A compositional, functional, and structural assessment of a 20 year post-mine restoration chronosequence.

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Abstract

The extraction of minerals has the potential to bring about drastic transformations to topography, soil physical and chemical properties, hydrology, and biological assemblages. The science of restoration ecology aims to facilitate the reassembly of plant communities in these highly altered environments to resemble a target or reference state in a timeframe that is acceptable to stakeholders and policy makers. The outcomes of restoration practices are the product of interactions between environmental drivers and management intervention techniques through time, yet the long term effects of these interactions have rarely be quantified and may provide valuable implications for leading-practice in ecological restoration. This thesis examines a 19 year-old, post-mining Banksia woodland restoration chronosequence in order to further current understanding of how ecosystems develop in restored sites, in terms of three major ecosystem attributes; 1) plant community composition (species richness and abundance); 2) ecosystem function, and; 3) vegetation structure.

The development of plant community composition through time, and the effects thereon of environmental drivers (rainfall regimes during the establishment phase, site aspect, slope) and management intervention techniques (substrate ripping and properties of reconstruction materials) were assessed in terms of four commonly employed restoration criteria; species richness, plant density, vegetation cover, and similarity to reference sites. Irrespective of environmental drivers or management intervention techniques, vegetation cover increased through time, while plant density and species richness declined. Compositional similarity to reference communities remained relatively unchanged. Within the confines of these trends, rainfall and ripping treatments interacted to significantly affect restoration criteria; species richness and plant density were greatest when rainfall in the first winter immediately following site restoration was low, and then followed by a high summer rainfall. The most effective ripping depth was dependent on rainfall, with deep-ripped sites performing best when rainfall was high, and shallow ripped sites performing best under low-mean rainfall conditions.

The development of hydrological function through time was assessed and the effects of substrate ripping treatments and association with vegetation cover quantified. Measurements of saturated hydraulic conductivity (infiltration rate) and penetration resistance were recorded in restored sites aged 0, 3, 7 and 15 years, along with restored
sites aged 0 and 15 years that had been deep-ripped. Classified, remotely-sensed aerial imagery was used to measure vegetation cover within a given radius of sampling points, in order to determine the effect of vegetation on hydrological properties. Infiltration rate was only mildly affected by time, but was significantly increased by deep ripping. Penetration resistance at the surface was generally unaffected by time or treatment, but was significantly lower at depth under deep-ripping. Penetration Resistance in shallow-ripped sites was slower to restore, but was equal to deep-ripped sites by ~ 10 years. Vegetation cover was associated with a significant yet mild increase in infiltration rate and penetration resistance at the surface, but did not affect penetration resistance at depth. The presence of proteoid root mats was found to significantly decrease infiltration rate, and may have confounded vegetation – infiltration rate relationships.

In addition, the development of vegetation structure through time in restored Banksia woodlands was assessed using classified, remotely-sensed aerial imagery in order to identify predictable trends that may be used to establish benchmarks of restoration success. Pattern development was then related to changes in hydrological function using hydrological data collected from the sites in the previous chapter. Selected landscape metrics were used to characterize changes in the spatial attributes of vegetation pattern in time, and in response to ripping treatments. All metrics were significantly correlated with time. Total cover, proximity index, and weighted mean patch size were all generally strong predictors of hydrological function, while nearest neighbour tended to be a weak predictor.

The results of the studies presented in this thesis provide important and new insights into the way in which restored systems develop through time, and in response to interacting environmental drivers and management intervention techniques. Understanding the long term effects of these interactions is critical for guiding leading-practice ecological restoration in the future.
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Restoration ecology in a post-mining context is primarily concerned with ecosystem reconstruction; more so than almost any other disturbance event, the extraction of minerals has the potential to bring about drastic transformations to topography (Rokich et al. 2001), soil physical and chemical properties (Chong and Cowsert 1997), hydrology (Archer et al. 2002, Shukla et al. 2004), and biological assemblages (Herath et al. 2008). The field of restoration ecology, however, is still in its infancy (Zedler 2007), and the long term efficacy of restoration treatments remain unclear. The evaluation of restoration is critical to the future of ecological restoration (Palmer et al. 2005), and the study of restoration chronosequences facilitates evaluation of large temporal ranges in a time frame relevant to research objectives. However, whilst chronosequences have long been employed in successional studies, they have been less commonly used for assessing restoration (Anderson et al. 2007). The field of Restoration ecology and the recording and reporting of restoration practices have improved markedly in recent years (Suding. 2011), providing a cumulative collection of data and restored site chronosequences that can now be used to further our understanding of restoration.

Success in restoring disturbed landscapes is largely dependent on the degree of disturbance and the ecological complexities of the system. Young landscapes, typically characterized by relatively fertile soils, and common and widespread species with good dispersal mechanisms (Hopper 2009), have been successfully restored in as little as 30 years with minimal intervention (Prach and Hobbs 2008, Prach et al. 2009, Tropek et al. 2010). Reassembling plant communities that resemble natural analogues in old, infertile, and biodiverse landscapes can be substantially more challenging. These systems commonly exhibit high levels of endemism and local rarity, a reduced capacity for seed dispersal, specialist resource acquisition strategies, and substrate bioengineering processes that have evolved over tens of thousands of years (Hopper 2009). Restoration of these systems may require significant management intervention.
and theoretically take hundreds of years to reach a natural analogue state (Prach and Hobbs 2008).

One such example of these ancient, highly evolved systems is Mediterranean Banksia woodlands of south-west Australia. Prior to human settlement, Banksia woodlands occupied the majority of the Swan Coastal Plain in south-west Australia. The woodlands occur on deep, well-drained sandy soils that in parts exceed 800,000 years old (Hopper and Gioia 2004), and are characterized by extreme nutrient impoverishment and low water-holding capacities (Rokich et al. 2001). Combined with a rainfall regime that imposes seasonal drought for up to four months of the year, these conditions have shaped the evolution of more than 2000 species (Hopper and Gioia 2004). Today, more than 80% of the original woodlands have been cleared for urban development and the woodlands are classified as a threatened ecological community (Hopper and Burbidge 1989). Several sand-mining operations are also present throughout the Swan Coastal Plain, providing an opportunity for restoration. Indeed, as mining activities in these areas are completed, government regulatory standards stipulate that the land be restored to resemble the natural Banksia woodland communities that existed previously, presenting a host of issues to be addressed by restoration practitioners and resource developers.

Effects of mining

Mining disturbances typically create entirely new surfaces and substrates—many of which are required to be restored. Clearing of the original vegetation removes shade and organic materials, and increases light, wind, and temperature (Hungerford and Babbitt 1987, Sork 1987), all of which may contribute to increased water stress in water-limiting environments. In Banksia woodland systems, alterations to soil physical properties below the surface are believed to be the most significant obstruction to restoration success (Enright and Lamont 1992, Rokich et al. 2001). Significant increases to soil impedance in restored soils commonly result from mechanical compaction, natural settling processes following substrate disturbance (Kozlowski 1999, Archer et al. 2002), or the creation of new landscapes in which the substrate is compressed (Leroueil and Vaughan 1990). Soil impedance has been identified as a major driver of plant mortality in restored sites (Burrows 1986, Enright and Lamont 1992, Rokich et al. 2001, Maestre et al. 2003). The extent to which a soil may become compressed is
determined by the soil’s physical and chemical properties, including texture, pH, cation-exchange capacity, clay particle thickness, organic matter content, chemical composition and soil water content, all of which affect the strength of the cohesive forces between soil particles (Assouline et al. 1997; Kozlowski 1999). In particular, the water content of the soil is thought to enhance the mobility of fine particles whereby the spaces between larger particles are filled by fine particles, increasing bulk density and compaction (Mullins et al. 1987). Increased soil impedance increases the resistance roots must overcome in order to move through a substrate, severely restricting plant access to nutrient and water resources (Kozlowski 1999). Given that even in some undisturbed systems natural rainfall events have been found to support only 6 establishment events in 97 years (Harrington 1991), substrate alterations in post-mine landscapes that further reduce plant available water could be catastrophic. Reducing compaction is therefore critical to restoration success in Banksia woodland systems.

**Restoring post-mine soils**

Remediation of soil impedance may be attempted via the addition of substrate amendments such as organic matter (Benigno et al. 2012b), but is more commonly carried out by ripping the soil profile using agricultural style tynes or ploughs (Chong and Cowsert 1997, Benigno et al. 2012a). The effectiveness of ripping as a technique for reducing soil impedance in the long term however, has been questioned; Rokich et al (2001) found that ripped soils returned to pre-ripped levels of soil compaction within 6 years, while Chong and Cowsert (1997) noted that the benefits of deep ripping dissipated after 3 years. Despite this, ripping continues to be widely employed, and the long term effects and interactions with developing plant communities have not yet been quantified.

Following initial colonization, remediation of soil physical properties may be continued by plant root action, soil microbial activity, and macro fauna (Greenwood and McKenzie 2001). These plant-soil interactions increase soil pore space and create channels of preferential flow, generating facilitative feedbacks in the form of enhanced infiltration rates and reduced bulk density (Shukla et al. 2004), which may in turn, play a large role in determining the spatial arrangement of plants in a landscape. Studies on these interactions however, are dominated by systems characterized by a higher proportion of small particles than are found in the virtually pure sand substrates.
inhabited by Banksia woodlands, and further studies are required to assess if these responses hold true.

**Facilitation, competition, and vegetation pattern formation**

The spatial arrangement of plants in a landscape is the product of numerous interacting factors; seed dispersal patterns, resource availability, herbivory, and allelopathy and other biotic/ abiotic factors may all contribute to determine which areas will be occupied by plant life (Moles and Westoby 2004). In climates ranging from Mediterranean to arid however, the availability of water is widely recognized as the key cause of germination and establishment failure (Herrera 1992, Moles and Westoby 2004, Padilla and Pugnaire 2007, López et al. 2009) and hence the spatial organisation of plant communities. Facilitative processes that enhance water-availability in water-limited systems have received increased attention in recent times (Bruno et al. 2003), particularly in arid and semi-arid systems (Facelli and Brock 2000, Puigdefábregas 2005). In these resource-limited systems, facilitative interactions occur when individuals benefit from their proximity to other plants.. Commonly referred to as “nurse plants” these often older, larger, and/or more robust species, provide a radiation and wind shield by which evaporative pressures beneath the canopy are lessened (Breshears et al. 1998). Additionally, the surface roughness associated with the presence of plants has been shown to capture nutrients and sediments transported across or eroded from inter-patches, leading to the development of what have been coined, “islands of fertility” (Ludwig and Tongway 1995). The concentration of resources deposited around vegetated patches in conjunction with organic litter from the vegetation itself results in higher levels of microbial activity, improving soil aggregate formation, hydraulic characteristics (particularly infiltration rates), and fertility (Archer et al. 2002, Shukla et al. 2004, Puigdefábregas 2005). These processes in turn generate a positive feedback mechanism whereby the concentration of resources captured by vegetated patches facilitates the continued growth of vegetation within the patch and subsequently enhances the capacity of the patch to capture more resources in the future (Facelli and Brock 2000, Puigdefábregas 2005). Beyond the canopy extent, competitive interactions with the roots from neighbouring patches predominate, restricting the establishment of new individuals and creating vegetation gaps or interpatches (Borgogno et al. 2009). These areas, devoid of significant vegetation and exposed to
rain drop strikes, may develop physical or biological crusts which promote rainfall runoff (Rietkerk and Van de Koppel 2008) and provide clear ways for wind-borne organic material to pass through (Bastin et al. 2002).

At the landscape scale, the interaction of heterogeneous infiltration characteristics associated with run-on and run-off zones has been found to generate distinct, biphasic vegetation patterns (Klausmeier 1999). On hillslopes, these may take the form of regular patterns of patches and gaps such as spots, bands, and tiger stripes (Figure 1), and are largely determined by the slope gradient and annual rainfall (Valentin et al. 1999). On flat ground, small, heterogeneous, topographical variations may produce large variations in plant density and subsequently generate irregular patterns in vegetation (Klausmeier 1999). Crucially, the infiltration rate of the soil must be low enough such that overland flow can occur (HilleRisLambers et al. 2001). For example, sloped areas characterised by uniformly deposited Aeolian sands on the Niger Plateau are devoid of regular pattern (Valentin et al. 1999). Hence, the processes driving vegetation pattern in topographically variable Banksia woodlands on sand substrates are likely to be complex.

Figure 1: Examples of vegetation pattern observed in water-limiting environments. From left to right; Somalia, SW Niger, and Zambia (Borgogno et al. 2009).

Vegetation pattern and Ecosystem function

The role played by vegetation in resource acquisition and local hydrology is indicative of a close association between vegetation pattern and ecosystem function. The definition of ecosystem function commonly refers to the capacity of a landscape to capture and retain sufficient resources such that the system is self-sustaining (Bastin et
al. 2002). Indeed, this relationship forms the foundation of Landscape Functional Analysis (LFA); a well-established procedure for monitoring rangelands and restored landscapes (Tongway and Hindley 2004). The relative sizes of vegetation patches and interpatches however, are a product of resource availability, whereby landscapes with more available resources can support larger vegetation patches, and increasing resource availability will eventually result in adjacent patches coalescing (Figure 2). The size, shape, orientation and distribution of vegetation patches have a significant effect on a landscape’s capacity to retain resources (Bastin et al. 2002), whereby many small patches of vegetation will capture and contain a greater amount of resources than the same area of vegetation arranged as a few large patches. Likewise, larger, closely packed vegetation patches will capture more resources than a site with smaller, sparsely distributed patches (Bastin et al. 2002). Whilst the relationship between vegetation pattern and ecosystem function at the landscape scale has been well-established in arid and semi-arid systems, it is less well understood in Mediterranean Banksia woodland systems where the relationship is likely to be confounded by highly porous, sandy substrates, and increased annual rainfall (Thompson et al. 2010).

Figure 2: Increasing resource availability (i.e. rainfall) causes vegetation (dark) patches to increase and coalesce, while gaps (light) decrease. Eventually, the matrix reverses such that vegetation is the dominant land cover. Image adapted from (D’Odorico et al. 2006).
Vegetation pattern analysis

The identification of large scale vegetation patterns and their association with ecosystem functional characteristics has coincided with the advancement and increased availability of aerial imagery. Indeed, prior to the availability of aerial vegetation maps, ecologists were largely ignorant to the presence of vegetation patterns, such as bands and spots, and to the landscape functional processes that drive pattern formation. Hence, understanding pattern at the landscape scale has in turn facilitated our increased understanding of function at the landscape scale, and is thus an important tool for understanding the development of restored landscapes. To date, this has rarely been done.

The recognition of vegetation pattern has been enhanced by the development of remote sensing and analysis tools, of which spectral analysis has played a dominant role (Kerr and Ostrovsky 2003). A commonly used vegetation index for distinguishing vegetation from other land cover types is the Normalized Difference Vegetation Index (NDVI). NDVI is a measure of “greenness” and is based on the knowledge that chlorophyll in leaves absorbs solar radiation from the photosynthetically active radiation (PAR) portion of the EM spectrum (400 – 700 nm; Figure 3) during the process of photosynthesis (Tucker 1979, Kerr and Ostrovsky 2003). Wavelengths of radiation beyond this range, in particular, infrared, are not sufficient to be utilized in the synthesis of organic molecules and are reflected/scattered by the leaf as a means of preventing heat stress (Dobrowski et al. 2005). Conversely, land cover types such as bare soil or water bodies, absorb relatively larger amounts of near infrared radiation (Nemani and Running 1997). Hence, by examining the relative absorption and reflectance of spectra collected by remote sensing instruments it is possible to discriminate land cover types (e.g. vegetation) from surrounding materials (e.g. soils).
Figure 3: Materials with significantly different chemical and structural characteristics emit significantly different spectral signatures (available at: http://www.cps-amu.org/sf/notes/m1r-1-8.htm)

A host of metrics used to quantify and describe the spatial distribution of landscape elements have evolved and are widely used tools in the field of landscape ecology. These may be broadly divided into textural (pixel-based) metrics and structural (patch-based) metrics. Pixel-based metrics quantify the intermixing of different-valued pixels within an image, while patch-based metrics aim to interpret structural characteristics within an image, such as the size and shape of a feature (Herold et al. 2003). Of the hundreds of metrics available, a large degree of redundancy is present (Cushman et al. 2008) and selection of appropriate landscape metrics requires a thorough understanding of the limitations of the metric, particularly in relation to the scale and extent of the area under examination (Uuemaa et al. 2009). For example, problems arise when attempting to compare pixel-based metrics with different spatial resolution, or patch-based metrics in sites of different extent (Leitão et al. 2006).

Measuring restoration success

Restoration projects rarely include explicitly defined endpoints or formulas for assessing success or failure (Zedler 2007) and there has been much debate over which criteria should be assessed to determine restoration outcomes (Palmer et al. 2005). In 2004, The Society for Ecological Restoration International (SER) published a primer for restoration, listing nine ecological attributes of restored systems (SER. 2004) that can be broadly reclassified into three major ecosystem attributes; 1) composition (species richness and abundance); 2) ecosystem function, and ; 3)vegetation structure. It has
been suggested that the measurement of all three of these criteria can reflect the recovery trajectory of restored systems, yet in an evaluation of published articles relating to the measurement of restoration success, Ruiz-Jaen et al (2005) found that whilst the majority of articles assessed at least one of the above criteria, assessment of all three was rare. Furthermore, the monitoring phase of most restoration projects rarely lasts more than five years (Ruiz-Jaen and Mitchell Aide 2005), despite the recovery of some ecological processes taking decades (Hopper 2009).

