observation and occurring within 64 m²: *Moraea bellendenii, Ursinia anthemoides* subsp. *anthemoides, Eriocephalus africanus var. africanus, Salvia africana-caerulea* and *Othonna arborescens*.

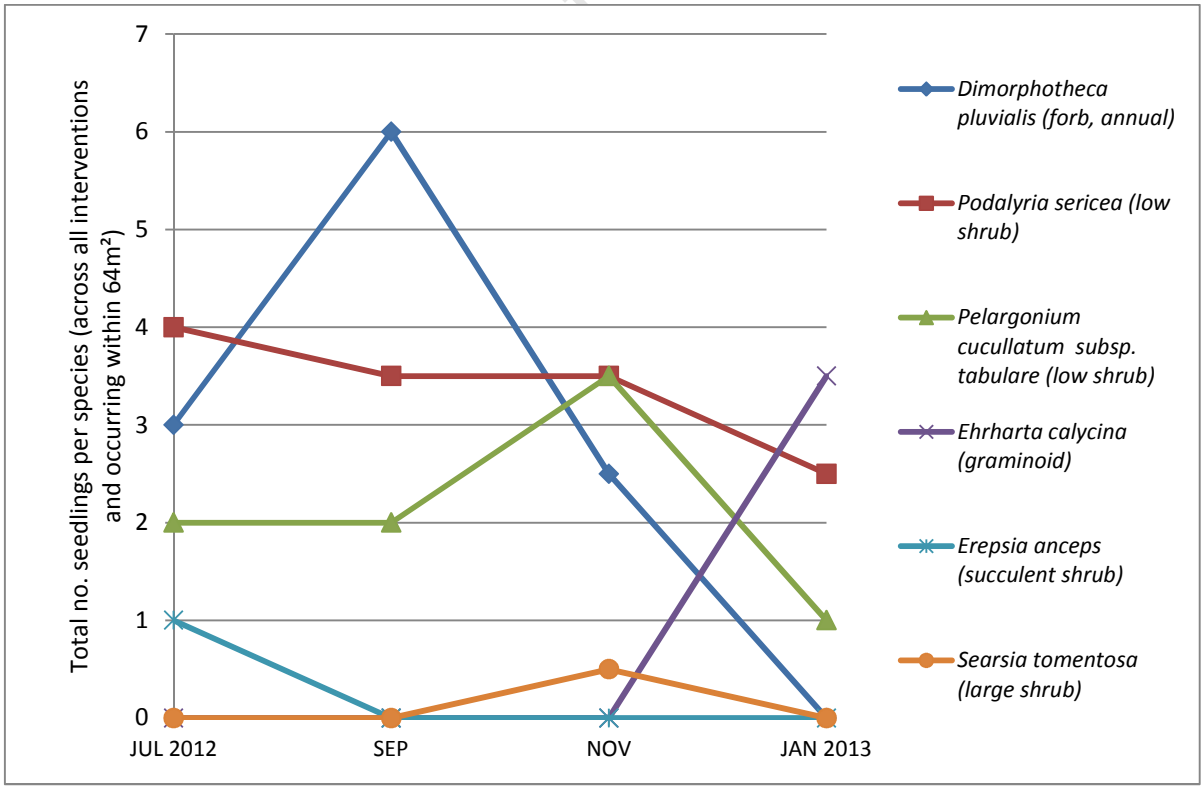

Figure 3.8d: Total number of seedlings per species, across all interventions per observation and occurring within 64 m²: *Dimorphotheca pluvialis, Podalyria sericea, Pelargonium cucullatum* subsp. *tabulare, Ehrharta calycina, Erepsia anceps* and *Searsia tomentosa*.

The median number of seedlings (per m²) per intervention over the four observations is provided below (Figure 3.9).

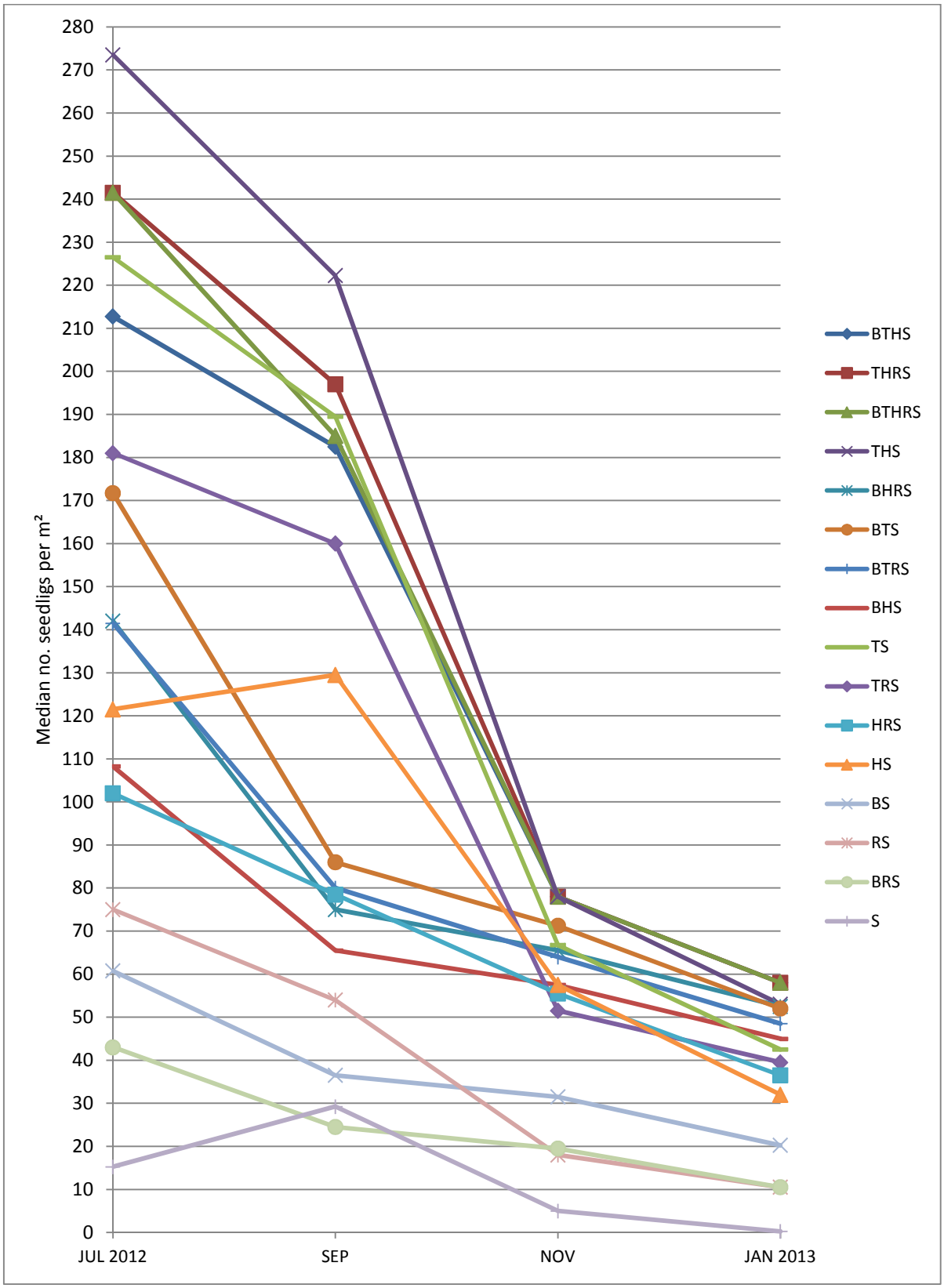

Figure 3.9: The median number of seedlings (per m²) per intervention over the four observations.

Canopy cover correlation: seeded species, existing indigenous species, weed species

Canopy cover of the seeded species was significantly positively correlated with that of the existing indigenous species for the November 2012 and January 2013 observations (r=0.268, p=0.033; and, r=0.406, p=0.001 respectively). The correlation was greatest in January yet was still reasonably weak (low to moderate magnitude) (refer Appendix 3.9).

The canopy cover of the seeded species with weed species over the three observations for September 2012, November and January 2013 was significantly negatively correlated (r=–0.466, p=<0.001; r=–0.559, p=<0.001 and, r=–0.656, p=<0.001 respectively). The correlation increased over time and was greatest in the final observation (January 2013) and was of moderate magnitude. This relationship was indicative of seedling cover increasing with time since sowing whilst the weed cover, composed predominantly of alien, annual grasses, declined dramatically in the January 2013 census.

The correlation of canopy cover of the existing indigenous species with weed species for November 2012 was significant and negative yet of a low magnitude (r=–0.321, p=0.010).

Vegetation structure

Figures 3.10a-c illustrate the median percentage canopy cover for each of the 16 seeded interventions for September 2012, November 2012 and January 2013 with respect to seeded species, existing indigenous species and weed species whilst figures 3.11a-c illustrate the breakdown of the seeded species into growth forms (graminoids, geophytes, forbs, succulent shrubs, low shrubs and tall shrubs).

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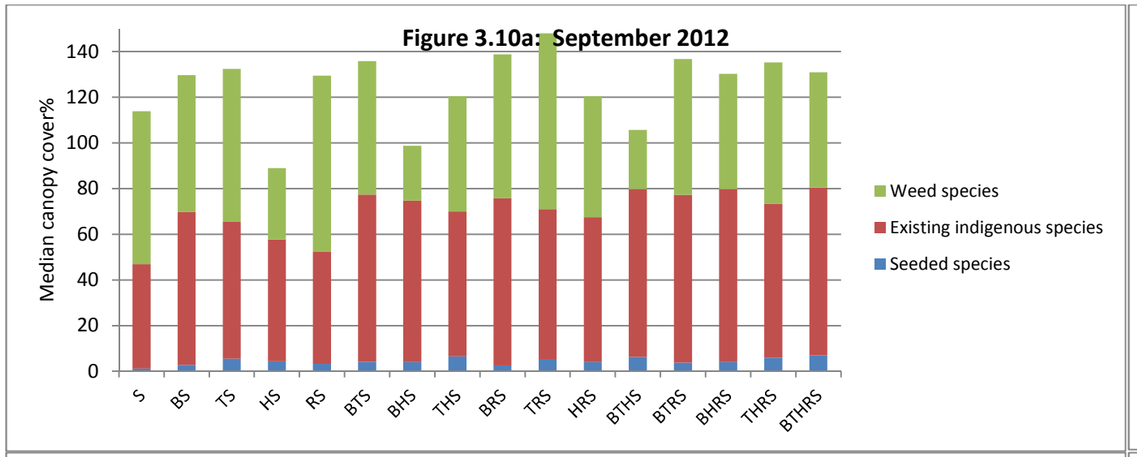

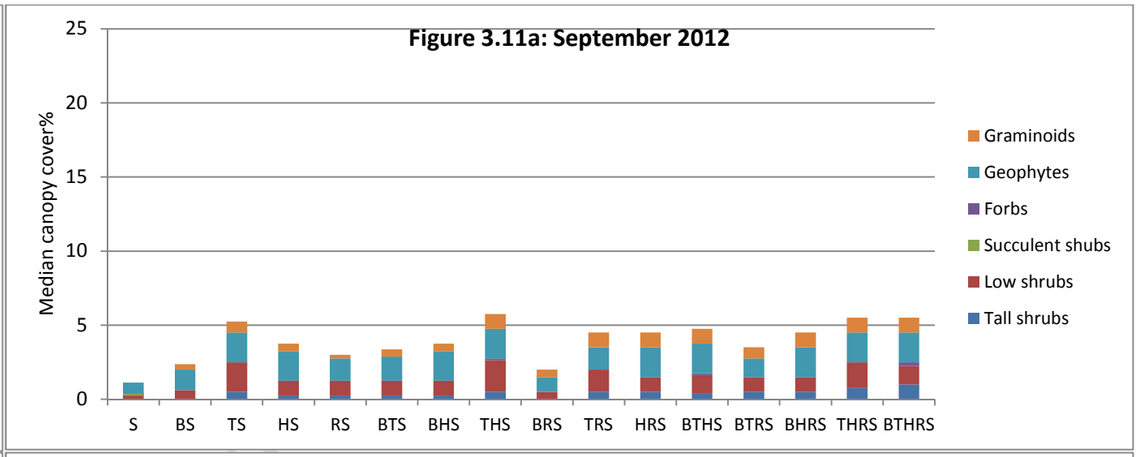

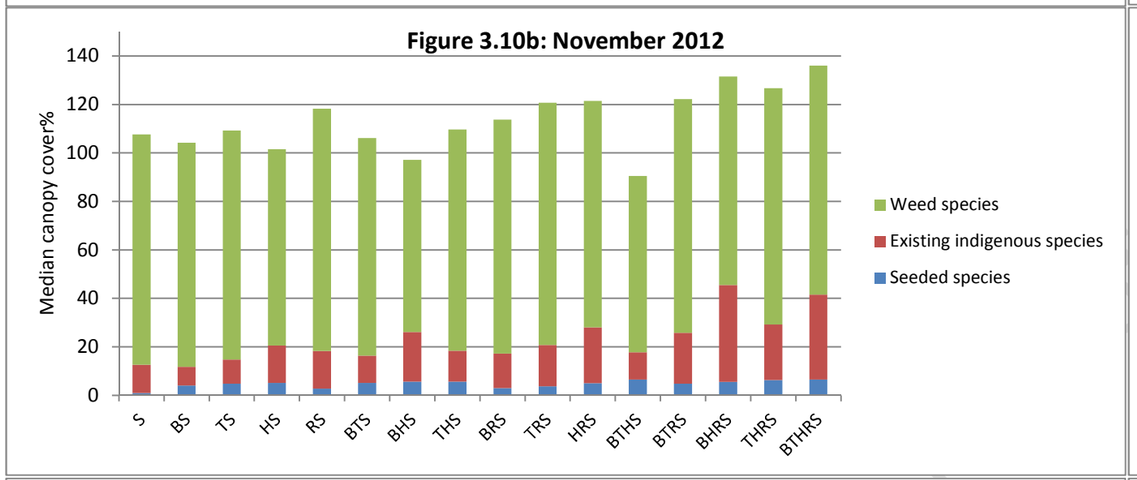

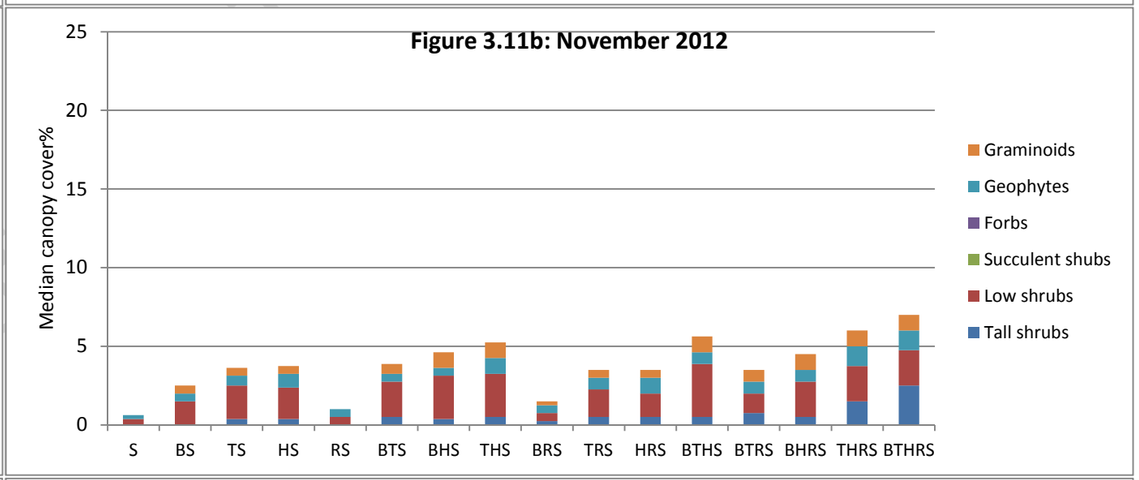

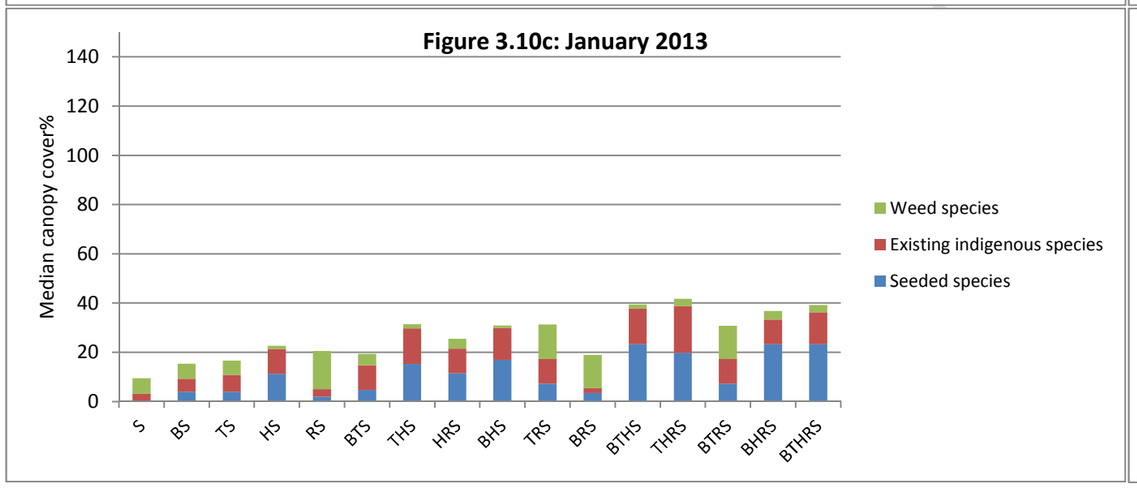

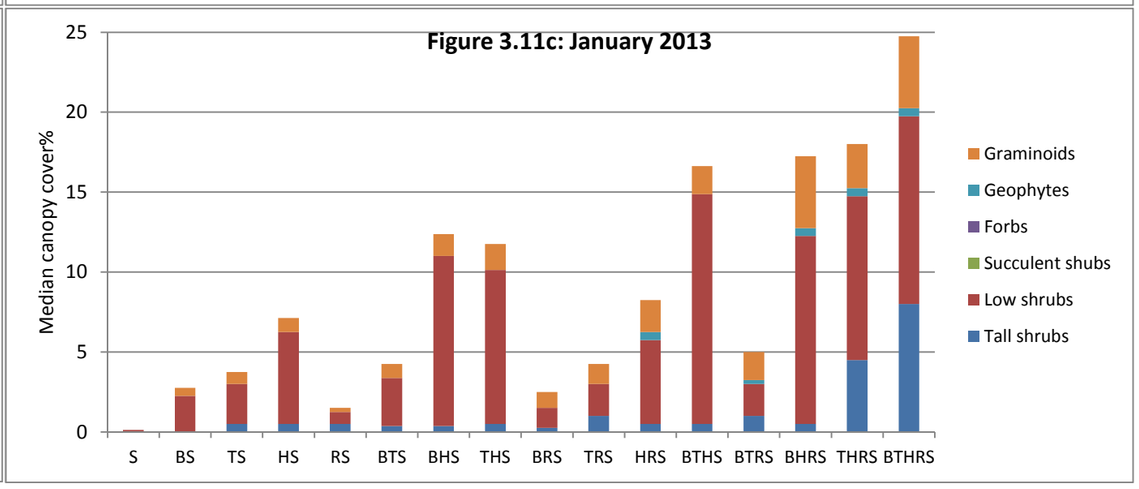

Figures 3.10a-c: Median percentage canopy cover for each of the 16 seeded interventions for September 2012, November 2012 and January 2013 with respect to seeded species, existing indigenous species and weed species.

Figures 3.11a-c: Median percentage canopy cover of the seeded species per growth form: graminoid, geophyte, forb, succulent shrub, low shrub and tall shrub.

Canopy cover: seeded species, existing indigenous species and weed species

In general the weed species cover peaked in November 2012 and thereafter declined dramatically in January 2013 due to the weed species being predominantly alien, annual grasses. The existing indigenous species cover was greatest in September 2012, declined dramatically in November 2012 and declined further in January 2013. With the exception of one perennial existing indigenous species (*Senecio pubigerus*) which increased in cover considerably from November 2012 to January 2013, the bulk of the existing species, being ephemeral geophytes, were largely dormant by the November 2012 census. The seeded species component contributed a very small proportion of the canopy cover make-up in September 2012, increasing in November 2012 and again notably in January 2013 where it contributed over half of the canopy cover of the total living biomass for some interventions (this despite the forb and geophytic growth forms of the seeded species being almost absent in the January 2013 census).

At the intervention-scale, tillage and rodent-exclusion promoted weed cover whilst herbicide-application and burning had significant inhibitive effects on weed cover and these effects are observable in Figures 3.10a-c. Existing indigenous species cover was significantly positively affected by B, T and H and these effects are emphasized when comparing them with the relatively low existing species cover associated with S-only. For the seeded species, S alone was the least effectual with respect to canopy cover. One can observe the positive effects of BS, TS and particularly HS whilst RS had an inhibitive effect.

Canopy cover: growth form breakdown within the seeded species

Increased cover in January 2013 can largely be attributed to increased cover in the low shrub growth form category and to lesser extents in the graminoid and large shrub categories. When plotting mean values, all six growth forms were present in several interventions in both the September 2012 and November 2012 censuses. However, when plotting medians (Figures 3.11a-c) the forb and succulent growth forms dropped away from the November 2012 census. By January 2013, whether plotting medians or means, a maximum of four growth forms were plotted for several interventions: low and tall shrubs, graminoids and geophytes (which largely still occurred within the rodent-exclusion plots most likely due to low light conditions).

3.7 Discussion

Responses to the interventions were variable with some performing poorly and others effectively. The interventions had strengths and weakness in different areas and they resulted in different response directions and magnitudes. The impacts of the single-factor interventions on seeded species, existing indigenous species and weed species were unexpected in some respects and contrary to the predicted ecological-response model. As the existing indigenous species component in the study area was deficient both with respect to seedling density, species richness, canopy cover and growth-form representation, the significantly positive impact seeding had on overall community attributes emphasizes the critical role it can play in redressing these limitations through augmentation of the existing indigenous community. Despite the considerable contribution of seeding to restoration

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outcomes, the implementation of seeding alone, without the implementation of at least another intervention to reduce the annual invasive alien grass component, generally resulted in poor recruitment and performance. Most significantly, this portion of the study revealed that implementation of the two-factor interventions (seeding in conjunction with another of the main interventions: B, T, H or R) had more merit than S-only, yet implementation of the three- to five-factor interventions (seeding implemented in conjunction with two or more factors) generally resulted in outcomes that were dramatically more effective. Experimentation testing seeding and multiple-factor interventions is relatively novel in the context of renosterveld restoration and, in addition to other local renosterveld and fynbos studies, the outcomes from this study require comparison with other Mediterranean-climate ecosystems.

3.7.1 Seeding

There is a large body of literature documenting a wide range of restoration outcomes from projects failing to achieve adequate seed germination (Holmes 2002, Midoko-Iponga 2004, Pausas et al. 2004, deFalco et al. 2012, Heelemann 2012, Vallejo et al. 2012) to projects achieving higher levels of germination and thus contributing to the attainment of desirable ecosystem attributes, including species richness, community structure and function (Jones and Hayes 1999, Allen et al. 2000, Holmes 2001, Holmes 2005, Joubert et al. 2009, Vallejo et al. 2009). A review of studies assessing the outcomes of seed introduction found that at least one of the introduced species was recorded in at least one of the sites in the majority of projects (53%) (Turnbull et al. 2000). Whether attributing seedling germination and establishment to the lottery model (Warner and Chesson 1985) or to the competition/colonisation trade-off (Tilman 1994) models of seed limitation, recruitment following species introduction by sowing affirms the existence of the regeneration niche and as a result implicates dispersal limitation as a factor resulting in the absence of the species (Turnbull et al. 2000).

In this study, seeding had profoundly positive effects with respect to all of the seeded species criteria and significantly contributed to overall seedling density and canopy cover (both in the order of a fifth) and species richness (increasing by approximately a third). Despite the positive ramifications of the burn, the game camp community of existing indigenous species remained relatively depauperate, particularly with respect to species richness and the severe over-representation of geophytes. The addition of the seeded species therefore made a dramatic contribution to overall community attributes (including the establishment of perennial graminoids, forbs, succulents, low and tall shrubs to an otherwise geophytic community) and thus emphasizes the valuable role of seed augmentation in ecological restoration. The importance of seed augmentation to redress dispersal limitation and/or seed-bank depletion is recognised both internationally (Turnbull et al. 2000, Pywell et al. 2002, Pywell et al. 2003, Standish et al. 2007, del Moral et al. 2008, Clewell and Aronson 2013) and locally in fynbos and renosterveld studies (Holmes and Richardson 1999, Holmes 2002, Holmes 2005, Holmes 2008, Memiaghe 2008).