Restoration criteria are commonly assessed in relation to reference sites, but what constitutes an appropriate reference sites is often not clear (Palmer et al. 1997). In many instances the aim is to return a site to a close approximation of its condition prior to being disturbed. However the original condition is not always known and major disturbances such as mining may result in a new landscape to be restored that bears no topographical resemblance to the original site. Hence, in these cases it may be more appropriate to select a reference site that more closely resembles the topographical and hydrological characteristics of the reconstructed site. Understanding natural analogues and the physical, chemical and biological changes brought about by the mining and restoration process will help guide restoration practitioners in selecting appropriate reference targets.

**Thesis Aims**

Restoration practitioners are required to restore degraded post-mine landscapes, and in order to do this, clear targets based on appropriate natural analogues must be established, and a thorough understanding of the system-specific interactions between key environmental drivers and management intervention techniques through time, developed. This thesis aims to;

1. Understand the effects of environmental drivers, management intervention techniques, and their interactions through time, on plant community composition.
2. Assess the effects of environmental drivers, management intervention techniques, and plant-soil interactions through time, on the restoration of hydrological function.
3. Examine the development of vegetation spatial pattern in restored sites through time, and the effects of management intervention techniques
4. Identify the relationship between vegetation spatial pattern and hydrological function in restored Banksia woodland systems

Thesis Outline

This thesis examines a 19 year-old, post-mining Banksia woodland restoration chronosequence in order to further current understanding of how ecosystem trajectories develop in restored sites in terms of three major ecosystem attributes; 1) plant community composition (species richness and abundance), 2) ecosystem function, and 3) vegetation structure. These three subjects broadly form the structure of this thesis, which is presented as a compilation of manuscripts that are proposed for publication.

Chapter Two examines how environmental drivers (rainfall regimes during the establishment phase, site aspect, slope) and management intervention techniques (substrate ripping and properties of reconstruction materials) effect the development of restoration across a 19 restoration chronosequence in comparison to different natural analogue sites (restoration targets). Development is assessed in terms of four commonly employed restoration criteria; vegetation cover, plant density, species richness, and similarity to reference sites. Statistical modelling is employed to quantify and illustrate the effects of these drivers on restoration criteria.

Chapter Three investigates the development of hydrological function in terms of saturated hydraulic conductivity (infiltration rate) and soil impedance across a restoration chronosequence. Restored sites aged 0, 3, 7 and 15 years, along with restored sites aged 0 and 15 years that had been deep-ripped were assessed, with a focus on the effect of vegetation cover (bioremediation), ripping treatments (management intervention), and the overall effect of time on hydrological properties. The development of hydrological function in restored sites through time is measured against reference targets to determine the factors that most affect the restoration of hydrological function. Classified remotely-sensed aerial imagery is employed in order to measure vegetation cover. Statistical modelling is used to quantify and illustrate the effects of vegetation cover and ripping treatments on hydrological function.
Chapter Four explores the development of vegetation spatial pattern at the landscape scale in restored Banksia woodlands in order to identify predictable trends that may be used to establish benchmarks of restoration success, and identify sites that are underperforming. The development of pattern is then related to changes in hydrological function using hydrological data collected in Chapter Two, enabling pattern to be employed as a rapid indicator of functional development in restored sites. The program, “Fragstats 4.0” is used to assess classified remotely-sensed aerial imagery in order to characterize pattern according to vegetation pattern metrics.

Chapter Five integrates and discusses the findings from the previous three experimental chapters. In particular, it examines the common themes of time (site age), and ripping treatments, and their effects on species composition, hydrological function, and vegetation structure, and summarizes these findings in order to make recommendations to improve restoration outcomes.
CHAPTER TWO

Do natural environmental drivers and restoration intervention methods influence trajectories of restoration programs?

INTRODUCTION

Post-mined landscapes, in almost all forms, are substantially altered from their original, pre-mined physical, chemical, hydrological and biological states. Restoration practitioners are often confronted with significantly altered hydrological regimes (Shukla et al. 2004) in sites devoid of the facilitative effects of nurse plants (Brooker et al. 2008), exposed to higher evaporative pressures via sun and wind exposure (Hungerford and Babbitt 1987, Sork 1987), and characterized by compacted substrates (Rokich et al. 2001). Particularly in drought affected and arid to semi-arid lands, plant germination, establishment and survival is inherently driven by plant-available water, and consequently natural recruitment events are highly dependent on rainfall (Yates and Hobbs 1997, Chesson et al. 2004). Indeed, modelling perennial shrub establishment in sandy soils in a semi-arid system highlighted that episodically varying soil moisture conditions had facilitated only six establishment events in 97 years (Harrington 1991).

For southern hemisphere ecosystems, the location of this study, some authors suggest that higher and more reliable rainfall conditions associated with El Nino Southern Oscillation (ENSO) models may be used to guide practitioners as to the most suitable years to conduct restoration activities (Holmgren and Scheffer 2001, Sitters et al. 2012). Outside the window of opportunity that ENSO events could provide, plant-available water may be increased at restoration sites through the implementation of management intervention techniques including the manipulation of site characteristics that facilitate increased water infiltration (Chong and Cowsert 1997), retention (Benigno et al. 2012b), and access (Rokich et al. 2001), and/or reduce evaporative water losses (Hungerford and Babbitt 1987). These techniques have been widely studied at the establishment phase, yet the long term efficacy and interactions with environmental drivers are less well-understood.

Whilst direct monitoring of a site through time is the most reliable and informative approach to understanding the development of the system (Walker and del
Moral 2009), examination of a restoration chronosequence may also provide powerful insights into system responses to restoration practices (Anderson et al. 2007). Studies of ecosystem succession have often utilized landscape chronosequences (in a space-for-time substitution) to assess change, yet restoration, often defined as the “manipulation of succession” and its effectiveness, have rarely been studied via restoration chronosequence (Walker et al. 2007). This is largely due to an absence of accurate recording of site reconstruction details, restoration practices implemented, and consistent monitoring techniques allowing for restoration to be assessed or attributed to any given factor. In more recent times, the importance placed on restoration outcomes by regulators and stakeholders, and raised expectations on the type and quality of restored landscapes by the community, has resulted in more detailed and accurate reporting, facilitating a more robust study of restoration chronosequences.

The aim of this study was to determine the comparative importance of natural drivers and management intervention techniques in determining the trajectory of plant community development in restored systems through time. We assessed compositional changes across a 19 year, post-mining, restoration chronosequence of Banksia woodland located in the biodiversity hotspot, of south-west Australia according to four criteria commonly employed as indicators of restoration success; vegetation cover, plant density, species richness, and similarity to natural analogues. Statistical modelling was conducted to understand the significance of variables that determine restoration outcomes.

We hypothesized that community development will lead to increasing cover and decreasing plant density (due to competitive interactions), and an associated decline in species richness and similarity. Banksia woodlands and shrublands in southwest Australia occur in Mediterranean systems characterised by summers of low rainfall and high vapour deficits which have been identified as a major determinant of plant mortality in restored sites (Enright and Lamont 1992). We hypothesized that restoration success (high species richness, plant cover, plant density and similarity with natural analogue reference sites) would be higher in sites with high rainfall, and in particular, high summer rainfall, in the first two years following restoration. Additionally, we expected that sites that experience relatively lower evaporative pressures would perform better than sites exposed to higher evaporative demand. We predicted that sites with reduced soil compaction, a critical issue associated with reduced root growth in restored Banksia woodlands (Rokich et al. 2001, Benigno et al. 2012a), would have greater
restoration success. Species composition in undisturbed Banksia woodlands varies with topographical position (Groom et al. 2000), and hence, we expected that species contributing most to cover would differ between sites. To assess the effect of topography on restoration criteria, natural analogue reference targets were selected in both high and low positions in the landscape.

METHODS

Site Description

The study was conducted at the Rocla Quarry Products sand extraction facility, located approximately 30 km north-east of Perth, Western Australia (31.7730° S, 115.8610° E); an area that has been the subject of intensive restoration research for the past 15 years (e.g. Rokich and Dixon 2007, Benigno et al. 2012a, Benigno et al. 2012b). The facility is located on the deep siliceous sands of the Bassendean Dune System of the Swan Coastal Plain, characterized by nutrient-poor, leached acidic podzols with low water-holding capacity (Dodd and Heddle 1989, McArthur et al. 1991). The topsoil sand typically comprises 94% coarse sand and 1% clay and whilst higher in nutrients than lower horizons, is still regarded as nutrient impoverished (McArthur et al. 1991). The climate is warm Mediterranean, with an annual rainfall of approximately 680 mm, 7% of which falls during the summer months on average (Figure 1), and high summer temperatures (~ 35°C in February) and low winter temperatures (~18°C in July) (Bureau of Meteorology. 2013). Low summer rainfall and high vapour deficits have been identified as a major determinant of plant mortality in restored sites (Enright and Lamont 1992). Native vegetation is a Banksia woodland with three dominant trees; Banksia attenuata (R.Br), B. menziesii (R.Br), and Eucalyptus todtiana (F. Muell), and a diverse midstorey of shrubs and understorey of herbaceous species (Dodd et al. 1984).
Figure 1: Rainfall history across the restoration chronosequence. July-November rainfall is the amount of rain that newly restored sites received following seeding. (Bureau of Meteorology. 2013).

The Mining and Restoration Process

Mining at the study site is typically focussed on dune features (as opposed to swales between dunes) which may exceed 30 m in height. Vegetation is cleared, and topsoil is precision stripped (top 0.1 m of substrate) using laser-guided equipment, Contemporary methods see the topsoil direct-transferred to sites awaiting restoration, although it is likely that stockpiling was practiced in the past (no site-specific data available). The extent of mining is limited by the seasonal high water table mark, with extraction of resource generally ceasing within 5 m of this point. Prior to 1997, overburden material (slightly leached grey-quartz sands of the upper soil profile that usually extend for approximately 1 meter below the topsoil (McArthur and Bettenay 1960) was also removed and then returned in the reconstruction process. Research conducted in 1996, however, suggested that overburden contributed to the development of soil compaction which was found to have significant negative effects on seedling survival (Rokich et al. 2001). From 1997 onwards, overburden was no longer used, and instead, a single 10 cm layer of topsoil was spread across the floor of the mined area to
be restored. An exception was made in 2011, whereby an experimental site was allocated to re-examine the effects of overburden. Following the application of topsoil, the site is then “ripped” (ploughed) as a means to alleviate high levels of bulk density and soil impedance encountered on the pit floor. The specifications of this practice, however, have varied throughout the mine’s restoration history. Deep-ripping using a D9 dozer with a single tine at 1m intervals (~80 cm depth) was introduced as standard procedure in 1997, following field experiments in 1996 in which deep-ripping was found to alleviate soil compaction (Rokich et al. 2001). This practice was discontinued from 2000 onwards and replaced with 15 – 30 cm ripping at 20cm spacing, using a Caterpillar 963 Traxcavator. In 2010, an experimental plot was deep-ripped to facilitate further research into the comparative outcomes of deep ripping and shallow tilling. Batters, the constructed sloped transition zone between restored land from within the previous mining pit, and adjacent, natural Banksia woodland located above, are not ripped, as they are replaced materials and less subject to compaction than exposed in situ substrates. A standard, locally-sourced seed mix comprised of species that may not exist in sufficient viable numbers in the replaced topsoil seed bank (i.e., species with canopy-stored seeds) are broadcast by hand across the site. Seed broadcasting is timed to coincide with the opening rains of the growing season (usually May - June).

**Floristic survey**

A floristic survey was conducted across all restored sites from 1993 to 2011 (n = 87 quadrats), and in two natural analogue sites at high (~ 30m above groundwater) and low (~ 5-10 m above groundwater) landscape positions (n = 10 each). For each restored site, three replicate 20m x 20m quadrats were established. All perennial species present within each quadrat were recorded, along with an estimate of percent cover. A smaller nested quadrat, located in the south-west corner of each 20m x 20m quadrat was established to assess stem density. Two scales of nested quadrat were used depending on the age of the restored site; sites aged five years or less contained a 5m x 1m quadrat, while sites aged 6 years and older, as well as natural analogue sites, contained a 5m x 5m quadrat (Figure 2). This accounted for the decline in plants per unit area as individual plant size increases with age, and hence, ensured sufficient sample sizes in density counts.
Environmental drivers and management intervention techniques

Floristic survey data was related to site information (site type, ripping treatment, use of overburden) provided by Rocla Quarry Products. Aspect (Table 2) was assessed from a Digital Elevation Model (DEM) image, and converted into a linear north-south gradient (northness) and an east-west gradient (eastness) by applying cosine and sine transformations respectively (Guisan et al. 1999, Kumar et al. 2006). Site type distinguishes sloped batters from flat sites on the pit floor. Disturbance depth distinguishes deep-ripped pit floor sites and batters from shallow ripped and unripped pit floor sites. Rainfall in the first two years following restoration was determined using rainfall records from nearby weather stations (Wanneroo 9105 and Mariginiup 9249; Bureau of Meteorology. 2013). Site age incorporates all other factors and processes not explicitly assessed that change with time; e.g. individual plant growth and mortality, increases in soil microbial activity, etc. A summary of site characteristics is provided in Table 3.
<table>
<thead>
<tr>
<th>Factor</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site age</td>
<td>Years since site reconstruction and seeding</td>
</tr>
<tr>
<td>Aspect NS</td>
<td>Site aspect expressed as a numerical value in terms of northerly orientation. $N = 1$, $NE/NW = 0.71$, $E/W = 0$, $SE/SW = -0.71$, $S = -1$</td>
</tr>
<tr>
<td>Aspect WE</td>
<td>Site aspect expressed as a numerical value in terms of westerly orientation. $W = 1$, $NW/SW = 0.71$, $N/S = 0$, $NE/SE = -0.71$, $E = -1$</td>
</tr>
<tr>
<td>Type</td>
<td>Pit floor or batter – is the reconstructed site located on what was previously the pit floor (likely to be flat, subjected to higher levels of vehicle movement, and over-consolidated), or a batter (sloped transition area between pit floor and adjacent, unmined land)</td>
</tr>
<tr>
<td>Disturbance</td>
<td>shallow (&lt;30cm), deep (~80 + cm)</td>
</tr>
<tr>
<td>Overburden</td>
<td>Yes/no – was overburden (the material removed from ~0.1 – 1m below the original soil surface and characterized by higher concentrations of small particles and organic matter, returned to the site (before topsoil replacement) as part of the site reconstruction process?</td>
</tr>
<tr>
<td>Wet 1</td>
<td>Rainfall (mm) received in July – November (post topsoil placement) in first year</td>
</tr>
<tr>
<td>Dry 1</td>
<td>Rainfall (mm) received in summer (December – February) in first year following restoration</td>
</tr>
<tr>
<td>Wet 2</td>
<td>Rainfall (mm) received in March - November in second year following restoration</td>
</tr>
<tr>
<td>Dry 2</td>
<td>Rainfall (mm) received in summer (December-February) in second year following restoration</td>
</tr>
</tbody>
</table>
Table 3: Individual site characteristics across the restoration chronosequence. Number in Site ID is site age (years since establishment). N = 3 for each treatment. For explanations of other terms, see Table 2.

<table>
<thead>
<tr>
<th>Site ID</th>
<th>Type</th>
<th>Aspect</th>
<th>Disturbance depth</th>
<th>Over-burden</th>
<th>Rainfall</th>
</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td>shallow</td>
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<td>497</td>
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<td>floor</td>
<td>0</td>
<td>shallow</td>
<td>no</td>
<td>497</td>
</tr>
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<td>1-C</td>
<td>floor</td>
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<td>shallow</td>
<td>no</td>
<td>497</td>
</tr>
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<td>floor</td>
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<td>deep</td>
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<td>deep</td>
<td>no</td>
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<td>deep</td>
<td>no</td>
<td>448</td>
</tr>
<tr>
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<td>batter</td>
<td>0</td>
<td>deep</td>
<td>no</td>
<td>448</td>
</tr>
<tr>
<td>3-C</td>
<td>batter</td>
<td>S</td>
<td>deep</td>
<td>no</td>
<td>448</td>
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<tr>
<td>4-A</td>
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<td>shallow</td>
<td>no</td>
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</tr>
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<td>batter</td>
<td>SW</td>
<td>deep</td>
<td>no</td>
<td>348</td>
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<tr>
<td>5-A</td>
<td>batter</td>
<td>N</td>
<td>deep</td>
<td>no</td>
<td>476</td>
</tr>
<tr>
<td>5-B</td>
<td>batter</td>
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<td>shallow</td>
<td>no</td>
<td>476</td>
</tr>
<tr>
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<td>no</td>
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<td>shallow</td>
<td>no</td>
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<td>shallow</td>
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<tr>
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<td>batter</td>
<td>E</td>
<td>shallow</td>
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</tr>
<tr>
<td>12-B</td>
<td>floor</td>
<td>0</td>
<td>deep</td>
<td>no</td>
<td>443</td>
</tr>
<tr>
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<td>batter</td>
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<td>shallow</td>
<td>no</td>
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</tr>
<tr>
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<td>deep</td>
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<td>deep</td>
<td>no</td>
<td>357</td>
</tr>
<tr>
<td>15-B</td>
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<td>deep</td>
<td>no</td>
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</tr>
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<td>no</td>
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</tr>
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<td>16-B</td>
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<td>deep</td>
<td>no</td>
<td>572</td>
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<tr>
<td>18</td>
<td>floor</td>
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<td>shallow</td>
<td>yes</td>
<td>347</td>
</tr>
<tr>
<td>19</td>
<td>floor</td>
<td>0</td>
<td>shallow</td>
<td>yes</td>
<td>401</td>
</tr>
</tbody>
</table>

Restoration criteria

The four restoration criteria were derived from floristic survey data and remotely sensed aerial imagery. Species richness and plant density were calculated from count data. For each individual restored site, Bray-Curtis Similarity was calculated from presence/absence data in relation to each of the 20 reference sites (high = 10, low = 10) and the mean calculated for each landscape position. This mean value was used in analyses as ‘Similarity’ (high and low). Total vegetation cover of 20m x 20m quadrats
was assessed using remotely-sensed, Near Infrared Red, aerial imagery, classified using the Normalized distribution Vegetation Index (NDVI) at 10cm spatial resolution, collected on 15 September, 2011 (Survey Graphics, Perth). The classification was ground-truthed using 282 digital images of known points, achieving 82% accuracy. Aerial imagery was not available for high reference sites, and classification of sites less than three years-old was not deemed accurate enough for inclusion in this study (for further details, see Chapter 3).