This study found that the severe infestation of alien, annual grasses suppressed seedling recruitment. This finding is consistent with other seed-based renosterveld restoration studies (Holmes 2002, Midoko-Iponga 2004) and numerous Mediterranean-climate ecosystem studies

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(Eliason and Allen 1997, Stylinski and Allen 2001, Standish et al. 2007, Beyers 2009). In this study, seedling suppression by grasses was evident due to the seeding-only intervention resulting in the lowest performances with respect to all of the seeded species criteria. These poor performances are mostly likely the result of the considerable cover of the annual alien grass component which dominated when not reduced by the implementation of another intervention such as burning or herbicide. In general, the performance of seeding-only dramatically improved for all criteria when implemented in combination with one or more of the other interventions. This finding is consistent with the Holmes (2005) study which found that the implementation of seeding-only, without another intervention to reduce weed cover (for instance ploughing or ploughing with herbicide), resulted in notably poorer seedling recruitment. The underlying mechanism of such dynamics evident in this study is most likely the reduction in weed cover, and thus an increase in the availability of microsites and a reduction in competition, achieved by the two- to five-factor interventions facilitating enhanced seedling recruitment. This indicates that dispersal limitation (i.e. seed availability) was not the only factor inhibiting species colonisation but that the near overwhelming alien grass component plays a large role in reducing the number of available colonisation sites and exerting competitive pressure.

In this study, median seedling density per m^2 notably exceeded density in similar seed-based restoration studies (Holmes 2002, Holmes 2005). In the two year trial period of the Holmes (2002) study, non-significant differences between the seeded and unseeded plots were recorded and at the first six month census there was an average of five seedlings per m^2. Seedling density was notably higher in the Holmes (2005) study with an average of 80 seedlings per m^2 in the ploughed plots, but declined dramatically after the first summer measuring five seedlings per m^2 within three years. There are numerous study variables making comparisons inaccurate yet the difference in seedling densities can most likely principally be attributed to the variation in soil preparation treatments, for example, the intervention THS in this study compared with only burning and mulch application in the Holmes (2002) study. It was in fact the limited seed germination success in the Holmes (2002) study which precipitated recommendations for further investigation of seeding in conjunction with soil preparation treatments such as fire, tillage and herbicide-application. In addition to soil preparation variables, the difference in seed-sowing rates most likely contributes to the disparity: in the Holmes (2002) study, Holmes (2005) study and this study, the respective rates sown were 50 kg uncleaned seed, 300 kg uncleaned seed and approximately 57 kg semi-cleaned seed per hectare.

To gain insight into how seedling density and richness of the restored plots in this study compares with natural renosterveld vegetation, a comparison was carried out with burnt and unburnt remnants of Swartland Shale Renosterveld occurring in Tygerberg Nature Reserve (refer Appendix 3.10). The comparison revealed that this study had just less than three times the density of indigenous seedlings than the natural vegetation. Much of this can be attributed to a high abundance of *Oxalis* seedlings in this study which were generally slender and clumped *en masse* contributing to the outcome. Additional factors possibly responsible for the outcomes include variable vegetation age (the unburnt Tygerberg plots were in excess of 40 years), discrepancies in the levels of predation and the soil-preparation techniques implemented in this study. The same data sets were compared with respect to

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seedling richness per m². In this study, species richness was just over 70% of that in the Tygerberg plots, attributable in part due to the number of species selected for reintroduction being limited to just 31 species.

Seeding has an apparent, although non-significant, negative effect on existing species density and cover and may be due to increased competition for resources. The significant positive effect seeding has on existing indigenous species (increasing richness from 5.0 to 5.5 species per m²) and seeming positive effect in weed canopy cover are difficult to explain. The seed sown on site was mixed with horticultural sand and moistened with a fungicide-water solution, and although seemingly minor additives to the seeded plots, their effects are not known (refer Table 3.6 Ecological-response model: summary explanations).

3.7.2 Burning

Burning stimulated recruitment and increased performance of all aspects of the existing indigenous species and these finding are in keeping with studies in fire-driven ecosystems both in other parts of the world (Keeley 1991, Hester and Hobbs 1992, Bell 2001, Allen et al. 2000, Safford and Harrison 2004) and in fynbos and renosterveld ecosystems (Holmes and Richardson 1999, Midoko-Iponga 2004, Musil et al. 2005, Rebelo et al. 2006, Milton 2007).

Of concern are the findings that both density and richness of the seeded species were significantly inhibited in the burnt plots. These results were unexpected as post-fire conditions of high levels of nutrients (such as nitrogen, phosphorus and cations) (Brown and Mitchell 1986, Stock and Lewis 1986, Musil and Midgley 1990) light and water generally promote seed germination (Bond and van Wilgen 1996). The increased performance of existing indigenous species may have increased resource competition exerting an inhibitive effect on seeded species. The leachate of the dominant annual alien grass species (*Avena fatua*) has been shown to have an allelopathic effect on seed germination (Tinnin and Muller 1972) and the ash of this species may have acted as a germination suppressant. In addition, the effect of the diesel-petrol mix used in the burn, although applied to small areas, cannot be discounted (refer Table 3.6). It must be reiterated that all of the seed used in this experiment (for both the unburnt and burnt plots) was smoke-treated prior to sowing, which points therefore to an environmental variable/s rendering recruitment in the burnt plots for some reason less suitable than recruitment in the unburnt plots. Further to fire resulting in a flush of nutrients (lasting four to nine months for phosphorus and nitrogen respectively (Allsopp and Stock 1992)), fire has been found to reduce the propagule density of vesicular-arbuscular mycorrhiza and thus the level of seedling infectivity (Klopatek et al. 1988, Vilarino and Arines 1991, Allsopp and Stock 1995). Mycorrhizal infection in seedlings occurs in the majority of species (Fenner and Thompson 2005) and is most likely critical for healthy seedling development as soil phosphorus uptake in non-mycorrhizal seedlings has been shown to be inhibited (Allsopp and Stock 1995). Renosterveld vegetation has high levels of infectivity (Allsopp and Stock 1994). However, as the correlation between mycorrhiza colonisation and soil temperature as a result of fire has been found to be positive and of a large magnitude (Klopatek et al. 1988) it is unlikely that the quick autumn fire consuming predominantly grass biomass resulted in soil temperatures high enough to affect propagule density. Intense fires in

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fynbos have in addition been known to cause soil water repellency (Scott and van Wyk 1992, Bond and van Wilgen 1996) and again, the characteristics of the on-site fire make it unlikely that the burn caused repellency. Increased levels of rodent predation in a post-fire environment (Bond 1984) can in this instance be discounted as rodent-exclusion in the burnt area (BRS) results in an even lower seedling density than the burning and seeding intervention (BS). Furthermore, burning significantly reduced weed cover thereby reducing direct competition. The many inhibitive effects of dense leaf litter on seedling emergence and establishment are widely reported (Facelli and Pickett 1991, Facelli 1994, Lenz at el. 2003) yet in this study the net effect of conditions in the unburnt plot (whether merely less inhibitive or possibly even facilitative) resulted in greater seedling recruitment than conditions in the burnt plots. Midoko-Iponga (2004) hypothesized that grass litter, albeit the result of herbicide-application, provided a mulching effect which increased soil moisture retention and consequently resulted in higher seed germination. Therefore, looking to the question of soil exposure and moisture retention, in this study, due to greater exposure to direct sunlight and wind, the soil in the burnt site may have retained moisture for shorter periods than the unburnt site which may have consequently experienced less moisture variability due to the shading effect of the leaf litter layer. The relatively more exposed soil surface and greater moisture fluctuations in the burnt site, despite a generally good season of rainfall, may have inhibited seed germination (refer Table 3.6).

In this study, burning significantly reduced weed cover which was comprised predominantly of alien, annual grasses. Controlling invasions of similar annual grasses has been achieved to some extent particularly where fires are carefully timed to intercept grass seed set (Allen et al. 2000, D'Antonio and Meyerson 2002, Midoko-Iponga 2004, Keeley 2006). However, in fire-driven Mediterranean-type ecosystems in general, fire has been found to more commonly promote than inhibit invasive grasses (Minnich and Dezanni 1998, van Rooyen 2003, Milton 2004, Musil et al. 2005, Keeley 2006, Memiaghe 2008) making the post-fire application of herbicide necessary where the invasive species in question are fire-adapted. It is suggested that the control of alien, annual grass by fire may be a temporary response, as depending on the species involved and the seed longevity traits, alien grass occurrence may return to pre-fire levels (Brooks and Pyke 2001, Keeley 2006). These findings emphasize the importance of long-term monitoring to determine more accurately the case-specific effect of fire on the weed species within a community over time. The existing indigenous species and some aspects of seeded species, promoted by the burn, may in addition have exerted competitive pressure on the weed component (refer Table 3.6).

3.7.3 Tillage

Excepting seedling height of the seeded species, which had a non-significant response to tillage, tillage stimulated large, apparent, positive responses for the remaining seeded species criteria and, although non-significant, for all of the existing species criteria. These responses are consistent with numerous seed-based studies where seedling establishment increases with cultivation-type disturbance (Holmes and Foden 2001, Holmes 2005, Joubert et al. 2009, Turnbull et al. 2000; deFalco et al. 2012). The majority of the existing indigenous species on site were geophytes and their performance was most likely promoted by the disturbance from the tillage, as in renosterveld, small

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bulbils from geophytes (for example *Oxalis* species) may be released and dispersed short distances in response to small and large mammal activity and ploughing (Shiponeni 2003b, Walton 2006) (refer Table 3.6).

Despite the initial reduction in weed cover observed soon after tillage, cover rapidly recovered and within eight months was ultimately promoted by the disturbance. This finding is consistent with an extensive body of studies as both locally (van Rooyen 2003, Milton 2004, Muhl 2008, Sharma et al. 2010) and in other Mediterranean-climate regions (Dowling 1996, Stylinski and Allen 2001, Allen et al. 2000, D'Antonio and Meyerson 2002) invasive annual grasses are extensively found to increase with disturbance. This promotion of weed cover and likely competition for resources may have been responsible for the apparent reduction in seedling height (refer Table 3.6).

In keeping with other interventions such as burning (Musil et al. 2005) and herbicide-application (Holmes 2008), the transience of the positive effects of tillage giving way to longer-term negatives show that successful restoration, while not the focus of this study, clearly requires a long-term view of on-going engagement and persistent management.

3.7.4 Herbicide-application

With the exception of a non-significant negative impact on the density of existing indigenous species, the application of herbicide resulted in the desired outcomes with respect to all criteria. Both locally and in other Mediterranean-climate vegetation types, numerous studies successfully reduce alien species cover with herbicide-application (Midoko-Iponga 2004, Holmes 2005, Musil et al. 2005, Cox and Allen 2008, Holmes 2008, Memiaghe 2008), yet, in keeping with other interventions, such as burning and tillage, several report the temporary effectiveness of herbicide and thus follow-up treatment is widely advocated (Cione et al. 2002, Holmes 2008, Standish et al. 2008). The competitive effect of alien, annual grasses has been shown both in-field and experimentally to have an inhibitory effect on renosterveld species (Midoko-Iponga et al. 2005, Musil et al. 2005, Muhl 2008, Sharma et al. 2010) and in this study, the significant reduction in weed cover, and thus the reduction in grass-induced competition, was most likely the driver of the high performances of both the seeded species and existing indigenous species (refer Table 3.6). The removal of these alien, annual grasses in other renosterveld studies has similarly been found to promote indigenous species performance (Midoko-Iponga 2004, Musil et al. 2005). In addition to the positive ramifications of reduced grass competition, the removal of the above ground biomass of alien grasses, leaving the below ground root biomass to decay, has also been found to enhance the soil nutrient status (Campbell et al. 1991, Midoko-Iponga 2004) and may have been a factor further promoting the performance of the indigenous species.

3.7.5 Rodent-exclosure

During the eight month observation period, the level of seed and seedling predation was evidently negligible on this site as seedling density of the seeded species was in fact lower inside the rodent-exclusion cages than outside. This finding may be in part explained by the small-mammal survey on

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Devil's Peak which found that the game camp supported the lowest rodent density, most likely due to the low structural diversity of the alien, annual grass and forb community (Dreyer 2012). Generally, rodent densities are higher in autumn yet in the game camp the density was higher in spring (Dreyer 2012). As the controlled burn was implemented in autumn (March 2012), these population dynamics may reflect an initial decline in numbers associated with the autumn fire followed by population recovery approximately six months later. These small-mammal population dynamics in response to fire are consistent with some, but not all, studies (Midgley and Clayton 1990, van Hensbergen et al. 1992, Bond and van Wilgen 1996).

The lower seedling densities within the cages, both respect to the seeded and existing indigenous species, may have resulted from the shading effect of the cages which were found to cut out more than half of the light (although less than the glasshouse which cut out 75% of the light) (refer Table 3.6). The low light conditions within the cages may have reduced the maximum soil temperature thus affecting seed germination as has been found in other studies where low light conditions affected seed germination (Bond 1984, Pausas et al. 2004, Castro et al. 2005). Apparent, though non-significant, positive effects on seedling height of the seeded species and cover of existing indigenous species may similarly be attributed to the relatively soft, low-light and low-wind conditions within the cages (refer Table 3.6) which may have in these respects functioned like brush packs (Milton 2001, Holmes 2005) or tree shelters (Pausas et al. 2004).

The microclimate conditions within the cages favoured high canopy cover of weed species . The dominant invasive grass species, *Avena fatua*, *Briza maxima* and *B. minor*, are C3 grasses (Milton 2004) which have an affinity for low light conditions (Cowling 1983) (refer Table 3.6). This very high weed cover would most likely have exerted competitive pressure on the seeded species and existing indigenous species.

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3.7.6 Ecological-response model: summary explanations

A summary table (Table 3.6) of possible explanations for disparities between the actual experimental intervention responses and the predicted responses.

Table 3.6: Annotated ecological-response model, summary explanations. Actual responses differing from expected responses are indicated in red. Significant effects:* for p<0.05, ** for p<0.01 and *** for p<0.001.

INTERVENTION	SEEDED SPECIES	predicted	actual	EXISTING INDIGENOUS SPECIES	predicted	actual	WEED SPECIES	predicted	actual	
BURNING	Density	+	− *	Density	+	+ ***				Possibly due to allelopathic effect of grass ash, effects of diesel-petrol mix, increased exposure and lower moisture retention, and, increased competition from existing indigenous species.
	Richness	+	− ***	Richness	+	+ ***				
	Cover	+	+	Cover	+	+ *	Cover	+	− ***	Potentially a short-term effect. Possibly due to increased competition from seeded and existing indigenous species.
	Height	+	+							
TILLAGE	Density	+	+ ***	Density	−	+ ***				Probably due to disturbance promoting the dispersal of *Oxalis* bulbils.
	Richness	+	+ ***	Richness	−	+ ***				
	Cover	+	+ ***	Cover	−	+ *	Cover	+	+	Most likely due to the promotion of weed species and increased levels of resource competition.
	Height	+	−							
HERBICIDE	Density	+	+ ***	Density	−	−				Possibly due to reduced competition with weeds.
	Richness	+	+ ***	Richness	−	+ *				
	Cover	+	+ ***	Cover	−	+ ***	Cover	−	− ***	Possibly due to alteration of micro-climate within cages e.g. low light, low soil temperature, wind protection.
	Height	+	+ ***							
RODENT-EXCLUSION	Density	+	− **	Density	0	− **				Possibly due to low light conditions within cages promoting alien C3 grasses.
	Richness	+	−	Richness	0	0				
	Cover	+	− *	Cover	0	+	Cover	0	+ ***	
	Height	+	+							
SEEDING	Density	+	+ ***	Density	0	−				Difficult to explain. Negative responses possibly due to increased competition with seeded species and weed species. Both negative and positive responses possibly due to the untested effect of the seed-sand-fungicide mix.
	Richness	+	+ ***	Richness	0	+ *				
	Cover	+	+ ***	Cover	0	−	Cover	0	+	
	Height	+	+ ***							

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3.7.7 Intervention interactions

Of the 16 seeded interventions, seeding-only consistently performed the worst. However, when seeding was implemented in combination with another of the factors (B, T, H or R), the outcomes were vastly improved. Despite the notable merit of these two-factor interventions, none resulted in the desired responses for all of the measurement criteria due to downfalls in one or more of the performance areas. The intervention HS is the best performing of the two-factor interventions in this regard but none the less produced a minor reduction in the density of existing indigenous species. In contrast, six (of the 11) three- to five-factor seeded interventions (BHS, THS, BTHS, BHRS, THRS and BTHRS) produced the desired responses in all respects with positive responses for all of the seeded species and existing indigenous species criteria as well as having an inhibitive effect on weed cover. This view of enhancing intervention outcomes through the implementation of multiple factors *in lieu* of one factor is supported locally by renosterveld and fynbos studies (Midoko-Iponga 2004, Holmes 2005, Midoko-Iponga et al. 2005, Musil et al. 2005, Memiaghe 2008, Joubert et al. 2009) as well as international studies (Eliason and Allen 1998, Allen et al. 2000, Pywell et al. 2002, Pausas 2004, Standish et al. 2007, Standish at al. 2008).

Due to the considerable range in effect sizes for each of the eight performance areas it is difficult to effectively draw meaningful comparisons with respect to identifying the overall effectiveness of each intervention. Consequently, a systematic evaluation of each intervention across all performance criteria is undertaken in chapter four to determine a measure of its relative performance and overall effectiveness.

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3.7.8 Seedling survival

Despite the considerable proportion of deciduous seedlings as well as the relatively hot and dry summer period, the survival rate of just over a quarter of the total number of seedlings was high compared with other studies, in part due to the short duration of this study which limits the accuracy of comparisons. Similar seed-based studies had approximately 6% survival after three years (Holmes 2005), and nearly zero% due to poor winter rains (Midoko-Iponga 2004). Survival rates of studies using propagated seedlings transplanted onto site (as opposed to sown seed) are notably higher (Holmes 2002, Midoko-Iponga 2004) and these levels of survivorship are more comparable to those attained in this study.

An increase in litter has been found to increase the abundance of seedling predators (such as plant-chewing arthropods) and cotyledon consumption (Facelli 1994). A fair number of arthropods, millipedes and grasshoppers, were observed particularly during the September 2012 census and these species may have fed on seedlings affecting seedling survival. In the September census, evidence of seedling predation was observable particularly for many of the *Ornithogalum thyrsoides* seedlings which were grazed to soil level, the succulent species (*Othonna arborescens*) and to a lesser extent the *Chrysocoma coma-aurea/Felicia filifolia* seedlings and perennial grass species (*Tenaxia stricta* and *Cymbopogon marginatus*).

Through the empirical measurement of intervention responses, this chapter set out to make a theoretical contribution through reflecting on an anticipated ecological-response model towards building the conceptual framework for restoration protocols in Peninsula Shale Renosterveld and potentially other renosterveld types. In this study, alien, annual grass cover was controlled to a degree, predominantly by interventions implementing herbicide and/or burning, but despite these measures of control, weed cover was generally high. Establishment of indigenous shrub cover has been found to successfully exclude invasive, annual grasses (Cione et al. 2002, Heelemann 2012) and in this study the ability of the indigenous species component within each community to compete with the alien grass component over the longer-term, beyond the year scope of this study, is critical to future restoration outcomes. Holmes and Richardson (1999) recommend long-term monitoring of a restoration site to ascertain how the species and community establish with respect to composition, structure and function and in addition whether species complete life-cycles as a measure of how sustainable and self-perpetuating the restored community is likely to be. This study provides empirical information capturing the responses to the 31 interventions over a period of eight months since sowing. The span of this study therefore captures the foundational stage of restoration namely seedling recruitment with respect to seed germination, seedling establishment and seedling survival. Some of the interventions have resulted in very good performances and at this point in time appear to have the potential to develop into self-perpetuating restored communities with a semblance of ecosystem composition, structure and function. Assessing each of the interventions with respect to longer-term performance will provide a more accurate measure of intervention success and will place decision makers in a better position with respect to how to proceed with the implementation of restoration over larger areas.

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Photo plate 3.9: The low-light conditions inside the rodent-exclusion cages promoted canopy cover of the alien, annual grasses. The cages were lifted off to enable data collection. Photograph taken September 2012.

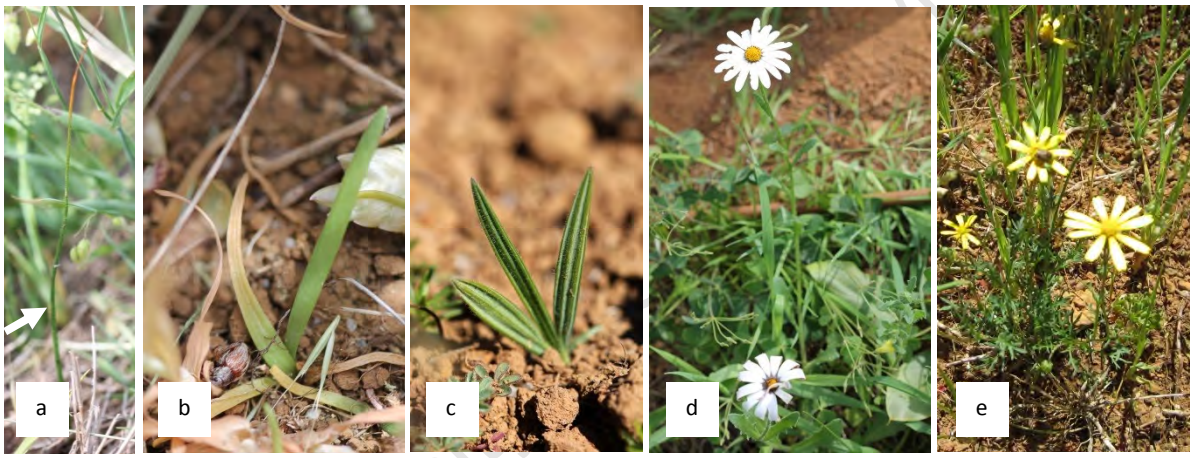

Photo plate 3.10: Some of the deciduous and annual species predominantly recorded in the September and November 2012 observation (a) *Trachyandra muricata* (b) *Ornithogalum thyrsoides* (c) *Babiana fragrans* (d) *Dimorphotheca pluvialis* and (e) *Ursinia anthemoides* subsp. *anthemoides*.

Photo plate 3.11: An unburnt seeded plot where the alien annual grasses (here *Briza maxima*) have remained dominant, exerting competitive pressure on the desirable seeded species (visible are *Dimorphotheca pluvialis, Chrysocoma coma-aurea* and *Athanasia crithmifolia*) (November 2012).

Photo plate 3.12: Burnt plots during the January 2013 census. Alien, annual grasses have died and visible green shrub cover is attributable largely to the seeded species.

Photo plate 3.13: Unburnt plots during the January 2013 census.

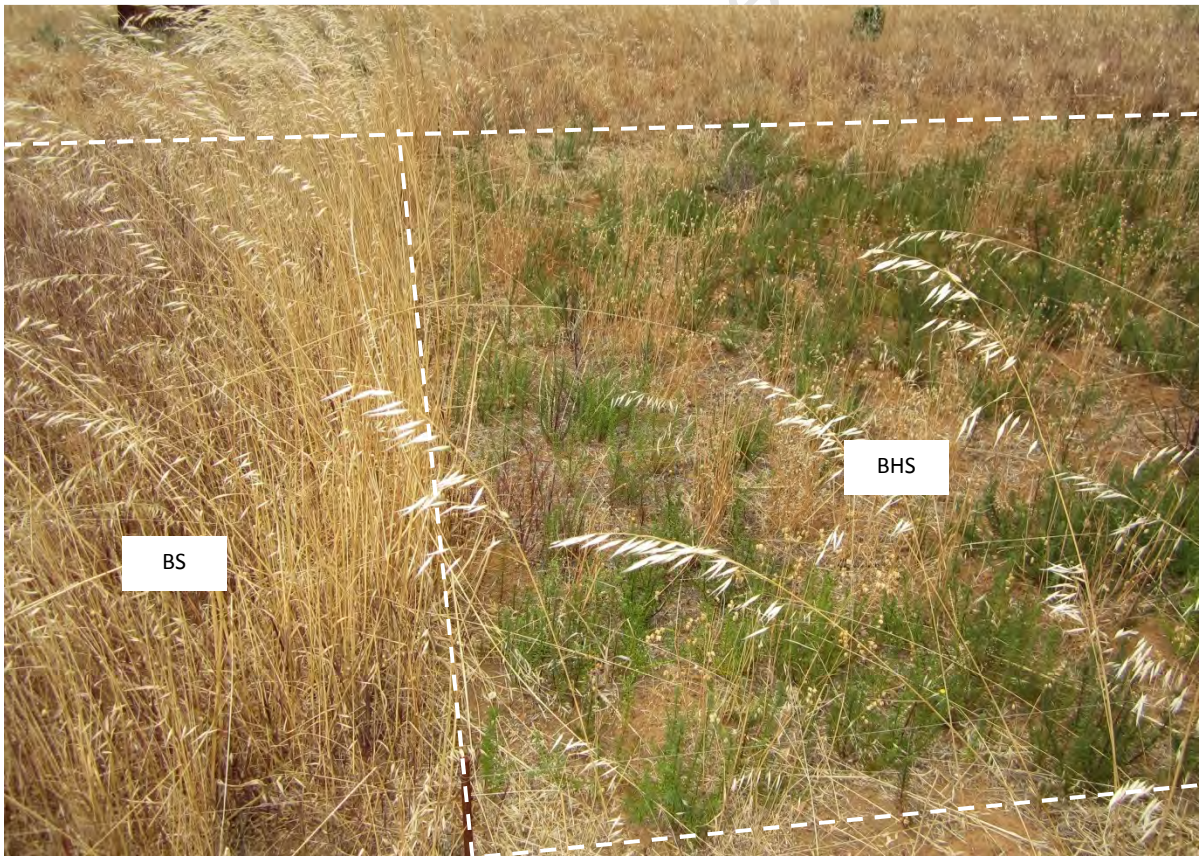

Photo plate 3.14: Burnt plots, including the seeded interventions BHS (right) and BS (left). BHS had relatively good canopy cover (with *Chrysocoma coma-aurea* as the dominant low shrub) whilst BS, without a herbicide component, was dominated by alien, annual grasses which suppressed recruitment of the seeded species. (January 2013).