Data analyses

To provide a visual indication of site similarity, non-metric, Multi-Dimensional Scaling (MDS) ordination was conducted using PC-Ord 5 (Grace and Urban 2002). Singletons were removed from ordination analysis to reduce the effect of rare species. Similarity within and between restored and reference sites (high and low landscape position) was quantified using Similarity Percentage (SIMPER) and assessed for significance using Analysis of Similarity (ANOSIM), in Primer v5 (CLARKE 1993). Direct comparisons of richness, density, similarity, and cover between restoration treatments (i.e. deep-ripping versus non-ripped, and overburden versus no overburden) in sites of the same age using the Kruskal-Wallis non-parametric comparison test in R V 3.0 (R Core Team 2013). Regression analysis was carried out to determine relationships between criteria (species richness, plant density, similarity, and cover). All statistics were tested at a critical value of \( p < 0.05 \), unless stated otherwise.

Generalized Linear Model (GLM) analyses were used to assess the effect of natural drivers and management intervention techniques on trajectories of species richness, plant density, vegetation cover, and Similarity. Linear and second-order polynomial terms of rainfall and age parameters were included to allow for both linear and non-linear relationships. In order to reduce model complexity, interactions were restricted to first and second-order interactions between rainfall and age parameters with all other measured variables. The ‘step’ function from the R package “MASS” (Venables and Ripley 2002) was used to select the parameters that comprised the final model, based on the lowest Akaike Information Criteria (AIC) value. Model predictions for selected variable combinations were plotted to illustrate their estimated effect on restoration outcomes.
RESULTS

Percent Cover

Vegetation cover displayed a clear positive trajectory through time (Figure 10; numbers on horizontal axis indicates site age), although substantial variation was evident within individual years. Percent cover in reference sites averaged (81 ± 7 %) which was significantly greater than all restored sites (p < 0.05). The highest mean cover score was recorded in 18 year-old sites (57 ± 11%), yet this was still approximately 30% less than low reference sites. These sites were flat, had not been ripped, and had incorporated overburden material into the reconstruction process. The highest cover measured in a single quadrat was also recorded in an 18 year-old site (79%). In a direct comparison of 16-year-old, deep-ripped versus non-deep-ripped sites, sites that had been deep-ripped had twice as much vegetation cover (69% ± 8.7) compared with sites of the same age that had not been ripped (27 ± 7.1) (p = 0.020).

Figure 10: Percent cover at sites within the restoration chronosequence and in the reference site. Means indicated by triangles, median values by horizontal bars, interquartile range by boxes and whiskers, and outliers (>1.5 times the interquartile range) by circles. Numbers in site labels indicate site age. Letters, where present, indicate a different treatment. See Table 3 for an explanation of treatments.
Percent Cover Modelling

The model accounted for 79% of the total variation in cover (Table 7). Site age was the largest determinant of cover, but did not interact significantly with any other terms. Under mean rainfall conditions, vegetation cover increased by approximately 2.5% per year (Figure 11a). Ripping had virtually no effect. Increasing rainfall from 350 mm to 500 mm in the first wet season resulted in a 25% increase in cover in deep ripped sites, while there was no effect on non-ripped sites (Figure 11b). Likewise, increasing first wet season rainfall from 350 mm to 500 mm increased vegetation cover in pit floor sites by nearly 10%, while, batter sites decreased by 10% (Figure 11d). Sites with overburden were modelled to have approximately 8% more cover than sites without (Figure 11c). Rainfall in the second wet season was associated with a decrease in vegetation cover (Table 4).

**Table 4:** Variables selected via stepwise regression modelling based on the lowest Akaike Information Criteria (AIC) value for predicting and explaining mean percent cover in restored sites. Level of significance is indicated by; 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

<table>
<thead>
<tr>
<th>Covariate</th>
<th>Coefficient</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>(Intercept)</td>
<td>28.400</td>
<td>0.687</td>
</tr>
<tr>
<td>Age</td>
<td>11.970</td>
<td>0.012  *</td>
</tr>
<tr>
<td>Age (2nd order)</td>
<td>0.618</td>
<td>0.004  **</td>
</tr>
<tr>
<td>Wet 1</td>
<td>0.163</td>
<td>0.113</td>
</tr>
<tr>
<td>Dry 1</td>
<td>0.567</td>
<td>0.013  *</td>
</tr>
<tr>
<td>Dry 1 (2nd order)</td>
<td>-0.005</td>
<td>0.024  *</td>
</tr>
<tr>
<td>Type batter</td>
<td>294.400</td>
<td>&lt; 0.001 ***</td>
</tr>
<tr>
<td>Aspect WE</td>
<td>12.420</td>
<td>0.161</td>
</tr>
<tr>
<td>Disturbance deep</td>
<td>-253.400</td>
<td>&lt; 0.001 ***</td>
</tr>
<tr>
<td>Overburden Yes</td>
<td>-112.700</td>
<td>&lt; 0.001 ***</td>
</tr>
<tr>
<td>Wet 2</td>
<td>-0.304</td>
<td>0.090  .</td>
</tr>
<tr>
<td>Wet 2 (2nd order)</td>
<td>&lt; 0.001</td>
<td>0.080  .</td>
</tr>
<tr>
<td>Dry 2 (2nd order)</td>
<td>-0.002</td>
<td>0.045  *</td>
</tr>
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<td>Age:Wet 1</td>
<td>-0.040</td>
<td>0.002  **</td>
</tr>
<tr>
<td>Wet 1:Type batter</td>
<td>-0.589</td>
<td>0.001  **</td>
</tr>
<tr>
<td>Wet 1:Disturbance deep</td>
<td>0.517</td>
<td>&lt; 0.001 ***</td>
</tr>
</tbody>
</table>
Figure 11: Vegetation cover; a) through time under deep and shallow disturbance and mean first year rainfall; b) the effect of disturbance depth under increasing first wet season rainfall; c) through time when sites include and exclude overburden material in site reconstruction and rainfall in the first year is at mean levels, and; d) on pit floor and batter sites when rainfall in the first wet season is increasing.

Cover – Species contributions

Species contributing most to cover were almost mutually exclusive between reference sites and restored sites. Slow-growing tree and shrub species contributed most to cover in the high reference sites, with *Banksia attenuata* and *Scholtzia involucrata* (Endl.) Druce cover scores ≥10 -20% in 5 out of 10 survey quadrats. *B. attenuata* cover was also relatively high in low reference sites, as was that of the small shrub *Hibbertia subvaginata*. Cover in restored sites was often dominated by a single, rapid growing, disturbance species (*Adenanthis cygnorum* Diels), particularly in sites that had been deep-ripped. Out of the 15 quadrats that had been deep-ripped in sites aged between 11 and 16 years, 10 of those were assessed to have a minimum of 5-10% cover by *A. cygnorum*), including 4 quadrats with 10-20% cover, and 1 site with 20-50% cover. Another disturbance species, *Kunzea glabrescens* Toelken, contributed
substantially to cover in sites less than 10 years old. Weed species only contributed to cover (1-5%) in 3 restored sites, and < 1% in all reference sites.

**Plant Density**

Mean plant density did not differ significantly in both high (6 ± 0.9 per m$^2$) and low (5 ± 0.5 per m$^2$) reference sites (p = 0.41), but were approximately twice that recorded in restored sites (3 ± 0.3 per m$^2$; p < 0.05) (Figure 5). Plant density in restored sites generally declined with site age away from the reference targets, with the oldest restored site (19 years) recording an approximately 75% lower mean plant density (1.4 ± 0.6 per m$^2$) than reference sites (p < 0.05). Despite this, a large degree of variation in plant density was evident within sites, and especially in sites less than 3 years old.

In a direct comparison of the effect of ripping treatments in one year-old sites, sites that were deep-ripped recorded a mean plant density more than twice that of non-ripped sites (9.4 ± 1.11 per m$^2$ and 3.9 ± 1.16 per m$^2$, respectively; p < 0.05). Plant density in 16 year-old deep ripped sites was 40% lower than in non-ripped sites, though a large degree of variation meant that this was not significant (1.12 ± 0.72 per m$^2$ and 1.96 ± 0.15 per m$^2$, respectively). The use of overburden in site reconstruction in one year-old sites almost doubled plant density (9.2 ± 2.68 per m$^2$ versus 5 ± 1.47 per m$^2$; p = 0.003).
Factors associated with variation in Plant Density

All explanatory variables were included in the final model of plant density which accounted for 70% of total variance between restored sites (Table 5). Plant density decreased with site age, irrespective of environmental drivers or management intervention techniques. Under mean rainfall conditions, density decreased by approximately 0.25 plants per m$^2$ each year (Figure 6a). Site aspect was a significant predictor of plant density, with easterly-orientated sites modelled to contain 1.7 additional species per m$^2$ than west-orientated sites, and 0.9 more plants per m$^2$ than flat sites (Figure 6c). Increasing rainfall in the first wet season decreased plant density in sites irrespective of east-west orientation (Figure 6b). Under mean first wet season rainfall, ripping treatment had virtually no effect (Figure 6a), however, varying rainfall regime interacted with ripping treatment significantly. Low first wet season followed by high first dry season rainfall produced plant densities that were three times greater than in sites that received high first wet season followed by low first dry season rain (Figure 6b).
However, density rapidly decreased with time in the low wet, high dry sites, while it declined at a lower rate in high wet, low dry sites. Increasing first dry season rainfall interacted with ripping to significantly increase plant density, with 80 mm of first dry rains approximately doubling plant density (Figure 6f). Increasing first wet season rainfall to 500 mm had the opposite effect, reducing plant density by almost 2 plants per m$^2$ in deep ripped sites, and 1.5 plants per m$^2$ in non-ripped sites. Overburden interacted with rainfall in the first wet season to increase plant density by approximately 5.5 plants per m$^2$ at 500 mm rainfall, while sites with no overburden decreased by almost 3 plants per m$^2$ when first wet season rainfall reached 500 mm (Figure 6d). When first wet season rainfall was low, batter sites were modelled to have 2 more plants per m$^2$ compared with pit floor sites. Increasing rainfall to 500 mm in the first wet season however, reduced the effect of site type, with both types having approximately 2 plants per m$^2$.

### Table 5: Variables selected via stepwise regression modelling based on the lowest AIC value for predicting and explaining plant density in restored sites. Level of significance is indicated by; 0 ‘****' 0.001 ‘***' 0.01 ‘**' 0.05 ‘.' 0.1 ‘ ’ 1

<table>
<thead>
<tr>
<th>Covariate</th>
<th>Coefficient</th>
<th>p</th>
</tr>
</thead>
<tbody>
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<tr>
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<td>&lt; 0.001***</td>
</tr>
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<td>&lt; 0.001***</td>
</tr>
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<tr>
<td>Aspect WE:Wet 2</td>
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<td>0.002**</td>
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Figure 6: Predicted plant density per m$^2$; a) through time under shallow and deep-ripping treatments and mean year rainfall; b) in sites with westerly, easterly, and no aspect when wet 1 rainfall is increasing and dry rain 1 is held constant; c) through time in sites with westerly, easterly, and no aspect under mean first year rainfall; d) with and without the incorporation of overburden with increasing rainfall in the first wet season; e) through time under deep and shallow disturbance and different first year rainfall regimes, and; f) under shallow and deep disturbance, mean first wet season rain and increasing first dry season rain.
Species richness

Species richness did not differ between high and low reference sites (average ± SE; 56 ± 0.3 and 53 ± 2.0 respectively), but was significantly lower (> 30%; p < 0.05) in restored sites on average than in reference sites (Figure 3). Overall, mean species richness did not follow a clear trajectory through time, although a slightly positive trend in the first six years followed by a decline was apparent. A large degree of variation was evident both within sites and between sites restored in the same year. The highest number of species recorded in a 20m x 20m quadrat occurred in a high reference site (69 species), whilst the highest recording in a restored site was in a 6 year-old, shallow-ripped site with no overburden (62 species). The lowest numbers of species recorded was 10, and occurred in a 3-year-old (2009) shallow-ripped site located on a batter.

In a direct comparison of sites of the same age, one year-old sites with overburden supported nearly 60% higher richness values than sites without overburden (48 ± 1.89 v. 29 ± 1.7; p < 0.05). Disturbance depth did not significantly affect richness, with 27 ± 2.0 and 30 ± 1.0 recorded in shallow disturbed and deeply disturbed sites respectively in a comparison of one year-old sites. Similarly, 16 year shallow-disturbed and deeply-disturbed sites recorded richness values of 35 ± 2.3 and 35 ± 3.0 respectively.
Factors associated with variation in Species Richness

The model selected with the lowest AIC accounted for 46% of total variance in species richness between restored sites (Table 4). Richness declined with site age irrespective of environmental drivers or management intervention techniques. Under mean first year rainfall conditions (wet 1 = 418 mm, dry 1 = 28 mm), species richness declined by 0.65 species per year (Figure 4a). Site aspect was the most important determinant of richness, with East-facing sites predicted to contain approximately 20 more species than west facing sites, and 15 more species than in flat sites, under mean first year rainfall conditions for the first year (Figure 4b). Likewise, north facing sites were predicted to have 6 more species than south facing sites, and 3 more than flat sites (Figure 4d). Deep disturbance reduced richness by 4 species (Figure 4a) under mean rainfall conditions, and was associated with decreased richness, regardless of rainfall. The use of overburden decreased richness by 5 species (Figure 4f). Increasing first wet season rainfall or first dry season rainfall had virtually no effect, but low first wet season followed by a high first dry season rainfall increased richness by 8, compared
with a high first wet season and a low first dry season. Increasing rainfall in the second wet season had a negative effect on species richness.

**Table 4:** Variables selected via stepwise regression modelling based on the lowest AIC value for predicting and explaining species richness in restored sites. Level of significance is indicated by: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

<table>
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</tr>
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</tr>
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<tr>
<td>Wet 1:Wet 2</td>
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<td>0.0272  *</td>
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Figure 4: Species richness; a) when disturbance is shallow and deep and mean first year rainfall; b) as affected by east-west orientation under mean first year rainfall; c) under high and low rainfall in the first year and shallow and deep disturbance depth; d) as affected by north-south orientation under mean first year rainfall; e) under varying first wet/first dry rainfall regimes in the first year and shallow and deep disturbance depth; f) with and without the incorporation of overburden material under mean rainfall in the first year.
Species Composition

Ordination of sites based on species presence/absence clearly grouped sites into restored, high reference, and low reference, and indicated that restored sites are less similar to low reference sites than to high reference sites, and have much greater variation (Figure 7a). ANOSIM indicated that high and low reference sites were markedly dissimilar (Global R = 0.81; p < 0.001%) and was supported by SIMPER analysis which showed similarity in high and low sites to be 49%.
Figure 7a) 2D MDS ordination showing similarity between restored (numbers indicate site age) and reference sites (H= high landscape position, L = low landscape position). Stress = 0.23; b) Mean Bray-Curtis restored site similarity based on presence/absence data with high reference sites, and; c) low reference sites. Means indicated by triangles, median values by horizontal bars, interquartile range by boxes and whiskers, and outliers (>1.5 times the interquartile range) by circles. Numbers in site labels indicate site age. Letters, where present, indicate a different treatment. See Table 3 for an explanation of treatments.
The top 10 species accounted for 14% dissimilarity between high and low reference sites, with the top contributor accounting for only 1.8%. Similarity was greater between the reference sites and restored sites than between reference sites in high and low landscape positions. According to SIMPER analysis, similarity between high reference sites and restored sites was 42%, while similarity between low reference sites and restored sites was 34%. Likewise, ANOSIM indicated that high reference sites and restored sites were less distinct (global R = 0.25, p < 0.005) than low reference sites and restored sites (global R = 0.69; p < 0.001).

No discernible trajectory was evident in the development of compositional similarity between restored sites and both high (Figure 7b) and low reference sites (Figure 7c). The greatest similarity with a high reference site (79%) was recorded in a 4 year-old, south-west facing batter (sloped) with no overburden, while the greatest similarity with a low reference site (52%) occurred in a 2 year-old south-west facing batter, with no overburden.

In a direct comparison of sites of the same age, similarity in one year-old deep-ripped and shallow-ripped sites was 59%, with the ten species contributing most to dissimilarity accounting for 36% of total dissimilarity. Likewise, 16 year-old deep-ripped and not-ripped sites were 57% similar. Twenty five percent of total dissimilarity was attributed to 10 species. The incorporation of overburden material in site reconstruction was associated with significant increases in site similarity to both high and low reference sites. Similarity to high reference sites increased from 35 ± 1.4 in sites without overburden, to 51 ± 1.3 in sites with overburden (p < 0.001), while similarity to low reference sites increased from 32 ± 1.7 to 44 ± 1.2 (p = 0.003).

Factors associated with variation in Similarity

Predictive modelling highlighted significant differences between landscape positions (Table 6). The modelled parameters accounted for 40% of the total variation in mean similarity to high reference sites, and 42% to low reference sites. When rainfall was held at mean chronosequence values for wet 1 and dry 1, both age and ripping treatment had no effect on similarity to both landscape positions (Figures 9a and 9b). Increasing rainfall in the first wet season had virtually no effect on restored site
similarity to high or low reference sites, nor did increasing rainfall in the first dry season improve similarity to low reference sites (Figure 6 d).

**Table 6:** Variables selected via stepwise regression modelling based on the lowest AIC value for predicting and explaining mean Bray-Curtis similarity in restored sites to high and low reference sites. Level of significance is indicated by: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

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<th>p</th>
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<td>Aspect WE</td>
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Under mean first year rainfall conditions, similarity to high and low reference sites did not change with site age (Figure 9a, b). High first wet season rainfall followed by low first dry season rainfall was associated with a 5 % increase in similarity to high reference sites over 20 years, while low first wet season followed by high first dry season rainfall decreased similarity to high reference sites by approximately 17 % over 20 years (Figure 9c). A similar yet milder response was modelled in low reference sites (Figure 9d). The use of overburden decreased similarity to high reference sites, and increasing first wet season rainfall exacerbated this (Figure 9e). Similarity to low reference sites was reduced by approximately 5% in sites with overburden, but increasing first wet season rainfall did not affect this (Figure 9f). Similarity to high reference sites was 5% greater in batter sites compared to pit floor sites when first wet season rainfall was low, but this effect diminished as rainfall increased.
Figure 9: Predicted Bray-Curtis similarity; a) to high reference sites under shallow and deep ripping when rainfall is held at mean values for first year; b) to low reference sites under shallow and deep ripping when rainfall is held at mean values for first year; c) to high reference sites under high wet 1, low dry 1, and low wet 1, high dry 1 rainfall regimes; d) to low reference sites under high wet 1, low dry 1 and low wet 1, high dry 1 rainfall regimes; e) to high reference sites when overburden is included and excluded in site reconstruction, under increasing first dry season rainfall; f) to low reference sites when overburden is included and excluded in site reconstruction, under increasing first dry season rainfall.
Correlation between restoration criteria

Species richness was significantly correlated with all other criteria (Table 8). The relationship was strongest between richness and low reference site similarity. Density and cover were also significantly correlated, while cover was not related to similarity in either high or low reference sites.

Table 8: Correlation between assessed criteria. Sim: high/low refers to Bray-Curtis similarity with reference sites in high and low landscape positions, respectively.

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<td>Sim: low</td>
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<tr>
<td>Cover</td>
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<td>0.002</td>
<td>0.18</td>
<td>&lt;0.001</td>
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DISCUSSION

As hypothesized, species richness and plant density decreased with site age, while vegetation cover increased. Similarity to reference sites remained relatively constant through time. Changing rainfall regimes and implementing restoration treatments did not affect these overall patterns. Within the confines of these trends, however, natural drivers and management intervention techniques exhibited a notable influence on restoration outcomes.

This project hypothesized that restoration success (high species richness, plant density, similarity with natural analogue reference sites, and vegetation cover) would be greater following establishment in years with relatively high rainfall, and in particular, summer rainfall, in the two years immediately following site reconstruction. In fact, whilst vegetation cover increased with rainfall, increasing rainfall in the first and second wet seasons was associated with a decrease in richness and a decrease in plant density. This suggests that increasing individual plant size per unit area in response to high rainfall may result in density-dependent mortality. The subsequent reduction in individuals per unit area has a follow-on effect, whereby samples of finite individuals will result in fewer species by chance alone (Stevens and Carson 1999). Regression
analysis showing that plant density and richness are significantly negatively correlated with cover supports this concept. Species cover data indicates a prevalence of *Adenantheros cygnorum* – a large, dense, disturbance species shrub, up to 4m in height (Western Australian Herbarium 1998), common throughout older restored sites, but largely absent from reference sites. Its abundance and large expanse of cover in restored sites is likely to have a significantly negative effect on indicators of restoration success.

If declining plant density and richness in restored sites is indeed the product of competition, then higher plant density and richness in natural analogues is related to plant community structure – how the physical space is occupied by plants. Even the oldest restored sites (19 years) had not yet developed the distinct vertical structure that is characteristic of Banksia woodlands. Hence, tree species in the restoration chronosequence are still occupying and competing for resources in the same space as understorey species. Continued growth through time and the development of vertical stratification could free up recruitment space (i.e. niches) for new germinants to occupy, both in terms of numbers of species, and in the number of individuals of species. Additionally, the development of vertical structure may increase facilitative feedbacks and interactions (Facelli and Brock 2000, D’Odorico et al. 2006). Furthermore, across the total area of restored land surveyed (3.48 ha), total richness was approximately 40% greater than in high or low reference sites (0.4 ha each), indicating that lower richness in restored sites may not be due to lack of habitability, but rather, is density-dependent. Recruitment in this system, however, may be dependent on environmental queues including disturbance events such as fire (Yates and Hobbs 1997), or particular rainfall regimes (Harrington 1991), or their combination. This, combined with the fact that many species in these systems have poor dispersal capabilities (Hopper 2009), may mean that further management interventions (i.e. controlled burning, infill seeding of species, etc.) may be required to assist in achieving reference targets.

The importance of summer rainfall however, in determining restoration outcomes was supported. Plant density and richness were optimal when rainfall in the first winter/spring was low (~350 mm), followed by high summer rainfall. In addition to seedling establishment requirements, the increase in plant density and richness under this rainfall regime may be due to a reduction in the relative abundance of species that respond to high rainfall in the first winter/spring and later out-compete other species. Additionally, it also suggests that seedling mortality may be high in freshly restored sites when summer rainfall in the first year is low. This is supported by Burrows et al
(1990), who found that eucalyptus recruitment in dry Mediterranean and semiarid woodlands is dependent on above average rainfall in the first summer following germination.

Additionally, we expected that sites that experience relatively lower evaporative pressures would perform better than sites exposed to higher evaporative demand. We found that sites with a more southerly aspect, and hence, exposed to less solar radiation and evaporative demands, did indeed support higher species richness than north-orientated or flat sites, but that a north aspect had little effect on other criteria. An easterly orientation was a more important positive determinant of species richness and plant density. Hot, prevailing easterly winds blow from inland Western Australia, particularly during the summer months (Bureau of Meteorology) and it may be possible that in newly restored sites where surface roughness is low, winds may be a larger determinant of evaporative pressure than solar radiation. This may be a case of increasing environmental stress/reduced resource availability imposing niche-limiting pressures upon species resulting in increased species diversity (Stevens and Carson 1999). At the same time, a westerly orientation was associated with increased vegetation cover in this study. West-orientated sites are likely to receive more rainfall due to rain coming from the west, and this, combined with reduced evaporation from wind may result in westerly sites being dominated by larger individuals, and consequently, reduced species richness (Stevens and Carson 1999). Several mechanisms have been suggested to explain this relationship, including interspecific competitive exclusion (e.g. Austin 1990, Huston and DeAngelis 1994) and assemblage level thinning (Oksanen 1996). Deciphering which mechanism is driving this response is beyond the scope of the present study, however it seems likely that competitive processes may be driving restoration criteria on sloped sites.

In assessing if management intervention techniques drive the trajectory of plant community development, we predicted that sites with reduced soil compaction, a critical issue associated with Banksia woodland restoration success (Enright and Lamont 1992, Rokich et al. 2001), would have greater restoration success. This included sites that were deep-ripped, batter sites, and/or sites that did not incorporate overburden into the site during reconstruction. The effect of disturbance depth was largely dependent on the rainfall regime, with deeply-disturbed sites requiring high summer rainfall in the first year to positively affect criteria relative to shallow disturbance. That under mean rainfall conditions, shallow disturbed sites produced higher density scores than deeply-
disturbed sites, suggests that the benefits of deep disturbance (reduced bulk-density/soil impedance at depth) are nullified by a reduction in water retention due to increased porosity (Gwenzi et al. 2011) when rainfall is not high. Richness was not significantly affected by disturbance depth, indicating that increases in density associated with deep disturbance are the product of an increase in the relative abundance of species already present in shallow disturbance sites. The use of overburden in the substrate reconstruction process did decrease richness and similarity but was associated with an increase in plant density and cover. However, in a direct comparison study of one year-old sites, sites incorporating overburden material into the reconstructed substrate had 40% higher richness, 45% higher plant density, and approximately 30% increase in similarity to both high and low reference sites, compared to sites that did not contain overburden. In the summer immediately following the vegetation survey, very high plant mortality was observed in these sites containing overburden. These sites had received particularly low first year rainfall, and observational evidence suggests they were highly compacted. This is supported by and confirms the previous study by Rokich et al (2001) that found overburden significantly contributed to higher levels of soil compaction. Ripping treatments could be refined in an attempt to gain the benefits of overburden whilst reducing the risk of developing compaction.

The study showed that restored sites were more similar to natural reference sites located in high landscape positions than in low landscape positions. This finding is not particularly surprising; the mining process is limited by the depth to groundwater, and hence, low lying areas are generally avoided. The topsoil used in the restoration process is sourced from areas cleared for mining, resulting in the soil seedbank being comprised of plant species that occur topographically high in the landscape. The resulting plant composition is subsequently more similar to that of high landscape positions. Restored sites, however, are typically situated low in the landscape and are topographically and hydrologically more similar to a low landscape position. It has already been stated that vegetation structure in the oldest restored sites has not yet reached maturity, and how the differences in landscape functional attributes associated with position may affect species composition in the long term remains to be seen.

This study highlighted the importance of rainfall and the complexities of its interactions with management intervention techniques in determining restoration outcomes. Plant-water deficit is a major driver of plant mortality across water-limiting systems and the ability to predict high rainfall events via ENSO could be used to select
the most appropriate restoration treatment (Sitters et al. 2012). The declining trend in plant density and associated decrease in species richness within the assessed timeframe of this study are likely linked to an underdeveloped vertical structure, and hence raise questions as to the appropriateness of the temporal scale in which restoration is currently commonly assessed (Ruiz-Jaen and Mitchell Aide 2005). However it must be noted that whilst the modelled parameters accounted for the majority of variation in plant density and vegetation cover (70% and 79%, respectively), only a relatively small proportion of the total variation was accounted for by species richness (46%), similarity to high reference sites (40%), and similarity to low reference sites (42%). These criteria are all inherently related to the diversity and viability of the seed material used in restoration, and may indicate that topsoil collection and handling, and seed mix composition, may require attention.

Restoration chronosequences are a useful tool for understanding the long term effects of natural drivers and management intervention techniques on restoration outcomes, but are often limited by a lack of treatment replication, controls, or continuity. For this reason, care should be taken when interpreting model outputs. Whilst necessarily governed by the economic and practical realities of industry, restoration should be conducted with as much scientific rigour as possible in order to contribute to the growing discipline of restoration ecology and its ability to feedback practical advice to improve on-ground outcomes.

Implications for restoration

- Old, biodiverse vegetation systems may take long periods (>100 years) to resemble reference states, and criteria such as richness and density may be governed by the development of structure. Indicators of restoration success should be indexed accordingly.
- The effectiveness of management intervention techniques varies with and is determined by their interactions with changing rainfall regimes.
- High mass, disturbance-based species may persist in a restored system for many years and have a negative effect on restoration criteria. Management intervention to remove these species from the system may be beneficial to restoration outcomes.
• Adaptive management provides a powerful tool in achieving restoration success in biodiverse ecosystems.
CHAPTER THREE

Influence of ripping and vegetation cover on the restoration of hydrological function through time in post-mining sand substrates.

INTRODUCTION

Ecosystem function commonly refers to the biotic and abiotic interactions that are the basis for self-maintenance in an ecosystem and may be broadly defined as the sequestering and transformation of energy, nutrients, and moisture (Clewell et al. 2004). Restored ecosystem function is listed by the Society for Ecological Restoration as one of the nine attributes of a restored landscape (Clewell et al. 2004), yet functional restoration is rarely assessed owing to the relatively slow recovery times of functional attributes compared with other restoration criteria (e.g. diversity and structure) (Chambers et al. 1994, Ruiz-Jaen and Mitchell Aide 2005). In systems ranging from arid to Mediterranean climates where plant-water deficit is a major driver of plant mortality (Enright and Lamont 1992), restoring the hydrological component of ecosystem function is particularly critical to restoration success, but is currently poorly understood (Rokich et al. 2001, Benigno et al. 2012a).

Post-mining landscapes often have significantly altered hydrological regimes as a result of substrate disturbance. Mining may result in drastic topographic changes, often creating new hydrological landscapes where surface soils are comprised of material that had been previously buried at depth and thus was associated with high bulk density (Leroueil and Vaughan 1990). In addition, mechanical compaction via vehicle movement and natural settling processes following site reconstruction may also significantly alter soil physical properties (Kozlowski 1999), resulting in substrates characterized by high bulk density and penetration resistance, reduced infiltration and water retention capacity, and a discontinuous macropore network (Archer et al. 2002). The cumulative effect of these substrate alterations is to decrease plant-available water, and to reduce the ability of plant roots to move through the substrate to access soil-moisture. In water-limited environments, these changes have been identified as major drivers of seedling mortality (Burrows 1986, Enright and Lamont 1992, Rokich et al. 2001, Maestre et al. 2003, Benigno et al. 2012b).
In undisturbed landscapes, soil hydrological properties have been shown to be correlated with vegetation cover, particularly in arid and semi-arid systems where vegetation patches have been highly correlated with decreased soil temperature and bulk density, and increased stem flow, soil moisture, microbial activity, porosity soil aggregate formation, and infiltration rate (Facelli and Brock 2000, Puigdefábregas 2005, Barbier et al. 2008). The relationships between hydrology and vegetation under changing rainfall and substrate characteristics, however, are less well understood. In a metadata analysis of changes in infiltration rates across climate and soil gradients, Thompson et al (2010), highlights that the positive correlations between vegetation cover and infiltration appears to decline as annual rainfall increases, and likewise, as clay content decreases. For example, hydrological processes that drive plant-soil-water interactions (Puigdefábregas 2005) are likely to behave very differently in sand substrates where infiltration processes prevail over overland flow due to relatively higher hydraulic conductivity, and low water retention enhances the possibility of water stress between rain events (Ceballos et al. 2002).

In disturbed landscapes the restoration of hydrological functioning is also likely to be correlated with the presence of vegetation, involving numerous dynamic biotic and abiotic interactions (Greenwood and McKenzie 2001). Soil structural characteristics, including bulk density, penetration resistance and infiltration rates in restored mine site soils have been observed to regenerate via plant root activity (Shukla et al. 2004, Breshears et al. 2005). Increased soil organic carbon (SOC), associated with vegetation inputs, increases soil microbial and macrofaunal activity, which in turn increases soil porosity, decreases penetration resistance and can modify water retention (Greenwood and McKenzie 2001, Shukla et al. 2004, Thompson et al. 2010). Additionally, restoration procedures including land forming and topsoil application replace natural soil-forming processes (Shukla et al. 2004) and may significantly influence hydrological function.

The implementation of management intervention techniques, including substrate ripping and the addition of soil amendments (Rokich et al. 2001, Benigno et al. 2012b), may also heavily influence hydrological characteristics in restored sites. Substrate ripping in particular, has been employed to hasten substrate regeneration, alleviating soil compaction, increasing infiltration rates, and enabling root elongation (Ashby 1997, Benigno et al. 2012b). However, studies have shown that in some systems, mechanically ripped soils may soon return to pre-ripped levels of soil compaction.
(Benigno et al. 2012b), whilst the effect on hydrological properties in the longer term remains unclear.

The aim of this study was to assess the development of hydrological properties through time in restored, post-mining landscapes in order to understand the role of biotic activity and management intervention techniques in restoring hydrological function. In particular, we examined the effects and interactions of vegetation cover and substrate ripping treatments (both deep and shallow) in relation to two key indicators of hydrological function; saturated hydraulic conductivity \( (K_s) \) – a measure of infiltration that determines water inputs into a system, and; penetration resistance (PR; at the surface and at depth) – a measure of the resistance plant roots must overcome to move through the soil profile and hence, a large determinant of relative plant-available water. The study examined restored Banksia woodland sites aged between 0 and 15 years under different ripping treatments, and compared them with a nearby natural analogue reference site, in the sandy soils of the Swan Coastal Plain, located in south-west Western Australia with a warm Mediterranean climate. We hypothesized that hydrological function would improve with time (relative to reference targets), and that the presence of vegetation and implementation of deep-ripping techniques would contribute significantly to this improvement.

**METHODS**

**Study Area**

The study was conducted at the Rocla Quarry Products sand extraction facility, located approximately 30 km north-east of Perth, Western Australia \((31.7730°\ S, 115.8610°\ E)\). The facility is located on the deep siliceous sands of the Bassendean Dune System of the Swan Coastal Plain (McArthur et al. 1991), and experiences a Mediterranean climate characterized by hot, dry summers and cold, wet winters. Approximately 80% of the annual rainfall (680 mm) falls between May and October (Bureau of Meteorology. 2013).