Photo plate 3.15: Some of the seeded species recorded in January 2013: (a) *Podalyria sericea* (b) *Salvia caerulea-africana* (c) *Chrysocoma coma-aurea, Helichrysum patulum* and *H. cymosum* (d) *Eriocephalus africanus* (e) *Pelargonium cucullatum, Athanasia crithmifolia* and *Chrysocoma coma-aurea* (f) *Searsia tomentosa* (g) *Chironia baccifera* (h) *Cymbopogon marginatus* (i) *Tenaxia stricta* visibly grazed (j) and (k) *Ehrharta calycina* inflorescence and leaves respectively (l) *Erepsia anceps*.

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3.8 Conclusion

To optimise restoration outcomes, ecological restoration efforts need to acknowledge and incorporate the ecological factors responsible for driving community regeneration (Holmes and Richardson 1999). Since knowledge of the ecology of pristine renosterveld is limited (Rebelo 1995) and renosterveld restoration studies are few, yet growing, experimentation to test the effects of ecosystem drivers is necessary for informing an ecological-response model for restoration in Peninsula Shale Renosterveld.

Seeding was imperative to shift the community from an alien, annual grass state to a shrubland state and to enhance overall community attributes including a greater variety of growth forms and significantly increased species richness, seedling density and canopy cover. Combinations of interventions proved most successful, where over half of the three- to five-factor seeded interventions resulted in a full set of desired outcomes, and in light of an otherwise overwhelming dominance of annual, alien grasses, this finding confirms that competitive dynamics, in addition to seed availability, play a large role in community recovery and must be addressed. The ecological-response model proved a useful means of considering standard ecological factors such as competition, disturbance,

nutrients, dispersal and predation and is a means of thinking through the ecological aspects of why elements gave rise to certain outcomes. Although the anticipated responses were largely confirmed by the experimental responses, there were several unexpected negative outcomes. Some of the measures that were successful in the span of this study are noted in the literature for promoting negative effects down the line, making longer-term monitoring essential for increasing the accuracy and relevance of intervention effectiveness and the ecological-response model.

Further to the theoretical contribution of the experimental ecological-response model, this chapter generates empirical measurement for the effectiveness of each of the 31 interventions in terms of several performance criteria and, among interventions, the analyses have identified where the different strengths and weaknesses lie and the order of their magnitude. In chapter four, the magnitude of these responses are systematically evaluated across all of the criteria so as to gain insight into overall intervention effectiveness.

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3.9 References

Allen, E.B., Eliason, S.A., Marquez, V.J., Schultz, G.P., Storms, N.K., Stylinski, C.D., Zink, T.A. and Allen, M.F. 2000. *What are the Limits to Restoration of Coastal Sage Scrub in Southern California?* 2nd Interface Between Ecology and Land Development in California. In: J.E. Keeley, M.B. Keeley and C.J. Fotheringham, (eds). USGS Open-File Report 00-62, Sacramento, California. In press.

Allen, E.B., Cox, R.D., Tennant, T., Kee, S.N. and Deutschman, D.H. 2005. Landscape restoration in southern California forblands: Response of abandoned farmland to invasive annual grass control. *Israel Journal of Plant Sciences,* 53, pp. 237-245.

Allsopp, N. and Stock, W.D. 1992. Functional Perspective of ecosystems. In: Cowling, R. M. (ed) *The Ecology of Fynbos: nutrients, fire and diversity.* Cape Town: Oxford University Press, pp. 241-259.

Allsopp, N. and Stock, W.D. 1994. VA mycorrhizal infection in relation to edaphic characteristics and disturbance regime in three lowland plant communities in the South Western Cape, *South Africa. Journal of Ecology*, 82, pp. 271-279.

Allsopp, N. and Stock, W.D. 1995. Relationships between Seed Reserves, Seedling Growth and Mycorrhizal Responses in 14 Related Shrubs (Rosidae) from a Low-Nutrient Environment. *Functional Ecology*, 9(2), pp. 248-254.

Bassett, I. E., R. C. Simcock, and N. D. Mitchell. 2005. Consequences of soil compaction for seedling establishment: Implications for natural regeneration and restoration. *Austral Ecology*, 30, pp. 827- 833.

Bell. D.T. 2001. Ecological Response Syndromes in the Flora of Southwestern Western Australia: Fire Resprouters versus Reseeders. *The Botanical Review*, 67(4), pp. 417-440.

Beyers, J.L. 2009. Non-native and Native Seeding. In: A. Cerda and P.R. Robichaud, (eds). *Fire Effects on Soils and Restoration Strategies.* Enfield: Science Publishers, pp. 321-336.

Bond, W. J. 1984. Fire Survival of Cape Proteaceae: Influence of Fire Season and Seed Predators. *Vegetatio*, 56(2), pp. 65-74.

Bond, W.J. and van Wilgen, B. W. 1996. *Fire and Plants. Population and community ecology, Series 14.* London: Chapman and Hall.

Boucher C. 1995. Overview of Management of Renosterveld. In: A. B. Low and F. E. Jones (eds). *The sustainable use and management of Renosterveld remnants in the Cape Floristic Region.* Proceedings from a symposium. FCC Report 95/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp. 47-50.

Boucher, C. 1983. Floristic and structural features of the coastal foreland vegetation south of the Berg River, western Cape Province, South Africa. *Bothalia,* 14(3&4), pp. 669 – 674.

Bragg, C. J., J. D. Donaldson, and P. G. Ryan. 2005. Density of Cape porcupines in a semi-arid environment and their impact on soil turnover and related ecosystem processes. *Journal of Arid Environments*, 61, pp. 261-275.

Brits, G. J. 1986. Influence of fluctuating temperatures and H202 treatments on germination of *Leucospermum cordifolium* and *Serruria florida* (Proteaceae) seeds. *South African Journal of Botany*, 52, pp. 286-290.

Britton, P. and Jackelman, J. 1995. Biological Management of Renosterveld: Some questions and lessons from Signal Hill and Devil's Peak. In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region. Proceedings of a symposium.* FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp.58-62.

Brooks M.L. and Pyke D.A. 2001. Invasive plants and fire in the deserts of North America. In: K.E.M. Galley and T.P. Wilson (eds).2000. *Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species.* Fire Conference: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication no. 11, Tall Timbers Research Station, Tallahassee, pp. 1–14.

Brown, G. and Mitchell, D. T. 1986. Influence of fire on the soil phosphorus status in sand plain lowland fynbos, south-western Cape. *South African Journal of Botany*, 52, pp. 67-72.

University of Cape Town

Brown, N. A. C. and Botha, P. A. 2004. Smoke seed germination studies and a guide to seed propagation of plants from the major families of the Cape Floristic Region, South Africa. *South African Journal of Botany*, 70, pp. 559-581.

Brown, N. A. C., Botha, P. A., Johnson, T. and Prosch, D. S. 2000. *Propagation of Cape Wildflowers From Seed.* Paper presented at the Fifth International Botanic Gardens Conservation Congress.

Brown, N. A. C., van Staden, J., Johnson, T. and Daws, M., I. 2003. A Summary of Patterns in the Seed Germination Response to Smoke in Plants from the Cape Floral Kingdom. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice.* Kew: Royal Botanic Gardens, pp. 563-574.

Brudvig, L. A. 2011. The Restoration of Biodiversity: Where has research been and where does it need to go? *American Journal of Botany*, 98, pp. 549-558.

Campbell, B.D., Grime, J.P., Mackey, J.M.L., and Jalili, A. 1991. The quest for a mechanistic understanding of resource competition in plant communities: the role of experiments. *Functional Ecology,* 5, pp. 241-253.

Carson W.P. and Peterson C.J. 1990. The role of litter on an old-field community: impact of litter quantity in different seasons on plant species richness and abundance. *Oecologia,* 85, pp. 8–13.

Castro, J., Zamora, R., Hodar, J.A. and Gomez, J.M. 2005. Ecology of seed germination of *Pinus sylvestris* L. at its southern, Mediterranean distribution range. *Invest Agrar: Sist Recur For*, 14 (2), pp. 143-152.

Cione, N.K., Padgett, P.E. and Allen, E.B. 2002. Restoration of a Native Shrubland Impacted by Exotic Grasses, Frequent Fire, and Nitrogen Deposition in Southern California. *Restoration Ecology*, 10(2), pp. 376-384.

Clewell, A.F and Aronson, J. 2013. *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession, second edition.* Washington: Island Press.

Cowan, O. S. 2013. *The Peninsula Shale Renosterveld of Devil's Peak: Phytosociology, System Drivers and Restoration Potential.* MSc thesis, University of Cape Town.

Cowling , R. M. 1984. A syntaxonomic and synecological study in the Humansdorp region in the Fynbos Biome. *Bothalia,* 15, pp. 175-227.

Cowling R.M., Pierce S.M. and Moll E.J. 1986. Conservation and utilization of South Coast Renosterveld, an endangered South African Vegetation Type. *Biological Conservation,* 37, pp. 363-377.

Cowling, R.M. 1983. The Occurrence of C_3 and C_4 Grasses in Fynbos and Allied Shrublands in the South Eastern Cape, South Africa. *Oecologia,* 58(1), pp. 121-127.

Cox, R.D. and Allen E.B. 2008. Stability of exotic annual grasses following restoration efforts in southern California coastal sage scrub. *Journal of Applied Ecology,* 45, pp. 495-504.

Crookes, D.J., Blignaut, J.N., de Wit, M.P., Esler, K.J., le Maître, D.C., Milton, S.J., Mitchell, S.A., Cloete, J., de Abreu, P., Fourie (nee Vlok), H., Gull, K., Marx, D., Mugido, W., Ndhlovu, T., Nowell, M., Pauw, M. and Rebelo, A. 2013. System dynamic modelling to assess economic viability and risk trade-offs for ecological restoration in South Africa. *Journal of Environmental Management,* 120, pp. 138-147.

D'Antonio, C. and Meyerson, L.A. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. *Restoration Ecology,* 10, pp. 703-713.

De Villiers, C.C., Driver, A., Clark, B., Euston-Brown, D.I.W., Day, E.G., Job, N., Helme, N.A., Holmes, P.M., Brownlie, S. and Rebelo, A.B. 2005. *Fynbos Forum ecosystem guidelines for environmental assessment in the Western Cape.* Fynbos Forum. Cape Town: Conservation Unit, Botanical Society of South Africa.

DeFalco, L.A., Esque, T.C., Nicklas, M.B. and Kane, J.M. Supplementing Seed Banks to Rehabilitate Disturbed Mojave Desert Shrublands: Where Do All the Seeds Go? *Restoration Ecology*, 20(1), pp. 85-94.

University of Cape Town

Del Moral, R., Walker, L. R. and Bakker, J. P. 2007. Insights Gained from Succession for the Restoration of Landscape Structure and Function. In: L.R. Walker, J. Walker and R.J. Hobbs, (eds). 2007. *Linking Restoration and Ecological Restoration.* New York: Springer Science.

Donaldson, J., Nänni, I., Zachariades, C. and Kemper, J. 2002. Effects of habitat fragmentation on pollinator diversity and plant reproductive success in renosterveld shrublands of South Africa, *Conservation Biology,* 16(5), pp. 1267-1276.

Dowling P.M. 1996.The ecology of Vulpia. *Plant Protection Quarterly,* 11, pp. 204–206

Dreyer, R. 2012. *Ecological Functioning and Restoration potential of the Peninsula Shale Renosterveld of Devil's Peak: Understanding the small mammal community*. BSc Hons thesis, University of Cape Town.

Eliason, S.A. and Allen, E.B. 1998. Exotic Grass Competition in Suppressing Native Shrubland Re-establishment. *Restoration Ecology,* 5(3), pp. 245-255.

Facelli J.M. 1994. Multiple indirect effects of plant litter affect the establishment of woody seedlings in old fields. *Ecology,* 75, pp. 1727–1735.

Facelli J.M. and Pickett S.T.A. 1991. Plant litter: its dynamics and effects on plant community structure. *Botanical Review*, 57, pp. 1–32.

Fenner, M. 2005. *The Ecology of Seeds*. Cambridge: Cambridge University Press.

Groves, R.H., Austin, M.P. and Kaye, P.E. 2003. Competition between Australian native and introduced grasses along a nutrient gradient. *Austral Ecology*, 28, pp. 491-498.

Hartmann, H. T. and Kester, D. E, (eds). 2011. *Plant propagation: Principles and practices*. Upper Saddle River: Prentice Hall.

Heelemann 2012. *Renosterveld remnants- Current ecological situation and restoration perspectives*. PhD thesis, University of Stellenbosch.

Heelemann, S., Krug, C.B., Esler, K.J., Reisch, C. and Poschold, P. 2013.Soil seed banks of remnant and degraded Swartland Shale Renosterveld. *Applied Vegetation Science,* pp.1-13.

Heelemann, S., Krug, C.B., Esler, K.J., Reisch, C. and Poschold, P.2012. Pioneers and Perches—Promising Restoration Methods for Degraded Renosterveld Habitats? *Restoration ecology*, 20(1), pp. 18-23.

Hester, A.J. and Hobbs, R.J. 1992. Influence of fire and soil nutrients on native and non-native annuals at remnant vegetation edges in the Western Australian wheatbelt. *Journal of Vegetation Science*, 3, pp. 101-108.

Hobbs R.J. 2001. Synergisms among habitat fragmentation, livestock grazing, and biotic invasions in southwestern Australia. *Conservation Biology*, 15, pp. 1522–1528.

Hobbs, R. and Cramer, V. 2008. Restoration Ecology: Interventionist Approaches for Restoring and Maintaining Ecosystem Function in the Face of Rapid Environmental Change. *Annual Review of Environment and Resources,* pp. 39-61.

Hobbs, R. and Harris, J. 2001. Restoration ecology: Repairing the Earth's ecosystems in the new millennium. *Restoration Ecology*, 9(2), pp. 239-246.

Holmes, P. M. 2001. Shrubland Restoration Following Woody Alien Invasion and Mining: Effects of Topsoil Depth, Seed Source, and Fertilizer Addition. *Restoration Ecology*, 9(1), pp. 71-84.

Holmes, P. M. 2002. *Renosterveld Restoration on Cultivated Lands at Eerste River, Western Cape: Preliminary Findings of a Pilot Study,* unpublished report to Lafarge Quarries, May 2002.

Holmes, P. M. 2005. Results of a lucerne old-field restoration experiment at the Fynbos-Karoo interface. *South African Journal of Botany,* 71, pp. 326-338.

Holmes, P. M. and Cowling, R. M. 1997. Diversity, composition, and guild structure relationships between soil-stored seed banks and mature vegetation in alien plant-invaded South African fynbos shrublands. *Plant Ecology,* 133, pp. 107-122.

University of Cape Town

Holmes, P. M. and Richardson, D. M. 1999. Protocols for restoration based on Recruitment Dynamics, Community Structure, and Ecosystem Function: Perspectives from South African Fynbos. *Restoration Ecology,* 7(3), pp. 215-230.

Holmes, P.M and Newton, R.J. 2004. Patterns of Seed Persistence in South African Fynbos. *Plant Ecology,* 172(1), pp. 143-158.

Holmes, P.M. 2008. Optimal ground preparation treatments for restoring lowland Sand Fynbos vegetation on old fields. *South African Journal of Botany,* 74, pp. 33-40.

Holmes, P.M. and Foden, W. 2001. The effectiveness of post-fire soil disturbance in restoring fynbos after alien clearance. *South African Journal of Botany*, 67(4), pp. 533-539.

Jones, A.T. and Hayes, M.J. 1999. Increasing floristic diversity in grassland: the effects of management regime and provenance on species introduction. *Biological Conservation*, 87, pp. 381–390.

Joubert, L., Esler, K.J. and Privett. D.D.J. 2009 The effect of ploughing and augmenting natural vegetation with commercial fynbos species on the biodiversity of Overberg Sandstone fynbos on the Agulhas Plain, South Africa. *South African Journal of Botany*.

Keeley, J.E. 1991. Seed Germination and Life History Syndromes in the California Chaparral. *Botanical Review,* 57(2), pp. 81-116.

Keeley, J.E. 2006. Fire Management Impacts on Invasive Plants in the Western United States. *Conservation Biology*, 20(2), pp. 375-384.

Kemper, J., Cowling, R.M., Richardson, D.M., Forsyth, G.G. and McKelly, D.H. 2000. Landscape fragmentation in South Coast Renosterveld, South Africa, in relation to rainfall and topography. *Austral Ecology,* 25, pp. 179-186.

Kemper, J., R. M. Cowling, and D. M. Richardson. 1999. Fragmentation of South African renosterveld shrublands: effects on plant community structure and conservation implications. *Biological Conservation,* 90, pp. 103-111.

Kilian and Cowling, R. M. 1992. Comparative Seed Biology and Co-Existence of Two Fynbos Shrub Species. *Journal of Vegetation Science*, 3(5), pp.637-646.

Klopatek, C.C., DeBano, L.F. and Klopatek, J.M. 1988. Effect of simulated fire on vesicular-arbuscular mycorrhizae in pinyon-juniper woodland soil. *Plant and Soil*, 109, pp. 245-249.

Krug, C. B., R. M. Krug, D. Midoko-Iponga, B. A. Walton, S. J. Milton, and N. N. Shiponeni. 2004. *Restoration of west coast renosterveld: facilitating the return of a highly threatened vegetation type*. In: Proceedings of the 10th international conference on Mediterranean climate ecosystems, Rhodes, Rhodos, Greece, pp. 107-119.

Krug. B.B. 2004. *Practical Guidelines for the Restoration of Renosterveld.* University of Stellenbosch.

Kruger, F.J. and Bigalke, R.C. 1984. Fire in Fynbos. In: P. Booysen and N.M. Tainton, (eds). *Ecological effects of fire in South African ecosystems.* Berlin: Springer-Verlag, pp. 67- 114.

Le Maître, D. and Midgely, J. J. 1992. Plant reproductive ecology. In: R.M. Cowling, (ed). *The ecology of Fynbos: nutrients, fire and diversity.* Cape Town: Oxford University Press, pp. 135-174.

Lenz T.I., Moyle-Croft, J.L. and Facelli J.M. 2003. Direct and indirect effects of exotic annual grasses on species composition of a South Australian grassland. *Austral Ecology,* 28, pp. 23–32.

Levyns, M.R. 1935. Germination in some South African Seeds. *The Journal of South African Botany*, 1, pp. 161-170.

Levyns, M.R. 1956. Notes on the biology and distribution of the Rhenoster bush. *South African Journal of Science*, 52, pp. 141-143.

McDowell, C.R. 1988. *Factors affecting the conservation of renosterveld by private landowners.* Ph.D. thesis, University of Cape Town.

University of Cape Town

McDowell, C.R. 1995. Grazing and Renosterveld Management. In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region. Proceedings of a symposium.* FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp.63-80.

Memiaghe, H. R. 2008. *Old field restoration: vegetation response to soil changes and restoration efforts in Western Cape Lowlands.* MSc thesis. University of Stellenbosch.

Microsoft Office. 2010. *Microsoft Excel professional 2010*. Washington, USA.

Midgely, J. J. and Clayton, P. 1990. Short-term effects on small mammal populations in southern Cape coastal mountain fynbos. *South African Forestry Journal*, 153, pp. 27-30.

Midgley, J., Anderson, B., Bok, A. and Fleming, T. 2002. Scatter-hoarding of Cape Proteaceae nuts by rodents, *Evolutionary Ecology Research*, 4, pp. 623- 626.

Midoko-Iponga, D. 2004. *Renosterveld Restoration: The role of competition, herbivory and other disturbances.* MSc Thesis, University of Stellenbosch.

Midoko-Iponga, D., Krug, C.B. and Milton, S.J. 2005. Competition and herbivory influence growth and survival of shrubs on old fields: implications for restoration of threatened renosterveld shrubland, *South Africa. Journal of Vegetation Science,* 16, pp. 685-692

Mills, A.J. and Fey, M.V. 2004. Effects of vegetation cover on the tendency of soil to crust in South Africa. *Soil Use and Management,* 20(3), pp. 308-317.

Milton, S. J. 2004. Grasses as invasive alien plants in South Africa. *South African Journal of Science,* 100, pp. 69-75.

Milton, S. J. 2007. *Rapid Assessment Method for Renosterveld*. Renosterveld Assessment Workshop, unpublished report.

Minnich, R. A. and R. J. Dezzani. 1998. Historical decline of coastal sage scrub in the Riverside-Perris Plain, California. *Western Birds,* 29, pp. 366-391.

Muhl, S. 2008. *Alien grass invasion in Renosterveld: Influence of soil variable gradients*. MSc thesis, Stellenbosch University.

Musil, C.F. and Midgley, G.F. 1990. The relative impact of invasive Australian acacias, fire and season on the soil chemical status of a sand plain lowland fynbos community. *South African Journal of Botany*, 56, pp. 417-419.

Musil, C.F., Milton, S.J. and Davis, G.W. 2005. The threat of alien invasive grasses to lowland Cape Floral diversity: an empirical appraisal of the effectiveness of practical control strategies. *South African Journal of Science,* 101(9/10), pp. 337-344.

Pausas, J.G., Blade, C., Valdecantos, A., Seva, J.P., Fuentes, D., Alloza, J., Vilagrosa, A., Bautista, S., Cortina, J. and Vallejo, R. 2004. Pines and oaks in the restoration of Mediterranean landscapes of Spain: New perspectives for an old practice – a review. *Plant Ecology,* 171, pp. 209-220.

Pywell, R. F, Bullock, J., M., Roy, D., B. Warman, L., Walker, K., J. and Rothery, P. 2003. Plant traits as predictors of performance in ecological restoration. *Journal of Applied Ecology*, 40, pp. 65–77.

Pywell, R.F., Bullock, J.M., Hopkins, A., Walker, K.J., Sparks, T.H., Burke, M.J.W. and Peel, S. 2002. Restoration of species-rich grassland on arable land: assessing the limiting processes using a multi-site experiment. *Journal of Applied Ecology*, 39, pp. 294–310.

R Core Team. 2013. *A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria, available from http://www.R-project.org/.

Radloff, G.T. 2008. *The ecology of large herbivores native to the coastal lowlands of the Fynbos Biome in the Western Cape, South Africa*. PhD thesis, Stellenbosch University.

Rebelo, A.G. 1992. Preservation of biotic diversity. In: R.M. Cowling, (ed). *The Ecology of Fynbos: nutrients, fire and diversity.* Cape Town: Oxford University Press, pp. 309-344.

University of Cape Town

Rebelo, A.G. 1995. Renosterveld: Conservation and Research In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region. Proceedings of a symposium.* FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp.32-42.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. and Rutherford, M, C. 2006. Fynbos Biome. In: L. Mucina, L. and M.C. Rutherford, (eds). *The Vegetation of South Africa, Lesotho and Swaziland.* Pretoria: South African National Biodiversity Institute.

Rusch, U. 2012. The Cape Spiny Mouse: Robin Hood amongst villains. *Veld and Flora*, 98(2), pp. 66-67.

Safford, H.D. and Harrison, S. 2004. Fire Effects on Plant Diversity in Serpentine vs. Sandstone Chaparral. *Ecological Society of America*, 85(2), pp. 539-548.

Scott, D. F and van Wyk, D. B. 1992. The Effects of Fire on Soil Water Repellency, Catchment Sediment Yields and Streamflow. In: Van Wilgen, B.W., Richardson, D.M., Kruger, F..J. and Van Hensbergen, H.J. (eds). *Fire in South African mountain Fynbos: Ecosystem, Community and Species Response at Swartboskloof.* Ecological studies 93. Berlin: Springer-Verlag, pp216-239.

Sharma, G.P., Muhl, S.A., Esler, K.J., Milton, S.J., 2010. Competitive interactions between the alien invasive annual grass Avena fatua and indigenous herbaceous plants in South African Renosterveld: the role of nitrogen enrichment. *Biological Invasions,* 12, pp. 3371-3378.

Shiponeni, N.N. 2003a. *Dispersal of seeds as a constraint of revegetation of old fields in renosterveld vegetation in the Western Cape, South Africa.* MSc thesis, University of Stellenbosch.

Shiponeni, N.N. 2003b. How Renosterveld banks its seeds. *Veld and Flora*, 89(3), pp. 32-33.

Shiponeni, N.N. and Milton, S.J. 2006. Seed dispersal in the dung of large herbivores: implications for restoration of Renosterveld shrubland old fields. *Biodiversity and Conservation,* 15(10), pp. 3161-3175.

South African Weather Service SAWS 2013. *Temperature and precipitation data for Groote Schuur, Kirstenbosch, The Royal Observatory and Molteno Reservoir for 2008 to 2013*. Extracted 2013/02/05.

Standish, R.J., Cramer, V.A. and Hobbs, R.J. 2008. Land-use legacy and the persistence of invasive *Avena barbata* on abandoned farmland. *Journal of Applied Ecology,* 45, pp. 1576-1583.