The native vegetation comprises small to medium shrub species from the Myrtaceae, Fabaceae and Epacridaceae families, with scattered dominant tree species including *Banksia attenuata* R. Br, *Banksia menziesii* R. Br, and *Banksia ilicifolia* R. Br. Banksia are Proteaceae and therefore produce thick lateral expanses of proteoid root
mats. Whilst the mats are short-lived (Shane et al. 2004), the fibrous root mat remains are long-lasting and may cover substantial areas (Jeffrey 1967). The role of root mats in nutrient acquisition is well documented e.g. (Jeffrey 1967, Grierson and Attiwill 1989, Lamont and Bergl 1991, Shane et al. 2004), yet their effect on hydrological properties is currently unclear.

The quartz sands in this area are of significant economic value as sources of silica and building materials and have been extracted commercially since the late 1980’s. Prior to sand extraction, native vegetation is cleared and topsoil (top 10 cm) is collected. The topsoil sand typically comprises 94% coarse sand and 1% clay and whilst higher in nutrients than lower horizons, is still regarded as nutrient impoverished (McArthur et al. 1991). The extraction process involves the removal of dunes to 30 m in height, leaving an entirely new surface that is situated only 2 - 5 m above the groundwater level (Rokich et al. 2001). Post-extraction, restoration methodologies have evolved since the early 1990’s where restoration consisted of emplacing overburden material (leached grey-quartz sands of the upper soil profile that usually extend for approximately 1 m below the topsoil) and topsoil (McArthur and Bettenay 1960). In 1996, overburden was no longer returned to the soil profile as research indicated that it significantly contributed to the development of compaction (Rokich et al. 2001). Additionally, deep-rippling to approximately 1 m depth was introduced to further assist in the alleviation of soil compaction. In 2003, deep-rippling was replaced by shallow-tilling (approx. 15 – 30 cm). Following these site preparations, topsoil is spread to a depth of approximately 10 cm across the site, and a locally-sourced seed mix comprised of species that may not exist in sufficient viable numbers in the replaced soil seed bank (within the topsoil), are dispersed by hand. Seed dispersal is timed to coincide with the commencement of winter rains, the so-called “break of season” (usually early June).

Restoration activity over the last 20 years has produced a chronosequence, enabling research into changes in hydrological functioning over time, and assessment of the effects of biological activity (vegetation cover) and management intervention techniques (substrate ripping). Restoration sites aged 0, 3, 7 and 15 years were assessed (Figure 1), that included sites aged 0 and 15 years that had been subjected to deep-rippling (~ 1 m) using a Caterpillar D12 bulldozer equipped with a one meter tyne, while sites aged 0 – 7 years were shallow tilled (30 cm) using a Caterpillar 963 Traxcavator, and a 15 year-old site that had not been ripped or tilled. We classified sites that had been shallow-tilled or not ripped as not ripped.
Figure 1: Map depicting mine layout and location of sampling sites across restoration chronosequence and reference sites. 1 = 0 years, deep rip, 2 = 0 years, shallow rip, 3 = 3 years, shallow rip, 4 = 7 years, shallow rip, 5 = 15 years, no rip, 6 = 15 years, deep rip, 7 = reference.

**Sampling Strategy**

Within each restoration site, a 28 m x 28 m grid with sample points every 4 m was established (Figure 2). Ten randomly selected points within the grid system were then selected to have 3 satellite sample points located on random azimuths 0.5, 1.0, and 2.0 m (Gwenzi et al. 2011) from the principal point, equating to a total of 94 sample points at each site. This allowed for variation to be assessed at a variety of spatial scales, as well as introducing a random component to the sampling strategy.
Figure 2: Example of sampling strategy utilized at each site across the restoration chronosequence and the reference site. Each site consisted of 94 sample points; 64 points in a 4m x 4m grid array (indicated by circles), and 30 satellite points (indicated by triangles) divided into 10 subsets of 3 located on random azimuths from a randomly allocated grid point, at 0.5, 1.0, and 2.0m distances. Numbers indicate dimensions in meters.

Data Acquisition

Saturated hydraulic conductivity ($K_s$) and penetration resistance (PR) were used as indicators of ecosystem function and assessed at each sample location in spring, 2011. $K_s$ was measured at the soil surface using a Phillip-Dunne permeameter according to the field protocols outlined in (Muñoz-Carpena 2002, Munoz-Carpena et al. 2002), and calculated using an automated program, “Estimation of field saturated hydraulic conductivity (V:1.0)” (Muñoz-Carpena 2002). The method is a falling-head approach that requires the permeameter to be installed just below the soil surface. To facilitate this, a 20 mm “cookie cutter” extension was attached to the base of the permeameter to remove the surface soil thus reducing the potential for water pressure to cause upwards water flow and leakage, violating the assumptions of the method. Penetration resistance was measured using a CP20 cone penetrometer (RMIK) at 2 cm intervals, to a maximum depth of 50 cm. In instances where soil resistance prevented the full insertion of the penetrometer, the maximum value measured was extrapolated for the remaining portion of the 50 cm.
The location of each sample point was recorded to an accuracy of < 0.10 m using a Trimble Pathfinder Differential Global Positioning System (DGPS) with “Tornado” antenna. A visual assessment of the presence/absence (absence < 50% > presence) of superficial roots at each sample point (defined by the circumference of the permeameter) was recorded in order to statisticallyanalyse the effect of roots on Ks.

Vegetation cover maps were produced from high spatial resolution, Colour Infrared (CIR), and Red, Green, Blue (RGB) imagery. Imagery was collected on 15 September, 2011, using a Zeiss LMK 2000 camera with 152 mm lens and Kodak Aerochrome III Infrared film 1443, with a spectral range of 200-900 nm (Survey Graphics, Perth, Western Australia). Data was collected in two runs at 1:4000 comprising a total of 30 images. Imagery was controlled by airborne DGPS and transformed using ground control points to a horizontal accuracy of < 5 cm. Imagery was scanned at 12 micron output pixel size which equated to a ground equivalent of 4.8 cm pixel resolution. The frames were colour-balanced and seamlessly mosaicked to produce a single orthomosaic for analyses.

Images were resampled to 10.0 cm pixel resolution to reduce data volumes for processing. ERDAS Imagine 2011 (ERDAS) was used to apply a Normalized Distribution Vegetation Index (NDVI) classification, enabling a clear distinction between living biomass (vegetation patches) and dead biomass/bare soil (gaps). A supervised classification was then applied to the NDVI image in order to create a binary representation of these two categories (Figure 3). Inspection of the high-resolution aerial images in conjunction with 282 colour photographs taken at ground level of sample points from sites aged 3, 7, and 15 years were used to ground truth the final classification. A classification accuracy of 82% was achieved.
Figure 3: Binary classification created from CIR aerial imagery depicting vegetation cover at each test site along the restoration chronosequence, excluding sites aged 0 years, which were devoid of vegetation. Vegetation is represented by grey and bare soil/non-vegetation by white. Black markers indicate hydrological sampling points.
Data analyses

To determine the relationship between vegetation and soil hydrological properties, vegetation cover at sample points were assessed at two scales; up to 25 cm from the sample point, and up to 50 cm from the sample point. These buffer zones were created to account for the fact that the influence of vegetation on soil hydrological characteristics below-ground, may exceed the extent of the above-ground biomass (Thompson et al. 2010). Additionally, buffers alleviated error associated with georeferencing and classification. Buffers were created using ArcGIS 10.0 (ESRI), with the percentage of pixels classified as vegetation within each buffer being used as a measurement of vegetation cover. No significant differences in % cover were noted between the two spatial scales selected and only data pertaining to 25 cm is presented herein. Linear regressions were carried out between vegetation cover and \( K_s \) and penetration resistance values at each sample point to determine the effect of cover on hydrological function. Mean values of hydrological properties at each site were calculated and compared to identify the effect of ripping treatments.

Statistical Analyses

Comparison of hydrological properties (\( K_s \), \( PR \) 10cm, \( PR \) 50 cm) between sites was conducted using the Kruskal-Wallis post-hoc multiple comparison function from the “pgirmess” package (Giraudoux 2011) in the statistical analyses program, R V 3.0 (R Core Team 2013). Stepwise regression of Generalized Linear Models (GLM’s) (‘mgcv’ package; Wood 2001) were used to identify and select the variables that best predicted hydrological properties, based on the lowest Akaike Information Criteria (AIC) score. Statistical modelling was used to illustrate interactive effects of vegetation and ripping treatment over time. Predictions pertaining to 100% vegetation cover were omitted from analysis of fresh (0 year) restored sites.

RESULTS

Within-site variation of hydrological properties

Coefficient of Variance (CV) in \( K_s \) generally increased in time up to seven years and then declined (Table 1). In 0-year-old sites, CV was 40% greater in deep-ripped sites than in non-deep-ripped sites, and approximately equal in deep-ripped sites aged 0
and 15 years. The greatest variation (7 year-old site) was approximately twice that recorded in the reference site. CV of PR 10 was similar for restored sites < 7 years old, after which CV declined. Again, variation in the 7 year-old site was twice that of the reference site. CV of PR 50 was lowest in the 0 year-old, shallow-ripped site, which was also markedly lower than in the 0 year-old, deep rip site. Fifteen year-old sites showed the highest variation, and were at least three times that of the reference site.

Table 1: Coefficient of Variance values of hydrological properties in sites of different age and disturbance depth (cm). $K_s$ is saturated hydraulic conductivity (m day$^{-1}$), PR 10 and PR 50 are penetration resistance at 10 cm and 50 cm respectively.

<table>
<thead>
<tr>
<th>Property</th>
<th>Site Age and Ripping Type</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>$K_s$</td>
<td>0 deep 3 7 15 15 deep</td>
<td>50 29</td>
</tr>
<tr>
<td>PR 10</td>
<td>48 51 52 60 29 31</td>
<td></td>
</tr>
<tr>
<td>PR 50</td>
<td>7 53 16 67 74 64</td>
<td>17</td>
</tr>
</tbody>
</table>

Effect of cover and ripping on $K_s$

Mean $K_s$ in deep-ripped sites did not differ significantly with time since restoration, and was approximately equal to the average $K_s$ calculated for the reference site (Figure 4a). $K_s$ was approximately 50% higher in 0 year-old deep-ripped sites in comparison to 0 year-old shallow-ripped sites ($p < 0.05$). In shallow-ripped sites, $K_s$ reduced by approximately half in the first three years and did not change significantly by 15 years, remaining significantly lower than the mean $K_s$ recorded in the fresh and 15 year-old, deep-ripped site, and the reference site ($p < 0.05$). No significant correlations were evident between vegetation cover and $K_s$, in either the reference site or any of the restored sites (Figure 4b).
Figure 4: a) Correlation between $K_s$ and vegetation cover in deep and shallow ripped sites of different ages and in the reference site; b) $K_s$ values in selected sites from the restoration chronosequence and reference site. Ripping treatments are denoted as either “shallow” or “deep.” Means indicated by triangles, median values by horizontal bars, interquartile range by boxes and whiskers, and outliers (>1.5 times the interquartile range) by circles.

Modelling indicated that deep ripping was the most significant determinant of $K_s$ (Table 2). According to the model, $K_s$ declined by approximately 25% and 45% in deep and non-deep ripped sites respectively over the first 7 years (Figure 5). $K_s$ in both ripping treatments was modelled to remain relatively constant thereafter, with some
suggestion that $K_s$ may increase slightly from 12 years. Modelling sites with vegetation cover set to 100% produced a marked increase $K_s$ over the first 10 years.

**Table 2**: Variables selected via step-wise regression analysis to explain and predict $K_s$, as determined by the lowest AIC value. "::" indicates an interaction.

<table>
<thead>
<tr>
<th>Covariate</th>
<th>Coefficient</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>2.7</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age</td>
<td>-0.3</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age (2\textsuperscript{nd} order)</td>
<td>&lt;0.1</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>% Cover</td>
<td>&lt; -0.1</td>
<td>0.021</td>
</tr>
<tr>
<td>Ripping deep</td>
<td>1.4</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age:% Cover</td>
<td>&lt;0.1</td>
<td>0.054</td>
</tr>
<tr>
<td>Age:Ripping deep</td>
<td>&lt;0.1</td>
<td>0.011</td>
</tr>
<tr>
<td>Age (2\textsuperscript{nd} order):% Cover</td>
<td>&lt;0.1</td>
<td>0.088</td>
</tr>
</tbody>
</table>

**Figure 5**: Modelled effects of vegetation cover and ripping depth on $K_s$ through time.

**The effect of roots on $K_s$**

Superficial roots (surface ~ 3-4 cm) were widely distributed throughout the reference site (48% of sample points), and to a lesser extent in the 15 year-old restored sites with both deep (29% of sample points) and shallow (27 % of sample points)
ripping treatments. High root density was significantly related with reduced saturated hydraulic conductivity (p < 0.05). Mean $K_s$ in 15 year-old restored sites and the reference sites was 35% lower (2.6 m day$^{-1}$) in areas with low root density than sample points classified as having dense roots (4.0 m day$^{-1}$).

**Penetration resistance at 10 cm depth**

Mean penetration resistance in the soil surface (10 cm depth) was approximately three times lower in the shallow-ripped fresh site compared with the deep-ripped fresh site, and approximately two times lower than penetration resistance values recorded for subsequent years and treatments (Figure 6b). PR 10 cm increased significantly (p < 0.05) between 0 and 3 years and then did not change significantly in shallow ripped sites up to 15 years old. Both shallow and deep ripped sites aged 15 years did not differ from the reference. PR 10 cm was not significantly correlated with $K_s$ ($R^2 < 0.01$), nor the presence of proteoid root mat. No significant correlations were evident between vegetation cover and PR 10 cm in either the reference site or any of the restored sites (Figure 6a).
Figure 6a) Penetration resistance values at 10 cm depth in selected sites from the restoration chronosequence and reference site. Ripping treatments are denoted as either “shallow” or “deep.” Means indicated by triangles, median values by horizontal bars, interquartile range by boxes and whiskers, and outliers (>1.5 times the interquartile range) by circles. b) Correlation between PR 10 cm and vegetation cover in deep and shallow ripped sites of different ages and in the reference site.

Penetration resistance in sites that had been deep-ripped was significantly greater than sites that had not been deep-ripped (Table 3). Vegetation cover and site age were also associated with a significant increase in penetration resistance at 10 cm. Site age and deep ripping interacted to decrease penetration resistance (p < 0.05), as did site age and vegetation cover (p < 0.05). The model explained 25% of total variation in penetration resistance at 10 cm depth. Predictive modelling indicated that the effect of deep ripping and non-deep ripping on penetration resistance in the soil surface stabilizes.
at around 15 years (Figure 7). Likewise, vegetation cover effects are no longer significant 15 years after restoration initiation.

Table 3: Variables selected via step-wise regression analysis to explain and predict PR 10 cm, as determined by the lowest AIC value. “2nd order” indicates a second order polynomial term. “::” indicates an interaction.

<table>
<thead>
<tr>
<th>Covariate</th>
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<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
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<tr>
<td>Age</td>
<td>83.6</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age (2nd order)</td>
<td>-4.0</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>% Cover</td>
<td>5.7</td>
<td>0.022</td>
</tr>
<tr>
<td>Ripping deep</td>
<td>487.3</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age:% Cover</td>
<td>-2.0</td>
<td>0.004</td>
</tr>
<tr>
<td>Age:Ripping deep</td>
<td>-30.1</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age(2nd order):% Cover</td>
<td>0.1</td>
<td>0.003</td>
</tr>
</tbody>
</table>

Figure 7: Modelled effects of vegetation cover and ripping treatments in restored sites through time.

Penetration resistance at 50 cm depth

No significant correlations were evident between vegetation cover and penetration resistance at 50 cm, in either the reference site or any of the restored sites (Figure 8). Mean penetration resistance at 50 cm depth was nearly twice as high in the fresh shallow ripped site as in the fresh deep ripped site (p < 0.05)(Figure 8b). Shallow
ripped sites indicated a decreasing trend in penetration resistance at 50 cm over the first 7 years following restoration, while deep ripped sites did not change significantly over time (P < 0.05). Penetration resistance in 7 year-old, shallow-ripped sites, and both shallow and deep-ripped, 15 year-old sites did not differ significantly from each other, but were all significantly higher than the reference site. A high degree of variation was evident in shallow ripped sites after 3 years and deep-ripped sites, compared to the reference site.
Figure 8a) Penetration resistance values at 50 cm depth in selected sites from the restoration chronosequence and reference site. Ripping treatments are denoted as either “shallow” or “deep.” Means indicated by triangles, median values by horizontal bars, interquartile range by boxes and whiskers, and outliers (>1.5 times the interquartile range) by circles; b) Correlation between PR 50 cm and vegetation cover in deep and shallow ripped sites of different ages and in the reference site.

Modelling suggests that vegetation cover did not significantly affect penetration resistance at 50 cm in restored sites and was not included in the final model. The final model explained 30.9% of the total variance (Table 4). Even when young sites were removed (0-7 years inclusive), vegetation cover was still not significantly correlated with penetration resistance at 50 cm (p > 0.05). The interactive effect of time and deep
ripping was the most important in determining penetration resistance at 50 cm. The effects of deep ripping are significant in the early years following restoration, but converge with shallow-tilling after 12-13 years (Figure 9). In the reference site, a significant correlation was identified between penetration resistance at 50 cm and vegetation cover (p < 0.05), in which increasing vegetation cover from 0% to 100% was modelled to increase PR 50 by approximately 200 KPa (not shown).