Standish, R.J., Cramer, V.A., Wild S.L and Hobbs, R.J. 2007. Seed dispersal and recruitment limitation are barriers to native recolonization of old-fields in western Australia. *Journal of Applied Ecology,* 44, pp. 435-445.

Stanway, R. 2007. Alien grass infestation in renosterveld fragments: effect on threatened life-history types and potential controls. *South African Journal of Botany*, 73(2), pp. 313-314.

Stock, W.D. and Lewis, O.A.M. 1986. Soil Nitrogen and the Role of Fire as a Mineralizing Agent in a South African Coastal Fynbos Ecosystem. *The Journal of Ecology*, 74(2), pp. 317-328.

Stuart, C. and Stuart, T. 2007. *Field Guide to Mammals of Southern Africa*, South Africa: Struik.

Stylinski, C.D. and E. B. Allen. 2001. Lack of native species recovery following severe exotic disturbance in southern California shrublands. *Journal of Applied Ecology,* 36, pp. 544-554.

Terblanche, K. 2011. *Meeting conservation goals: passive restoration of peninsula shale renosterveld following the clearance of alien invasive forestry species.* BSc Hons thesis, University of Cape Town.

Tilman, D. 1994. Competition and biodiversity in spatially structured habitats. *Ecological Society of America*, 75, pp. 2-16.

Tinnin, R.O. and Muller, C.H. 1972. The Allelopathic Influence of Avena fatua: The Allelopathic Mechanism. *Bulletin of the Torrey Botanical Club*, 99(6), pp. 287-292.

Turnbull, L.A., Crawley, M.J. and Rees, M. 2000. Are Plant Populations Seed-Limited? A Review of Seed Sowing Experiments. *Oikos*, 88, pp. 225-238.

University of Cape Town

Vallejo, R.V., Allen, E.B., Aronson, J., Pausas, J.G., Cortina, J. and Gutierrez, J.R. 2012. Restoration of Mediterranean-type Woodlands and Shrublands. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.130-144.

Vallejo, V.R., Serrasolses, I., Alloza, J.A., Baeza, M.J., Blade, C., Chirino, E., Duguy, B., Fuentes, D., Pausas, J.G., Valdecantos, A. and Vilagrosa, A. 2009. Long-term restoration strategies and techniques. In: A. Cerda and P.R. Robichaud, (eds). 2009. *Fire effects on soils and Restoration Strategies.* Enfield: Science Publishers, pp. 373-398.

Van Hensbergen, H. J., Botha, S. A. and Forsythe, G. G. 1992. Do small mammals govern vegetation recovery after fire in fynbos? In: Van Wilgen, B.W., Richardson, D.M., Kruger, F..J. and Van Hensbergen, H.J. (eds). *Fire in South African mountain Fynbos: Ecosystem, Community and Species Response at Swartboskloof.* Ecological studies 93. Berlin: Springer-Verlag, pp. 182-202.

Van Rooyen S. 2003. *Factors affecting alien grass invasion into West Coast Renosterveld fragments.* MSc thesis, University of Stellenbosch.

Van Staden, J., A. K. Jäger, M. E. Light, and B. V. Burger. 2004. Isolation of the major germination cue from plant-derived smoke. *South African Journal of Botany,* 70, pp. 654-659.

Van Wilgen, B.W. and Forsythe, G.G. 1992. Regeneration strategies in fynbos plants and their influence on the community boundaries after fire. In: Van Wilgen, B.W., Richardson, D.M., Kruger, F..J. and Van Hensbergen, H.J. (eds). *Fire in South African mountain Fynbos: Ecosystem, Community and Species Response at Swartboskloof.* Ecological studies 93. Berlin: Springer-Verlag, pp. 54-80.

Vila M., Lloret F., Ogeri, E. and Terradas J. 2001. Positive fire-grass feedback in Mediterranean basis woodlands. *Ecological Management*, 51, pp. 94–112.

Vilarino, A. and Arines, J. 1991. Numbers and viability of vesicular-arbuscular fungal propagules in field soil samples after wildfire. *Soil Biology and Biochemistry*, 23, pp. 1083-1087.

Vlok, J.H.J. 1988. Alpha diversity of lowland fynbos herbs at various levels of infestation by alien annuals. *South African Journal of Botany*, 54(6), pp. 623-627.

Von Hase, A., Rouget, M., Maze, K., Helme, N. 2003. *A Fine-Scale Conservation Plan for the Cape Lowlands Renosterveld: Technical Report.* Cape Town: Cape Conservation Unit, Botanical Society.

VSN International Ltd. 2012. *Genstat statistical package 15th edition*. Hemel Hempstead, UK.

Walton, B. J. 2006. *Vegetation patterns and dynamics of renosterveld at Agter-Groeneberg Conservancy, Western Cape, South Africa.* MSc thesis. University of Stellenbosch.

Warner, R. R. and Chesson, P. L. 1985. Coexistence mediated by recruitment fluctuations: a field guide to the storage effect. *American Society of Naturalists*, 125, pp. 769-787.

Whisenant, S.G. 2002. Terrestrial systems. In: Perrow, M.R. and Davy, A.J. (eds). 2002. *Handbook of Ecological Restoration.* Cambridge: Cambridge University Press, pp. 83-105.

Wilson, D., Stock, W.D. and Hedderson, T. 2009. Historical nitrogen content of bryophyte tissue as an indicator of increased nitrogen deposition in the Cape Metropolitan Area, South Africa. *Environmental Pollution,* 157, pp. 938-945.

Young J.A. and Allen F.L. 1997. Cheat grass and range science 1930–1950. *Journal Range Management,* 50, pp. 530–535.

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Appendix 3.1: Field experiment design

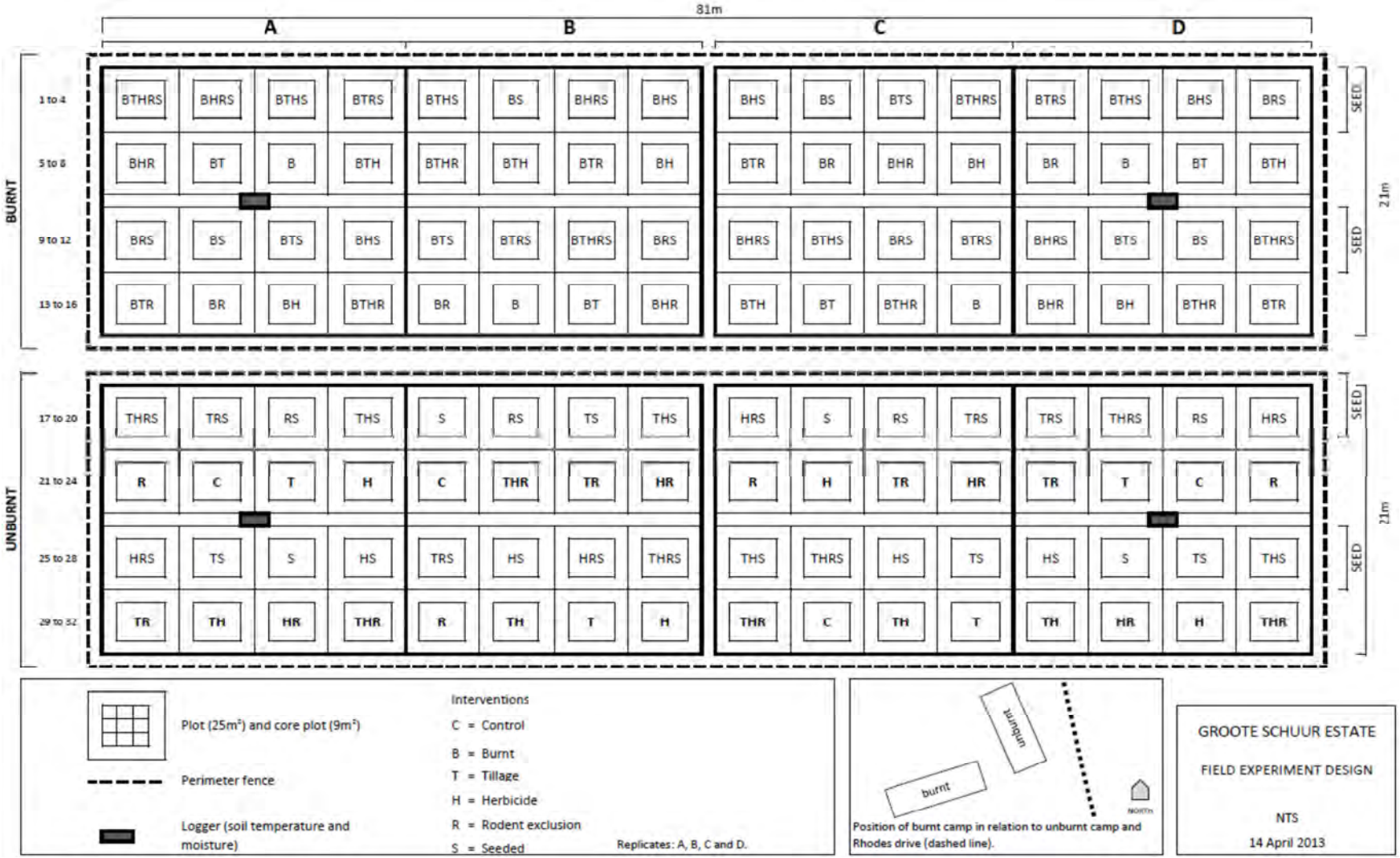

Figure 3.12: Field experiment design.

Appendix 3.2: Analysis of seeded species: summary information derived from anova for each criterion.

Table 3.7: Analysis of seeded species: summary information derived from ANOVA for each criterion.

CRITERIA	TRANSFORMATION	POWER	RESIDUALS
DENSITY	Fourth root power transformation	For an F-ratio of 8.73 at a significance level of 0.050, the power is 0.837 (to detect an effect size of 0.32).	Residual distribution is acceptably normal. There are no influential outliers and there is a slight increase in variance which is an indication of reduced precision for larger values of the response.
RICHNESS	Square root power transformation	For an F-ratio of 8.29 at a significance level of 0.050, the power is 0.818 (to detect an effect size of 0.22).	The histogram of residuals is peaked but acceptable, the outliers are shown. Plotted residual values against fitted values show slight increasing variance which is an indication of reduced precision for larger values of the response. The following plots were flagged as outliers: January 2013 observation of plot C12 (with intervention BTR) and January 2013 observation plot C28 (intervention T). No disturbance due to guineafowl scratchings was recorded for plot C12, however, an average of 23% guineafowl damage (between the two subplots) for C28 was recorded, possibly accounting for the low species count. To further assess the outliers, the fitted values were plotted against Cook's statistics and the two outliers were not considered influential.
COVER	Fourth root power transformation	For an F-ratio of 8.08 at a significance level of 0.050, the power is 0.807 (for a difference of 0.18 to be detected).	No large residuals were flagged by the Anova. The histogram of residuals is slightly skewed to the right but acceptable, the outlier is shown. The fitted values were plotted against Cook's statistics and no influential outliers were identified. The fitted-values plot indicates slight increasing variance (which is an indication of reduced precision for larger values of the response).
HEIGHT	Fourth root power transformation	For an F-ratio of 8.31 at a significance level of 0.050, the power is 0.816. (for an effect size of 0.11 to be detected).	An outlier was reported for plot A25 from January 2013 (HRS) due to a high average height. Cook's statistics were plotted against the fitted values and the outlier was found to be influential and its value deleted and retained as a missing value.

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Appendix 3.3: Analysis of existing indigenous species: summary information derived from ANOVA for each criterion.

Table 3.8: Analysis of existing indigenous species: summary information derived from ANOVA for each criterion.

CRITERIA	TRANSFOR-MATION	POWER	RESIDUALS
DENSITY	Square root power transformation	For an F-ratio of 8.08 at a significance level of 0.050, the power is 0.805 (for an effect size of 1.22 to be detected).	The following observations were identified as outliers: plot A14 (intervention BR) due to a relatively high density and plot C32 (intervention T) due to a relatively low density. Residuals plotted against fitted-values show a random scatter. Residual values were plotted against Cook's statistics and the outliers were not identified as influential.
RICHNESS	Untransformed	For an F-ratio of 8.16 at a significance level of 0.050, the power is 0.808 (for a difference of 0.71 to be detected).	No large residuals were reported. Residuals are normally distributed. The residuals plotted against fitted values show a random scatter.
COVER	Third root power transformation	For an F-ratio of 8.34 at a significance level of 0.050, the power is 0.821 (for a difference of 0.35 to be detected).	No large residuals were flagged by the general anova. The residuals are slightly skewed but acceptable. The residuals plotted against the fitted values show a random scatter.

Appendix 3.4: Analysis of weed species: summary information derived from ANOVA for canopy cover

Table 3.9: Weed species: summary information derived from ANOVA for canopy cover.

CRITERIA	TRANSFOR-MATION	POWER	RESIDUALS
COVER	Arcsine transformed	For an F-ratio of 9.29 at a significance level of 0.050, the power is 0.856 (to detect an effect size of 0.12).	Residual histogram is peaked but acceptable. The residuals plotted against fitted values indicate that the error variance does not appear to be constant where the large responses are less variable than small responses. Cook statistics were plotted against the residuals and outliers were not found to be influential.

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Appendix 3.5: Analysis of soil moisture: summary Kruskal-Wallis results

Table 3.10: Analysis of soil moisture: summary Kruskal-Wallis results.
Testing the effects of burning, seeding and logger location on soil moisture. Significant effects are in bold and underlined.

Factor	x^2	d.f.	P value		Median	Lower quartile	Upper quartile
B	4.28	1	**0.039**	0	0.179	0.116	0.255
				1	0.159	0.0945	0.228
S	1.58	1	0.209	0	0.169	0.105	0.248
				1	0.158	0.097	0.23
Location	22.5	3	**< 0.001**	Burnt block A	0.159	0.0963	0.229
				Burnt block D	0.155	0.0865	0.228
				Unburnt block A	0.155	0.083	0.213
				Unburnt block D	0.239	0.139	0.339

Appendix 3.6: Analysis of light: summary Kruskal-Wallis results

Table 3.11: Analysis of light: summary Kruskal-Wallis results.
Testing the effects of the glasshouse and rodent-exclusion cages on mean levels of light (× 10 µmol.m-2.s-1). Significant effects indicated in bold and are underlined.

Factor	x^2	d.f.	P value		Median	Lower quartile	Upper quartile
Glasshouse	44.26	1	**< 0.001**	Outside glasshouse	198	196	199
				Inside glasshouse	47.5	47	49
Cage	44.26	1	**< 0.001**	Outside cage	219	216	220
				Inside cage	97	91	99

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Appendix 3.7: Analysis of seeding: summary Kruskal-Wallis results

Table 3.12a: Summary Kruskal-Wallis results testing the effect of seeding (unseeded and seeded) on seeded species with respect to density, richness, cover and height. Significant effect in bold and underlined.

SEEDED SPECIES (seeded plots only)								
CRITERIA	FACTOR	x^2	d.f.	P value		Median	Lower quartile	Upper quartile
Density	Seeding	298.300	1	≤ **0.001**	unseeded	0	0	0
					seeded	38.75	6.5	125
Richness	Seeding	298.300	1	≤ **0.001**	unseeded	0	0	0
					seeded	4	2	6
Cover	Seeding	204.100	1	≤ **0.001**	unseeded	0	0	0
					seeded	3	0.75	6.5
Height	Seeding	250.900	1	≤ **0.001**	unseeded	0	0	0
					seeded	6.38	3.628	10.32

Table 3.12b: Summary Kruskal-Wallis results testing the effect of seeding on overall density, richness and cover (i.e. existing indigenous species plus seeded species). Significant effects in bold and underlined.

Factor	x^2	d.f.	P value		Median	Lower quartile	Upper quartile
Density	8.116	1	**0.004**	Unseeded plots (existing indigenous species only)	206.5	126	288.8
				Seeded plots (existing indigenous species plus seeded species)	244	156.5	389.2
Richness	19.23	1	≤ **0.001**	Unseeded plots (existing indigenous species only)	5	4	6.75
				Seeded plots (existing indigenous species plus seeded species)	6.5	4.3	11.8
Cover	4.258	1	**0.039**	Unseeded plots (existing indigenous species only)	17	4.3	42
				Seeded plots (existing indigenous species plus seeded species)	20.1	6.5	48.5

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Appendix 3.8: Analysis of seedling height

Table 3.13a: Summary Kruskal-Wallis results testing the effect of location (glasshouse versus field) on seedling height. The significant result is indicated in bold and is underlined.

Factor	x^2	d.f.	P value		Median	Lower quartile	Upper quartile
Location	6.465	1	**0.011**	Glasshouse	2.863	1.368	4.28
				Field	6.517	3.817	7.667

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Table 3.13b: Mean seedling height (cm) over the three observations: Sep 2012, Nov and Jan 2013 (four, six and eight months after sowing respectively)

FAMILY	SPECIES	GROWTH FORM	GLASSHOUSE TRIAL		FIELD TRIAL		GLASSHOUSE HEIGHT AS A% OF FIELD HEIGHT
			Mean height	±SD	Mean height	±SD	
APIACEAE	*Arctopus echinatus*	Forb	0.5	0.25			
ASTERACEAE	*Athanasia crithmifolia*	Tall shrub	1.1	0.39	11.7	9.83	9.1
IRIDACEAE	*Babiana fragrans*	Geophyte	3.8	0.55	6.3	2.03	59.4
GENTIANACEAE	*Chironia baccifera*	Low shrub	1.0	0.32	2.2	0.78	46.6
ASTERACEAE	*Chrysocoma coma-aurea*	Low shrub	2.7	2.21	8.5	6.85	31.6
POACEAE	*Cymbopogon marginatus*	Graminoid	4.1	4.18	5.9	3.59	69.2
ASTERACEAE	*Dimorphotheca pluvialis*	Forb (annual)	7.2		6.8	4.44	105.4
POACEAE	*Ehrharta calycina*	Graminoid	4.4	1.61	6.0	15.71	74.1
MESEMBRYANTHE-MACEAE	*Erepsia anceps*	Succulent shrub	3.2	3.37			
ASTERACEAE	*Eriocephalus africanus var africanus*	Low shrub	2.9	0.88	8.6	5.06	34.2
ASTERACEAE	*Felicia filifolia*	Low shrub	1.1	0.52	see Chrysocoma above		
ASTERACEAE	*Helichrysum cymosum* subsp. *cymosum*	Low shrub	3.1	4.18	4.1	3.30	75.3
ASTERACEAE	*Helichrysum patulum*	Low shrub	1.6	1.84	4.3	3.58	37.4
MALVACEAE	*Hermannia hyssopifolia*	Low shrub					
HYACINTHACEAE	*Lachenalia fistulosa*	Geophyte	0.4	0.30	0.1	0.32	285.0
MESEMBRYANTHE-MACEAE	*Lampranthus emarginatus*	Succulent shrub	1.5	0.76			
POACEAE	*Tenaxia stricta*	Graminoid	7.5	5.31	6.7	3.11	112.0
IRIDACEAE	*Moraea bellendenii*	Geophyte	1.8	1.18	3.6	2.41	49.9
MYRSINACEAE	*Myrsine africana*	Tall shrub	1.9	0.45			
HYACINTHACEAE	*Ornithogalum thyrsoides*	Geophyte	0.2	0.00	0.4	0.52	50.0
ASTERACEAE	*Othonna arborescens*	Succulent shrub	4.6	1.64	7.1	3.33	64.2

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GERANIACEAE	*Pelargonium cucullatum* subsp. *tabulare*	Low shrub	2.8	1.55	11.5	7.54	24.2
POACEAE	*Pentaschistis airoides* subsp. *airoides*	Graminoid (annual)					
FABACEAE	*Podalyria sericea*	Low shrub	13.6	8.10	7.3	4.19	186.3
MESEMBRYANTHE-MACEAE	*Ruschia rubricaulis*	Succulent shrub					
LAMIACEAE	*Salvia africana-caerulea*	Low shrub	2.6	1.11	6.7	7.77	38.4
ANACARDIACEAE	*Searsia laevigata var. villosa*	Tall shrub					
ANACARDIACEAE	*Searsia tomentosa*	Tall shrub			1.0		
POACEAE	*Themeda triandra*	Graminoid	8.3	10.85			
ASPHODELACEAE	*Trachyandra muricata*	Geophyte	1.1	0.34	3.2	2.57	35.6
ASTERACEAE	*Ursinia anthemoides* subsp. *anthemoides*	Forb (annual)	4.1	1.19	8.0	8.45	51.5

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Appendix 3.9: Canopy cover correlation: Spearman's rank summary results

Table 3.14: Canopy cover correlation: Spearman's rank summary results.
Results for percentage canopy cover of seeded species, existing indigenous species and weed species.

SEEDED : EXISTING			
	r	T approximation	T probability
SEP 2012	-0.083	-0.656	0.514
NOV	0.268	2.186	0.033
JAN 2013	0.406	3.501	0.001
SEEDING : WEEDS			
	r	T approximation	T probability
SEP 2012	-0.466	-4.146	<0.001
NOV	-0.559	-5.314	<0.001
JAN 2013	-0.656	-6.848	<0.001
EXISTING : WEEDS			
	r	T approximation	T probability
SEP 2012	-0.233	-1.884	0.064
NOV	-0.321	-2.669	0.010
JAN 2013	-0.224	-1.812	0.075

Appendix 3.10: Summary of comparative seedling density and species richness values

Table 3.15: Summary of comparative seedling density and species richness values.
Summary seedling density and species richness data from this study across the 16 seeded interventions (median values from the September 2012 census) and plots from Tygerberg Nature Reserve (median values from the Spring 2011 census). Tygerberg Nature Reserve data from the north facing plots (of vegetation burned the preceding March and unburnt vegetation over 40 years old) were used for the analysis. Tygerberg Nature Reserve data were collected and provided by Penny Glanville.

	DENSITY	RICHNESS
	Median (lower quartile, upper quartile)	Median (lower quartile, upper quartile)
This study: Seeded species (n=64)	55.3 (11.4, 206.6)	6.0 (2.0, 9.0)
This study: Existing indigenous species (n=64)	205.5 (127.9, 274.6)	5.8 (4.0, 7.0)
This study: Combined (seeded species plus existing indigenous species) (n=64)	317.0 (191.8, 453.4)	11.8 (7.4, 15.0)
Tygerberg Nature Reserve north plots (burnt and unburnt) (n=12)	110.0 (76.8, 126,3)	16.5 (11.8, 22.0)

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4 Intervention feasibility: determining intervention cost- effectiveness for inclusion in landscape-scale restoration strategies

4.1 Abstract

Worldwide, ecological restoration is an endeavour characterised by unpredictability, risk and substantial costs. Despite restoration being implemented throughout the world, assessment of restoration cost-effectiveness has been conducted for few studies. Further to these wide-spread challenges, the ecology of pre-disturbance renosterveld is poorly understood and the ecological requirements of renosterveld species are in general largely unknown, contributing to the limited success achieved in seed-based renosterveld restoration to date.

The need to implement ecological restoration of Peninsula Shale Renosterveld has been identified as central to redressing its conservation-target shortfall. Experimentation designed to further understanding of the ecological response of the ecosystem has been central to this research project and interventions were implemented to test their effects on community recruitment. The intervention responses were variable making comparison of the intervention outcomes complex. In order to more accurately compare overall intervention performance, this portion of the study evaluated the different performance criteria for each intervention to ascertain an overall measure of intervention effectiveness. Assessment of the intervention effectiveness indices in conjunction with cost per hectare determined a measure of intervention cost-effectiveness as an indication of intervention feasibility.

Plotting each of the 32 interventions on the cost-effectiveness plane identified the potentially feasible interventions for large-scale implementation. Of these, interventions preferred on the grounds of cost-effectiveness included: BTHS, THS, BHS and HS of the seeded interventions and BTH, BH and B of the unseeded interventions. This portion of the study found that for both seeded and unseeded interventions, there was a strong positive relationship between both the number of factors per intervention and cost per hectare, and, the number of factors per intervention and effectiveness. However, the relationships between cost per hectare and intervention effectiveness were non-significant, a reflection that the most effective interventions were not necessarily the most costly.

For large-scale restoration of Peninsula Shale Renosterveld, this cost-effectiveness model may be invoked in conjunction with consideration of the site-specific needs (the restoration agendas, goals and associated objectives) and the available budget towards selecting the most appropriate intervention for inclusion into a viable restoration strategy for restoration at scale. This systematic approach to evaluating the most feasible interventions reduces some of the risks associated with restoration by simplifying decision-making in an otherwise complex terrain and provides a greater degree of assurance placing decision-makers in a stronger position to implement well-informed restoration strategies.

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4.2 Introduction

Despite progress in restoration ecology practice and science over the past 30 years, there remain significant uncertainties in the field (van Andel and Aronson 2012). The outcomes and ecosystem trajectories of ecological restoration efforts are often difficult to predict (Clewell and Aronson 2013) contributing to the high level of uncertainty and risk associated with restoration projects (Crookes et al. 2013). There are in addition multiple risk factors associated with restoration where project outcomes may be considerably affected for example by a scarcity of information pertaining to the ecological requirements of the plant species selected for re-introduction (Crookes et al. 2013) or a dry season following species re-introductions (Cione et al. 2002, Midoko-Iponga 2004, Holmes 2005). Furthermore, as ecological processes often play out differently at scale (Wiens 1989), for instance alien invasions (Roura-Pascual et al. 2009), predicting restoration outcomes, particularly when scaling up from small experimental plots to landscape-scale areas, is likely to contribute to outcome unpredictability. For renosterveld, knowledge of the ecological processes in pristine ecosystems is poor due to large herbivore extinctions within 200 km of Cape Town by the early 1700s and extensive transformation from the late 1800s (Rebelo 1995). There is a paucity of information pertaining to the ecological requirements of renosterveld species in general, however, understanding of this nature for Peninsula Shale Renosterveld has been furthered through phytosociological and historical land-use studies for the vegetation on Signal Hill (Joubert 1991) and Groote Schuur Estate (Terblanche 2011, Cowan 2013). Despite the existence of contemporary, ecosystem-specific information, ecological restoration of renosterveld in general remains considerably challenging as reflected by the limited success achieved to date, particularly with respect to seed-based efforts. Restoration is costly (Bullock et al. 2011, Crookes et al. 2013) and intervention intensity and thus costs are contingent on the nature of ecosystem degradation (Milton et al. 2003) and may be prohibitively expensive in large areas (Holmes and Richardson 1999, Bullock et al. 2011). Worldwide, restoration is frequently undertaken yet cost-effectiveness analysis has been carried out in very few studies (Bullock et al. 2011, TEEB 2011, Birch et al. 2010). In response to the need to address the risk factors and uncertainties associated with restoration, and in the knowledge that budget is most often the largest constraint (Macmillan et al. 1998), this portion of the study assesses intervention effectiveness in association with intervention cost to determine intervention cost-effectiveness. The determination of intervention cost-effectiveness may thus reflect a measure of intervention feasibility with respect to the viability of scaling-up interventions for inclusion in landscape-scale restoration strategies. In assessing intervention effectiveness, an understanding of the theoretical framework relevant to the idea of effectiveness is required and in this respect the introduction addresses what projects strive to achieve and how one measures success (restoration agendas, goals, attributes and objectives). Thereafter, some of the issues pertaining to restoration cost are addressed.

4.2.1 Restoration agendas, goals and objectives

Ecological restoration is underpinned by several sets of values or agendas such as ecological, socio-cultural and economic values (MA 2005, Young et al. 2007, TEEB 2010). Restoration agendas

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originate from different motivational standpoints where, in broad terms, the ecological agenda seeks to recover ecosystem integrity and health, socio-economic agenda seeks to recover ecosystem services and cultural agenda seeks to unite communities and institutions (van Andel et al. 2012, Clewell and Aronson 2013). In addition to the ecological, economic, social and cultural imperatives, van Andel and Aronson (2012) incorporate the political and legislative motivations driving ecological restoration. In practice, despite restoration projects generally focussing on one or two sets of values, consideration and realisation of all agendas renders the restoration more meaningful (Clewell and Aronson 2013).

Setting restoration goals and objectives is one of the most important steps in ecological restoration and needs to incorporate the interests of all stakeholders (Tongway and Ludwig 2012, van Andel and Aronson 2012) and thus reflect all agendas (Clewell and Aronson 2013). The realisation of goals in turn requires the regular monitoring and evaluation of the objectives using appropriate methods (SER 2004, Ntshotsho et al. 2011). Goals and objectives need to be based on a thorough understanding of the restoration options and associated costs (Hobbs and Harris 2001). The numerous uncertainties associated with emerging ecosystems (van Andel and Aronson 2012) particularly in the context of a changing climate (Hobbs 2012) frequently result in unanticipated restoration outcomes (Gaertner et al. 2012). Ecological restoration should aim to recover the historic trajectory of an ecosystem (Clewell and Aronson 2013) and *in lieu* of aspirations to recreate an exact replica of the historic reference site (Jackson and Hobbs 2009, Perring et al. 2013), it is more realistic to attain just some of the attributes of the reference site (SER 2004, Tongway and Ludwig 2012) particularly those characteristics that are desirable for the ecosystem in the future (Pfadenhauer and Grootjans 1999, Suding and Leger 2012, Perring et al. 2013). Whether attributing the assembly of a restoring community to the theory of multiple equilibria (i.e. state and transition models) (Milton et al. 1994) or to the theory of nonequilibria (i.e. patch dynamics) (Pickett and White 1985), more than one possible restoration outcome needs to be acknowledged (Clewell and Aronson 2013). Thus, in defining restoration goals, instead of a single, strictly-defined restoration outcome, it is advisable to include several likely attributes that are scientifically-measurable (SER 2004, Hilderbrand et al. 2005).

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4.2.2 Ecological attributes

Despite equal importance of socio-economic and cultural agendas with the ecological agenda, the initial focus in most restoration efforts is on achieving the ecological attributes as, sequentially, their realisation is often a necessary first phase for facilitating the subsequent realisation of the remaining attributes (Clewell and Aronson 2013). Ecological attributes are attained through the implementation of restoration interventions designed to redress the biophysical degradation of a particular site (Clewell and Aronson 2013). Species composition, community structure and ecosystem function are widely recognised as core ecological elements in restoration (Holmes and Richardson 1999, van Andel and Aronson 2012, van de Koppel et al. 2012). The SER (2004) outline nine ecological attributes as appropriate for realising restoration goals and Clewell and Aronson (2013) elaborate on these and propose 11 attributes which they categorise as those that are directly realised through the implementation of biophysical interventions (landscape context, appropriate abiotic environment,

species composition and community structure) and those that manifest indirectly in response to the realisation of the direct attributes (ecological functionality, historic continuity, ecological complexity, self-organisation, resilience, self-sustainability and biosphere support). The four directly attained attributes (Clewell and Aronson 2013) are briefly addressed below.

Considering the landscape context of a restored site, successful restoration outcomes are promoted though ecological integration and connectivity at the landscape scale (Holmes 2005, Hobbs 2012, van Andel et al. 2012, van de Koppel et al. 2012, Clewell and Aronson 2013). A restoration site is found to benefit from the supporting landscape which facilitates flow of matter and movement of organisms (van Diggelen et al. 2012) whilst buffering potential threats (Clewell and Aronson 2013). Consequently, the surrounding landscape and land use may influence restoration success (Holmes and Richardson 1999). Restoration efforts in proximity to transformed landscapes are rendered more challenging than those in an untransformed context and require a more careful formulation of the restoration strategy and most likely post-implementation management that is more intensive and required over a longer time frame (Holmes and Richardson 1999).

An abiotic environment capable of supporting a restored community is critical to restoration efforts (Holmes and Richardson 1999, Memiaghe 2008, van Andel and Aronson 2012, Clewell and Aronson 2013). As discussed in Chapter 1, threshold models of ecological states of impairment and transitions, originally proposed by Whisenant (1999), emphasize the necessity of correcting the abiotic limitations of a restoration site as a priority to ensure that biotic interventions are only implemented once the abiotic environment is in a state capable of supporting the biotic changes (Hobbs and Harris 2001) thus promoting resource efficiency and restoration success.

Species composition plays a critical role in successful restoration as it determines many of the other ecological attributes, for instance, community structure, ecological functionality, historic continuity, ecological complexity, self-organisation, resilience and self-sustainability (Clewell and Aronson 2013). In aiming to achieve desirable species composition in a restoring community, Clewell and Aronson (2013) recommend species selection on the grounds of occurring in the reference site (due to the likelihood of co-adaption), species that are representative of each of the functional groups, and, a number of species that is comparable with the species richness of the reference site. Comprehensive species composition is desirable and promotes ecosystem resilience but aiming to establish as many species as those occurring at the reference site may not be realistic or financially viable (Currie et al. 2009).

The structure of a restored community is a function of the species composition where the successful establishment of plant species, with respect to sizes, life forms, abundance and spatial distribution, shapes the nature of the emergent community structure (Clewell and Aronson 2013). Establishing a community that has the potential to develop into a community with structural diversity bears relevance to the initial species selection and seed collection process.

4.2.3 Cost components to restoration

Restoration is costly (Crookes at al. 2013) and in addition to the expense of the actual intervention implementation, there are numerous supplementary aspects of a restoration project, each element

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having a cost factor contributing to the total cost. The SER (2004) describes a comprehensive ecological restoration project in six phases, each of the phases having costs relating to time (fees/wages), capital expenditure (equipment and tools) and consumables (materials including biotic resources such as seed). The phases and deliverables involve conceptual planning (feasibility study), preliminary tasks (budget, ecological inventory and impairment history, baseline monitoring, reference model, goals and objectives, permitting, interested parties engagement), implementation planning (implementation plan, training and skill development), implementation tasks (intervention implementation), post-implementation tasks (monitoring, maintenance, adaptive management), and evaluation and publicity (final report, technical presentations, articles) (Clewell and Aronson 2013).

As degradation severity increases so too do the restoration costs (Rees et al. 2007) as the costs are a function of the degradation extent, duration and intensity (Hobbs 2012). In practice the costs of ecological restoration, particularly of large areas in excess of 5 hectare (Holmes and Richardson 1999), may be prohibitively high (Macmillan et al. 1998). One of the principle challenges in ecological restoration is to cover the costs of a project and be cost-effective (Bullock et al. 2011). In recent years, an approach labelled 'ecological economics' is empirically motivating the restoration of natural capital in the political decision-making arena (Daly and Farley 2010). Scientific data are increasingly validating the sustainable use of ecosystems, and therefore ecosystem restoration, as more economically profitable than if used unsustainably or not restored (Cowling et al. 2003, Frazee et al. 2003, Rees et al. 2007, Aronson and van Andel 2012). When evaluating both the monetary and non-monetary benefits of ecological restoration derived from ecological and socio-economic aspects (de Groot et al. 2010), increasingly, investments into ecological restoration have been shown to make economic sense with respect to return on investment and cost-effectiveness (Aronson and van Andel 2012 citing de Groot et al. unpubl. MS; Elmqvist et al. unpubl. MS). In this light, implementing restoration should be seen as an investment with good returns and not simply as a cost (de Wit et al. 2009, Aronson and van Andel 2012 citing de Groot et al. unpubl. MS). Further to the net benefits of successful restoration, implementing a Payment for Ecosystem Services (PES) scheme is another way of incentivising restoration practices (de Abreu 2011). In South Africa the Working for Water programme is a flagship PES scheme functioning to alleviate poverty and restore ecosystem services (water provision) through the removal of invasive plant species (Turpie et al. 2008).

In acknowledgment of the challenges associated with restoration ecology, the experimentation in this study has functioned to test the effects of restoration interventions on community recruitment and thus plays a critical role in generating an ecological-response model (Hobbs and Harris 2001) for restoration of Peninsula Shale Renosterveld. In so doing, the ecological-response model increases the predictability of intervention outcomes (Suding and Leger 2012) thereby reducing some of the risk associated with unpredictability. The outcomes for each of the 32 interventions were varied with respect to response direction and response magnitude. Furthermore, interventions exhibited various combinations of strengths and weaknesses in the eight different performance areas (density, richness, height and canopy cover of the 31 seeded species; density, richness and canopy cover for the existing indigenous species; and, canopy cover of the weed species). To better compare this variability in intervention responses, this portion of the study, assesses each of the 32 interventions in a simple and

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systematic manner whereby the effect sizes for each of the performance criteria have been evaluated to determine an overall index of intervention effectiveness. Assessed in combination with cost per hectare, each intervention has been expressed in terms of cost-effectiveness as a measure of intervention feasibility.

4.3 Methods

A pragmatic approach to assessing restoration costs is required where the marginal benefits of the proposed restoration effort need to be greater than the marginal costs of implementation, in order to motivate and justify restoration efforts (Farley and Gaddis 2007). Cost-effectiveness analysis is applicable where monetary value is not easily assigned to the measure of the restoration outcomes (the effects) (Emerton and Bos 2004, Rees et al. 2007). Cost-effectiveness analysis is used to assess each option in terms of cost and effectiveness (Macmillan et al. 1998, Rees et al. 2007) to determine a cost-effectiveness index which facilitates easy comparison among the options and thus decision-making on the basis of selecting a preferred restoration strategy to deliver on project objectives in a cost-effective manner (Gittinger 1982, National Research Council 2005).

To ascertain the intervention cost-effectiveness index for each of the 32 seeded interventions (including the control), no additional experimentation was required for this portion of the study as the analysis made use of existing data from the field experiment (presented in Chapter 3). The field-trial data (evaluated to reflect intervention effectiveness) were considered in conjunction with the intervention cost per hectare. The project costs, with respect to the actual expenditure, the number of personnel and the work hours per task, were recorded throughout the study.

4.4 Data collection

No new data were collected for this portion of the study which made use of existing data collected from the field trial in July 2012, September 2012, November 2012 and January 2013. The project costs including actual expenditure, the number of personnel and work hours per task were recorded throughout the duration of the study.

4.5 Data analysis

4.5.1 Intervention cost

Actual expenditure (including wages, capital costs and consumables) was compiled in a bill of quantities for each of the main interventions (refer Appendices 4.1 to 4.5 for cost breakdowns). The costs of interventions that are comprised of two or more factors were calculated on an additive cost basis (i.e. cost of HS=cost of H + S). The costs for tillage, herbicide and seeding were scaled up *pro rata* to calculate cost per hectare. The cost of the burn was provided by the Newlands Working on Fire base and due to the small size of the area burnt in this study relative to the basic 'start up' costs involved, in calculating the cost per hectare, only the additional cost of more fuel was added to the

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actual cost items as the cost of burning the experimental site is effectively the same as burning 1 hectare. Due to the relatively large expense of the rodent-exclusion cages, the cost was multiplied by a factor of only two for the purposes of maintaining a realistic inter-intervention comparison.

Wages varied according to role and although the seed collection volunteers and I were not paid, it was decided to include the cost of our hours into the model so as to reflect a more realistic cost scenario for similar work being carried out in future by labourers and a team supervisor respectively. The actual cost of the Technical Officer from the UCT Zoology department (who assisted with construction of the rodent-exclusion cages) was a subsidised rate of R65 per hour but his time has been incorporated into the costing at R120 per hour again to reflect a more realistic fee. The following wage costs were incorporated into the costing model (refer Table 4.1).

Table 4.1: Role and wage rates as incorporated into the costing model.

ROLE	R / HOUR
Seed collection assistant	R 65.00
Seed collection volunteers (x 15 but usually averaging one to two volunteers per collection)	R 22.50
UCT Technical Officer	R 120.00
Labourers (x 1 to 2 for intervention implementation)	R 22.50
Field data-collection assistant	R 50.00
My time (when directly involved with labour for intervention implementation and supervision)	R 120.00

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4.5.2 Intervention effectiveness

The effectiveness of interventions was evaluated for both the 16 seeded interventions as well as for 16 unseeded interventions (including the control). Untransformed data from each of the criteria measured in the field experimentation (median values for density, richness, canopy cover and height of the seeded species; density, richness and canopy cover of the existing indigenous species; and, canopy cover of the weed species) were divided into categories in accordance with the scale of the response and allocated scores ranging from one to 14 (Table 4.2). The seeded species criteria were weighted (allocated twice the value) due to the important role that species reintroductions by seed is required to play in ecological restoration (Holmes and Richardson 1999, Holmes 2002, Hufford and Mazer 2003, Holmes 2005, Krauss and He 2006, Broadhurst et al. 2008, Mortlock 2008), particularly in areas such as the game camp in Groote Schuur Estate where the seed bank is largely depleted (Cowan 2013), the functional groups of the existing indigenous species are poorly represented (species being almost entirely geophytic), the area being dominated by annual invasive alien grasses and situated far from relatively untransformed remnants which function as a source of seed.

4.5.3 Cost-effectiveness

To show the graphic relationships between intervention cost per hectare and the intervention effectiveness indices, these variates were plotted on a cost-effectiveness plane for each of the 32 interventions where the 16 seeded interventions and the 16 unseeded interventions (including the

control) have been plotted separately due to the large cost differential associated with the seeding intervention.

4.5.4 Correlation among variates: number of factors per intervention, intervention cost per hectare, intervention effectiveness

Separate nonparametric correlation tests (Spearman's rank correlation) (for the seeded and unseeded interventions and control) were run on three variates: the number of factors per intervention, intervention cost per hectare and intervention effectiveness.

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Table 4.2: Measuring intervention effectiveness.
Categories (dividing the effect range of the untransformed medians) and scores for each of the eight criteria measured in the field experimentation.

CRITERIA		UNIT	SCORE											
			(0)	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(10)	(12)	(14)
SEEDED SPECIES	Density	seedlings/m²	<35		35-55		55-75		75-95		95-115	115-135	135-155	>155
	Richness	species/m²	<4		4-4.5		4.5-5		5-5.5		5.5-6	6-6.5	6.5-7	>7
	Canopy cover	% cover/m²			<4		4-5		5-6		6-7	7-8	8-9	>9
	Height	cm/seedlings/m²					<8		8-10		10-12	12-14	14-16	>16
EXISTING INDIGENOUS SPECIES	Density	seedlings/m²	<150	150-200	200-225	225-250	250-275	275-300	300-325	325-350	>350			
	Richness	species/m²	<4.5	4.5-5	5-5.5	5.5-6	6-6.5	6.5-7	7-7.5	7.5-8	>8			
	Canopy cover	% cover/m²		<20	20-25	25-30	30-35	35-40	40-45	>45				
WEED SPECIES	Canopy cover	% cover/m²		>95	90-95	85-90	80-85	75-80	70-75	<70				

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4.6 Results

4.6.1 Intervention cost

The costs (experimental cost and per hectare) for each of the main interventions and for each of the 32 interventions (additive costs of interventions comprised of two or more factors) are reported in Table 4.3 and 4.4 respectively.

Table 4.3: Experimental cost and cost per hectare for each of the main interventions.
Experimental costs are scaled up and rounded off to the nearest rand to calculate the cost per hectare except rodent-exclosure which was not scaled up to a hectare but multiplied by only two*.

MAIN INTERVENTIONS	EXPERIMENTAL COST	COST PER HA
Seeding	R 60 884.86	R 365 309
Burning	R 6 700.00	R 7 000
Tillage	R 4 770.00	R 28 620
Herbicide application	R 1 438.30	R 8 630
Rodent exclosure	R 59 729.71	* R 119 459

Table 4.4: Cost per hectare for each of the 31 interventions and control.

UNSEEDED		SEEDED	
Interventions	Cost per hectare	Interventions	Cost per hectare
C	R 0	S	R 365 309
B	R 7 000	BS	R 372 309
T	R 28 620	TS	R 393 929
H	R 8 630	HS	R 373 939
R	R 119 459	RS	R 484 769
BT	R 35 620	BTS	R 400 929
BH	R 15 630	BHS	R 380 939
TH	R 37 250	THS	R 402 559
BR	R 126 459	BRS	R 491 769
TR	R 148 079	TRS	R 513 389
HR	R 128 089	HRS	R 493 398
BTH	R 44 250	BTHS	R 409 559
BTR	R 155 079	BTRS	R 520 389
BHR	R 135 089	BHRS	R 500 398
THR	R 156 709	THRS	R 522 018
BTHR	R 163 709	BTHRS	R 529 018

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4.6.2 Intervention effectiveness

The workings (untransformed medians and corresponding scores) and total score (effectiveness index) for each of the 16 seeded interventions and the 15 unseeded interventions and control are reported in Table 4.5 and 4.6 respectively (refer Table 3.4 for the medians and inter-quartile range).

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Table 4.5: Effectiveness of seeded interventions.
Median effect size (untransformed data) and corresponding scores for each of the 16 seeded interventions. The total score is a measure of overall intervention effectiveness.

	SEEDED SPECIES								EXITING INDIGENOUS SPECIES						WEED SPECIES		TOTAL SCORE
	DENSITY		RICHNESS		COVER		HEIGHT		DENSITY		RICHNESS		COVER		COVER		
	seedlings /m²	score	Spp. / m²	score	% cover / m²	score	ht /seedling	score	seedlings / m²	score	Spp. / m²	score	% cover / m²	score	% cover / m²	score	
S	38.8	2	4	2	3.0	2	6.4	4	205.5	2	5.5	3	15.1	1	95	1	**17**
BS	34.5	0	3.5	2	3.3	2	7.5	4	264.8	4	6.5	5	15.8	1	92.5	2	**20**
TS	91.8	6	5.0	6	5.0	6	6.5	4	262.5	4	6.4	4	17.0	1	94.5	2	**33**
HS	64.0	4	5.0	6	5.8	6	8.6	6	211.3	2	5.7	3	20.0	2	81	4	**33**
RS	26.5	0	4.0	2	3.0	2	7.3	4	191.5	1	5.7	3	19.5	1	100	1	**14**
BTS	77.8	6	5.0	6	4.8	4	7.5	4	313.0	6	7	6	16.4	1	89.8	3	**36**
BHS	60.0	4	5.0	6	6.5	8	10.1	8	252.0	4	6.8	5	25.5	3	71	6	**44**
THS	169.3	14	6.3	10	7.0	10	8.5	6	264.8	4	6.6	5	20.0	2	91.3	2	**53**
BRS	25.5	0	4.0	2	3.0	2	7.8	4	248.5	3	6.8	5	19.5	1	96.5	1	**18**
TRS	82.5	6	5.5	8	4.8	4	6.2	4	261.5	4	6.8	5	22.8	2	100	1	**34**
HRS	61.5	4	5.5	8	5.3	6	9.2	6	214.5	2	5.9	3	26.5	3	93.5	2	**34**
BTHS	124.3	10	5.3	6	9.0	14	8.9	6	293.8	5	7.3	6	17.5	1	72.8	6	**54**
BTRS	60.5	4	5.0	6	4.3	4	7.6	4	284.5	5	7.4	6	25.0	3	96.5	1	**33**
BHRS	65.5	4	5.0	6	6.5	8	16.4	14	252.0	4	7.4	6	40.0	6	86	3	**51**
THRS	125.5	10	7.0	14	7.8	10	8.9	6	279.0	5	7.4	6	31.5	4	97.5	1	**56**
BTHRS	112.5	8	6.5	12	9.0	14	12.8	10	280.5	5	8.3	8	35.0	5	94.5	2	**64**

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Table 4.6: Effectiveness of unseeded interventions.
Median effect size (untransformed data) and corresponding scores for each of the 15 unseeded interventions and control (note: seeded species criteria are irrelevant and are dropped from the evaluation). The total score is a measure of overall intervention effectiveness.

	EXITING INDIGENOUS SPECIES						WEED SPECIES		TOTAL SCORE
	DENSITY		RICHNESS		COVER		COVER		
	seedlings / m²	score	Spp. / m²	score	% cover / m²	score	% cover / m²	score	
C	146.5	0	3.5	0	19.5	1	100	1	**2**
B	286.0	5	6.2	4	18.8	1	89.8	3	**13**
T	265.3	4	6.1	4	17.8	1	93	2	**11**
H	208.0	2	5.5	3	23.3	2	77.5	5	**12**
R	197.5	1	5.3	2	20.0	2	100	1	**6**
BT	350.8	8	6.9	5	21.8	2	86.8	3	**18**
BH	252.0	4	6.6	5	28.0	3	69.3	7	**19**
TH	264.8	4	6.3	4	22.0	2	81.5	4	**14**
BR	248.5	3	6.5	5	24.0	2	96.5	1	**11**
TR	257.5	4	6.2	4	23.8	2	100	1	**11**
HR	212.0	2	5.7	3	31.8	4	91	2	**11**
BTH	315.5	6	6.8	5	25.0	3	66.5	7	**21**
BTR	315.5	6	7.1	6	30.0	4	96.5	1	**17**
BHR	252.0	4	7.1	6	41.5	6	78	5	**21**
THR	279.0	5	6.7	5	33.5	4	94.5	2	**16**
BTHR	315.5	6	7.6	7	48.0	7	88.5	3	**23**

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4.6.3 Cost-effectiveness

Plotting the intervention effectiveness indices against intervention cost per hectare, for each of the interventions, graphically illustrated the relationships between the variates (refer Figure 4.1 and 4.2 for seeded and unseeded interventions respectively).

General relationships within quadrants of the cost-effectiveness plane

When plotted, all interventions with a rodent-exclosure component occurred above the *x* axis due to the expense of the exclusion cages, which in the context of this experimentation, renders such interventions unfeasible at the larger scale. Of the four quadrants, interventions that occurred in the lower right quadrant had the greatest potential as they were characterised by a relatively low cost per hectare in concert with relatively good performance due to a relatively high effectiveness index. Interventions that occurred within the upper left quadrant had the opposite characteristics (high cost and low effectiveness) and are therefore a waste of resources and unfeasible. The upper right quadrant (high cost and high effectiveness) and lower left quadrant (low cost and low effectiveness) contained potentially feasible interventions which warrant further consideration (in relation to the project specific information and objectives) before being discarded as viable options, particularly if an inexpensive alternative method for rodent-exclusion could be implemented *in lieu* of the costly exclusion cages.