Table 4: Variables selected via step-wise regression analysis to explain and predict PR 50 cm, as determined by the lowest AIC value.

<table>
<thead>
<tr>
<th>Covariate</th>
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<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>(intercept)</td>
<td>5042.1</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age</td>
<td>-465.7</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age (2\textsuperscript{nd} order)</td>
<td>19.2</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Ripping deep</td>
<td>-2400.5</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Age:Ripping deep</td>
<td>188.0</td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>

Figure 9: Modelled effects of ripping depth on PR 50 cm through time.
DISCUSSION

This study shows that parameters driving global patterns of hydrological function are not as important in restored sites where site age and substrate manipulations are significant confounding factors. Within-site variability of $K_s$ values recorded in this study were an order of magnitude less than has commonly been recorded in other studies of natural sandy soils and reconstructed ecosystems, where CV ranges from 100-400 (Bagarello 1997, Reynolds et al. 2000, Buczko et al. 2001, Salama et al. 2005, Botros et al. 2009). In restored sites, reduced variation in $K_s$ is likely attributable to site-wide, homogenous substrate treatments, such as substrate ripping to a uniform depth. Variation was observed to generally increase with age over the first seven years, and this may be the due to heterogeneous plant-soil interactions increasing as vegetation cover increases with site age. Additionally, whilst the upper limit of the range of $K_s$ was similar to those of other studies in deep alluvial sandy soils (0.05-14.5 m day$^{-1}$; Botros et al. 2009) and in another study on the Swan coastal plain (0.4- 7.3 m day$^{-1}$; Salama et al. 2005), the lower limit of the range (0.002 – 10.09 m day$^{-1}$) of $K_s$ recorded across the chronosequence in the present study was an order of magnitude lower than the previously mentioned studies. This may be indicative of higher bulk densities associated with post-mine soils (Chong and Cowsert 1997, Archer et al. 2002, Benigno et al. 2012a).

The effect of site age in determining $K_s$ in restored sites was heavily mediated by ripping treatment in the 15 year span covered by the restoration chronosequence. Site age was an important initial factor in predicting $K_s$, however this was a transient dynamic in soil physical properties in shallow-ripped sites, as no marked evolution in $K_s$ was apparent after the first 3-5 years. These apparent settling processes in the early years following site reconstruction may be attributable to particle sorting brought about by wetting and drying cycles (Kozlowski 1999, Thompson et al. 2010), which in turn increase bulk density/penetration resistance, and slow the rate of water movement through the soil (Shukla et al. 2004). Settling processes in deep-ripped sites were not examined in the present study due to limited replication of the treatment within the restoration chronosequence, and mean $K_s$ at 0 years was statistically equal to mean $K_s$ at 15 years. However, in a study of deep-ripping in a post-mined silt-loam, Chong and Cowsert (1997) noted that the effects of deep tillage declined over the first three years following site reconstruction. This presents two alternative possibilities; sorting of substrate materials via wetting and drying cycles is slower in a predominantly sand
system due to a greater proportion of coarse particles, or; biotic processes including root action and soil faunal activity (Greenwood and McKenzie 2001) may have restored $K_s$ to freshly ripped values by 15 years. After 15 years, $K_s$ in non-deep-ripped sites remained at half the rate recorded in natural analogues. The recovery of soil properties in post-mined landscapes is notoriously slow (Allen 1993, Ruiz-Jaen and Mitchell Aide 2005), yet a fundamental restoration objective is to hasten these processes (Hendrychová 2008). The present study found that deep ripping to approximately one meter produced mean $K_s$ rates statistically equal to those recorded in the natural analogue, and supports the employment of this treatment. Chong and Cowsert (1997) also found that ripping to 80cm resulted in the greatest increases in infiltration rate compared with shallower ripping treatments, stating that significantly lower $K_s$ in non-ripped sites was caused by the wetting front slowing once it reached the bottom of the tilled layer.

Mediterranean systems are water-limited, and we expected the presence of vegetation to be associated with increased $K_s$ (HilleRisLambers et al. 2001, Thompson et al. 2010). This trend was generally supported, although the effect was mild. Additionally, we noted that in older restored sites and natural analogue sites, patches of dense, fibrous, proteoid root mats were associated with significantly decreased $K_s$. It seems likely that this is the product of plant roots occupying soil pore spaces causing a reduction in the rate at which water can move through the substrate (Archer et al. 2002, Shukla et al. 2004) (Shukla et al. 2004). In a review of global trends of vegetation-infiltration feedbacks across environmental gradients, Thompson et al (2010), found that the correlation was strongest in substrates characterized by high clay fractions. In water-limited, sandy substrates, root morphologies that reduce infiltration rate may be advantageous to plants, enabling them to retain moisture in the rhizosphere for longer, hence increasing plant-available water.

Overall, penetration resistance increased with time in deep-ripped sites, and decreased in non-ripped sites, indicating that the substrates move towards equilibrium irrespective of the initial conditions. In non-ripped sites, penetration resistance on the surface increased with age in the first years (< 3 years) following reconstruction, confirming observations of Rokich et al (2001) who found that penetration resistance in the same system increased from 5 times that of natural analogues in newly reconstructed sites, to up to 10 times natural analogue values in the first 6 years after site reconstruction, and is indicative of natural settling processes following substrate
disturbance. Rokich’s study, however, did not assess beyond 6 years, and our finding show that, at least at the soil surface, penetration resistance does not change significantly beyond this point in time. It is unclear why penetration resistance at 10 cm was up to three times higher in fresh deep ripped sites than fresh non-deep ripped sites given the surface portion of the substrate is always highly disturbed in the reconstruction process, and is hence not a pre-existing condition. The most likely scenario is that heavy machinery involved in the site reconstruction process may have inadvertently recompacted some areas of the site (e.g. whilst turning), post ripping.

Penetration resistance at 50 cm depth in non-ripped sites also declined significantly in the first 5-7 years following restoration, and may be a result of processes associated with soil decompression following removal of overburden. Wetting and drying cycles, growth and decay of plant roots, and the action of soil animals then facilitate the recovery of soil physical properties to a new equilibrium (Greenwood and McKenzie 2001). This contradicts the findings of Rokich (2001), who found that penetration resistance at depth increased in the first 6 years following site reconstruction. Penetration resistance under deep-ripping in 0-year-old restored sites was significantly lower than 0 and 3 year-old sites that had not been ripped. From 7 - 15 years there was no difference between shallow and deep ripped sites, indicating that natural rebound processes are as effective in restoring hydrological function as management intervention techniques over this time period. Given that Banksias in natural conditions grow a tap root in excess of 70 cm in the first year (Benigno et al. 2012b), reducing penetration resistance in newly restored sites is likely to be critical to Banksia survival. However, even after 15 years, mean penetration resistance in both shallow and deep-ripped sites was significantly greater than in the natural analogue, often exceeding 2000 KPa. Penetration resistance values greater than 2000 KPa severely retard root penetration (Bengough and Mullins 1991, Bennie 1991) and have been associated with low root densities in a study in the same system (Rokich et al. 2001). These results suggest that substantially more time is required to restore hydrological function at depth.

We expected that vegetation cover would be related to decreased penetration resistance, as this has been a common association in previous studies of natural arid and semi-arid systems (Facelli and Brock 2000, Puigdefàbregas 2005). This was not supported in our study, in either restored or reference sites. At the surface, proteoid root mats, well beyond the extent of above ground biomass, may have affected penetration
resistance values, and confounded any attempt to predict soil hydrological properties based on a binary classification of vegetation patches and gaps. Penetration resistance deeper in the profile (50 cm) was significantly correlated with vegetation cover, although weakly, which may indicate that larger plants could establish in less dense soils or that high cover had a greater potential for deeper roots to weaken penetration resistance at depth.

This study confirmed that deep ripping as a remediation technique can rapidly restore hydrological function in post-mining landscapes. Within one year of deep-ripping, $K_s$ and penetration resistance values in restored sites were equal to reference targets, while $K_s$ in non-deep-ripped sites was half that of the reference site after 15 years. Furthermore non-deep-ripped sites took 3-7 years to restore surface penetration resistance to reference site levels. Penetration resistance at 50 cm was significantly reduced immediately following deep-ripping compared with non-ripped sites, and this could be critical for early root development and plant survival in water-limiting environments (Benigno et al. 2012b).

High-resolution, binary classification of remotely-sensed imagery was employed in this study to measure vegetation cover within an arbitrary 25 cm radius of hydrological sampling points. This methodology partially accounts for vegetation-hydrology feedbacks extending beyond the point of sampling (Rietkerk and Van de Koppel 2008), but still essentially assumes a binary relationship between the two. Some studies explicitly assessing the spatial distribution of soil hydrological properties have found that the effects decline with distance from the stem, and hence vegetation-infiltration associations should be assessed at the individual plant scale, rather than the patch scale (Barbier et al. 2008). The simplification of this relationship in the present study may have limited the accuracy of the results.

Soil compression is a major driver of seedling mortality in restored sites (Enright and Lamont 1992), and its alleviation, enhancing infiltration and reducing penetration resistance, particularly at depth, may contribute to significantly improving restoration outcomes. This study confirmed deep-ripping as a beneficial treatment for the rapid restoration of hydrological function in post-mine disturbance sites, where plant-soil interactions play a secondary yet nonetheless important role.
Implications for restoration

- Vegetation – soil feedbacks driving hydrological properties in many natural systems are confounded by substrate ripping treatments in restored landscapes.
- Infiltration rate is rapidly restored by deep ripping (≥ 80cm) of the substrate
- Deep-rip and rebound processes are both important for restoring hydrological function at depth.
CHAPTER FOUR

Relationships between vegetation spatial pattern, time, treatment, and hydrological function within a restoration chronosequence.

INTRODUCTION

The relationship between vegetation spatial pattern and ecosystem function is well established, particularly in arid and semi-arid systems where facilitative and competitive processes acting at various spatial scales produce distinctive regular patterns including spots, bands, and tiger stripes (Rietkerk and Van de Koppel 2008). Typically, facilitation occurs over a short-range resulting in the formation of vegetation patches, whereby neighbouring plant individuals benefit from being situated in close proximity to one another’s canopy, stem or root zone (Breshears et al. 1998). Patches, often referred to as “resource islands” (Schlesinger et al. 1996) play a critical role in concentrating resources through obstructing surface flows of wind and water, capturing runoff, nutrients and sediments, reducing evapo-transpiration pressure, and increasing infiltration rates (Ludwig et al. 1999). Competitive processes generally begin to predominate at a longer range due to the extension of lateral roots well beyond an individual’s above-ground biomass (Rietkerk and Van de Koppel 2008). Without the facilitative benefits of neighbouring plants, new individuals may struggle to establish in the “competitive zone,” which in turn may generate further negative feedbacks. Areas devoid of vegetation (inter-patches) are known as resource “sources” (Ludwig et al. 2005), and promote rainfall runoff and provide clear areas for wind-borne matter to pass through (Bastin et al. 2002).

The interplay of patch-gap dynamics has been described as a self-organising mechanism that acts to maximize the overall productivity of the landscape (Noy-Meir 1973, Ludwig and Tongway 1995). As resource availability increases, smaller source areas can sustain larger vegetation patches, eventually resulting in adjacent patches coalescing. Additionally, the size, shape, orientation and distribution of vegetation patches have a significant effect on a landscape’s capacity to retain resources (Bastin et al. 2002).

The relationship between landscape vegetation pattern and resource availability has been utilized as proxy for ecosystem function and is the foundation of the well-
known Landscape Functional Analysis (LFA), developed for rangeland and mine site restoration monitoring in Australia (Tongway and Hindley 2004). Indeed, the Society for Ecological Restoration’s (SER) primer on ecological restoration states that a restored ecosystem is one that contains “sufficient biotic and abiotic resources” to “sustain itself structurally and functionally” (Clewell et al. 2004). Implicit in this is the requirement for vegetation pattern (structure) in restored systems to follow a trajectory that culminates in a similar spatial arrangement to that of the target state/natural analogue. Deviations from this trajectory likely indicate a shift in resource availability relative to the natural analogue site.

The degree of change in resource availability in restored sites compared to natural analogues is dependent on the degree of disturbance that has occurred. Post-mining landscapes often undergo drastic substrate disturbance and may be characterised by highly altered hydrological regimes, whereby plant-available water is reduced. Often, these changes are linked to decreased infiltration rates and increased soil impedance (Shukla et al. 2004), which may in turn reduce root growth (Benigno et al. 2012a). In water-limited environments, this may be a key driver of plant mortality (Enright and Lamont 1992, Rokich et al. 2001), and is likely to manifest as changes in vegetation spatial pattern. Restoration treatments such as deep-ripping are often employed to ameliorate these conditions and increase resource (i.e. soil water) availability in the system (Chong and Cowsert 1997).

As the monitoring of restored land collectively increases, new cost and time effective techniques are required. However, the majority of research relating spatial pattern to ecosystem functional attributes has been restricted to arid and semi-arid systems, where pattern tends to consist of regular, discrete vegetation patches on a matrix of bare soil. Mediterranean systems represent an increase in resource availability compared with arid and semi-arid systems, yet experience resource-limitation during the hot, dry summer period. Although not as well-defined, vegetation pattern in these climates may exhibit patchiness, and hence can conceivably be used as a measure of function.

The aim of this paper was to assess, for the first time, whether irregular vegetation pattern, characteristic of undisturbed Mediterranean Banksia woodlands, develops in a predictable manner as a function of time and as a function of restoration treatment, in order to support the use of pattern metrics as a monitoring tool for
restoration in these systems. We compared vegetation pattern in undisturbed Mediterranean Banksia woodlands with the development of pattern in an 18 year-old, post-mining, restoration chronosequence. Plant community composition, and hence, vegetation structure in Banksia woodlands, is driven by plant-water availability and is known to vary with landscape topography (Groom 2004). To account for this variation, we compared restored sites with natural analogues in both high and low landscape positions. We hypothesized that pattern development would follow a predictable trajectory that could be characterized by selected landscape metrics as a product of time. Restored sites are typically situated low in the landscape, and hence we expected that vegetation pattern in restored sites would follow a trajectory towards pattern characteristic of a low landscape position. We then utilized data from a previous study (Chapter Two) to examine how hydrological attributes relate to vegetation spatial pattern, hypothesising that changes in landscape metrics would be accompanied by significant changes in hydrological function, but that function would also be heavily mediated by substrate ripping treatments.

METHODS

Study Area

The study was conducted at the Rocla Quarry Products sand extraction facility, located approximately 30 km north-east of Perth, Western Australia (31.7730° S, 115.8610° E). The facility is located on the deep siliceous sands of the Bassendean Dune System of the Swan Coastal Plain. These quartz sands hold significant economic value and have been extracted since the early 1990’s. The climate is Mediterranean, with an annual rainfall of approximately 680 mm (Bureau of Meteorology. 2013) (Figure 1). The native vegetation is classified as Banksia woodlands, comprised of three dominant overstorey species; Banksia menziesii R. Br, Banksia attenuata R. Br, and Eucalyptus todtiana F. Muell, along with a highly diverse community of shrubs and herbaceous species (Dodd et al. 1984).
Figure 1: Mean monthly rainfall (left vertical axis) and mean maximum monthly temperature (right vertical axis) at the study site, located 30 km north-east of Perth, Western Australia (Bureau of Meteorology. 2013).

The Mining and Restoration Process

Mining is typically focused on dune features (as opposed to the swales between dunes). In some cases, these dunes may exceed 30 m in height. Vegetation is cleared and the topsoil (top 10 cm) is direct transferred to sites awaiting restoration. The extent of mining is limited by the water table, with extraction generally ceasing within 2 – 5 m of this point (Rokich et al. 2001).

The restoration process begins with the spread of topsoil to a depth of approximately 10 cm across the site. The site is then ripped or tilled as a means of alleviating soil compression; a condition resulting from the new surface being previously buried many meters below the surface. Ripping was first introduced in 1996, and was carried out using a Caterpillar D9 bulldozer that ripped the substrate to approximately 50 - 80 cm (deep-ripping) with a single tyne at roughly 50 cm spacing. From 2000 onwards, deep-ripping was replaced by shallow-tilling (15 - 30 cm depth at 20 cm spacing) using a Caterpillar 963 Traxcavator. Following substrate reconstruction, a standard, locally-sourced seed mix is dispersed by hand across the site. The seed mix is comprised of species that may not exist in sufficiently viable numbers in the replaced topsoil. Seed dispersal is timed to coincide with the commencement of rains in autumn or early winter (usually early June), locally known as the “break of season”. Restoration
activities first began at the site in 1993, and have been carried out most years since, creating a restoration chronosequence spanning nearly 20 years (Figure 2).

Figure 2: Left: Aerial image of the active mine and the northern restoration sites (age range from 3 – 8 years) used in pattern analyses (indicated by black border). Reference site indicated by yellow border. Red border indicates high landscape position; Right: Aerial image of southern restoration sites (age range from 4 – 18 years) used in pattern analyses (indicated by black border).

Data Acquisition and Processing

Vegetation pattern was assessed using high spatial resolution, remotely-sensed, aerial imagery. Colour Infrared (CIR) and Red, Green, Blue (RGB) imagery was collected on 15 September, 2011 at 0.048 m pixel resolution and a horizontal accuracy of < 10 cm, using a Zeiss LMK 2000 camera with 152 mm lens by Survey Graphics, of Perth. The software ERDAS Imagine 2011 (ERDAS) was used to apply a Normalized Distribution Vegetation Index (NDVI) classification to the images. Based on the assessment of 282, georeferenced digital colour photographs taken at ground level and used for ground-truthing, an NDVI threshold value of 0.05 was found to accurately
distinguish living biomass (vegetation patches > 0.05) from dead biomass/bare soil (gaps <0.05) allowing all imagery to be binary classified (Figure 3). A mean classification accuracy of 82% was achieved. Georeferencing was conducted using a Trimble Pathfinder Differential Global Positioning System (DGPS), with an accuracy of <0.015 m. Imagery was resampled to 0.1 m pixel resolution to reduce data volumes for further processing. All restoration sites were separated according to age, and tracks, roads, boundary lines, etc. were removed from each site image. Shadow was minimized by collecting imagery around midday, and was present in only very small proportions in restored sites where vegetation was relatively low. A greater proportion of shadow was evident in natural analogue sites but its removal was deemed to have a greater effect on landscape metrics than its inclusion. In natural analogue sites where vegetation cover ranged from 65 – 85%, shadow was classified as vegetation, as it was more likely to fall upon vegetation than bare ground.

**Spatial Analyses**

Vegetation pattern in each site was determined using the spatial analysis program, Fragstats 4.1 (McGarigal et al. 2012). Sites were assessed in terms of the following metrics.

*Percent cover* is the proportion of the total site comprised of vegetation and is the most commonly used metric for describing vegetation development in restoration monitoring, and indeed, in landscape characterization. It is arguably the most intuitive indicator of resource availability in a system, as well as being a major determinant of a system’s capacity to capture and retain resources (Bastin et al. 2002). All other metrics are inherently related to cover.

\[
\text{cover} = P_i = \frac{\sum_{j=1}^{n} a_{ij}}{A} (100)
\]

where \(P_i\) = proportion of the landscape occupied by patch type (class) \(i\), \(a_{ij}\) = area (m\(^2\)) of patch \(ij\), and \(A\) = total landscape area (m\(^2\)) (McGarigal et al. 2012)
Weighted Mean patch size (WMPS) uses the frequency distribution of different patch sizes within a landscape to calculate the probability that a pixel within the image is occupied by a patch of a particular size, and the probability that a patch of that particular size occurs within the bounds of the image (landscape). Hence, it provides a measure of the most likely patch size that an organism, placed at random within the landscape, will find itself in (McGarigal et al. 2012)

\[
WMPS = \sum_{j=1}^{n} \left[ x_{ij} \left( \frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}} \right) \right]
\]

where i and j are patch types (patches and gaps), x is the proportional abundance, and a is the area (m²) (McGarigal et al. 2012).

Patch size coefficient of variance (Patch CV) is a second order statistic that provides a measure of patch size variability, and hence, landscape heterogeneity. Greater variability may indicate differences in underlying processes within the landscape (McGarigal et al. 2012), or be indicative of a diverse plant community, and/or the occurrence of recruitment events.

\[
Patch\ CV = \frac{SD}{MN} (100)
\]

where SD is the patch size standard deviation and MN is the mean patch size

Euclidean Nearest Neighbour (ENN) is determined by the shortest distance between the edge of a focal patch and the nearest edge of an adjacent patch, and is the mean of all such relationships within the image extent (McGarigal et al. 2012). A relatively high NN may indicate that larger source areas are required to support sinks. Larger gaps between vegetation patches enables wind and water to transport resources out of the landscape more readily, indicating reduced function (Bastin et al. 2002).

\[
ENN = h_{ij}
\]
where \( h_{ij} \) = distance (m) from patch \( ij \) to nearest neighbouring patch of the same type (class), based on patch edge-to-edge distance, computed from cell centre to cell centre (McGarigal et al. 2012)

**Proximity index** is equal to the sum of the vegetation patch area (m\(^2\)) divided by the nearest edge to edge distance squared (m\(^2\)) between the focal patch and all patches within a specified buffer distance of the focal patch (McGarigal et al. 2012). For the purpose of this study, a buffer distance of 2 m was selected, based on the largest recorded mean NN plus 1 standard error (3 year-old site). The Proximity index integrates vegetation patch area and spacing, and identifies sites with many sparsely distributed small patches from sites characterized by large, closely spaced patches (Bastin et al. 2002).

\[
\text{Proximity} = \sum_{s=1}^{n} \frac{a_{ijs}}{h_{ijs}^2}
\]

where \( a_{ijs} \) = area (m\(^2\)) of patch ijs within specified neighbourhood (m\(^2\)) of patch \( ij \), and \( h_{ijs} \) = distance (m) between patch ijs and patch ijs, based on patch edge-to-edge distance, computed from cell centre to cell centre (McGarigal et al. 2012).

**Landscape Contagion**, unlike NN and the Proximity Index, is a pixel-based metric, defined as the relative frequency of finding a pixel of type \( i \) next to pixel of type \( j \) (Li and Reynolds 1993), and provides an indication of landscape texture. It can be used in relation to either vegetation patches or gaps, depending on the focal question, and measures the extent to which patches/gaps are aggregated; for example, a landscape with a few large clumped patches/gaps will have a higher value of contagion for these elements than a landscape comprised of many small, dispersed patches/gaps (McGarigal et al. 2012). For the purpose of this analysis, we assessed bare ground (gap) contagion, as corridors of open ground can increase overland movement of resources through the landscape (Bastin et al. 2002).
\[
\text{contagon} = \left[ 1 + \frac{\sum_{i=k}^{m} \sum_{i=k}^{m} \left( \frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}} \right) \ln \left( \frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}} \right)}{2\ln(m)} \right] (100)
\]

where \( P_i \) = proportion of the landscape occupied by patch type (class) \( i \), \( g_{ik} \) = number of adjacencies (joins) between pixels of patch types (classes) \( i \) and \( k \), and \( m \) = number of patch types (classes) present in the landscape (McGarigal et al. 2012).

For the purpose of identifying the relationship between vegetation pattern development and time, all sites in the restoration chronosequence spanning from 3-18 years old were used. Vegetation in sites less than 3 years old was deemed too small to be accurately detected and classified, and was not included in the analysis. To overcome issues associated with comparing sites of different extents (Leitão et al. 2006), a grid of 20 m x 20 m quadrats was overlayed onto each site, and metric values based on the mean of each quadrat were assessed. Additionally, this provided a measure of within-site variation. Linear, power and exponential models were applied to assess correlation trends with time, and the final model selected based on the highest \( R^2 \) value.
Figure 3: Classified images of vegetation (dark grey) from the restoration chronosequence. The images are representative 20 m x 20 m subsets from each age and treatment.

The relationship between vegetation pattern and hydrological function was assessed by extracting a single 32 m x 32 m area of classified imagery from directly over the 28 m x 28 m hydrological sampling plan (Figure 4) for the years tested (see Methods – hydrological functional data).
Figure 4: Hydrological sampling points (black circles) employed to assess saturated hydraulic conductivity and penetration resistance across the restoration chronosequence and natural analogue reference site. Grey denotes vegetation, white denotes interpatches. Each site is approximately 32m x 32m.
Hydrological function data

Hydrological data was collected as described in Chapter Two. Shallow-ripped sites aged 3, 7, and 15 years, a 15 year-old deep-ripped site, and a natural analogue reference site, were assessed in terms of saturated hydraulic conductivity ($K_s$) and soil impedance. The sampling strategy was based upon the work of (Gwenzi et al. 2011). Briefly, a 28 m x 28 m grid with sample points every 4 m was established at each site (Figure 3). Ten randomly selected points within the grid system were then selected to have 3 satellite sample points located on random azimuths 0.5, 1.0, and 2.0 m from the principle point, equating to a total of 94 sample points at each site. This allowed for variation to be assessed at a variety of spatial scales, as well as introducing a random component to the sampling strategy.

Field Data Acquisition

Surface Saturated hydraulic conductivity ($K_s$-PD) was measured at each sample point using a Phillip-Dunne permeameter according to the field protocols outlined in Muñoz-Carpena et al (2001;2002), and Muñoz-Carpena and Álvarez-Benedí (2002). An automated program, “Estimation of field saturated hydraulic conductivity (V:1.0) (Muñoz-Carpena and Álvarez-Benedí, 2002) was used to process the data and calculate $K_s$. A 20 mm “cookie cutter” extension was attached to the base of the permeameter in order to overcome highly hydrophobic conditions by increasing the relative pressure head and reducing the movement of the water laterally through the soil profile.

A CP20 cone penetrometer (RMIC) was used to measure soil impedance at 2 cm intervals, to a maximum depth of 50 cm at each sample location. In instances where soil resistance prevented the full insertion of the penetrometer, the maximum value measured was extrapolated for the remaining portion of the 50 cm. The location of each sample point was recorded to an accuracy of < 0.10 m using a Trimble Pathfinder Differential Global Positioning System (DGPS).
RESULTS

The effect of reference landscape position

Reference sites in low landscape positions had greater values for almost all landscape metrics than reference sites located high in the landscape (Figure 5). Vegetation cover and patch size variation were 20% and 30% higher respectively, while WMPS and Proximity Index values were twice as high as those recorded for the high landscape position (p < 0.001). Landscape gap contagion, however, was 40% greater in high sites compared with low sites (p < 0.05), and Nearest Neighbour values did not differ significantly according to landscape position (p < 0.05).

The relationship between pattern, time and treatment

All landscape metrics were significantly correlated with site age and followed an exponential trend (Figure 5). Vegetation Cover was mildly correlated with site age ($R^2 = 0.71$; p < 0.001) (Figure 5a). Cover in the oldest site (18 years) was 24% below mean vegetation cover scores observed in high reference sites and 42% below mean cover in the low reference sites. The 15 year-old, deep-ripped sites had twice as much vegetation cover, on average, as sites of the same age that had not been ripped (p <0.05). Removal of the deep-ripped site increased the $R^2$ value from 0.63 (p < 0.001) to 0.72. Cover in the 11 year-old site was notably lower than expected (7 ± 3.2%); approximately half that of the next youngest site (8 years; 13%) and one third of the next oldest site (12 years; 21%). According to the modelled trend, cover in restored sites should reach that of high reference sites after 19 years, and low sites after 22 years.

Mean WMPS was weakly correlated with site age ($R^2 = 0.62$; p < 0.001) (Figure 5b). The greatest WMPS in restored sites occurred in the 15 year-old, deep-ripped site; and was 10 times the value recorded in the 15 year-old, shallow tilled site. WMPS in both the 13 year-old site, and 15 year-old, deep-ripped site were approximately equal in value to that of the high reference site, yet were only half of the value recorded in the low reference position. According to the modelled trend, restored sites should have a WMPS equal to the low reference site after approximately 23 years.

Variation in patch size increased significantly with site age (p < 0.001) (Figure 5c). Once again, the greatest value in a restored site was recorded in the 15 year-old,
deep ripped site. This was also higher than that of the high reference site, but approximately 20% less than was recorded in the low reference site, which restored sites are predicted to achieve after around 26 years.

Euclidean Nearest Neighbour (ENN) decreased significantly through time ($p < 0.05$), suggesting decreasing distances between patches, although the strength of this relationship was weak ($R^2 = 0.46$) (Figure 5d). The mean ENN value for the 11 year-old site was located 2.7 standard deviations above the trend line (reflecting larger inter-patch distances, and when removed as an outlier, improved model fit ($R^2 = 0.64$). By far the largest variation in NN distances within a site was recorded in the 3 year-old restored area. After 18 years, the mean ENN trend is approximately 13 cm greater than high restored sites, and 18 cm greater than low restored sites. ENN is expected to equal that of high reference sites at 25 years, and low reference sites at 30 years.
Figure 5: Mean pattern metric values across the restoration chronosequence and in high and low reference sites. SE for high and low reference sites (respectively) for each metric are as follows; a) percent vegetation cover (SE: 0.89, 1.06); b) WMPS (SE:8.21, 7.57); c) patch area coefficient of variance (SE:2.06, 1.31) ;d) Euclidean Nearest Neighbour distance (SE: 0.03, 0.05); e) Proximity index (SE: 113.39, 152.67), and; f) Landscape contagion (SE: 0.72, 1.59), across a restoration chronosequence. Dashed lines indicate values measured in natural analogues in high and low landscape positions.

Proximity index values were mildly correlated with time (R² = 0.56; p < 0.05), yet were not significant (P > 0.05) (Figure 5e). Eighteen years after restoration, proximity values were still only half that of high reference sites and a third of those
recorded in low reference sites. The most notable exception occurred in the 15 year-old, deep-ripped site (36.31 ± 31.05) which was statistically indistinguishable from the high reference site. According to the modelled trend, restoration sites should attain high reference site Proximity index values after 24 years, and low reference site values at 27 years.

Landscape contagion values decreased significantly with increasing site age (p < 0.001, $R^2 = 0.72$) (Figure 5f). The mean gap contagion trend in restored sites intersected high reference sites at 10 years, and low reference sites at 18 years. The 11 year-old site deviated from the trend more than any other year (2.3 standard deviations). Removal of this site as an outlier would improve the model fit ($R^2$ of 0.80). Contagion in restored sites reaches high reference site values after 10 years, and low reference site levels after 20 years.

The relationship between pattern and function

Infiltration

Cover was highly correlated with $K_s$ ($R^2 = 0.80; p = 0.04$) (Figure 6). WMPS, patch area CV, Proximity and Contagion all trended positively with $K_s$ ($R^2 = 0.73, 0.62, 0.83$ and $0.61$ respectively) although were not significant ($p > 0.05$; statistical power = $0.26, 0.17, 0.40$ and $0.16$ respectively). Nearest Neighbour trended negatively with $K_s$ ($R^2 = 0.47; p > 0.05$, power = 0.10).
Figure 6: Correlation between $K_s$ and landscape metrics.

**Penetration resistance 10 cm**

Soil impedance at 10 cm depth was strongly positively correlated with percent cover ($R^2 = 0.87; p = 0.02$), WMPS ($R^2 = 0.91; p = 0.012$) and the proximity index ($R^2 = 0.99, p= 0.016$) (Figure 7). Correlation was strong between Patch size CV, and soil impedance at 10 cm ($R^2 = 0.82$), yet statistical power was weak (power = 0.38), whilst NN and contagion were both weakly correlated ($R^2=0.28$; power = 0.06 and $R^2 = 0.37$; power = 0.07, respectively).
Vegetation cover and soil impedance values at 50 cm depth trended negatively ($R^2 = 0.73$), as did WMPS ($R^2 = 0.78$), yet were not statistically significant ($p = 0.154$, power = 0.26). Patch area CV and Proximity index values were both moderately correlated ($R^2 = 0.64$ and 0.71, yet lacked statistical power (0.18). Nearest Neighbour and Contagion metrics were weakly correlated with soil impedance at 50 cm ($R^2 = 0.38$ and 0.13, respectively).

**Penetration resistance 50 cm**

Figure 7: Correlation between PR 10 cm and landscape metrics.
Vegetation structure is a key attribute for defining and describing plant communities and is one of the Society for Ecological Restoration’s listed criteria for assessing restoration success (Clewell et al. 2004). Although commonly studied in systems defined by regular patterns of discrete patches and gaps (Facelli and Brock 2000, Rietkerk and Van de Koppel 2008, Borgogno et al. 2009), this study showed that irregular vegetation pattern in Mediterranean systems also develops according to a trajectory that is predictable as a function of time. Additionally, it illustrates how the efficacy of management intervention techniques such as substrate ripping can be
accurately and efficiently assessed, and how pattern analysis can be used to focus on-ground assessment of restoration.

In order to establish appropriate natural analogue reference targets, we first assessed natural variation of vegetation spatial pattern in undisturbed systems in relation to topographical position. Vegetation pattern in low landscape positions was characterised by higher cover, WMPS, patch size variability, and proximity index values, and lower gap contagion, compared with high landscape positions. Plant density in high and low natural analogues does not differ significantly in these systems (Chapter two), and hence differences in pattern may be attributed to commonly observed shifts in species distribution in response to a topographic (i.e. resource) gradient (e.g. Oliveira-Filho et al. 1994, Tateno and Takeda 2003). In Banksia woodland systems, species distribution is a function of rooting depth and plant-water relations resulting in a greater abundance of shallow-rooted shrub species in high landscape positions, relative to deep-rooted tree species lower in the landscape (Groom 2004). Reconstructed sites in the study area are typically situated low in the landscape, and hence, from a hydrological perspective, a low-lying site is the most appropriate natural analogue for vegetation structure. However, as noted in Chapter two, topsoil used in the restoration process and seed collections are preferentially sourced from high landscape positions, resulting in a species composition more similar to high rather than low topographical position, which in turn may affect vegetation pattern.

Significant correlations with time were evident for all vegetation pattern metrics, with trajectories clearly heading towards a reference state. This supports the use of pattern metrics as a tool for indicating restoration success in Mediterranean Banksia woodlands, based upon vegetation structure criteria. It is interesting to note that the presence of vegetation was generally not correlated with hydrological properties at the local/point scale in Chapter 2, yet closely associated at the landscape scale in this chapter. This suggests that constraining the effects of vegetation on soil properties to the extent of the above-ground biomass via a binary classification does not capture the effects of plant roots on hydrological properties that may well exceed the horizontal canopy (Thompson et al. 2010).