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Seeded interventions

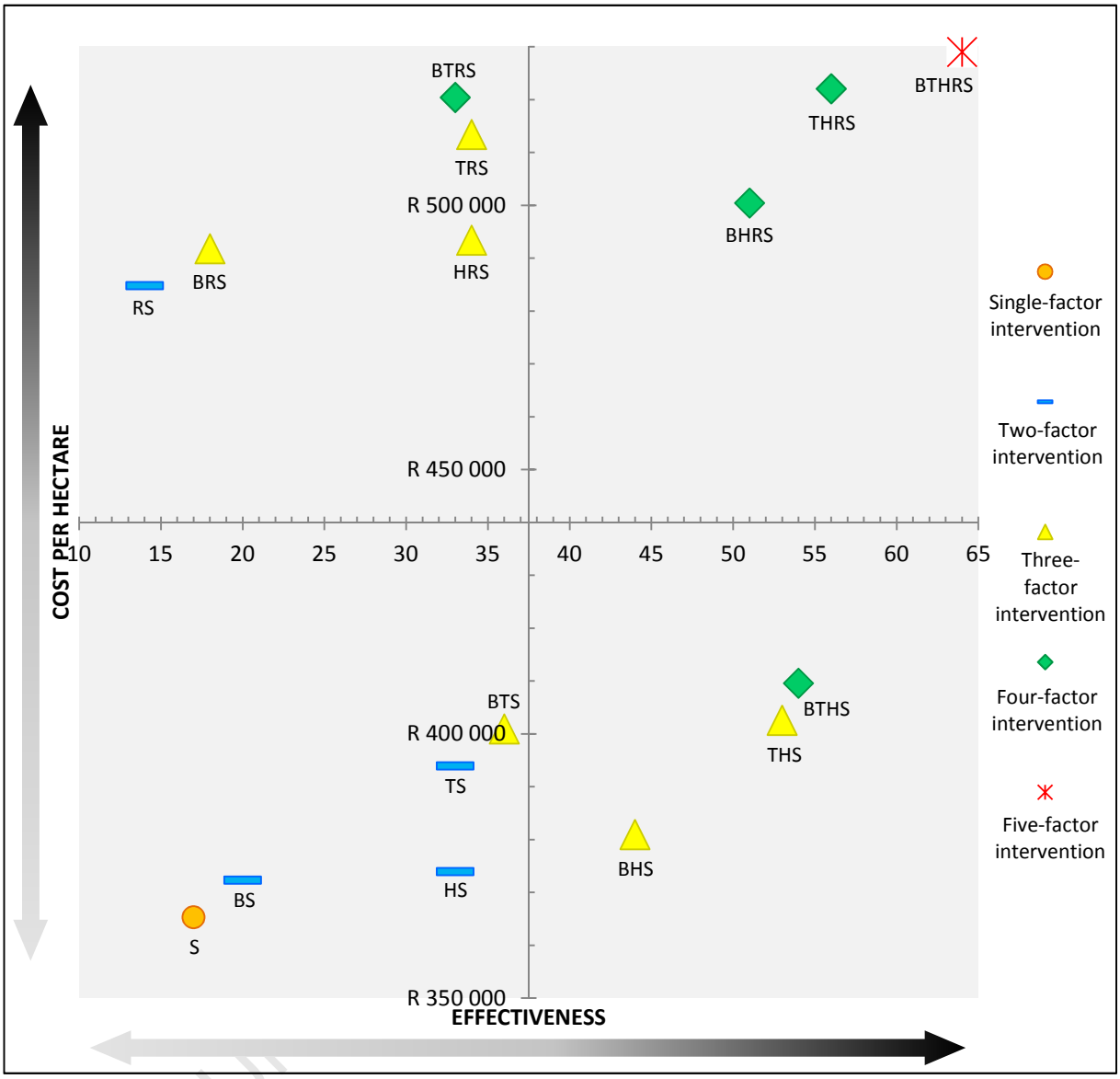

Figure 4.1: Cost-effectiveness plane for the 16 seeded interventions.
Indicating overall intervention effectiveness in relation to intervention cost per hectare and the number of factors per intervention.

Plotting on the cost-effectiveness plane (Figure 4.1) graphically depicted the following anticipated relationships: that as the number of factors per intervention increased, so too did the cost per hectare and as the number of factors per intervention increased so too did the intervention effectiveness index.

In the context of this experimentation, due to the following findings, interventions with a rodent-exclusion component are not viable for larger scale restoration: the very high rodent-exclusion cage cost (positioning all interventions with a rodent-exclusion factor above the *x* axis), the effect of the rodent population on seeded species seedling density was found to be negligible and that the cages were in fact found to have a significant negative effect on seeded species density among other criteria

(refer chapter 3). This said, the perimeter camp fences had been excavated 300 mm below ground level to exclude porcupines, the effect of which was not controlled for.

Seeding-only was the least costly and performed poorly with only RS being less effective. The five-factor intervention, BTHRS, was the most costly and most effective with an effectiveness index of 64. However, from a cost-effectiveness point of view there was not always a positive relationship between effectiveness and cost. Of the two- factor interventions HS was as effective as TS yet notably less costly and marginally more costly than BS yet notable more effective. Of the three-factor interventions, the two most effective interventions, THS and BHS, were amongst the three least costly interventions. Of the four-factor interventions, the second most effective intervention, BTHS (with 54), was only marginally less effective than THRS (56) and of the two, BTHS was considerably less expensive.

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Unseeded interventions and control

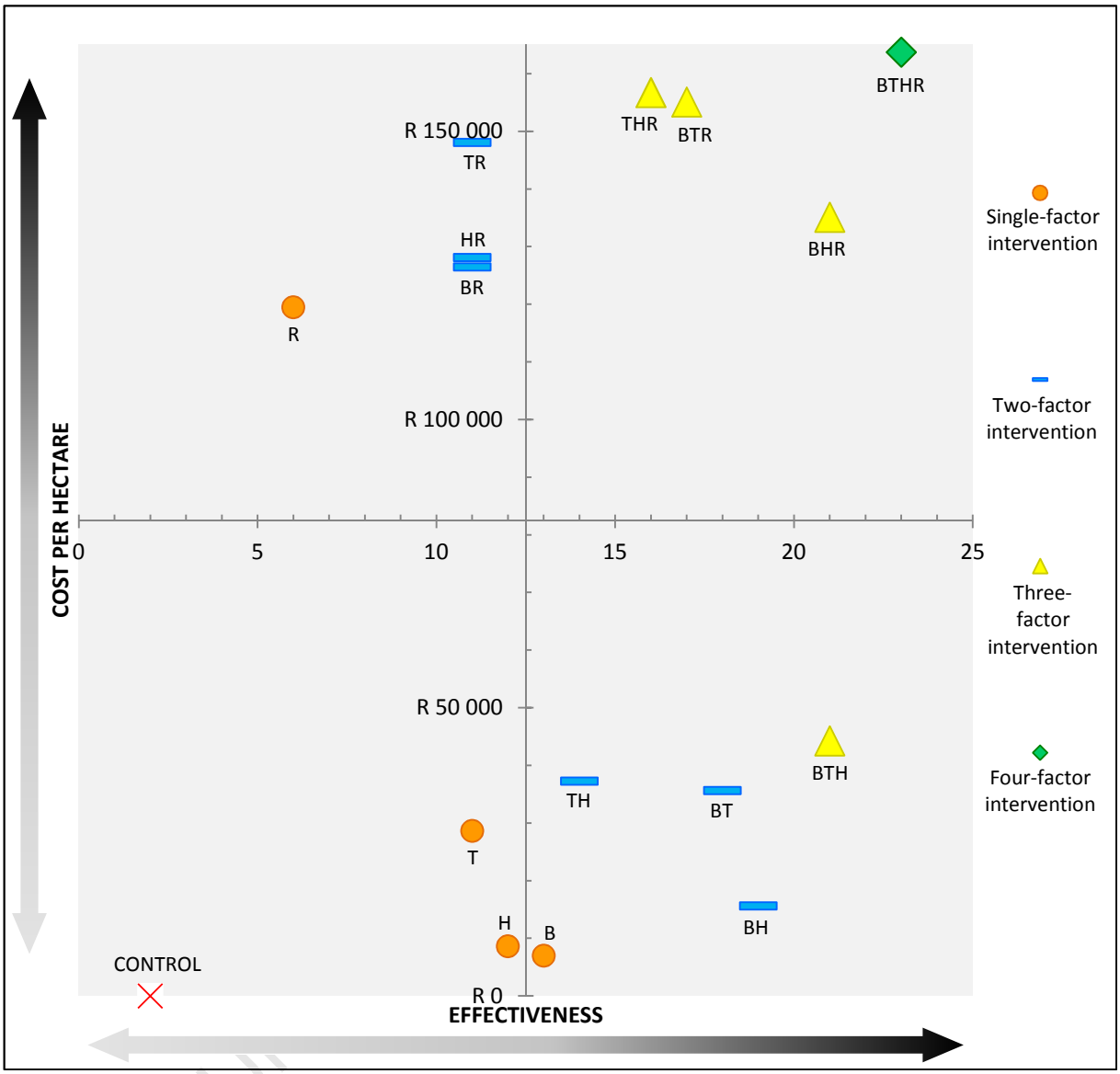

Figure 4.2: Cost-effectiveness plane for the 15 unseeded interventions and control.
Indicating overall intervention effectiveness in relation to intervention cost per hectare and the number of interactions per intervention.

The same general trends were plotted in the cost-effectiveness model for the unseeded interventions (Figure 4.2) as those depicted for the seeded interventions. Generally, as the number of factors per intervention increased, so too did the cost per hectare, and, an increase in the number of factors per intervention generally increased the intervention effectiveness index.

The treatment control (the do-nothing approach) had no cost and was the least effective, with an effectiveness index of just 2 points. The four-factor intervention, BTHR, was the most costly and most effective (with an effectiveness index of 23). As observed in the cost-effectiveness model for seeded interventions, the most effective interventions were not necessarily the most expensive. Of the four single-factor interventions, burning was the most effective yet least costly. Of the six two-factor interventions, BH (19 effectiveness index) was the most effective yet least costly. The three-factor

intervention BTH performs as well as BHR (both achieving an effectiveness index of 21) yet was distinctly the least costly of the three-factor interventions.

4.6.4 Correlation among variates: number of factors per intervention, intervention cost per hectare, intervention effectiveness

Results of the Spearman's rank correlation, for both the seeded and unseeded interventions, between the three variates (the number of factors per intervention, cost per hectare and intervention effectiveness) are reported in Table 4.7.

Table 4.7: Correlation results for the three variates: number of factors per intervention, intervention cost per hectare and intervention effectiveness.
Correlations for both the 16 seeded interventions and the 15 unseeded interventions and control. Significant effects are in bold and underlined.

	VARIATES	r	t approximation	t probability
SEEDED INTERVENTIONS	**Number of factors : cost per ha**	0.805	5.083	**<0.001**
	Number of factors : intervention effectiveness	0.785	4.744	**<0.001**
	Cost per hectare : intervention effectiveness	0.475	2.021	0.063
UNSEEDED INTERVENTIONS	**Number of factors : cost per ha**	0.805	5.083	**<0.001**
	Number of factors : intervention effectiveness	0.789	4.800	**<0.001**
	Cost per hectare : intervention effectiveness	0.334	1.324	0.207

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Considering the seeded interventions, two of the three correlations are significant. Both the correlations between the number of factors per intervention and cost per hectare, as well as the number of factors per intervention and intervention effectiveness are positive, significant at the 1% level, and, of a moderate to high magnitude (r=0.805 and r=0.785 respectively).

For the unseeded interventions, the correlations between the number of factors with cost and the number of factors with effectiveness were significant (p=<0.001 for both) and the magnitude of the relationships were strong (r=0.805 and r=0.789 respectively).

For both the seeded and unseeded interventions, the correlation of cost per hectare with intervention effectiveness were positive yet not significant (p=0.063 and p=0.207 respectively). This lack of significance indicates that the most effective interventions were not necessarily the most costly.

4.7 Discussion

In plotting the 31 interventions and control on the cost-effectiveness plane, the potentially feasible interventions were readily identified. There were strong positive relationships between the number of factors per intervention and cost per hectare, and, the number of factors per intervention and intervention effectiveness for both the seeded and unseeded interventions. This latter finding reiterates that single-factor interventions generally performed relatively poorly and that intervention effectiveness increased as did the number of factors per intervention. As budget is so often the most significant constraining factor in restoration (Macmillan et al. 1998), it is profoundly encouraging to establish that, for both seeded and unseeded interventions, the most effective interventions were not necessarily the most costly. This finding is reflected by non-significant positive correlations for the variates intervention cost per hectare and intervention effectiveness.

4.7.1 Landscape-level restoration costs

A meta-analysis of 96 restoration projects conducted by TEEB (2011) determined that the costs of restoration vary according to ecosystem type and costs have been found to range from several hundred to several thousand US$ per hectare for grassland, rangeland and forest restoration; several tens of thousands US$ per hectare for inland waters restoration; and, several million US$ per hectare for coral reef restoration (Bullock et al. 2011) equating roughly (in the order of ten) to several thousand to several tens of thousands; several hundred thousand and several tens of millions ZAR per hectare respectively.

It is useful to look at costs in other restoration projects for comparison, to explore surrogate interventions and gain insight into how the costs of this project fit into the cost spectrum. In addition to the rodent-exclosure intervention (the methodology of which is not feasible for large scale restoration), in this study, seeding contributed a considerable portion of the cost for each seeded intervention. Relative to similar local studies the cost of seeding per hectare is high predominantly because the labour rates were well above the minimum wage rate and a large portion of the time and capital expense items particular to the actual experimentation, such as the seed processing costs (which would not ordinarily form part of an actual restoration effort), were included in the total cost (refer Appendix 4.5a). As alternatives to the costing model, seed costing alternative 1 (refer Appendix 4.5b), removes the costs particular to the experimentation yet retains the same wage rates, whilst, seed costing alternative 2 (refer Appendix 4.5c), similarly removes the cost items particular to the experimentation and in addition reduces wages to the minimum wage rate. Both of these adjustments (seed costing alternatives 1 and 2) resulted in considerable reductions of nearly a third and 90 % of the original total cost per hectare respectively and would result in massive savings for seed-based restoration in larger areas.

Using the cost of seed per kg is a useful criterion for inter-project comparison despite the numerous variables most notably the seed being used in varying states from uncleaned to semi-cleaned (debris partially removed). In the Holmes (2002) study, the rate for uncleaned seed was approximately R 2 300 per uncleaned kg (4.05 kg uncleaned seed collected in 467 hrs at a supervised

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labour rate of R20 per hour). Seed collection efforts in the Holmes (2005) study, where a suction harvester was used, collected an equivalent of 300 kg uncleaned seed per hectare at a cost of R 14 000 per hectare (labour rate of R 80 per day) which resulted in an exceptionally low rate of approximately R 50 per uncleaned kg seed. For a commercial landscaping company, based in the Western and Northern Cape and in part specialising in restoration and seed collection, the direct cost price is approximately R 400 to R 800 per kg seed (clean seed equivalent) and incorporating mark-up the average species (R 650) is commercially available for approximately R 1 200 to R 1 400 per kg seed (clean seed equivalent) (D van Eeden, personal communication, February 28, 2013). For seed collected by hand in the Prince Albert area, the cost price inclusive of labour, transport, post-harvest processing and storage, ranged from R 150 to R 400 per kg (Milton 2010). In this study, approximately 9.6 kg semi-cleaned seed was collected in approximately 415 hours and using the actual wage rates equated to approximately R 3 500 per semi-cleaned kg per hectare, which is comparatively high, yet substituting the actual wage rates with the seed costing alternative 2 wage rates (minimum wage) equated to just less than R 500 per semi-cleaned kg per hectare, a cost that is more comparable with commercial seed costs.

In keeping with the explanations provided in chapter 2, additional to, *inter alia*, the genetic fitness of a source population (Mijnsbrugge et al. 2010) and the effects of resource limitation on seed traits (Fenner and Thompson 2005), there are numerous elements of the seed collection process, from pre-collection planning to post-collection processing and storage, which are inextricably linked to seed quality and thus restoration outcomes. Attention to detail and management in these respects was possible due to the relatively small scale of this project and is not necessarily realistic for landscape-level restoration. Seeds collected at the right time (Way 2003), processed without inflicting seed coat damage (Dickie and Stuppy 2003), stored in favourable conditions (Hay and Smith 2003), and treated with the necessary cues capable of alleviating dormancy are likely to exhibit a healthy physiological status, and, subject to favourable environmental conditions, are likely to germinate well in the field. Factors such as fumigation (to eliminate insect infestation), seed storage in the Kirstenbosch cold room (15 °C and 15 % equilibrium RH), seed cleaning and germination-cue expert advice provided by the staff from the Kirstenbosch Gardens and Seed Room and the Millennium Seed Bank at Kew and SANBI contributed to the preservation of seed viability and most certainly the generally good in-field performance.

4.7.2 Intervention surrogates and cost-reductions

Surrogate treatments can be sought for the interventions implemented in this study although their efficacy is untested. Despite an initial capital outlay, seed collection can be rendered more cost-efficient by using a suction harvester (Holmes and Richardson 1999, Holmes 2005, Milton 2010) and by targeting seed and leaf litter from the ground in addition to collecting from the plant canopies (Holmes 2005, Milton 2010). Suction harvesters (range from a small hand-held-, backpack-, or large tractor-drawn devices) (Holmes and Richardson 1999) and a widely-used, basic, hand-held model is a vacuum shredder (Holmes 2005) costing around R 4 500 and lasting approximately 30 machine days (Milton 2010).

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Not strictly surrogates for seeding but interventions that have been found to promote seed germination and seedling survival, and thus may support a less intensive (and therefore less costly) seeding intervention, include methods such as creating a variety of microsites (van der Merwe and Kellner 1999, Whisenant 2002, Holmes 2005, Milton 2010), establishing nurse plants (Beukman 1991, Holmes 2005, Padilla and Pugnaire 2006) and the installation of brush packs (van Rooyen 2000, Holmes 2005). In aiming to establish greater seedling density in restoration by seeding, Holmes (2005) suggested the creation of a variety of microsites within the restoration site for instance by mechanically pitting (Milton 2010) or tilling the soil surface (van der Merwe and Kellner 1999, Milton 2010) by placement of brush packs (Beukes 1999, van Rooyen 2000) (for example several dead shrubs placed on the soil surface) and mulch (Beukes 1999, Holmes 2002, Whisenant 2002, Holmes 2005). In addition to trapping seed and soil particles, brush packs were found to act as non-competitive surrogates for nurse plants and promoted vegetation recovery through sun and wind protection and increasing animal activity (Holmes 2005). Through facilitation, nurse plants have been found to promote seedling survival and growth in arid areas (Beukman 1991, Padilla and Pugnaire 2006). Propagation and establishment of seedlings in nodes, in conjunction with seeding, may be viable where guilds are difficult to establish by seed such as resprouting species (Holmes and Richardson 1999) and the larger, animal-dispersed shrub species (Holmes 2005). Fast-growing pioneer species, introduced by seed simultaneously with the other guilds, have also been found to play a nurse plant role in promoting survival of the slower-growing perennial seedlings in their shelter (Holmes 2005).

Due to the very small size of the burnt area in this study relative to the basic 'start up' costs involved, the cost of the burn would have been considerably reduced on the basis of the economy of scale (C Dilgee, personal communication, February 26, 2013). Whether in a private or public capacity, assistance with implementing prescribed burning is best sought through membership with a Fire Protection Association in partnership with one of the over 200 Working on Fire bases located across the country. The National Working on Fire Programme, predominantly funded by the Department of Water and Environmental Affairs, has numerous partners including the Working for Water Programme, Expanded Public Works Programme, national and provincial conservation agencies, local and district municipalities and each of the Fire Protection Associations (WoF 2013). For both state and private entities, membership with a Fire Protection Association is highly recommended due to the numerous benefits and cost incentives. As a member, all activities associated with a prescribed burn (for example preparation of fire breaks, burn implementation and post-burn observation) cost approximately R 1 000 per day for a team of 25 people. In areas where plant biomass is high, three to four days of post-burn observation may be required (P Prins, personal communication, February 26, 2013).

The cost of tillage was elevated as the work was done by hand due to the small scale of the experimental design. In practice, alleviating soil compaction with for instance a tractor-drawn rip tine and following up with a tine harrow to fine grade the soil surface creating safe sites for seed germination (Holmes 2005), is considerably more cost-effective for large areas. Pricing this process for the game camp for instance, in addition to an initial once-off cost of R 2 000 to transport the

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tractor to and from site, to rip and harrow one hectare would cost approximately R 900 per hectare including an operator, equipment and fuel (as approximately two hectares can be ripped and harrowed in a day at a rate of R1 800 per day) (D Haddad, personal communication, February 23, 2013).

The rodent-exclusion cages were useful in establishing that the effect of the rodent population on seed and seedling predation was negligible, however the method is unfeasible at a larger scale. This unfeasibility is illustrated for instance by BHRS which is considerably more costly than BTHS yet equally effective. More cost-efficient surrogate methods which have the potential to reduce the impact of rodents include sunken fences (excavated to 300 mm below ground level) if porcupine predation is anticipated, installing artificial bird perches to encourage small rodent predation (Milton 2001, Holmes 2002) and broadcasting chicken-feed seed mix outside of the restoration site until such time as the bulk of the indigenous seed has germinated (Holmes 2002). The installation of bird perches has the added benefit of promoting seed dispersal by frugivorous bird species (Heelemann et al. 2012). Further experimentation on the effects of fire on rodent population dynamics is required and, although not the case for gerbils (Holmes 2008), some studies have found that burning a site reduces the population size and/or granivory (Midgley and Clayton 1990, Bond and van Wilgen 1996, Dreyer 2012) for a period of approximately six to nine months (van Henbergen et al. 1992) and may be sufficient to reduce seed predation during the critical germination months.

4.7.3 Effectiveness

The variability in intervention responses makes comparison among interventions, and thus intervention selection, complex. The evaluation of the effect sizes across all criteria, as carried out in this chapter, thus provided greater insight into a cumulative measurement of intervention effectiveness and functions to aid identification and selection of an appropriate intervention that is comprised of the factors that are likely to result in the desired outcomes, towards achieving the restoration objectives.

The concept of restoration effectiveness, and thus restoration success, is contingent on the agendas and goals motivating the restoration efforts and the degree to which the corresponding attributes have been realised (Clewell and Aronson 2013). As previously stated, restoration outcomes are considerably more successful where multiple agendas are satisfied (Tongway and Ludwig 2012, van Andel and Aronson 2012), yet in practice, implementing interventions addressing the ecological agenda is often prioritised ahead of the socio-economic considerations as realisation of the ecological attributes is often necessary for the manifestation of the remaining attributes (Clewell and Aronson 2013). In considering restoration outcomes, Clewell and Aronson (2013) suggest that realisation of all 11 ecological attributes they propose (landscape context, appropriate abiotic environment, species composition, community structure, ecological functionality, historic continuity, ecological complexity, self-organisation, resilience, self-sustainability and biosphere support) equates to ecological restoration success. However achieving all of these attributes is likely to take many years (Clewell and Aronson 2013) and may not be appropriate to all projects or feasible (Hobbs and Harris 2001).

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Restoration projects achieve varying degrees of success and a meta-analysis of ecological restoration projects from numerous ecosystems worldwide found that biodiversity and ecosystem services were positively correlated and improved in the order of 44% and 25% respectively (Rey Benyas et al. 2009). Bullock et al. (2011) similarly found that restoration efforts are successful in increasing biodiversity and ecosystem services but caution that these outcomes should not automatically be assumed. In some local studies, treatment effects in some respects have been short-lived and become less or cease to be discernible beyond one or two years (Midoko-Iponga 2004, Musil et al. 2005, Krug and Krug 2007, Memiaghe 2008). Towards achieving species composition, community structure and ecosystem functioning in fynbos and fire-prone shrubland systems, Holmes and Richardson (1999) recommend monitoring goals such as certain measures of density, species richness and canopy cover for each of the functional groups for at least a two year period since the initial fire event. Where the goal of restoration is to achieve a fully functioning community, comparing the restored community with a reference community is advisable (Holmes and Richardson 1999). However, the focus of this study is narrowed by the objective to measure intervention effectiveness towards identifying the most effective intervention for incorporation into a standard restoration protocol where for instance seed from as many species as possible is harvested for reintroduction. As the aim was not to test which intervention resulted in a community most comparable to a reference community, but to specifically test intervention effectiveness with respect to seed germination criteria, the number of species was restricted to 31 to keep the seed physiological test aspect manageable yet ensure a growth form representation wide enough to adequately gauge intervention effectiveness. To measure longer-term attributes in restoring communities, a second fire event implemented five to 10 years after the initial fire is an appropriate period after which to ascertain measures of community resilience and self-sustainability (Holmes and Richardson 1999), where such attributes, as reflected for instance by subsequent seed output (Turnbull et al. 2000), are of great relevance to ascertaining long-term success (Hilderbrand et al. 2005, Hobbs 2012, Aronson and van Andel 2012, Clewell and Aronson 2013).

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4.7.4 Cost-effectiveness

The findings of the correlation tests analysing the relationship between cost and effectiveness indicate that the most effective interventions were not necessarily the most costly. The implication of these findings is reassuring for restoration efforts, as in practice the available budget more often than not constrains restoration scope (Macmillan et al. 1998) and thus limits the nature of the intervention incorporated into a restoration strategy.

The outcomes of this portion of the study indicated that, of the potentially feasible seeded interventions (BTHRS, THRS, BTHS, THS, BHS, TS, HS, BS and S), on the grounds of cost-effectiveness, the following interventions were preferred: BTHS, THS, BHS and HS. Although BTHRS and THRS were the two most effective interventions, the considerable cost of the rodent-exclosure cages in the context of this study rendered them unviable at the larger scale, yet substitution with a cost-efficient surrogate intervention may render the interventions feasible. HS was preferred over TS due to being as effective yet less costly. Although BS and S were potentially feasible, they were

dropped from the list of preferred interventions due to relatively low effectiveness indices compared with that of HS which was considerably more effective for a cost that was only marginally greater.

Should over the longer-term, these 'most effective' interventions develop poorly in relation to the invasive grass component, an alternative phased approach may be to hold the indigenous grass seed back on initial sowing, implement a rigorous grass-specific herbicide-application programme, and once effectively eradicated, reintroduce the indigenous grass species by seed. The grass seed reintroduction may then take place subject to the effective control of invasive annual grasses (which could be the following winter or even several years later), by which time the indigenous shrub species should be well established and the invasive grass element eradicated or reduced to a minimum (bearing in mind, however, the alien grass seed bank is likely to persist for a longer duration (Milton 2004)). This phased approach to species reintroduction in ecological restoration has been suggested by Pywell et al. (2003) for species that are poor performing, for instance due to a poor ability to establish or compete, yet are desirable elements to establish in a restoring community. This step-wise, species-reintroduction approach, implemented over several years, is proposed on the basis of the poor performing species being better able to establish once the environmental conditions are more suitable and the community exhibits greater stability (Pywell et al. 2003). An alternative phasing approach, more in alignment with fire-adapted ecosystems, could entail alien grass control for a number of years prior to the reintroduction of the full spectrum of seed into a post-fire environment (P Holmes personal communication July 27, 2013).

Where a seed bank is viable and/or functional types are well-represented, a restoration intervention using species re-introduction by seeding is no longer necessary. Of the five potentially feasible unseeded interventions (BTHR, BHR, BTH, BH and B) the preferred unseeded interventions from a cost-effectiveness perspective were BTH, BH and B (with BTHR and BTR being dropped due to the relatively costly rodent-exclusion component). In considering the 15 unseeded interventions, of the single factors, burning contributed the most to the high performance indices in general. Of the top four interventions, BH was the most cost-effective intervention without a tillage component, and consequently is associated with an intact, undisturbed topsoil layer. Tillage of previously cultivated land is appropriate for implementation in conjunction with species reintroduction by seed to promote seed germination (Holmes 2005, Milton 2007, this study) and tillage has been found to be effective in controlling weeds such as the herbaceous lucerne (Holmes 2005). However, ploughing (and by inference tillage) results in considerable change to substratum properties (Holmes and Cowling 1997), reduces species and life-form richness and slows renosterveld recovery (Walton 2006), and, renders the soil more susceptible to subsequent species invasion (Holmes and Cowling 1997, Kemper et al. 2000, van Rooyen 2003, Milton 2004) rendering tillage potentially inappropriate in a restoration context where the ecosystem is in a relatively untransformed state and seeding or weed control is not required. Where avoiding disturbance of the soil structure is necessary, BH becomes the preferred option. Given that BH was only marginally more expensive than B, yet was notably more effective, implementing BH on the grounds of cost-effectiveness would be preferable to implementing B alone. Yet, should there not be adequate budget to implement H, the implementation of B alone would still result in some degree of effectiveness and at minimal cost.

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However, it must be noted that where the relevant invasive species are fire-adapted this latter approach of burning in autumn without herbicide-application may ultimately promote the invasive species.

It is important to note that the herbicide used in this study was a broad spectrum post-emergent herbicide to which the predominantly geophytic community of existing indigenous species (primarily *Oxalis* species and *Zantedeschia aethiopica*), although in leaf when sprayed, was largely resilient. Under different baseline conditions, for instance where the seed bank is in a better condition, and where a good diversity of seedlings from a spectrum of functional groups emerge post-fire, the broad-spectrum herbicide used in this study could have had a negative effect on community recruitment. Thus, where desirable seedlings are present in the post-fire community, a more appropriate herbicide which targets the specific weed species is important, for instance a post-emergent gramicide such as Gallant Super by Dow Agro-Sciences for the control of annual and perennial grasses as applied in a type of West Coast Renosterveld by Midoko-Iponga (2004) and by Memiaghe (2008). This alternative however would still have an impact on the desirable indigenous grass species and, as a follow-up action to address this limitation perhaps a focussed re-introduction of grass species by seed would be necessary.

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4.8 Conclusion

This portion of the study simplifies the assessment of intervention performance through an evaluation of the variable intervention response criteria to produce an overall index of intervention effectiveness. Consideration of intervention effectiveness in conjunction with intervention cost, in the form of the cost-effectiveness plane, as populated by the data generated from the field experimentation, identified several empirically-validated, potentially feasible interventions.

This chapter found significant, strong, positive relationships between the number of factors per intervention and effectiveness, and, the number of factors per intervention and cost for both the seeded and unseeded interventions, however, the correlations between intervention cost and intervention effectiveness were non-significant. This relationship between cost and effectiveness is fortuitous as it indicates that the most effective interventions were not necessarily the most costly. On the grounds of cost-effectiveness, of the seeded interventions, the preferred interventions included BTHS, THS, BHS and HS with BTH, BH and B emerging as the preferred unseeded interventions.

This chapter provides decision-makers and restoration stakeholders with a tool, an intervention cost-effectiveness matrix, which may be used to guide restoration decisions. While the tool could be adapted to any ecological restoration situation, the empirical contributions here relate specifically to restoration in Peninsula Shale Renosterveld and have the potential to shape restoration strategies in other closely associated renosterveld types. Through the determination of intervention cost-effectiveness, potentially feasible interventions, which have been empirically validated, have been identified. Consideration of the potentially feasible interventions in concert with project budget, agenda, goals and objectives, places restoration stakeholders in a strong position to select the most appropriate intervention for inclusion in feasible, landscape-scale restoration strategies. A tool such as this has the potential to facilitate informed decision-making in a complex terrain based on

individual needs and circumstances and should go some way to addressing frequently reported stumbling blocks to successful restoration (van Andel and Aronson 2012, Clewell and Aronson 2013, Crookes et al. 2013).

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4.9 References

Aronson, J. and van Andel, J. 2012. Restoration Ecology and the Path to Sustainability. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.293-304.

Beukes P.C. 1999. *Livestock impact assessment and restoration strategies in the semi-arid Karoo*. Ph.D. thesis, University of Cape Town.

Beukman, R.P. 1991. *The role of nurse plants in the vegetation dynamics of the succulent Karoo*. MSc thesis, University of Cape Town.

Birch J.C., Newton, A.C., Alvarez Aquino, C., Cantarello, E., Echeverria, C., Kitzberger, T., Schiappacasse, I. and Tejedor Garavito, N. 2010. *Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services.* PNAS, 107(50), pp.21925-21930.

Bond, W.J. and van Wilgen, B. W. 1996. *Fire and Plants. Population and community ecology, Series 14.* London: Chapman and Hall.

Broadhurst, L. M., Lowe, A., Coates, D. J., Cunningham, S. A. McDonald, M., Vesk, P. A., and Yates, C. 2008. Seed supply for broadscale restoration: maximizing evolutionary potential. *Evolutionary Applications*, pp. 587–597.

Bullock, J.M., Aronson, J., Rey Banayas, J.M., Pywell, R. and Newton, A. 2011. Restoration of ecosystem services and biodiversity. *Trends in Ecology and Evolution,* 26, pp.541-549.

Cione, N.K., Padgett, P.E. and Allen, E.B. 2002. Restoration of a Native Shrubland Impacted by Exotic Grasses, Frequent Fire, and Nitrogen Deposition in Southern California. *Restoration Ecology*, 10(2), pp. 376-384.

Clewell, A.F and Aronson, J. 2013. *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession, second edition.* Washington: Island Press.

Cowan, O. S. 2013. *The Peninsula Shale Renosterveld of Devil's Peak: Phytosociology, System Drivers and Restoration Potential.* MSc thesis. University of Cape Town.

Cowling, R.M., Pressey, R.L., Rouget, M. and Lombard, A.T. 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. *Biological Conservation*, 112, pp. 191-216.

Crookes, D.J., Blignaut, J.N., de Wit, M.P., Esler, K.J., le Maître, D.C., Milton, S.J., Mitchell, S.A., Cloete, J., de Abreu, P., Fourie (nee Vlok), H., Gull, K., Marx, D., Mugido, W., Ndhlovu, T., Nowell, M., Pauw, M. and Rebelo, A. 2013. System dynamic modelling to assess economic viability and risk trade-offs for ecological restoration in South Africa. *Journal of Environmental Management,* 120, pp. 138-147.

Currie, B., Milton, S., J. and Steenkamp, J.C. 2009. Cost–benefit analysis of alien vegetation clearing for water yield and tourism in a mountain catchment in the Western Cape of South Africa. *Ecological Economics,* 68 pp.2574-2579.

Daly, H.E. and Farley, J. 2010. *Ecological Economics: Principles and Applications, 2nd edition.* Washington: Island Press.

De Abreu, P. 2011. *The effect of rehabilitation on ecosystem services in the semi-arid Succulent Karoo lowlands of the Little Karoo, South Africa.* M.Sc Thesis, University of Cape Town.

De Groot, R.S, Blignaut, J., Aronson, J., van der Ploeg, S. Elmqvist, T. and Farley, J. (unpubl. MS). *Restoration pays: evidence from the field.*

De Groot, R.S. Alkemade, R., Braat, L., Hein, L. and Willemen, L. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, 7, pp. 260-272.

De Wit, M. P., H. Van Zyl, D. J. Crookes, J. N. Blignaut, T. Jayiya, V. Goiset and B. K. Mahumani. 2009. Investing in Natural Assets. *A Business Case for the Environment in the City of Cape Town.* Report prepared for the City of Cape Town, South Africa.

University of Cape Town

Dickie, J. B. and Stuppy, W. H. 2003. Seed and fruit Structure. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice*. Kew: Royal Botanic Gardens, pp. 255-279.

Dreyer, R. 2012. *Ecological Functioning and Restoration potential of the Peninsula Shale Renosterveld of Devil's Peak: Understanding the small mammal community*. BSc Hons thesis, University of Cape Town.

Elmqvist, T., Setala, H., Handel, S. van der Ploeg, S., Aronson, J., Blignaut, J. and de Groot, R. (unpubl. MS). *Benefits of restoring ecosystems services in cities*, unpublished report.

Emerton, L. and Bos, E. 2004. Counting Ecosystems as an Economic Part of Water Infrastructure. Gland: IUCN.

Farley, J. and Gaddis, E. 2007. Restoring Natural Capital: An Ecological Economics Assessment. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business and Practice.* Washington: Island Press, pp. 17-27.

Fenner, M. and Thompson, K. 2005. *The Ecology of Seeds*. Cambridge: Cambridge University Press.

Frazee, S. R., R. M. Cowling, R. L. Pressey, J. K. Turpie, and N. Lindenberg. 2003. Estimating the costs of conserving a biodiversity hotspot: a case-study of the Cape Floristic Region, South Africa. *Biological Conservation,* 112, pp. 275-290.

Gaertner, M., Holmes, P.M. and Richardson, D. 2012. Biological Invasions, Resilience and Restoration. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.265-280.

Gittinger, P. 1982. *Economic Analysis of Agricultural Projects*, 2nd ed. EDI: John Hopkins University Press.

Hay, F. R. and Smith, R. D. 2003. Seed Maturity: when to collect seeds from wild plants. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds).*Seed Conservation: turning science into practice.* Kew: Royal Botanic Gardens, pp. 97-133.

Heelemann, S., Krug, C.B., Esler, K.J., Reisch, C. and Poschold, P.2012. Pioneers and Perches—Promising Restoration Methods for Degraded Renosterveld Habitats? *Restoration ecology.* 20(1), pp. 18-23.

Hilderbrand, R.H., Watts, A.C. and Randle, A.M. 2005. The Myths of Restoration Ecology. *Ecology and Society,* 10(1): 19.

Hobbs, R. and Harris, J. 2001. Restoration ecology: Repairing the Earth's ecosystems in the new millennium. *Restoration Ecology,* 9(2), pp. 239-246.

Hobbs, R.J. 2012. Environmental Management and Restoration in a Changing Climate. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition*. UK: Wiley-Blackwell, pp.23-29.

Holmes, P. M. 2002. *Renosterveld Restoration on Cultivated Lands at Eerste River, Western Cape: Preliminary Findings of a Pilot Study*, unpublished report to Lafarge Quarries, May 2002.

Holmes, P. M. 2005. Results of a lucerne old-field restoration experiment at the Fynbos-Karoo interface. *South African Journal of Botany,* 71, pp. 326-338.

Holmes, P. M. and Cowling, R. M. 1997. Diversity, composition, and guild structure relationships between soil-stored seed banks and mature vegetation in alien plant-invaded South African fynbos shrublands. *Plant Ecology,* 133, pp. 107-122.

Holmes, P. M. and Richardson, D. M. 1999. Protocols for restoration based on Recruitment Dynamics, Community Structure, and Ecosystem Function: Perspectives from South African Fynbos. *Restoration Ecology,* 7(3), pp. 215-230.

Hufford, K. M. and Mazer, S, J. 2003. Plant ecotypes: genetic differentiation in the age of ecological restoration. *Trends in Ecology and Evolution*, 18(3), pp. 147-155.

Jackson, S.T. and Hobbs, R.J. 2009, Ecological restoration in the light of ecological history. *Science,* 325(5940), pp. 567-569.

Joubert, C. 1991. *History and description of contemporary vegetation, Signal Hill, Cape Town.* MSc thesis, University of Cape Town.

University of Cape Town

Kemper, J., Cowling, R.M., Richardson, D.M., Forsyth, G.G. and McKelly, D.H. 2000. Landscape fragmentation in South Coast Renosterveld, South Africa, in relation to rainfall and topography. *Austral Ecology,* 25, pp. 179-186.

Krauss, S., L. and He, H., T. 2006. Rapid genetic identification of local provenance seed collection zones for ecological restoration and biodiversity conservation. *Journal for Nature Conservation*, 14, pp. 190-199.

Krug, C. B. and R. M. Krug. 2007. Restoration in Old Fields in Renosterveld: A Case Study in a Mediterranean-type Shrubland of South Africa. In: V. A. Cramer and R. J. Hobbs, (eds). *Old fields: Dynamics and restoration of abandoned farmland.* Washington: Island Press, pp. 265-285.

Macmillan, D.C, Harley, D. and Morrison, R. 1998. Cost-effectiveness analysis of woodland ecosystem restoration. *Ecological Economics,* pp. 313-324.

Memiaghe, H. R. 2008. *Old field restoration: vegetation response to soil changes and restoration efforts in Western Cape Lowlands.* MSc thesis. University of Stellenbosch.

Midgely, J. J. and Clayton, P. 1990. Short-term effects on small mammal populations in southern Cape coastal mountain fynbos. *South African Forestry Journal*, 153, pp. 27-30.

Midoko-Iponga, D. 2004. *Renosterveld Restoration: The role of competition, herbivory and other disturbances.* MSc Thesis, University of Stellenbosch.

Mijnsbrugge, K. V., Biscoff, A. and Smith, B. 2010. A question of origin: Where and how to collect seed for ecological restoration. *Basic and Applied Ecology,* 11, pp. 300-311.

Millennium Ecosystem Assessment (MA). 2005. *Ecosystems and Human Well-being: General Synthesis*. Washington: Island Press.

Milton, S. 2010. *Feasibility and benefits of veld rehabilitation following control of invasive Prosopis in the Calvinia area*, report prepared for Working for Water: Namakwa-District Municipality.

Milton, S. J. 2004. Grasses as invasive alien plants in South Africa. *South African Journal of Science,* 100, pp. 69-75.

Milton, S. J., W. R. J. Dean, M. A. du Plessis and W. R. Siegfried. 1994. A Conceptual Model of Arid Rangeland Degradation: The Escalating Cost of Declining Productivity. *BioScience,* 44, pp. 70–76.

Milton, S.J. 2001. Rethinking ecological rehabilitation in arid and winter rainfall regions of southern Africa. *South African Journal of Science,* 97, pp. 47–48.

Milton, S.J., Dean, W.R.J. and Richardson, D.M. 2003. Economic incentives for restoring natural capital in southern African rangelands. *Frontiers in Ecology and the Environment,* 1, pp. 247–254.

Mortlock, W. 2008. Local seed for revegetation: Where will all that seed come from? *Ecological Management and Restoration,* 1(2), pp. 93-101.

Musil, C.F., Milton, S.J. and Davis, G.W. 2005. The threat of alien invasive grasses to lowland Cape Floral diversity: an empirical appraisal of the effectiveness of practical control strategies. *South African Journal of Science,* 101(9/10), pp. 337-344.

National Research Council. 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington: The National Academies Press.

Ntshotsho, P., Reyers, B. and Esler, K.J. 2011. Assessing the evidence base for restoration in South Africa. *Restoration Ecology*, 19(5), pp. 578-586.

Padilla, F.M. and Pugnaire, F.I. 2007. Rooting depth and soil moisture control Mediterranean woody seedling survival during drought. *Functional Ecology*, 21, pp. 489-495.

Perring, M.P., Standish, R.J. and Hobbs, R.J. 2013. Incorporating novelty and novel ecosystems into restoration planning and practice in the 21st century. *Ecological processes*, 2(18), pp. 1-8.

University of Cape Town

Pfadenhauer, J. and Grootjans, A. 1999. Wetland restoration in Central Europe: aims and methods. *Applied Vegetation Science,* 2, pp. 95-106.

Pickett, S. T. A., and P. S. White. 1985. *The Ecology of Natural Disturbance and Patch Dynamics*. San Diego: Academic Press.

Pywell, R. F, Bullock, J., M., Roy, D., B. Warman, L., Walker, K., J. and Rothery, P. 2003. Plant traits as predictors of performance in ecological restoration. *Journal of Applied Ecology*, 40, pp. 65–77.

Rebelo, A.G. 1995. Renosterveld: Conservation and Research In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region.* Proceedings of a symposium. FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp.32-42.

Rees, W., Farley, J., Vesley, E-T. and de Groot, R. 2007. Valuing Natural Capital and the Costs and Benefits of Restoration. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 227-236.

Rey Benayas, J.M., Newton, A.C., Diaz, A. and Bullock, J.M. 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science* 325, pp. 1121-1124.

Roura-Pascual, N., Richardson, D.M., Krug, R.M., Brown, A., Chapman, A.R., Forsyth, G.G., Le Maître, D.C., Robertson, M.P., Stafford, L., Van Wilgen, B.W. Wannenburgh, A. and Wessels, N. 2009. Ecology and management of alien plant invasions in South African fynbos: Accommodating key complexities in objective decision making. *Biological Conservation*, in press.

Society for Ecological Restoration International Science and Policy Working Group (SER). 2004. *The SER International Primer on Ecological Restoration.* Tucson: Society for Ecological Restoration International.

Suding, K. and Leger, E. 2012. Shifting baselines: Dynamics of Evolution and Community Change in a Changing World. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.281-293.

The Economics of Ecosystems and Biodiversity (TEEB). 2010. In Pushpam Kumar (ed) 2010. *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations.* London: Earthscan.

The Economics of Ecosystems and Biodiversity (TEEB). 2011. ten Brink, P., (ed.). 2011. *The Economics of Ecosystems and Biodiversity: National and International Policy Making.* London: Earthscan.

Terblanche, K. 2011. *Meeting conservation goals: passive restoration of peninsula shale renosterveld following the clearance of alien invasive forestry species.* BSc Hons thesis, University of Cape Town.

Tongway, D.J. and Ludwig, J.A. 2012. Planning and Implementing Successful Landscape-scale Restoration. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.30-44.

Turnbull, L.A., Crawley, M.J. and Rees, M. 2000. Are Plant Populations Seed-Limited? A Review of Seed Sowing Experiments. *Oikos,* 88, pp. 225-238.

Turpie, J.K., Marais, C., Blignaut, J.N., 2008. The Working for Water Programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics,* 65, 788-798.

Van Andel, J. and Aronson, J. 2012, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell.

Van Andel, J., Grootjans, A.P. and Aronson, J. 2012. Unifying Concepts. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.9-22.

Van de Koppel, J., van Andel, J. and Biere, A. 2012. Populations: Ecology and Genetics. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.73-86.

University of Cape Town

Van der Merwe J.P.A. and Kellner K. 1999. Soil disturbances and increase in species diversity during rehabilitation of degraded arid rangelands. *Journal arid Environment,* 41, pp. 323–333.

Van Diggelen, R., Hobbs, R.J. and Miko, L. 2012.Landscape Ecology. In: J. van Ande and J. Aronson, (eds). 2012. *Restoration Ecology: the New Frontier, 2nd edition.* UK: Wiley-Blackwell, pp.45-58.

Van Hensbergen, H. J., Botha, S. A. and Forsythe, G. G. 1992. Do small mammals govern vegetation recovery after fire in fynbos? In: Van Wilgen, B.W., Richardson, D.M., Kruger, F..J. and Van Hensbergen, H.J. (eds). *Fire in South African mountain Fynbos: Ecosystem, Community and Species Response at Swartboskloof.* Ecological studies 93. Berlin: Springer-Verlag, pp. 182-202.

Van Rooyen S. 2003. *Factors affecting alien grass invasion into West Coast Renosterveld fragments*. M.Sc. thesis, University of Stellenbosch.

Van Rooyen, A.F., 2000. *Rangeland degradation in the southern Kalahari.* Ph.D. Thesis, University of Natal, Pietermaritzburg.

Walton, B. J. 2006. *Vegetation patterns and dynamics of renosterveld at Agter-Groeneberg Conservancy, Western Cape, South Africa.* MSc thesis. University of Stellenbosch.

Way, M. J. 2003. Collecting seed from non-domesticated plants for long-term conservation. In: R.D. Smith, J.B. Dickie, S.H. Linington, H.W. Prichard and R.J. Probert, (eds). *Seed Conservation: turning science into practice.* Kew: Royal Botanic Gardens, pp. 163-201.

Whisenant, S. G. 1999. *Repairing damaged wildlands*. Cambridge: Cambridge University Press.

Whisenant, S.G. 2002. Terrestrial systems. In: Perrow, M.R. and Davy, A.J. (eds). 2002. *Handbook of Ecological Restoration.* Cambridge: Cambridge University Press, pp. 83-105.

Wiens, J.A., 1989. Spatial scaling in ecology. *Functional Ecology,* 3, pp. 385–397.

Working on Fire. 2013. http://www.workingonfire.org/, accessed June 25, 2013.

Young, M, Hajkowicz, S., Gaddis, E. and de Groot, R. 2007. A Decision-Analysis Framework for Proposal Evaluation Natural Capital. In: J. Aronson, S. Milton and J. Blignaut, (eds). 2007. *Restoring Natural Capital: Science, Business, and Practice.* Washington: Island Press, pp. 237-248.

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Appendix 4.1: Burning: bill of quantities

Table 4.8: Burning: bill of quantities.

BILL OF QUANTITIES: BURNING	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
Working on fire team of 25 for 1 day	day	1	R 1 000.00	R 1 000.00
Vehicle hire (+- R600 per truck), foam, fuel (ignition and transport).	sum			R 3 500.00
Labour				
Drivers, contractor fire fighters and 2x management staff.	sum			R 2 200.00
Subtotal				**R 6 700.00**

Appendix 4.2: Tillage: bill of quantities

Table 4.9: Tillage: bill of quantities.

BILL OF QUANTITIES: TILLAGE	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
2x garden forks	No.	2	R 120.00	R 240.00
Gloves (x4)	No.	4	R 22.00	R 88.00
Raingear (x4)	No.	4	R 123.00	R 492.00
Boots (x2)	No.	2	R 130.00	R 260.00
Labour				
Time to till by hand with garden fork to depth of 200mm.	hrs	164	R 22.50	R 3 690.00
Subtotal				**R 4 770.00**

Appendix 4.3: Herbicide-application: bill of quantities

Table 4.10: Herbicide-application: bill of quantities.

BILL OF QUANTITIES: HERBICIDE	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
Herbicide (5lt of Glyphosate 360)	5lt	1	R 169.30	R 169.30
Syringe	sum			R 10.00
Surgical gloves	sum			R 20.00
Vapour mask refills	No.	1	R 165.00	R 165.00
Rental of 16lt Matabi backpack sprayer (with flat fan TK5 nozzle) and hood	per day	1	R 114.00	R 114.00
Labour				
Time to apply.	No.	8	R 120.00	R 960.00
Subtotal				**R 1 438.30**

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Appendix 4.4: Rodent-exclusion: bill of quantities

Table 4.11: Rodent-exclusion: bill of quantities.

BILL OF QUANTITIES: RODENT CAGES	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
Marker pens	No.	18	R 17.66	R 317.88
Mild steel sheets	No.	129	R 322.21	R 41 564.96
Pop rivets and washers	sum			R 1 405.62
Cutting discs	No.	20	R 11.00	R 220.00
Pegs (mild steel)	kg	125	R 18.24	R 2 280.00
Wire	No.	1	R 28.75	R 28.75
Labour				
Cage construction supervision (Technical officer)	hrs	64	R 120.00	R 7 680.00
To construct (2 pax)	hrs	257	R 22.50	R 5 782.50
To transport (2 pax)	hrs	20	R 22.50	R 450.00
To install (3 pax)				
PW	hrs	16	R 120.00	R 1 920.00
Others (x3)	hrs	48	R 22.50	R 1 080.00
Subtotal				**R 59 729.71**

Appendix 4.5: Seeding: bill of quantities

Appendix 4.5a: Costing as incorporated into costing model

Table 4.12: Seeding: bill of quantities as incorporated into costing model.
As plotted on cost-effectiveness plane.