In this study, pattern analysis suggests that restored sites are still substantially short of natural analogue reference targets. After 18 years, the oldest restored site was still significantly lower in vegetation cover, WMPS, patch size variation, and proximity...
index values than both high and low reference sites, as well as being significantly lower in mean gap contagion compared with low reference sites. Not only is this a product of larger individual plant sizes in reference sites, but plant density in restored sites is also approximately half that of low restored sites (Chapter Two). Similarly, Miller et al (2010) compared the spatial distribution of stems in 16 and 24 year-old restored Banksia shrubland with natural analogues and found that stem densities were approximately 50 - 75% of those recorded in reference sites. However, they also found that restored sites were characterized by a higher abundance of larger gaps. In our study, vegetation cover in the oldest restored sites was 30% lower than the low-lying natural analogue, yet gap contagion was approximately equal. This suggests that gaps are more abundant yet smaller in restored Banksia woodland sites than in the natural analogue, which may indicate that spatially heterogeneous hydrological characteristics associated with patch-forming facilitative processes common to natural, water-limited systems (Facelli and Brock 2000, Puigdefábregas 2005) may not be the dominant drivers of vegetation spatial pattern in restored sites. Rather, pattern in these restored sites may be confounded by the homogeneous application of restoration treatments such as substrate ripping which may reduce the benefits for individual plants of being situated in close proximity to their neighbours (e.g. increased infiltration, reduced penetration resistance). Natural settling processes (Kozlowski 1999, Archer et al. 2002) following substrate ripping may reduce these effects with time. Additionally, whilst this study focussed on hydrological drivers of spatial pattern (i.e. water resources), other resources such as soil nutrients are important drivers of richness and density in restored sites (Rokich and Dixon 2007), and are likely influencing the development of vegetation pattern also.

Currently, post-mining restoration monitoring obligations commonly do not exceed 5 years (Ruiz-Jaen and Mitchell Aide 2005). According to the trend analyses of this study however, restored sites will likely take up to 24 years to attain structural characteristics equal to the high reference site, and up to 30 years for the low reference site. This could be used to support a case for increasing the duration of operator responsibility. Analysis of remotely-sensed vegetation spatial pattern may provide cost-effective means of ongoing monitoring across a more ecologically suitable timeframe, alerting land managers of potential restoration issues that may arise.

Analysis of vegetation pattern clearly detected the effects of specific restoration treatments. The 15 year-old, deep-ripped site from this study is a case in point; this site
was characterised by significantly higher cover, WMPS, patch size variation, and proximity index values compared with the unripped site of the same age. This suggests that deep-ripping has contributed to increasing resource availability, most likely through reducing soil bulk density and facilitating ease of root movement through the substrate. However, as illustrated in Chapter Two, interactions between deep-ripping and high rainfall in the first year immediately following restoration can affect species composition which in turn may manifest as significant changes to vegetation structure. In this study, similarities in pattern metrics between the 15 year-old deep-ripped site and reference sites are primarily driven by the abundance of a large, post-disturbance colonizer shrub species (*Adenantheros cygnorum* Diels) in the restored site, and hence, structural characteristics should be assessed in conjunction with compositional data when assessing restoration development.

Additionally, the analysis of vegetation pattern may be used as a rapid and cost-effective indicator tool to detect areas where restoration issues may be present. For example, in the 11 year-old site, vegetation cover, WMPS, proximity, nearest neighbour, and gap contagion values deviated substantially from the expected range and may be indicative of a relative reduction in resource availability (e.g. insufficient seed, poor topsoil handling, erosional processes, or restricted access to water due to compaction. These findings can be used to direct further, on-ground, site enquiries.

Clear, broad trends were evident in the relationships between vegetation pattern and hydrological properties at the landscape scale, supporting the use of pattern metrics as an indicator tool of functional development in restored systems. Vegetation cover was significantly correlated with $K_s$ and PR 10 cm, but not PR 50 cm, and is indicative of increased plant-soil interactions in the substrate surface; in a study of restored sites in the same area, Rokich et al (2001) found that root mass in the top 10 cm of the substrate was four times greater than at 40 cm depth.

Both vegetation cover and the proximity index were highly correlated with hydrological function, and may be mutually redundant in this case. Mean proximity would likely be more informative than cover in situations where vegetation is less evenly dispersed across sites, and hence, may affect the sites capacity to capture resources. The strong correlation between proximity index and function found in this study contrasts with the findings of Bastin *et al.* (2002), who found that proximity index failed to accurately rank four sites of known functional status. Nearest Neighbour and
gap contagion were less effective in distinguishing hydrological function, in part due to low correlation and low statistical power. Furthermore, development of vegetation structure is still limited after 15 years in non-deep-ripped sites, and hence, sampling was biased towards lower cover values.

Vegetation spatial pattern in restored Mediterranean Banksia woodlands developed through time in a statistically significant and relatively predictable manner, supporting the employment of vegetation structure benchmarks, indexed against site age, as indicators of restoration success. Broad trends indicated a relationship between pattern and hydrological function which may be further resolved with additional study and increased replication. Given the rarity with which ecosystem function is reported in restoration projects (Ruiz-Jaen and Mitchell Aide 2005), in conjunction with the relative ease and cost-effectiveness of acquiring and analysing remotely sensed data, there is a strong case for regular cover assessments as a means of monitoring restored areas over a temporal scale appropriate to the time required for system development.

Implications for industry

- Irregular vegetation pattern in Mediterranean systems develops according to a predictable trajectory that can be used to track development of restored systems, and to assess the efficacy of restoration treatments.
- Deviations from the expected trajectory may be indicative of restoration issues and used to direct and focus further on-ground assessments.
- Clear, broad trends were evident between vegetation pattern and hydrological function, supporting the employment of pattern assessment as an indicator of hydrological restoration.
CHAPTER FIVE

General Discussion

The long term drivers of restoration success and efficacy of restoration intervention techniques are currently poorly understood (Menz et al. 2013). The study of restoration chronosequences is a useful tool for understanding ecological processes that operate on temporal scales outside the time constraints of conventional empirical research, and this study examined a 19 year-old, post-mine restoration chronosequence of Banksia woodland in a biodiversity hotspot, in Mediterranean south-west Western Australia. Restoration success in this study was assessed in terms of plant community composition, ecosystem function, and vegetation structure; individually, these ecosystems attributes are commonly employed in restoration monitoring but rarely are all three assessed together (Ruiz-Jaen and Mitchell Aide 2005). This approach facilitated a holistic examination of the long term drivers of ecosystem development and restoration success in post-mine disturbed sites.

Broadly, the three experimental chapters within this research thesis are structured around each of these ecosystem attributes; Chapter Two examined changes in plant community composition over time, and the effects of rainfall, site physical characteristics, and substrate manipulations, in terms of four criteria; species richness, plant density, similarity to natural analogue reference sites, and vegetation cover. Chapter Three assessed the drivers behind restoring hydrological function in post-mine sites over time, with particular focus on the effects of substrate ripping and plant-soil interactions in determining infiltration rate and soil impedance. Chapter Four examined the development of vegetation spatial pattern through time in Banksia woodland systems, and related that pattern to hydrological function.

This chapter aims to integrate and synthesize the findings of each study in order identify the overall effects of the common themes; time, environmental drivers, and restoration intervention techniques. It discusses the benefits and limitations of using restoration chronosequences as a tool for studying the development of restored systems through time, before addressing the suitability of other methodologies employed in this
research thesis. Finally, it provides general conclusions and recommendations from this body of work that may improve future restoration outcomes.

**Time**

Almost all the parameters assessed within this thesis were strongly mediated by time since establishment; in Chapter Two, vegetation cover was positively correlated with time, while plant density and species richness were negatively correlated. These overall relationships were consistent, regardless of environmental drivers or restoration intervention techniques, and are indicative of density-dependent mortality affecting restoration outcomes; as plant individuals grow (cover increase), competitive interactions reduce the number of individuals that can occupy a given area (decreasing plant density). Hence, samples of finite numbers of individuals will result in fewer species by chance alone (Stevens and Carson 1999). Even after 19 years, species richness and density were significantly lower than in natural analogue reference sites, and suggest that vertical structure in restored sites is underdeveloped relative to reference communities, and hence, competition for physical space is greater in restored sites than in reference sites. Substantially more time is required for restored sites to reach reference levels in these criteria. In fact, Chapter Four established that according to the current modelled trend through time, restored sites would take approximately 24 years to reach the equivalent structure of high reference sites, and up to 30 years to reach low reference sites. This begs the question, what is an appropriate time frame for which industry/restoration practitioners should be accountable and responsible for restoration monitoring and management, and should this time frame be set according to the complexities of the system? Certainly, 5 years, as is currently common (Ruiz-Jaen and Mitchell Aide 2005) may need to be revised.

In Chapter Three, natural settling processes in deep-ripped sites, and processes associated with rebound in shallow-ripped sites, had a significant effect on hydrological function through time. In Chapter Four, vegetation pattern characteristics (despite being highly irregular in comparison to pattern in arid and semi-arid systems), changed significantly, and predictably with time, supporting the employment of pattern metrics as a rapid technique for assessing vegetation structure as a criterion for restoration success, indexed accordingly for site age.
Environmental drivers

Plant-available water is the primary driver of biological activity in systems that experience drought-stress for at least part of the year (Moles and Westoby 2004) (arid, semi-arid, and Mediterranean biomes), but most importantly, rainfall during the summer/dry period was found to be critical in this study. Plant density and species richness were greatest when rainfall in the first winter immediately following site restoration was low, and then followed by high summer rainfall. These conditions appear to prevent the dominance of high cover, disturbance species that may otherwise out-compete many other species, as appears to be the case in the 15 year-old, deep-ripped site. These findings translated to the same 15 year-old deep-ripped site being characterized by the highest vegetation cover, Weighted Mean Patch Size, patch size variation, and proximity index values in Chapter Four, highlighting the association between rainfall and vegetation structure.

Likewise, site aspect plays an important role in determining plant available water and had a similar effect on plant community composition in this study. Easterly-orientated slopes were characterized by higher species richness and plant density than westerly-orientated slopes. We suggest that this is attributable to higher evaporative pressures from prevailing winds in easterly-orientated sites, and increased rainfall interception on west-facing slopes. These conditions likely result in larger individuals occupying westerly slopes which in turn promotes competitive interactions and density-dependent mortality, while niche-limiting pressures on easterly slopes provide habitat for a greater number of smaller species (Stevens and Carson 1999).

The globally observed trend of a positive correlation between vegetation cover and infiltration rate is strongest in mesic systems with substrates characterized by a high clay fraction, and declines with increasing rainfall and decreasing clay fraction (Thompson et al. 2010). This decline was apparent in the results of our study, whereby higher rainfall and sandy substrates with naturally high rates of infiltration, typifying Mediterranean Banksia woodlands, was significantly yet weakly associated with increased infiltration. Additionally, specialized root morphologies (i.e. proteoid roots) that extend well beyond the extent of above-ground biomass are also likely to be confounding a binary relationship between vegetation cover and hydrological properties.
Management intervention techniques

The effects of ripping depth on plant community composition were largely dependent on rainfall, with deep-ripped sites performing best under high rainfall conditions, and sites that had been shallow-ripped performing better under low-mean rainfall than high rainfall. As suggested in Chapter two, this is likely the product of increased soil water retention via smaller pore spaces in shallow ripped sites resulting in higher survivability during dry periods, whilst reduced soil impedance at depth associated with deep ripping facilitated greater root movement through the substrate and produced better results under high rainfall. We suggest that restoration practitioners could use ENSO predictions (Sitters et al. 2012) to predict seasons of high rainfall in order to guide ripping treatment selection. Alternatively, they could adopt a hybrid approach (both shallow and deep ripping), whereby ripping is applied in a heterogeneous fashion across the site. This would more closely mimic natural conditions, whereby heterogeneity in soil properties and hydrological function facilitates water harvesting, as well as providing a form of bet-hedging for variable rainfall.

In Chapter three we found that hydrological function is indeed heavily mediated by ripping treatment, and that commonly accepted relationships between vegetation and soil properties were not evident in a predominantly sand system, where whole-of site treatments (i.e. substrate ripping) create relatively homogenous hydrological conditions. Whilst vegetation cover had little effect on hydrological properties at the local scale (Chapter three), in Chapter four we identified strong trends between vegetation pattern and hydrological function. We suggest that ripping treatments drive hydrological function, which in turn create conditions that support higher vegetation cover, rather than vegetation mediating hydrology, as is often observed in natural systems (Thompson et al. 2010). Management should therefore identify the ripping treatment that is most suitable to both substrate and rainfall conditions.

The incorporation of overburden material was associated with increased richness and plant density in one-year-old sites, but a lack of replication and controls prevented direct comparison in older sites. Overburden is characterized by higher organic matter content relative to the material below (Rokich et al. 2001), and has been found to greatly increase water storage capacity in sandy soils (Sands et al. 1979). A disproportionately high rate of plant mortality was observed in the young overburden
site in the summer following the vegetation survey, aligning with the findings of Rokich (2001) who found that the incorporation of overburden material in site reconstruction was associated with increasing soil impedance, and perhaps may be attributed to increased natural settling processes following disturbance. Further research is required to ascertain whether modified ripping treatments, such as hybrid ripping, may counter these settling processes in sites reconstructed with overburden material whilst still providing the benefits to richness and density.

**Restoration chronosequences**

Chronosequences are a useful tool for assessing the development of restored sites over time (Anderson et al. 2007), but caution must be exercised when interpreting data outputs. Post-mine restoration is necessarily governed by the need for economic efficiency, and adaptive management whereby best practice restoration changes with time result in chronosequences that are often characterized by unequal replication, discontinuity, and lack of controls. This meant that in Chapter one, some of the parameters used to predict/explain plant community composition were poorly replicated (e.g. the use of overburden in the reconstruction process). Likewise, in Chapter two, only two deep-ripped sites (aged 0 and 15 years) were available for inclusion in the model; whilst the model provides a best estimate when interpolating between these two points, additional data is required before robust interpretation can be made.

Despite this, much can be gleaned from existing restoration chronosequences, and to omit studying them due to less than ideal replication is to pass up a valuable opportunity to contribute knowledge to the field of restoration ecology. Conducting future restoration trials with an element of scientific robustness (i.e. sufficient replication and controls) can enhance this contribution substantially.

**Methodology**

Chapters three and four utilized classified remotely-sensed imagery in order to discern vegetation cover and pattern. Whilst an exceptionally powerful tool, classified imagery is not without its limitations, particularly when attempting to recognize very small features, such as plant seedlings (e.g. < 40 cm²). Spectral signatures can be confounded when ground reflectance interacts with spindly or needle-like leaf
morphologies, producing a mixed-pixel effect (Sawaya et al. 2003). For this reason, we did not classify plants in sites < 3 years old. By this age, plant sizes were generally large enough to be detected confidently. Whilst classifying imagery at very high spatial resolution (i.e. 10 cm pixel size) provides a high degree of accuracy on the one hand, it also presents difficulties when georeferencing and ground-truthing data. Image noise and hence, the potential for classification error is amplified at this spatial resolution and may have significant impacts on results.

Shadow is also difficult to deal with, particularly as its classification and removal may substantially affect pattern metrics. Acquiring aerial imagery at midday when the sun is directly overhead minimizes but does not negate this issue (Sawaya et al. 2003). In this study, shadow was mainly present in older restored sites and reference sites, where taller vegetation produces more shadow. In these sites where vegetation cover far exceeds gaps, we deemed it most accurate to classify shadow as vegetation. Whilst providing the most accurate classification, this would also affect the results. Integrating LIDAR (Light Detection and Ranging) data to provide a 3-dimensional surface would enable vegetation to be classified according to height criteria as well as spectral criteria (Lefsky et al. 2002), and therefore address the issue of shadow.

Despite these challenges, classified remotely-sensed aerial imagery does provide restoration practitioners with an excellent tool for assessing vegetation change (growth and plant condition) and vegetation pattern at the landscape scale (Xie et al. 2008), as well as providing a rapid technique for assessing functional attributes over large spatial extents. In chapter four, pattern metrics suggested that vegetation structure in the 15 year-old, deep-ripped site was the most similar to reference targets. Whilst this was supported by hydrological function data, measures of Bray-Curtis similarity were no greater than in other sites, and species cover data indicated that the site was in fact dominated by a disturbance species, A. cygnorum that does not occur commonly in natural analogues. This highlights the fact that vegetation structure as a restoration criteria should not be considered in isolation as it may be misleading. Additionally, many pattern metrics may only be compared in sites of approximately equal extent (Leitão et al. 2006), which necessitated the need to subdivide sites into 20m x 20m subsets in this study. This process, however, is likely to also alter some metric outputs (e.g. mean patch size) and hence, is not without limitations.
Conclusions and recommendations

This thesis represents a rare, holistic examination of the manner in which restored post-mine disturbance landscapes develop through time. Ecosystem characteristics and processes are inherently interrelated and quantifying and understanding the effects of natural environmental drivers and management intervention techniques on plant community composition, hydrological function, and vegetation structure may assist practitioners to improve future restoration outcomes.

Key recommendations specific to Banksia woodland restoration identified in this study include:

- Hybrid ripping to promote site heterogeneity and mimic natural conditions.
- Incorporate overburden to increase soil-water retention and improve restoration outcomes
- Remove or reduce abundance of disturbance species when competitive effects begin to outweigh facilitative/restorative effects.
- Use vegetation pattern as an indicator of function and to direct on-ground monitoring.
- Conduct restoration with scientific rigour in order to contribute to developing science of restoration ecology.
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