BILL OF QUANTITIES: SEEDING	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
Cotton collecting bags	sum			R 30.00
Doom foggers	No.	5	R 17.99	R 89.95
Vapour mask	No.	1	R 206.10	R 206.10
Vapour mask refills	No.	1	R 165.00	R 165.00
Labels	No.	1	R 10.00	R 10.00
Boxes	No.	5	R 25.00	R 125.00
20lt buckets for mixing seed	No.	2	R 35.00	R 70.00
5lt containers	No.	64	R 10.57	R 676.48
Rake	No.	1	R 85.00	R 85.00
Broom	No.	1	R 75.00	R 75.00
Sand	m^3	1	R 130.85	R 130.85
Apron (fungicide)	No.	1	R 950.00	R 950.00
Brown paper bags	Pkt 100	1	R 96.48	R 96.48
Labour				
Time spent seed collecting:				

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Zukile	hrs	154	R 65.00	R 10 010.00
PW	hrs	186	R 120.00	R 22 320.00
Volunteers	hrs	75	R 22.50	R 1 687.50
Seed processing				
Insecticiding with Doom fogger (PW)	hrs	15	R 120.00	R 1 800.00
Cleaning seed (SANParks volunteers)	hrs	71	R 22.50	R 1 597.50
Cleaning seed, weighing and dividing (PW)	hrs	124	R 120.00	R 14 880.00
Seed hot water treatment (for *Myrsine africana, Arctopus echinatus* and *Podalyria sericea*) (PW)	hrs	2	R 120.00	R 240.00
Seed smoking				
PW	hrs	16	R 120.00	R 1 920.00
Others (x1)	hrs	16	R 22.50	R 360.00
Sowing and raking				
PW	hrs	16	R 120.00	R 1 920.00
Others (x4)	hrs	64	R 22.50	R 1 440.00
Subtotal				**R 60 884.86**

Appendix 4.5b: Costing alternative 1

Table 4.13: Seeding costing alternative 1.
Experiment-related cost items removed and actual wages retained.

BILL OF QUANTITIES: SEEDING ALT 1	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
Cotton collecting bags	sum			R 30.00
Doom foggers	No.	5	R 17.99	R 89.95
Vapour mask	No.	1	R 206.10	R 206.10
Vapour mask refills	No.	1	R 165.00	R 165.00
Labels	No.	1	R 10.00	R 10.00
Boxes	No.	5	R 25.00	R 125.00
20lt buckets for mixing seed	No.	2	R 35.00	R 70.00
Rake	No.	1	R 85.00	R 85.00
Broom	No.	1	R 75.00	R 75.00
Apron (fungicide)	No.	1	R 950.00	R 950.00
Brown paper bags	Pkt 100	1	R 96.48	R 96.48
Labour				
Time spent seed collecting:				
Zukile	hrs	154	R 65.00	R 10 010.00
PW	hrs	186	R 120.00	R 22 320.00
Volunteers	hrs	75	R 22.50	R 1 687.50
Seed processing				
Insecticiding with Doom fogger (PW)	hrs	15	R 120.00	R 1 800.00
Seed hot water treatment (for *Myrsine africana, Arctopus echinatus* and *Podalyria sericea*) (PW)	hrs	2	R 120.00	R 240.00
Seed smoking	hrs			
PW	hrs	16	R 120.00	R 1 920.00

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Others (x1)	hrs	16	R 22.50	R 360.00
Sowing and raking				
PW	hrs	16	R 120.00	R 1 920.00
Others (x4)	hrs	64	R 22.50	R 1 440.00
Subtotal				**R 43 600.03**

Appendix 4.5c: Costing alternative 2

Table 4.14: Seeding costing alternative 2.
Experiment-related cost items removed and minimum wage rates incorporated.

BILL OF QUANTITIES: SEEDING ALT 2	UNIT	QTY	RATE	AMOUNT
Materials and equipment				
Cotton collecting bags	sum			R 30.00
Doom foggers	No.	5	R 17.99	R 89.95
Vapour mask	No.	1	R 206.10	R 206.10
Vapour mask refills	No.	1	R 165.00	R 165.00
Labels	No.	1	R 10.00	R 10.00
Boxes	No.	5	R 25.00	R 125.00
20lt buckets for mixing seed	No.	2	R 35.00	R 70.00
Rake	No.	1	R 85.00	R 85.00
Broom	No.	1	R 75.00	R 75.00
Apron (fungicide)	No.	1	R 950.00	R 950.00
Brown paper bags	Pkt 100	1	R 96.48	R 96.48
Labour				
Time spent seed collecting:				
Zukile	hrs	154	R 11.38	R 1 752.52
PW	hrs	186	R 11.38	R 2 116.68
Volunteers	hrs	75	R 11.38	R 853.50
Seed processing				
Insecticiding with Doom fogger (PW)	hrs	15	R 11.38	R 170.70
Seed hot water treatment (*for Myrsine africana, Arctopus echinatus* and *Podalyria sericea*) (PW)	hrs	2	R 11.38	R 22.76
Seed smoking	hrs			
PW	hrs	16	R 11.38	R 182.08
Others (x1)	hrs	16	R 11.38	R 182.08
Sowing and raking				
PW	hrs	16	R 11.38	R 182.08
Others (x4)	hrs	64	R 11.38	R 728.32
Subtotal				**R 8 093.25**

5 General conclusion

5.1 Introduction

This study set out to determine intervention effectiveness, in terms of several seed germination criteria, with a view to informing the future drafting of feasible, landscape-scale, ecological restoration strategies for Peninsula Shale Renosterveld. Effectiveness of the 32 interventions (including the control) was measured in terms of seedling density, species richness, seedling height and canopy cover for the 31 seeded species; seedling density, species richness and canopy cover for the existing indigenous species; and, canopy cover for the weed species. An additional aspect of the study set out to determine the viability and germinability of the 31 harvested species, towards assessing each as a key restoration species. Intervention feasibility was also addressed through assessing intervention effectiveness in conjunction with intervention cost per hectare. This final chapter considers the key findings in relation to the original aim: to determine the effectiveness of each of the 32 interventions (including the control) with respect to several seed germination criteria. Thereafter, the chapter looks to the future in terms of the way forward for strategic ecological restoration of Peninsula Shale Renosterveld.

5.2 Key findings

Among the 31 selected species, the traits of germinability and to a lesser extent viability were variable. These findings emphasize the sense in ascertaining seed physiological status prior to embarking on large-scale harvesting and sowing so as to optimise restoration outcomes and circumvent costly failures (Lippitt et al. 1994). The majority of species exhibited moderate to high levels of viability and germinability. As germinability is a good indicator for likely in-field performance, it is of some concern that just under half of the species failed to achieve germinability of 50%. Low germinability may indicate poor seed quality or that the conditions required to stimulate germination were not met. Where the requirements for a taxon are relatively unknown (for example *Arctopus*), further experimentation will be necessary testing different pre-treatments under a range of conditions.

The majority of species occurred in the mid to high key-restoration-species index range. Growth forms, deemed critical for long term ecological functioning (Diaz and Cadibo 2001), were also well-represented. Although the model used here is particular to just 31 selected species occurring in Peninsula Shale Renosterveld, the effect size parameters and corresponding scoring values can easily be adjusted to accommodate other species in this or other ecosystems whilst the eight criteria remain relevant and important to seed collection in all future restoration efforts, demonstrating the value of the proposed model for future use in potentially mitigating some of the high time, cost and risk factors associated with any ecological restoration project (Macmillan et al. 1998, Crookes et al. 2013).

Seeding is imperative in a context such as this, where: the area requiring restoration is located some distance from a source of propagules (Krug and Krug 2007); the area exhibits old field

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characteristics due to an abundance of alien, annual grasses (Euston-Brown 2011) and an existing indigenous species element comprised predominantly of geophytes; and, the seed bank is relatively depleted (Cowan 2013), despite the significantly positive effect the burn had in stimulating existing indigenous species. Without seeding, restoration interventions failed to shift the trajectory of the community from one dominated, and most likely perpetuated, by alien, annual grasses towards a more desirable shrubland state composed of a greater diversity of growth forms. Across all seeded interventions, the sowing of just 31 species had profoundly positive impacts and managed not only to introduce growth forms but to considerably supplement the richness of existing growth forms.

In this study, recruitment of the seeded species in the seeded-only plots confirms the existence of the regeneration niche and points to vegetation recovery in the study area being inhibited by a lack of available seed (Turnbull et al. 2000). This implicates seed dispersal as a limiting factor to community recovery by acting as a barrier inhibiting the return to a more desirable shrubland state. However, since recruitment in these seeded-only plots was the poorest of all 16 seeded interventions with respect to all criteria, it can be concluded that the availability of seed is not the only factor responsible for poor community recovery.

This study draws the conclusion that the presence of alien, annual grasses plays a large role in impeding seedling recruitment of the seeded species. This conclusion is based on the finding that seeding performed the worst in the seeded-only plots and that germination was considerably improved when seeding was implemented in combination with one or more of the other factors such as burning, tillage, herbicide-application and/or rodent-exclusion. This finding is affirmed by the significant positive correlation between the number of factors per intervention and intervention effectiveness for both the seeded and unseeded interventions. The mechanism underpinning these dynamics most likely hinges on the heightened capacity of the multi-factor interventions to reduce the occurrence of alien grasses, increase the number of available colonisation sites and reduce resource competition (Fenner and Thompson 2005). In conclusion, the possible effects of rodent predation and unfavourable soil properties are negligible compared with the effects of dispersal limitation and competition from alien, annual grasses in acting as barriers to community recovery.

Predicted responses for the ecological-response model were largely confirmed by the experimental responses. Of the 19 unexpected responses, eight are beneficial to community attributes whilst 11 are detrimental. These detrimental outcomes include: the significant negative effect burning had on seeded species seedling density and species richness; the non-significant negative effect tillage had on seeded species seedling height; several effects due to rodent-exclusion (significant negative effects on seeded species density and cover and existing indigenous density; non-significant effect on seeded species richness; and, significant promotion of weeds species- these effects, however, were almost certainly due to the exclusion cages, which altered the internal microclimate, and not due to rodent exclusion *per se*); and, the non-significant negative effects seeding had on existing indigenous species density and canopy cover and the non-significant positive effect on weed cover. The adoption of an ecological-response model serves to highlight these outcomes and prompt interrogation of the ecological reasons for anticipated, and in particular unanticipated, outcomes, identifying those areas in need of further specific and detailed experimental work.

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Since ecological restoration is risky (Crookes et al. 2013) and outcomes are largely unpredictable (Clewell and Aronson 2013), at least some of the risk and unpredictability may be managed through implementing interventions with proven effectiveness and predictable costs. Plotting the overall effectiveness for each of the 32 interventions against cost per hectare identified the potentially feasible interventions. On the grounds of cost-effectiveness, preferred interventions were identified for landscape-scale implementation. Of the seeded interventions, BTHS, THS, BHS and HS were preferred whilst of the unseeded interventions, BTH, BH and B were preferred. As budgetary constraints are so often the greatest inhibiting factor (Macmillan et al. 1998), it is reassuring that the correlations between intervention cost per hectare and intervention effectiveness were not significant. This indicates that some relatively effective interventions are still cost-efficient and are potentially feasible restoration options for implementation at scale.

5.3 Future direction

5.3.1 Renosterveld species database

The value of the seed physiological tests and the key restoration species assessment is evident by the many practical benefits which derive from their outcomes and which are central to informing efficient and responsible restoration protocols. Some of these benefits include: targeting likely high-yield species for seed collection; adjusting the quantity of seed for collection based on knowledge of likely germinability; knowing when to initiate further germination-cue experimentation; promoting collections of genetic integrity yet adequate diversity; ensuring source population resilience; promoting seed collection 'ease'; and informing the timing and methodology of restoration implementation on the basis of optimal germination cues. However, some of the seed physiology tests may involve costly equipment, particularly laboratory-based tests, and are time-consuming (for instance species that typically germinate in the second year for example *Myrsine africana* exhibiting highest germinability in the ninth and final month of monitoring in this project) and as a result may not be easily implementable. In response to both the benefits and constraints, establishing a local database as a repository for seed physiological data for renosterveld species would bypass the need for each restoration practitioner to replicate these tests and promote time and cost efficiency and restoration outcomes. The database would need to be accessible to all prospective users (whether interested landowners, citizen-scientists or scientists) and enable both the up-take of information as well as the contribution of newly generated data. To ensure adequate quality control and integration, the database would realistically require management and would benefit from residence within an existing, active, expertise-rich, renosterveld conservation organisation.

5.3.2 Longer-term monitoring

There is considerable value derived from close attention to the early stages of seedling recruitment, including emergence, establishment and survival, yet the value of extending the monitoring period for this restoration experiment is unquestionable. Tipping the scales from an

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undesirable alien, annual grass dominated community to a more desirous shrubland community is central to realigning the community with a desired trajectory and to initiating effective restoration efforts. By the final data collection in January 2013, eight months since sowing, interventions resulted in variable overall performances, ranging from almost ineffectual to very good. Some interventions had created communities that, at that point in time, appeared to have the capacity to suppress the undesirable grass element and develop into communities with a semblance of composition, structure and function. For a more accurate and ultimately more useful measure of intervention effectiveness, long-term monitoring is needed to determine a more meaningful reflection of intervention performance. Updating the experimental ecological-response model with longer-term data would also be necessary to enhance theoretical understanding of community recovery in Peninsula Shale Renosterveld and potentially other renosterveld types and it is advisable that, upfront, projects consider the cost of this longer-term view.

In determining this longer-term view, one would need be mindful of how the small scale of intervention implementation in this study (25 m²) is likely to impact community development over time. Communities established at this small scale, in proximity to or surrounded by dense alien, annual grasses, are likely to experience edge effects (van Rooyen 2003) and are probably less likely to persist than if implemented at a larger spatial scale.

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5.3.3 Alternative approaches

It would be premature to presume, however, that the communities produced by the most effective and promising interventions at this point will continue to win the war on alien, annual grasses. Should over the longer-term, these most effective interventions slip back into grass-dominated states, alternative approaches to restoration will be needed. Two such proposed approaches involve prolonged weed management programmes. The first alternative approach involves, for example burning, tillage, herbicide-application and seeding (yet holding back the indigenous grass species), followed by a long-term, post-emergence, grass-specific herbicide-application (such as Agil by Makhteshim-Agan SA). Only once the alien grasses are eradicated, which may even be years later, and by which time other components of the community are likely to be well-established, should the introduction of the desirable grass seed take place. The second alternative approach, more in alignment with recruitment following fire, involves the implementation of, for example, burning, tillage and a long-term herbicide-application prior to the reintroduction of all seed (including indigenous grass seed), which is sown only once the alien grass component is eradicated (P Holmes personal communication July 27, 2013). In both cases the initial burning and tillage interventions would function to stimulate a flush of the majority of the alien grass seed bank, but not all of it. Therefore, assuming the use of a post-emergent, grass-specific herbicide, both of these alternatives, as would the 31 experimental interventions, would remain to some extent vulnerable to recruitment of the remaining alien grass seed bank should another fire or disturbance occur prior seed-bank depletion and as such would likely require continued post-fire/disturbance weed management. These alternative approaches, the second more so than the first, would have limited practicality on steep slopes and fine-textured soils vulnerable to erosion during heavy rain. Regular monitoring for

erosion (Britton and Jackalman 1995) as well as sensitivity of indigenous vegetation to herbicide (Krug and Krug 2007) should take place regularly and these potential effects mitigated where necessary.

5.3.4 Towards restoration strategies

The way forward for ecological restoration in Peninsula Shale Renosterveld is clear and promising. Baseline data for the ecosystem (contributed amongst others by Michell (1922), Britton and Jackelman (1995), Joubert (1991), Rebelo et al. (2006), Euston-Brown (2011), Terblanche (2011), Dreyer (2012) and Cowan (2013)) has been instrumental in the design of this experiment which has centred around determining an ecological-response model for the restoration of the ecosystem. Further to determining a response model, the effectiveness of each of the 31 restoration interventions has been evaluated, the cost per hectare calculated and the potentially feasible interventions and the preferred (i.e. cost-effective) interventions identified, placing stakeholders in a position to proceed with the formulation of restoration options. The next theoretical step, according to the strategic restoration framework proposed by Hobbs and Harris (2001), would be to draft restoration options jointly informed by the response model and by project goals, themselves compiled on the basis of response model input (Figure 5.1). A restoration option may thereafter be selected on the grounds of being appropriate to satisfy project goals, and, for being appropriate to the project and operational budget with respect to implementation, monitoring, evaluation and operational management. However, to raise the confidence-level of large-scale restoration decision-making, the considerably more prudent approach would be to suspend the stages of drafting goals and tabling restoration options, and instead, focus on obtaining longer-term data towards drafting a more accurate and meaningful ecological-response model. In fynbos and other fire-adapted shrublands, a five to 10 year period is an appropriate scale in which implement a second fire and thereafter to gauge a more accurate measure of vegetation recovery and longer-term attributes (Holmes and Richardson 1999). Once such longer-term monitoring data is fed back into a revised, verified and more robust ecological-response model for Peninsula Shale Renosterveld, the momentum for implementing the remaining steps of the strategic restoration framework may be rebuilt and the cyclical process set in motion. Restoration project planning requires a long-term vision with respect to management, monitoring and budgets to ultimately meet these objectives. As time is limited and there is a degree of urgency to arrest degradation and initiate ecological restoration, parallel approaches to larger-scale intervention implementation, where interventions take an experimental design, may be a good way to fast-track knowledge production and inform future restoration efforts.

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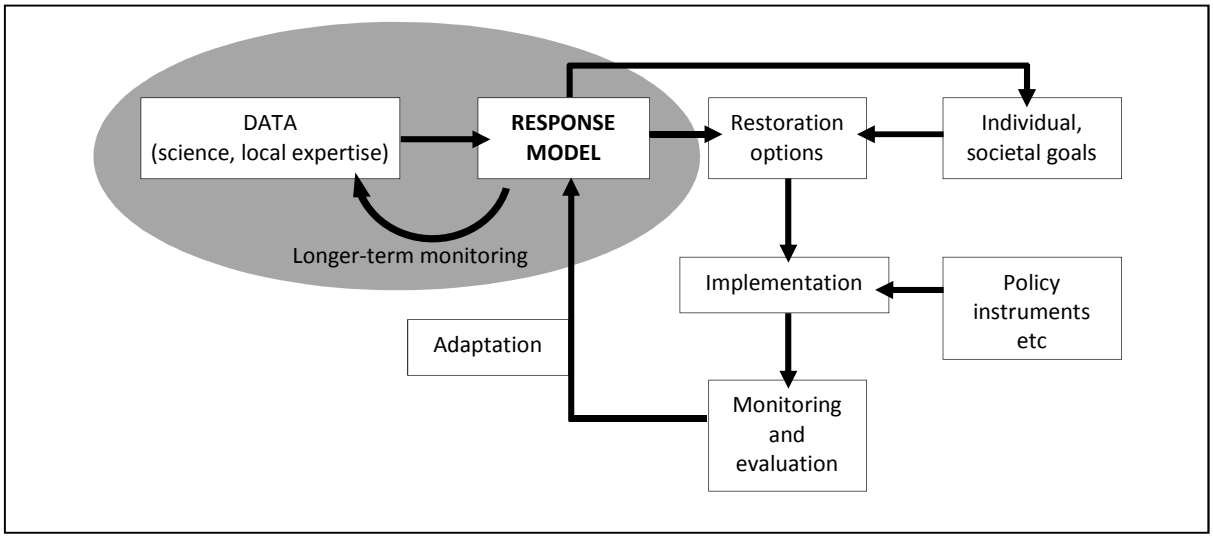

Figure 5.1: Conceptual framework for strategic ecological restoration efforts (adapted from Hobbs and Harris 2001) in relation to this and previous studies. Steps covered and addressed by previous studies and this study include the generation of data and formulation of an experimental ecological-response model. Longer-term monitoring is proposed to increase the accuracy of the response model prior to progressing to the remaining steps.

5.3.5 Concluding thoughts

The conservation status of Cape lowland renosterveld is dire (von Hase et al. 2003) and has been recognised as such for decades yet few inroads have been made into alleviating the critical conservation status of these ecosystems. Peninsula Shale Renosterveld represents just one of numerous lowland renosterveld types that are Critically Endangered, compromised to the point of falling short of their conservation targets and declared 'totally irreplaceable'. The findings of this study indicate a promising and rewarding way forward for ecological restoration of Peninsula Shale Renosterveld and in addition speak to a far broader set of biodiversity conservation stakeholders, those concerned with the conservation strategy of the CFR. This is however not to say that the way forward for ecological restoration in renosterveld will be without very real challenges. On the contrary, challenges derived from outcome unpredictability, risks and financial constraints are anticipated. Yet, this study has managed to generate successful seed-based experimental restoration outcomes for Peninsula Shale Renosterveld. These findings have the potential to expand an already-growing knowledge base and seek to propel forthcoming study and implementation of ecological restoration of renosterveld into a future where the remnants of these valuable ecosystems are buffered, enlarged, enriched and linked to healthier prospects.

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5.4 References

Britton, P. and Jackelman, J. 1995. Biological Management of Renosterveld: Some questions and lessons from Signal Hill and Devil's Peak. In: A.B. Low and F. E. Jones, (eds). 1995. *The sustainable use and management of Renosterveld remnants in Cape Floristic Region.* Proceedings of a symposium. FCC Report 1995/4. Flora Conservation Committee, Botanical Society of South Africa, Kirstenbosch, Cape Town, pp.58-62.

Clewell, A.F and Aronson, J. 2013. *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession, second edition.* Washington: Island Press.

Cowan, O. S. 2013. *The Peninsula Shale Renosterveld of Devil's Peak: Phytosociology, System Drivers and Restoration Potential.* MSc thesis. University of Cape Town.

Crookes, D.J., Blignaut, J.N., de Wit, M.P., Esler, K.J., le Maître, D.C., Milton, S.J., Mitchell, S.A., Cloete, J., de Abreu, P., Fourie (nee Vlok), H., Gull, K., Marx, D., Mugido, W., Ndhlovu, T., Nowell, M., Pauw, M. and Rebelo, A. 2013. System dynamic modelling to assess economic viability and risk trade-offs for ecological restoration in South Africa. *Journal of Environmental Management,* 120, pp. 138-147.

Diaz, S. and Cadibo, M. 2001. Vive la différence: plant functional diversity matters to ecosystem processes. *Elsevier Science*, 16(11), pp. 646-655.

Dreyer, R. 2012. *Ecological Functioning and Restoration potential of the Peninsula Shale Renosterveld of Devil's Peak: Understanding the small mammal community*. BSc Hons thesis, University of Cape Town.

Euston-Brown, D. 2011. *Description of natural vegetation on Groote Schuur Estate, Cape Town.* University of Cape Town, unpublished report.

Fenner, M. and Thompson, K. 2005. *The Ecology of Seeds*. Cambridge: Cambridge University Press.

Hobbs, R. and Harris, J. 2001. Restoration ecology: Repairing the Earth's ecosystems in the new millennium. *Restoration Ecology*, 9 (2), pp. 239-246.

Holmes, P. M. and Richardson, D. M. 1999. Protocols for restoration based on Recruitment Dynamics, Community Structure, and Ecosystem Function: Perspectives from South African Fynbos. *Restoration Ecology,* 7(3), pp. 215-230.

Joubert, C. 1991. *History and description of contemporary vegetation, Signal Hill, Cape Town*. MSc thesis, University of Cape Town.

Krug, C. B. and R. M. Krug. 2007. Restoration in Old Fields in Renosterveld: A Case Study in a Mediterranean-type Shrubland of South Africa. In: V. A. Cramer and R. J. Hobbs, (eds). *Old fields: Dynamics and restoration of abandoned farmland.* Washington: Island Press, pp. 265-285.

Lippitt, L., Fidelibus, M. W. and Bainbridge, D. A. 1994. Native Seed Collection, Processing, and Storage for Revegetation Projects in the Western United States. *Restoration Ecology*. 2(2), pp. 120-131.

Macmillan, D.C, Harley, D. and Morrison, R. 1998. Cost-effectiveness analysis of woodland ecosystem restoration. *Ecological Economics*, pp. 313-324.

Mitchell, M.R. 1922. Some observations on the effects of bush fire on the vegetation of Signal Hill. *Transactions of the Royal Society of South Africa,* 10(4), pp. 213-235.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. and Rutherford, M, C. 2006. Fynbos Biome. In: L. Mucina, L. and M.C. Rutherford, (eds). *The Vegetation of South Africa, Lesotho and Swaziland.* Pretoria: South African National Biodiversity Institute.

Terblanche, K. 2011. *Meeting conservation goals: passive restoration of peninsula shale renosterveld following the clearance of alien invasive forestry species.* BSc Hons thesis, University of Cape Town.

Turnbull, L.A., Crawley, M.J. and Rees, M. 2000. Are Plant Populations Seed-Limited? A Review of Seed Sowing Experiments. *Oikos,* 88, pp. 225-238.

Van Rooyen S. 2003. *Factors affecting alien grass invasion into West Coast Renosterveld fragments*. M.Sc. thesis, University of Stellenbosch.

Von Hase, A., M. Rouget, K. Maze, and N. A. Helme. 2003. *A fine-scale conservation plan for Cape Lowlands Renosterveld.* Full Report. Cape Town: Conservation Unit. Botanical Society of SA.

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